Office Memorandum

To: Members of the Executive Board — February 27, 2014

From: The Secretary

Subject: **Getting Energy Prices Right—from Principle to Practice—Supplementary Chapters and Appendices**

The attached paper containing supplementary chapters and appendices is being issued as a supplement to the report on Getting Energy Prices Right—from Principle to Practice (FO/DIS/14/29, 2/27/14).

It is intended that this supplement together with the main paper will be published in July 2014.

Questions may be referred to Mr. Keen (ext. 34442), Ms. Perry (ext. 36392), and Mr. Parry (ext. 39724) in FAD.

This document will shortly be posted on the extranet, a secure website for Executive Directors and member country authorities.

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Supplement 1

INTERNATIONAL MONETARY FUND

Getting Energy Prices Right: From Principle to Practice

Supplementary Chapters and Appendices

February 2014

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Chapter 1. Energy Systems, Environmental Problems, and Current Fiscal Policy: A Quick Look

Fossil fuels are used pervasively to generate electricity, power transportation vehicles, and provide heat for buildings and manufacturing processes. Fuel combustion produces CO_2 emissions and various local pollutants, while use of transportation vehicles also causes road congestion, accidents and (less importantly) pavement damage.

This chapter provides a quick look at energy systems, elaborates on their major environmental impacts, and discusses existing fiscal provisions affecting energy. Although the information here is not directly relevant for estimating corrective fuel taxes, it provides broader context and suggests why corrective taxes, and their impacts, are likely to differ considerably across countries.

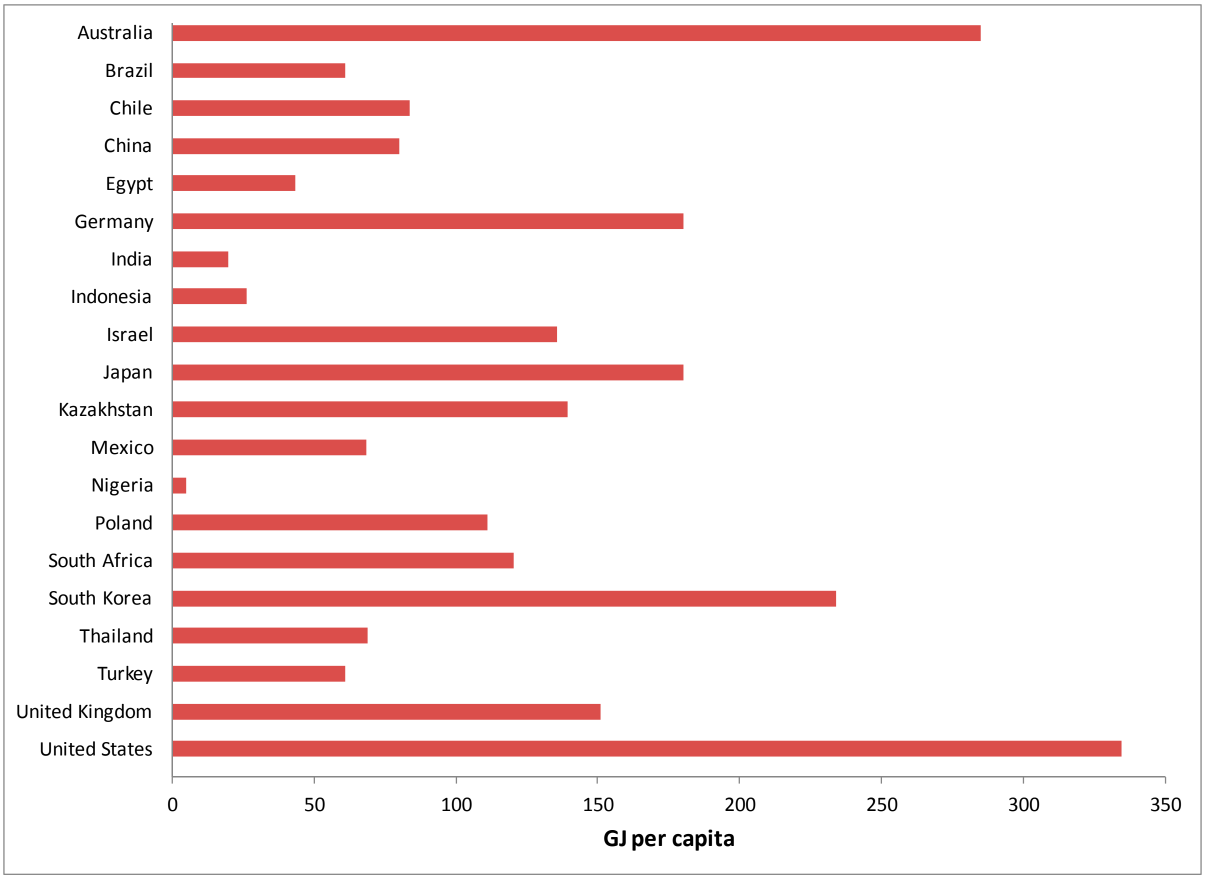
A. Overview of Energy Systems

Although (insofar as possible) this volume presents results for 144 countries, a focus on 20 countries is used to illustrate how corrective taxes and their impacts vary with per capita income, fuel mixes, population density, road fatalities, and so on. This subsection provides some basic statistics for these countries for the year 2010 (or the nearest year for which data is available).

Figure 1.1 shows primary energy consumption (i.e., the energy content of fossil and other fuels prior to any transformation into electricity) in gigajoules (GJ) per capita. Energy consumption is highest in the United States and roughly half as high in countries like Germany, Japan, and the United Kingdom. At the other end, energy consumption per capita in India, Indonesia, and Nigeria is 8 percent or less of that in the United States.

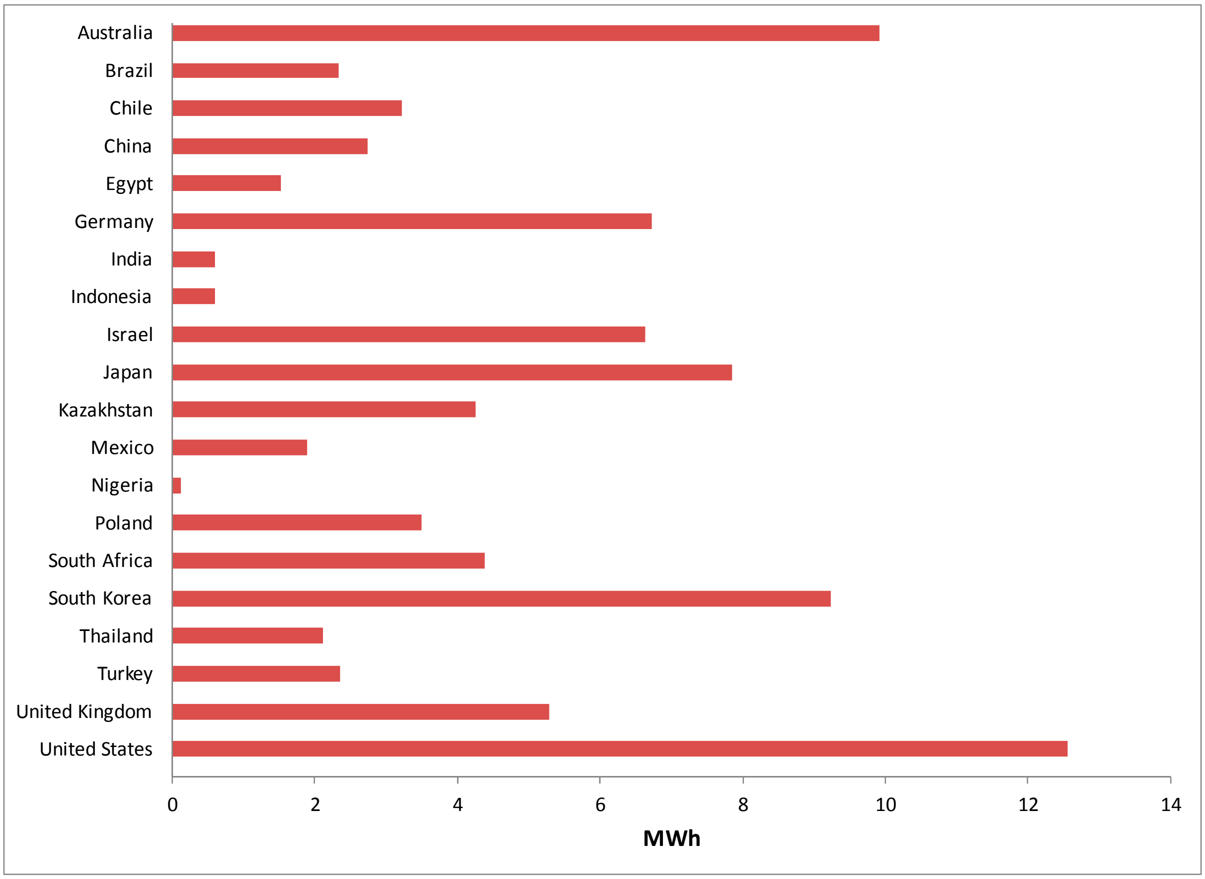
These differences primarily reflect differing reliance on electricity and motor vehicles. As indicated in Figure 1.2, relative differences in electricity consumption per capita broadly follow those (in Figure 1.1) for total energy consumption per capita. In the United States, for example, people tend to live in relatively large-sized homes (implying higher energy use), while in Indonesia and India, about 35 percent of the population lacks access to electricity, and about 50 percent in Nigeria (World Bank, 2013).

Figure 1.1. (Primary) Energy Consumption Per Capita, Selected Countries, 2010



Sources: US EIA (2013).
Notes: Primary energy consumption is the energy content of fossil and other fuels prior to any transformation into power generation.

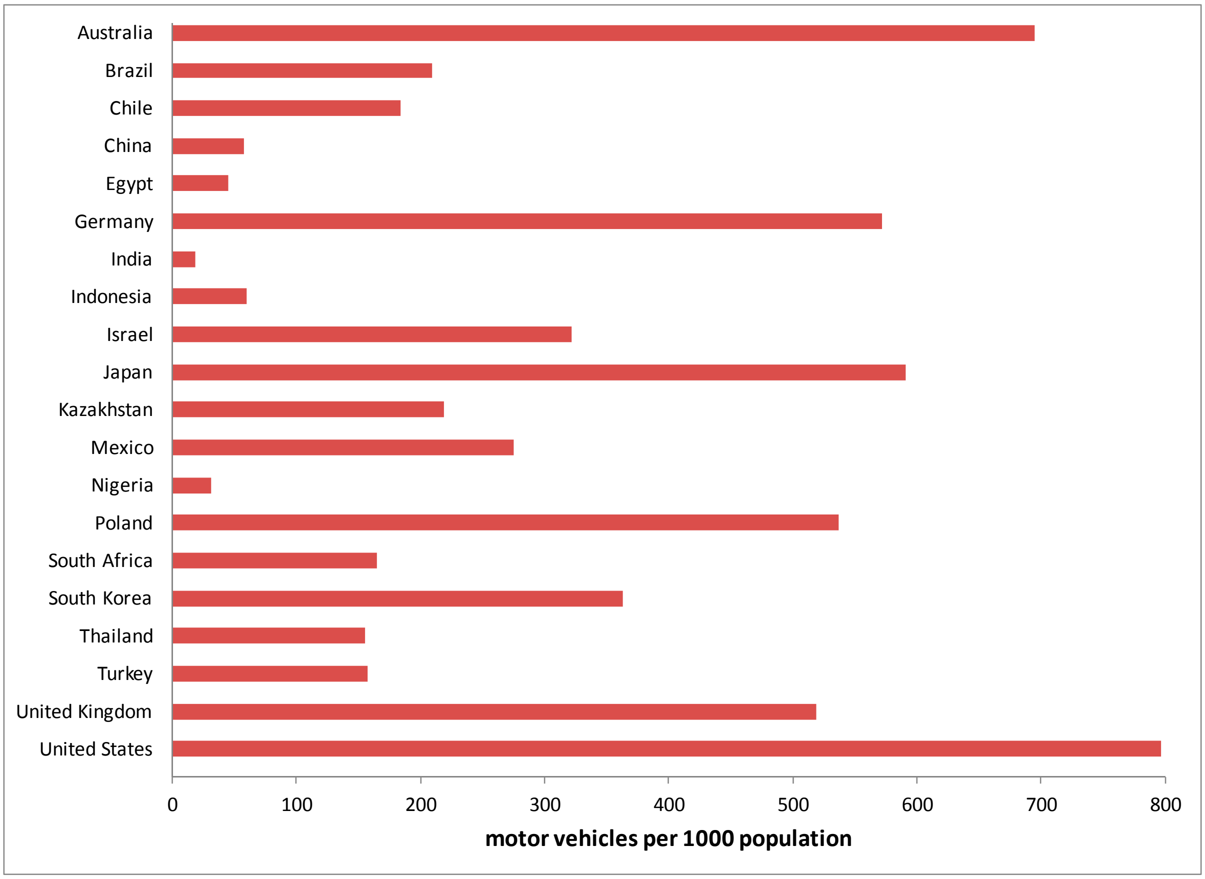
Figure 1.2. Electricity Consumption Per Capita, Selected Countries, 2010



Sources: US EIA (2013).
Notes: Electricity consumption includes residential and industrial uses.

Similarly, countries with lower per capita energy consumption also tend to have lower vehicle ownership rates (Figure 1.3). In the United States and Australia, for example, there are about 800 and 700 motor vehicles per thousand people respectively, while in China, Egypt, India, Indonesia, and Nigeria, there are fewer (in some cases far fewer) than 100 vehicles per thousand people.

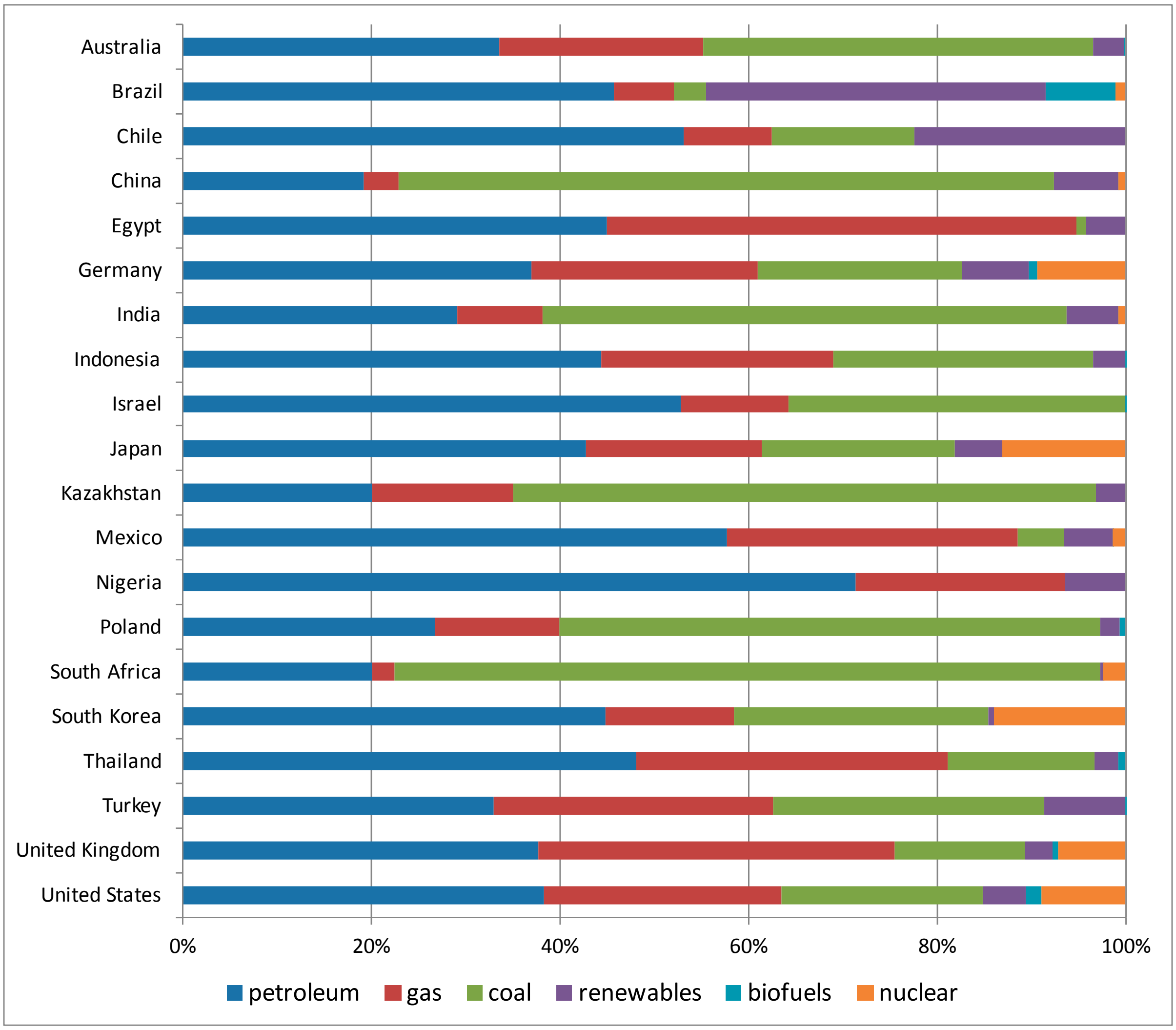
Figure 1.3. Motor Vehicle Ownership Rates, Selected Countries, 2010 (or thereabouts)



Sources: World Bank (2013).
Notes: Motor vehicles include cars, trucks, and buses. However two-wheelers (which are used pervasively in many Asian countries) are not included in the data.

The scale of environmental problems also depends critically on a country's fuel mix, and again there are large differences, as indicated in Figure 1.4. For example, coal constitutes more than half of total energy consumption in China, India, Kazakhstan, Poland, and South Africa, but 5 percent or less in Brazil, Egypt, Mexico, and Nigeria. Petroleum varies from 19 percent of energy consumption in China to 71 percent in Nigeria. And natural gas varies from 2 percent in South Africa to 50 percent in Egypt. To varying degrees, countries rely on renewables (wind, solar, hydro, etc.), though there are challenges to their growth (e.g., the intermittent supply from wind and solar power and the mismatch between their ideal location and urban centers).

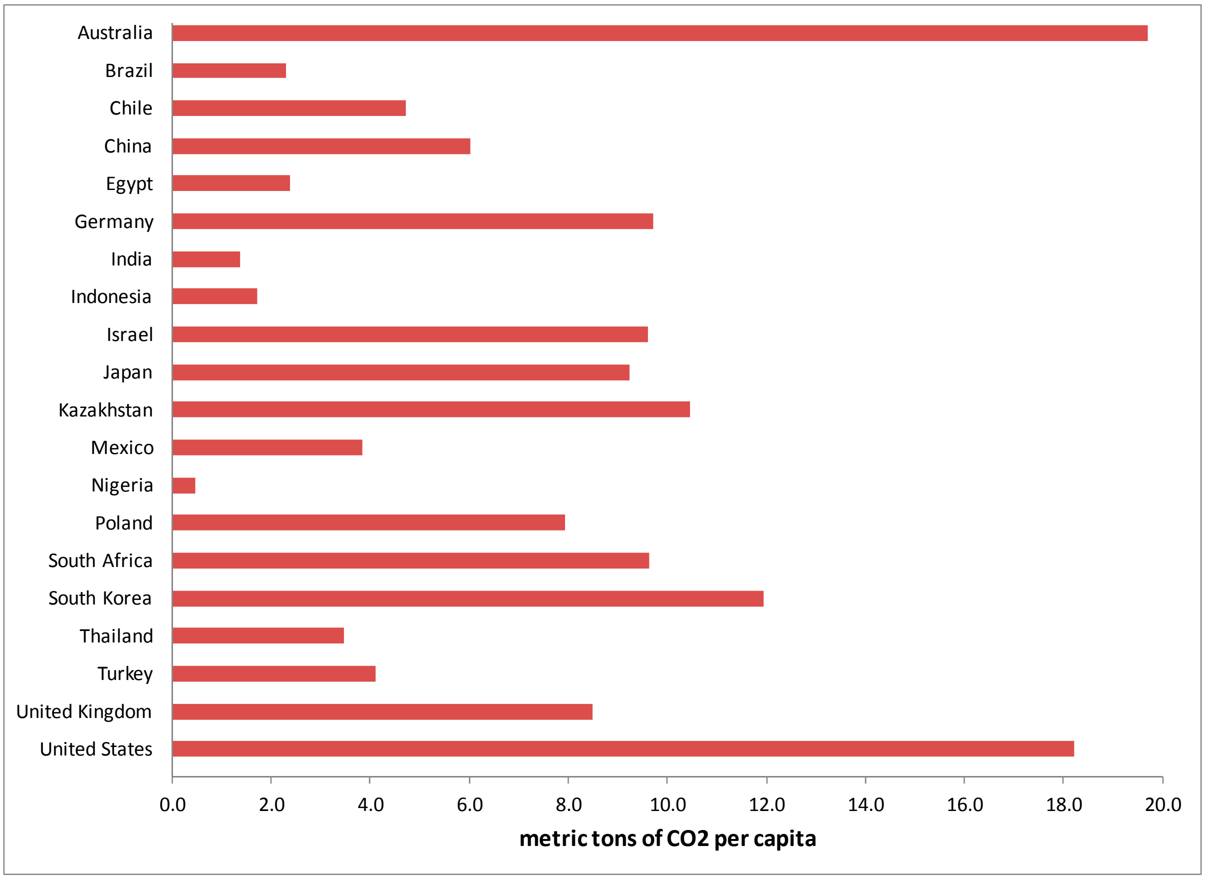
Figure 1.4. Share of Final Energy Use by Fuel Type, Selected Countries, 2010 (or thereabouts)



Sources: US EIA (2013)
Notes: The figure shows the share of primary energy (from direct fuel combustion) and secondary energy (primarily power generation) attributed to different fuels, where fuels are compared on an energy equivalent basis.

Differences in energy consumption per capita, in particular, but also in fuel mixes, explain differences in (energy-related) CO_2 emissions per capita, shown in Figure 1.5. For example, annual emissions per capita are almost 20 (metric) tons in Australia and United States, countries that use a lot of energy and also have relatively emissions intensive fuel mixes.

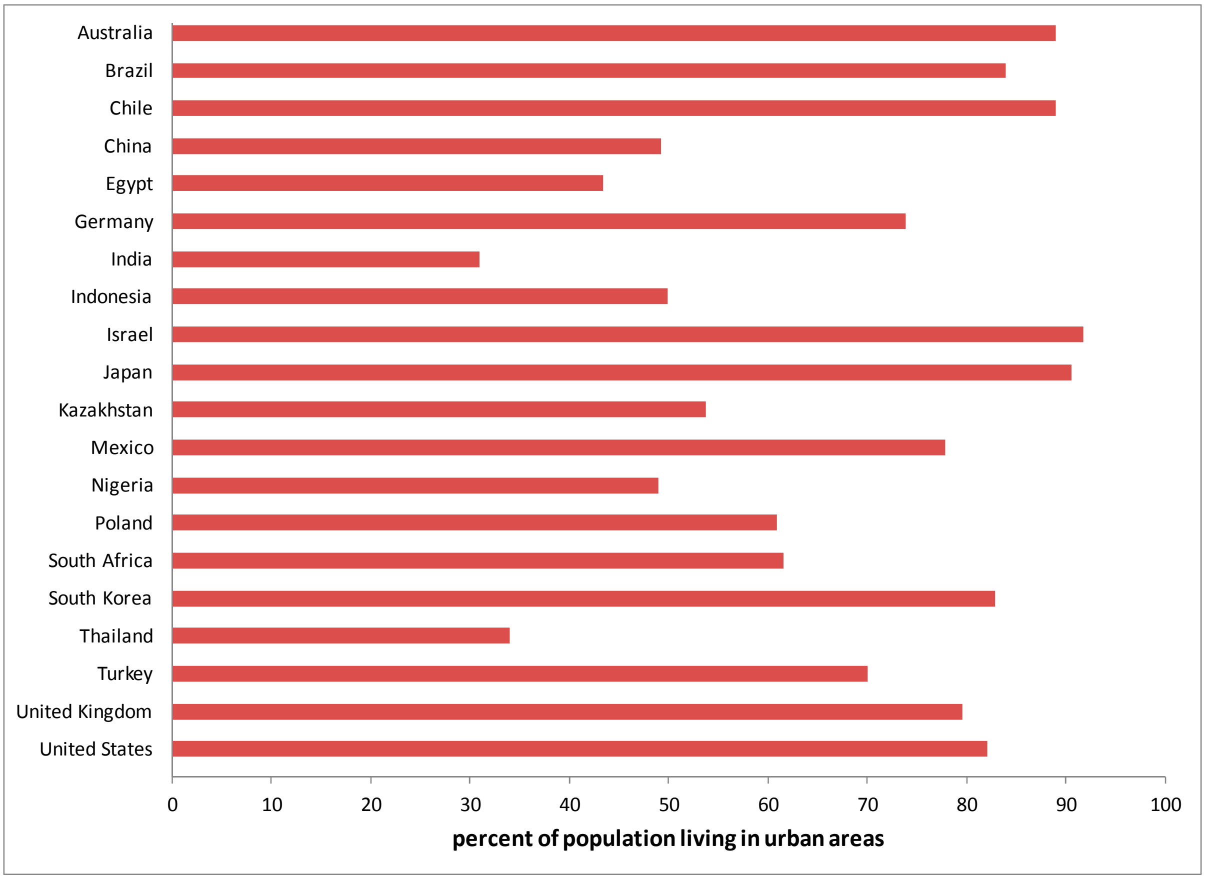
Figure 1.5. CO_2 Emissions Per Capita, Selected Countries, 2010



Sources: US EIA (2013).

The severity of environmental problems also depends on population density (greater density generally implies more people exposed to local air emissions and more crowded road systems), which again varies considerably across countries. For example, the share of the population living in urban areas varies from about 90 percent in Australia, Chile, Israel, and Japan to less than 40 percent in India and Thailand (Figure 1.6).

Figure 1.6. Urban Population, Selected Countries, 2010



Sources: World Bank (2013).
Notes: Urban population refers to the share of people living in urban areas as defined by national statistical offices.

B. Environmental Side Effects

Fossil fuel use is associated with a variety of environmental side effects, or 'externalities'. An (adverse) externality occurs when the actions (e.g., fuel combustion) of individuals or firms imposes costs on others that the perpetrators do not take into account. Externalities call for policy intervention, principally charges that are (a) directly targeted at the source of environmental harm and (b) set at levels to reflect environmental damages (Chapter 1 of the main report).

Here the main externalities of concern for this study are discussed, namely CO_2 emissions, local air pollution, and broader costs of vehicle use. Some further environmental problems are discussed in Box 1.1.

Box 1.1. Broader Environmental Effects Beyond the Study Scope

A variety of other costs associated fossil fuel production and use are not considered here, due to one or more of the following factors (see, e.g., NRC 2009, Ch. 2, for further discussion):

- these costs might be taken into account by individuals and firms
- they may be modest relative to the environmental damages estimated below
- they might be too difficult to quantify
- and they may call for policies other than fuel taxes

Examples include:

Additional pollutants. Carbon monoxide (CO), a by-product of fuel combustion, reduces oxygen in the bloodstream, posing a danger to those with heart disease, but when released outside its concentration is usually not sufficient to cause significant health effects. Lead (Pb) emissions cause neurological effects, especially for children, with potentially significant impacts on lifetime productivity (e.g., Grosse and others, 2002; Zax and Rees, 2002). However, lead has been, or is being, phased out from petroleum products in many countries. Various other toxics (e.g., benzene) are generally not released in sufficient quantities to cause health damages that are significant relative to those from the pollutants considered here.

Upstream environmental impacts. Environmental impacts occurring during fuel extraction and production include:

- de-spoiling of the natural environment (e.g., mountaintop removal for coal, accidents at oil wells)
- wastes from fuel processing (e.g., slurry caused by ‘washing’ raw coal)
- emissions leaks during fuel storage (e.g., due to corrosion or evaporation at underground tanks at refineries and gasoline stations)
- further leakage during transportation (e.g., spills from oil tankers).

However, per unit of fuel use, these damages appear to be small relative to those estimated here (e.g., Jaramillo, Griffin, and Mathews, 2007; NRC, 2009, Ch. 2) and these problems call for other interventions (e.g., double hull requirements for tankers, mandatory insurance for accident costs, requirements that mined areas be returned to their pre-existing vegetative state) rather than fuel taxes.

Occupational hazards. For fossil fuel extraction industries occupational hazards include, for example, lung disease from long-term exposure to coal dust, coal mine collapses, explosions at oil rigs. Individuals may account for these risks however, when choosing among different occupations (a long-established literature in economics suggests that higher-risk jobs tend to compensate workers through higher wages, see, for example, Rosen, 1986). And to the extent policy intervention is warranted (perhaps because individuals understate risks), more targeted measures (e.g., workplace health and safety regulations) would be more efficient than fuel charges.

Indoor air pollution. Indoor air pollution causes an estimated 2.7 million deaths worldwide each year (e.g., Burnett and others, 2013). For example, in low-income countries burning coal in poorly ventilated cooking stoves or open fires can create serious pollution-related health problems (e.g., Ezzati, 2005). Raising consumer coal prices may not be the best policy to deal with indoor air pollution, however (not least because it may promote equally harmful use of biomass), at least until cleaner energy sources (e.g., charcoal, natural gas, electricity, or even processed coal that burns more cleanly), and better technologies (e.g., better ventilated stoves), are made available.

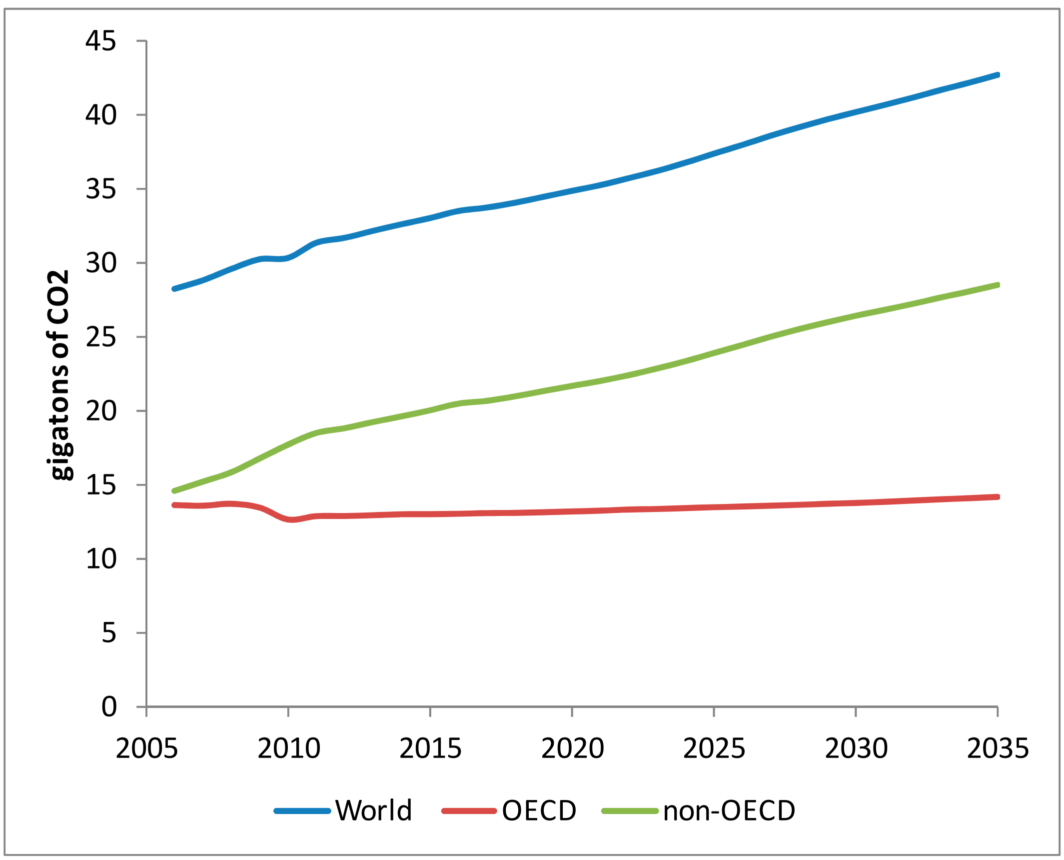
Energy security. While energy security concerns often motivate policies to reduce domestic consumption of oil and other fuels, quantifying a reasonable fuel tax level in this regard is challenging. Some studies (e.g., Brown and Huntington, 2010) suggest the costs (not taken into account by the private sector) due to the vulnerability of the macroeconomy to oil price volatility are not especially large (at least for the United States). More generally, dependence on oil supplies from politically volatile regions may realign a country’s foreign policy away from globally desirable objectives toward one focused on promoting access to oil markets (e.g., CFR, 2006), though rapid development of non-conventional oil (e.g., shale oil) may be alleviating these concerns.

CO_2 emissions

CO_2 emissions from fossil fuel combustion are, by far, the largest source of global greenhouse gas (GHG) emissions. Here emissions trends and the scientific basis for human-induced global warming are briefly summarized. For an in-depth discussion, see successive reports of the Intergovernmental Panel on Climate Change (IPCC), most recently the assessment of the science in IPCC (2013).

Global, energy-related CO_2 emissions have increased from around 2 billion tons in 1900 to around 30 billion tons today and, in the absence of mitigating measures, are projected to increase to almost 45 billion tons by 2035 (Figure 1.7). Emissions from non-OECD countries overtook those from OECD countries around 2005, and are projected to account for two-thirds of the global total by 2035.

Figure 1.7. Projected Global, Energy-Related CO_2 Emissions



Sources: US EIA (2011), Table A10.

Roughly 50 percent of CO_2 releases accumulate in the global atmosphere, where they remain, on average, for about 100 years: consequently, atmospheric CO_2 concentrations have increased from pre-industrial levels of around 280 parts per million (ppm) molecules to current levels of about 400 ppm. Accounting for other greenhouse gases (GHGs), such as methane and nitrous oxide (from agricultural and industrial sources), and expressing them in lifetime warming equivalents to CO_2, atmospheric concentrations in CO_2 equivalents are now about 440 ppm. In the absence of substantial emissions mitigation measures, GHG concentrations are expected to reach 550 ppm (in CO_2 equivalent) by around the middle of this century, and continue rising thereafter (e.g., Aldy and others, 2010; Bosetti and others, 2012).

IPCC (2013) estimates that global average temperatures have risen by 0.85°C since 1880 and is 95 percent certain that the main cause is fossil fuel combustion and other man-made greenhouse gases (rather than other factors like changes in solar radiation and heat absorption in urban areas). However, due to lags in the climate system (i.e., gradual heat diffusion processes in the oceans) temperatures are expected to continue rising, even if concentrations were stabilized at current levels. As indicated in Figure 1.8, if GHG concentrations were stabilized at 450, 550, or 650 ppm respectively, the eventual mean projected warming (over pre-industrial levels) is 2.1, 2.9, and 3.6°C respectively.[1] Alternatively, contemporaneous warming is expected to reach around 3–4°C by the end of the century, though actual warming could be substantially higher (or lower) than this (IPCC 2007, Bosetti and others 2012).

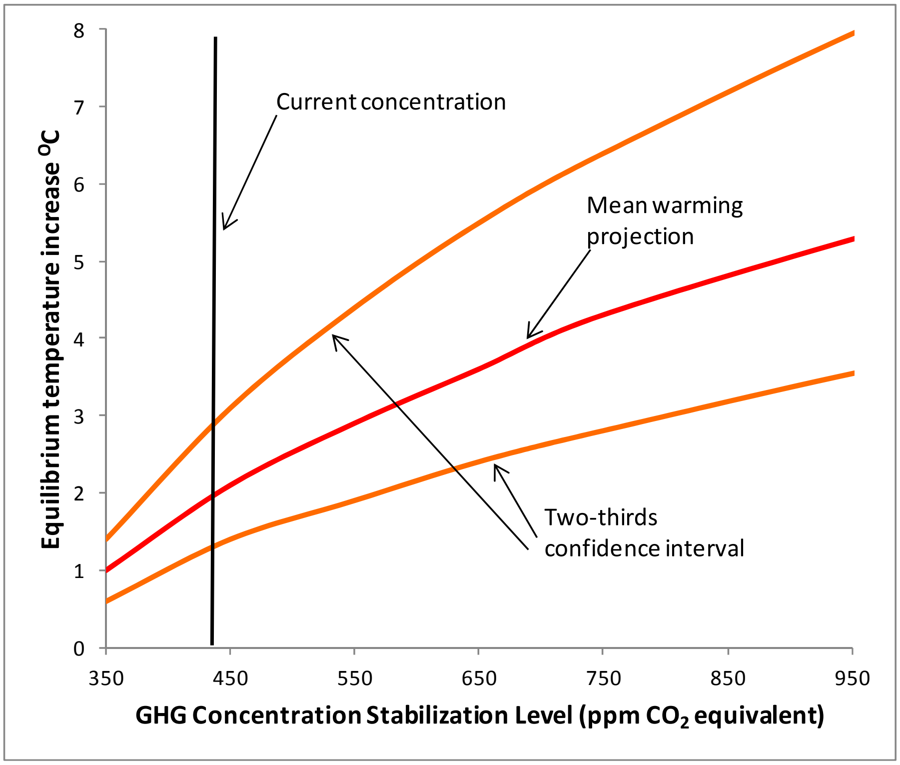
The climatic consequences of warming include changed precipitation patterns, sea level rise (due to thermal expansion of the oceans and melting sea ice), more intense and perhaps frequent extreme weather events, and possibly more catastrophic outcomes like runaway warming, ice sheet collapses, or destruction of the marine food chain (due to warmer, more acidic oceans). Considerable uncertainty surrounds all of these effects, not least the potential for feedback effects (e.g., releases of methane from thawing permafrost tundra, less reflection of sunlight as glaciers melt) that might compound warming.

Policies to price the carbon content of fossil fuels (or otherwise mitigate CO_2 emissions) are needed, because at present households and firms are generally not charged for the future climate change damages resulting from these emissions.[2]

[1] For perspective on the scale of these changes, current temperatures are about 5°C higher than at the peak of the last Ice Age around 20,000 years ago when the climate was radically different, covering much of the Northern Hemisphere in ice.

[2] Other policies are also needed, but are largely beyond the scope here. These include policies to reduce emissions from international aviation and maritime emissions (see Keen and others 2013) and land use (see Mendelsohn and others 2012); measures to enhance clean technology development (see Chapter 2); adaptation to climate change (e.g., coastal defenses, shifting to hardier crop varieties); development of last-resort technologies (e.g., to remove CO_2 from the atmosphere or to 'manage' solar radiation through sunlight-deflecting particles) for possible deployment in extreme scenarios; and mobilization of financial assistance for developing countries (see de Mooij and Keen 2012).

Figure 1.8. Projected Long-Run Warming Above Pre-Industrial Temperatures from Stabilization at Different GHG Concentrations



Sources: IPCC (2007), Table 10.8.
Notes: Figure shows the projected rise in global temperature (once the climate system has fully adjusted, which takes several decades) over pre-industrial levels if atmospheric GHG concentrations are stabilized at different levels. The most recent assessment (IPCC 2013) slightly lowered the bottom end of the confidence interval for a doubling of CO_2 equivalent.

Local air pollution

Unless it is priced to reflect environmental damages, local air pollution from fuel combustion is also excessive from society's perspective. Here the sources of air pollution and their environmental impacts are discussed.

Sources of air pollution

Fossil fuel use results in both 'primary' pollutants, emitted during fuel combustion, and 'secondary' pollutants, formed subsequently from chemical transformations of primary pollutants in the atmosphere. In terms of pollution-related health effects—the main source of environmental damage—potentially the most important pollutant is fine particulates or $PM_{2.5}$

(particulates with diameter up to 2.5 micrometers), as these permeate the lungs and bloodstream. $PM_{2.5}$ is directly emitted when fuels, like coal, are combusted, but is also formed indirectly from chemical reactions in the atmosphere involving certain primary pollutants.

As regards coal, the most important pollutants are directly emitted $PM_{2.5}$, and sulfur dioxide (SO_2) which reacts in the atmosphere to form $PM_{2.5}$. Fine particulates are also formed from nitrogen oxide (NO_x) emissions, but generally in much smaller quantities, because NO_x emission rates are generally lower than for SO_2 and they are less reactive. Emission rates per unit of energy can vary considerably across different coal types and a number of newer coal plants in many countries incorporate emissions control technologies (both factors should be considered in setting coal taxes).

Natural gas is a much cleaner fuel than coal, as it produces a minimal amount of SO_2 and direct $PM_{2.5}$ emissions, though it does generate significant amounts of NO_x. Motor fuel combustion also produces NO_x and, additionally, diesel fuel combustion causes some SO_2 and direct $PM_{2.5}$ emissions. Motor fuel combustion also releases volatile organic compounds (VOCs), which react with NO_x in the presence of sunlight to form ozone (O_3), a major component of urban smog. Ozone has health effects, though the link with mortality is much weaker than for fine particulates (damages from ozone are not considered here).[3]

Environmental damages

Local air pollution damages are potentially large, and have been estimated at around 1 percent of GDP for the United States, and almost 4 percent for China.[4] These damages range from impaired visibility and non-fatal (heart and respiratory) illness to building corrosion and reduced agricultural yields (when pollutants react with water to form acid rain). However, a number of studies suggest that, by far, the main damage component (and the component this study focuses on) is elevated risks of premature (human) mortality.[5]

It is well established in the epidemiological literature that long-term exposure to $PM_{2.5}$ is associated with increased risk of lung cancer, chronic obstructive pulmonary disease, heart disease (from reduced blood supply), and stroke (Burnett and others, 2013; HEI, 2013; Humbert and others, 2011; Krewski and others, 2009). Seniors, infants, and people with pre-

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[3] This ground-level ozone is distinct from stratospheric ozone, which blocks cancer-causing, ultra-violet radiation. Stratospheric ozone depletion is caused by man-made chemicals, but these have now been largely phased out (Hammitt, 2010).

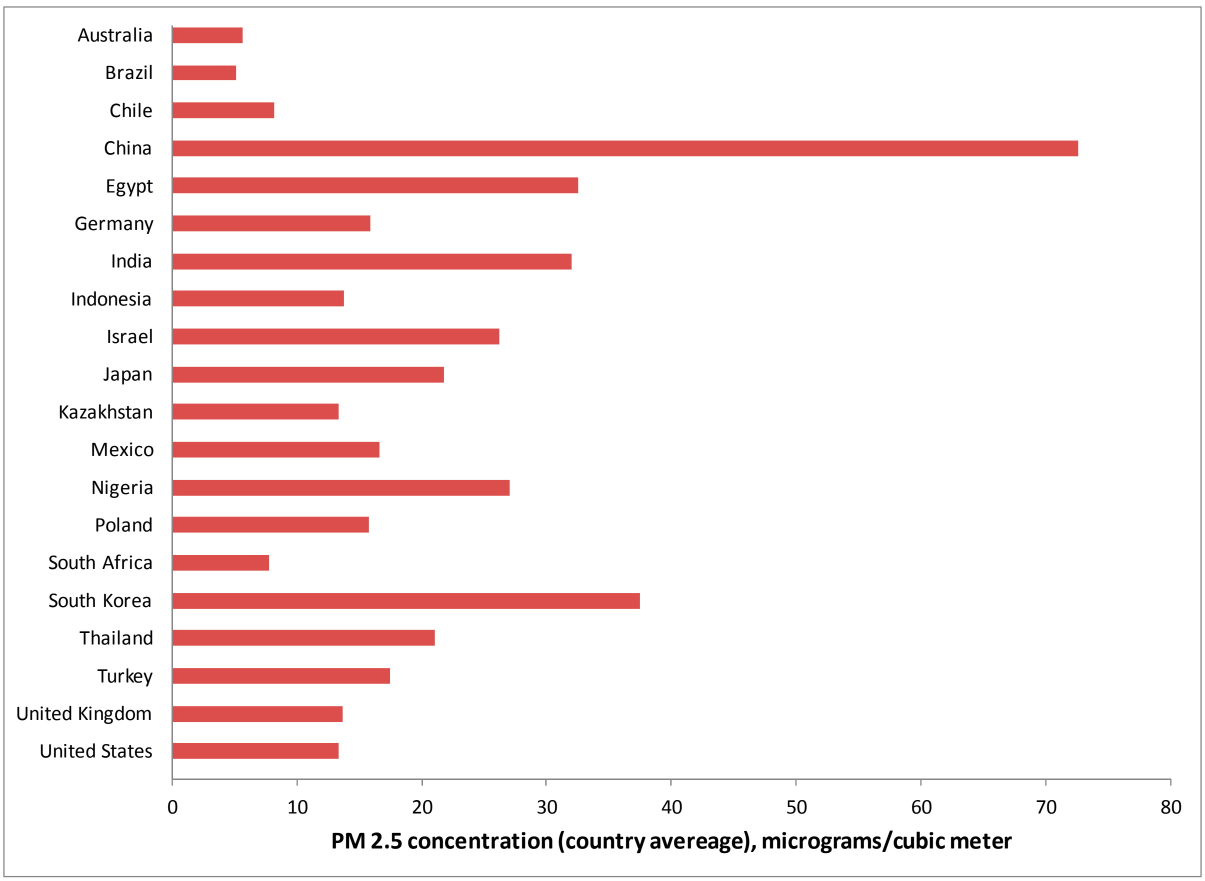
[4] See NRC (2009), Muller and Mendelsohn (2012), and SEPA/WB (2007).

[5] For example, studies for China, Europe, and the United States find that mortality impacts typically account for 85 percent or more of the total damages from local air pollution (e.g., US EPA, 2011; EC, 1999; NRC 2009; SEPA/WB, 2007; Watkiss, Pye, and Holland, 2005).

existing health conditions (e.g., those who have suffered strokes or who are suffering from cardiovascular disease) are most susceptible (Rowlatt and others, 1998).

Figure 1.9 shows ambient $PM_{2.5}$ concentrations for selected countries, averaged across regions, in 2010. For many countries (e.g., Germany, Indonesia, Kazakhstan, Mexico, Poland, United Kingdom, United States) average $PM_{2.5}$ concentrations are between about 10 and 20 micrograms/cubic meter. Some countries have average $PM_{2.5}$ concentrations of less than 10 micrograms/cubic meter (e.g., Australia, Brazil, and South Africa, where the coastal location of cities helps to disperse pollution). But in other countries $PM_{2.5}$ concentrations can be much greater: for example, between 30 and 40 micrograms/cubic meter in Egypt, India, and South Korea and, strikingly, over 70 micrograms/cubic meter in China.

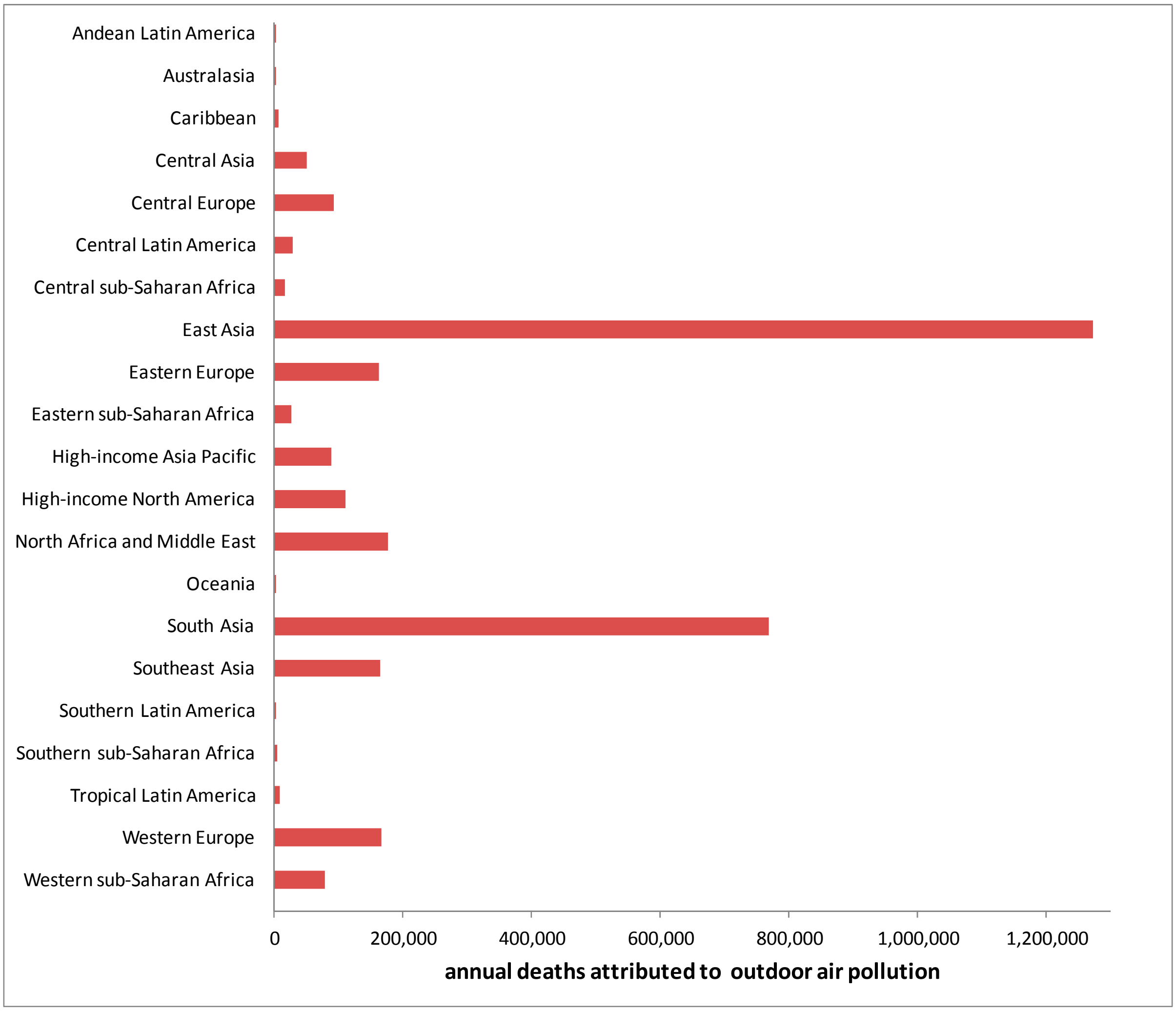
Figure 1.9. Air Pollution Concentrations, Selected Countries, 2010



Sources: Brauer and others (2012).
Notes: Data is an average of regional pollution concentrations (weighted by population shares) within a country, where regional observations are based on satellite data (concentrations for specific urban centers can be much higher than national averages).

Figure 1.10 shows estimated deaths by region attributed to local (outdoor) air pollution in 2010. Worldwide, deaths were 3.2 million and were especially concentrated in East Asia (about 1.3 million) and South Asia (about 0.8 million).

Figure 1.10. Air Pollution Deaths by Region, 2010



Sources: Burnett and others (2013).
Notes: Figure shows estimated deaths from outdoor (ambient) air pollution and excludes deaths from indoor air pollution (see Box 1.1). Data by country will be available soon, but was not available at the time of writing.

Fuel taxes should not necessarily be highest in countries with the worst pollution however, as the extra health risks posed by additional pollution do not necessarily depend on the existing pollution concentration (Chapter 1 of the main report). And appropriate taxes depend, for example, on the size and composition of the exposed population and on how health risks are valued (which might vary with income levels). Environmental taxes likely, however, have relatively larger impacts in high-pollution countries, where there is greater scope for reducing pollution.

Broader externalities related to motor fuel use

Use of fuel in motor vehicles is associated with some further side effects that should be factored into tax design, the most important of which are traffic congestion and traffic accidents (road damage plays a more minor role in corrective fuel taxes).

Traffic congestion

Traffic on roads where speeds are below free-flow levels is generally excessive: unless they are charged for road use, motorists will not account for their own impact on adding to congestion and slowing speeds for other road users.[6] This applies irrespective of complementary policies (e.g., investment in road or transit capacity, improved coordination of traffic signals), though these improvements can lower the appropriate charge for congestion (e.g., by alleviating bottlenecks).

Traffic congestion varies dramatically across urban and rural areas, and across time of day. It has been estimated, for example, that drivers in the London rush hour impose costs on others equivalent to US$10 per liter of fuel, through their contribution to traffic congestion (Parry and Small, 2009). Congestion is best addressed through taxes on vehicle km driven on busy roads, with rates varying over the course of the day with prevailing traffic levels (Chapter 1 of the main report). Until such charges are comprehensively implemented (e.g., using Global Positioning Systems) however, it is appropriate to reflect congestion costs that motorists impose on others in fuel taxes (e.g., Parry and Small, 2005).

The appropriate fuel charges are likely to vary considerably across countries, even if travel delays were valued in a similar way. Figure 1.11, which shows registered vehicles (cars, trucks, buses) per km of nationwide road capacity, provides some, albeit very crude, sense of this. For example Germany, Japan, Mexico, Poland, and the United Kingdom, have far more vehicles per km of road capacity than the United States, implying that a much greater portion of nationwide driving likely occurs under congested conditions in the former countries.

Traffic accidents

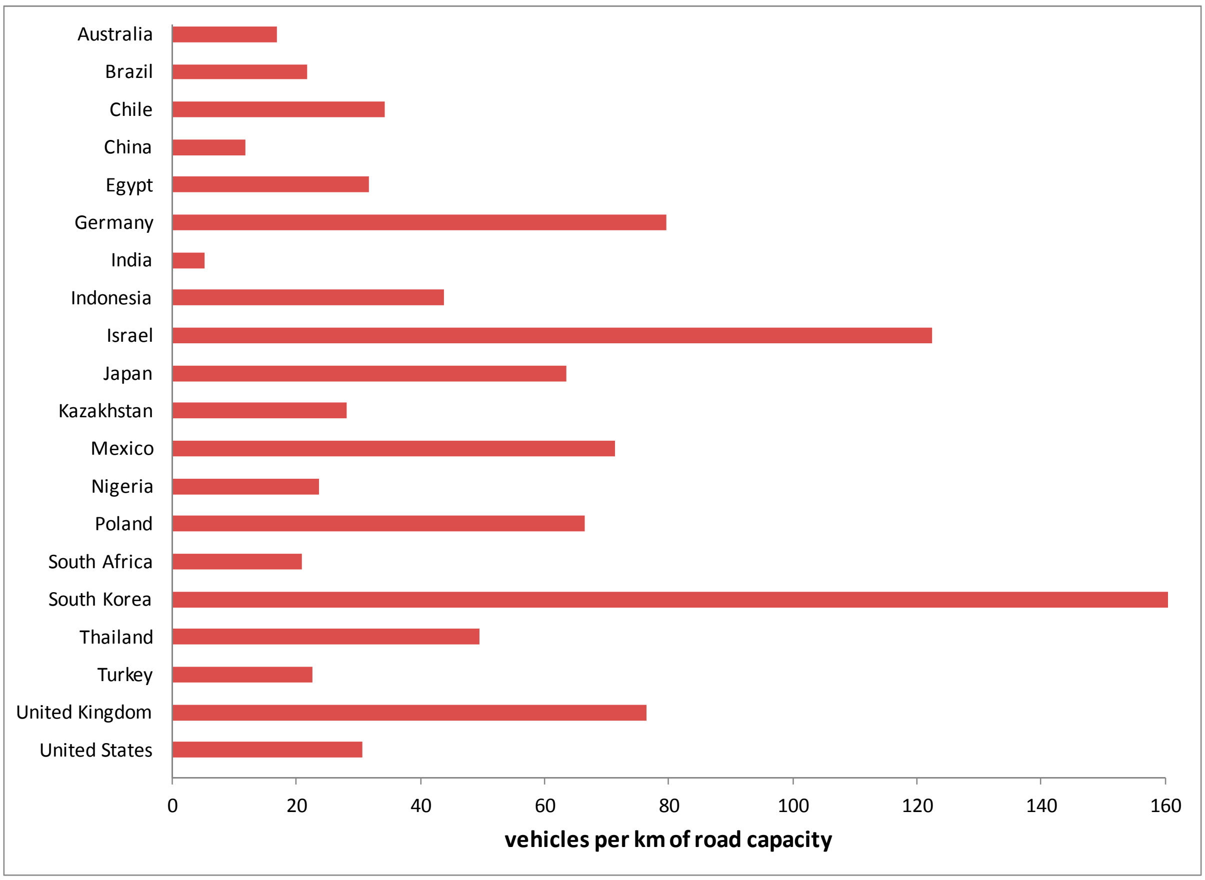
Another side effect of vehicle use is traffic accidents. Although drivers should take into account some accident costs (e.g., injury risks to themselves), other costs (e.g., injury risks to pedestrians, property damage, medical costs borne by third parties), are not taken into account implying excessive driving from a societal perspective. Again, this applies irrespective of other measures (e.g., drunk driver penalties, airbag and seatbelt mandates,

[6] See, for example, Arnott, Rave, and Schöb (2005), Lindsey (2006), Litman (2013), and Santos (2004).

traffic medians), though these measures lower the appropriate charge for accidents (e.g., by reducing fatality rates).

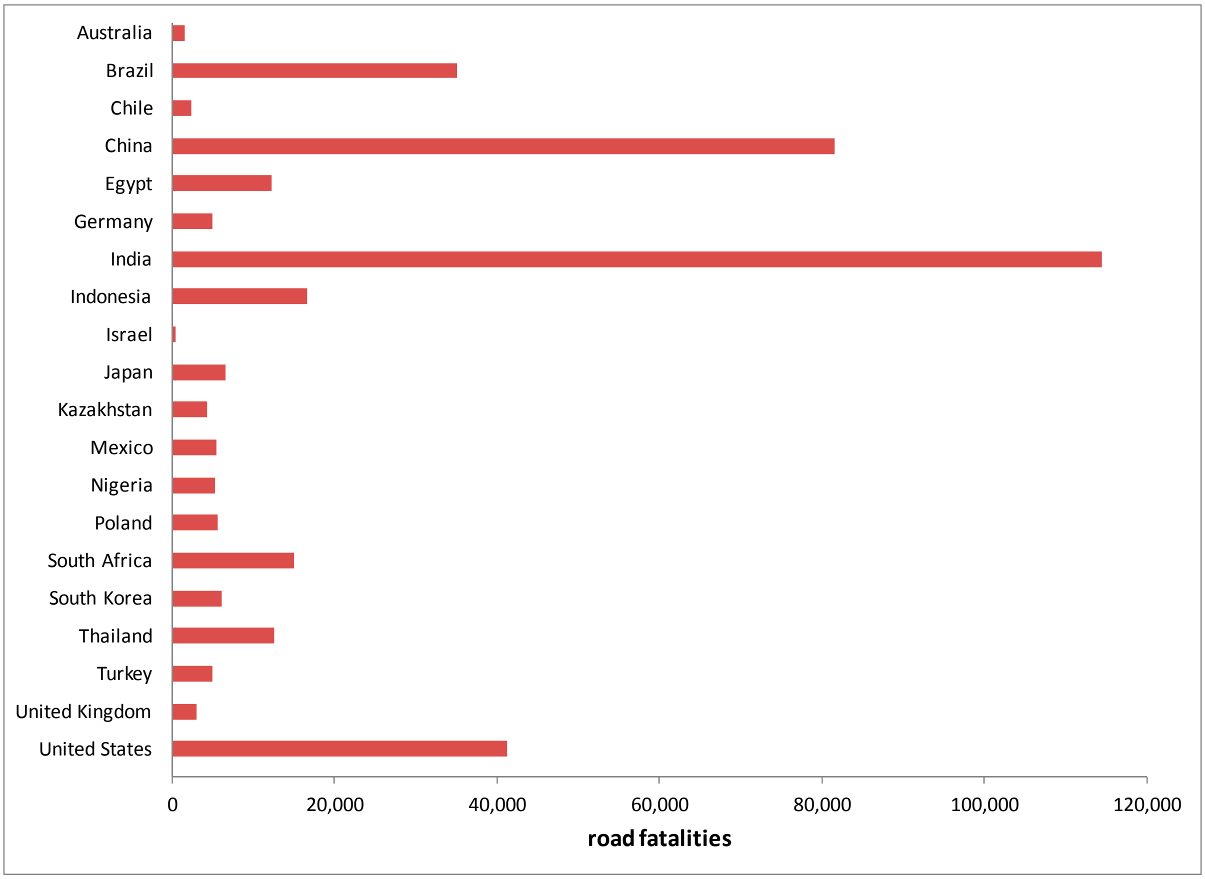
Figure 1.12, which shows the number of road fatalities in 2007, gives some sense of the problem. In India, for example, there were about 114,000 road deaths; in China about 82,000; and even in South Africa (which has 4 percent of the population as China), there were about 15,000 fatalities (and these figures could substantially understate the problem in developing countries—see Chapter 3).

Figure 1.11. Vehicles and Road Capacity, Selected Countries, 2007 (or thereabouts)



Sources: IRF (2009). See Chapter 3 below for more details.
Notes: Road capacity includes both paved and unpaved roads. Vehicles include cars, buses, and trucks but not two-wheelers.

Figure 1.12. Road Deaths, Selected Countries
(2007)

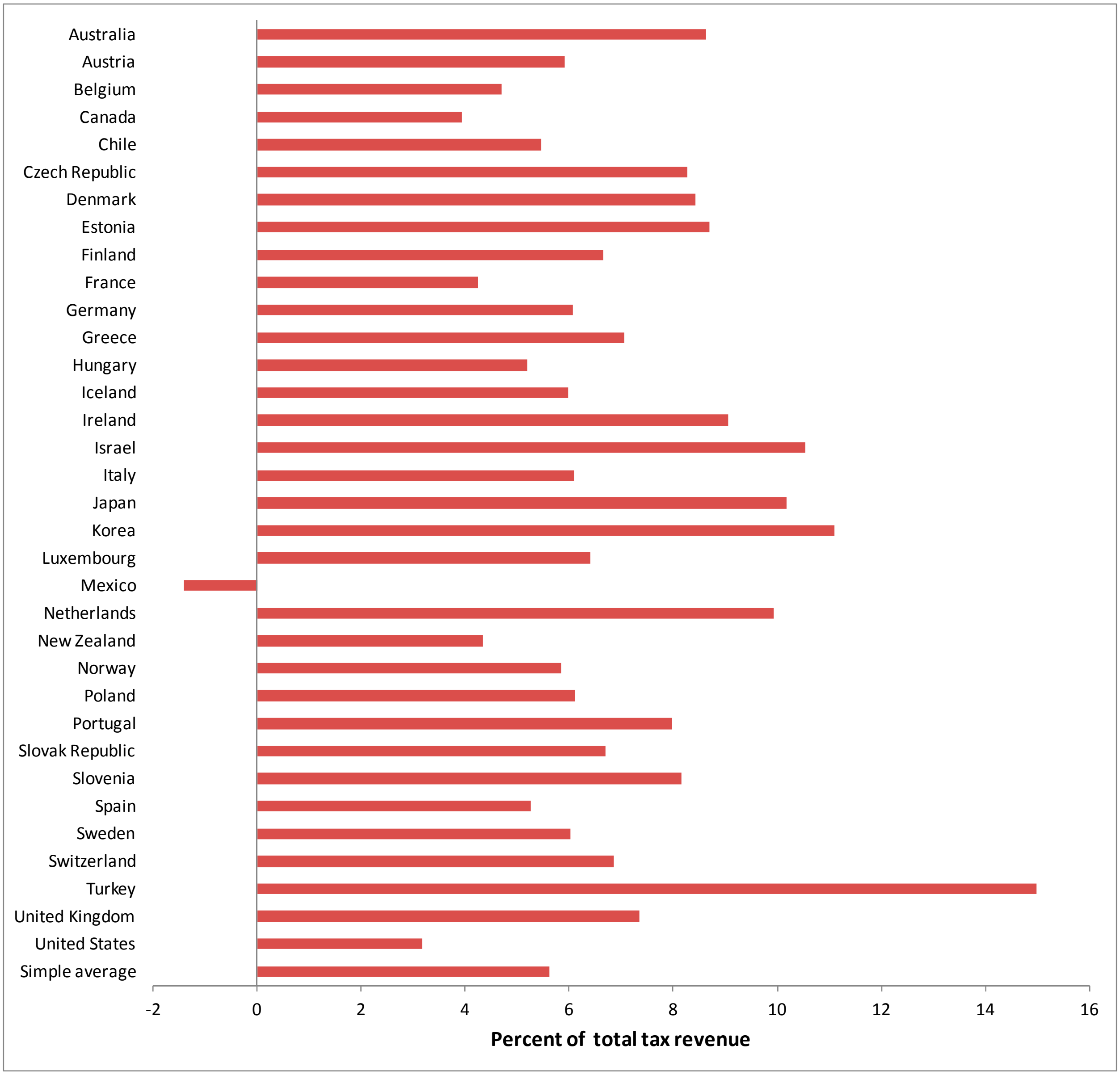


Sources: IRF (2009).
Notes. These figures may understate road fatalities in developing countries due to under-reporting (see Chapter 3). WHO (2013a) suggests, for example, traffic deaths in India and China are much larger, and the global total deaths is 1.3 million.

C. Fiscal Policies Currently Affecting Energy and Transportation

For data quality reasons, the discussion of tax policies in this subsection focuses on OECD countries (estimates of taxes/subsidies by fuel product for all countries are provided in Appendix to Chapter 2 of the main report). Among these countries, revenue from environmentally-related taxes see (Figure 1.13) averaged around 6 percent of total tax revenue in 2010, varying between 15 percent of revenue in Turkey, to 3 percent in the United States, and about minus 1.5 percent in Mexico where petroleum was subsidized significantly in 2010 (prior to the recent liberalization).

Figure 1.13. Revenue from Environmentally-Related Taxes as Percent of Total Revenue in OECD Countries, 2010 (or thereabouts)

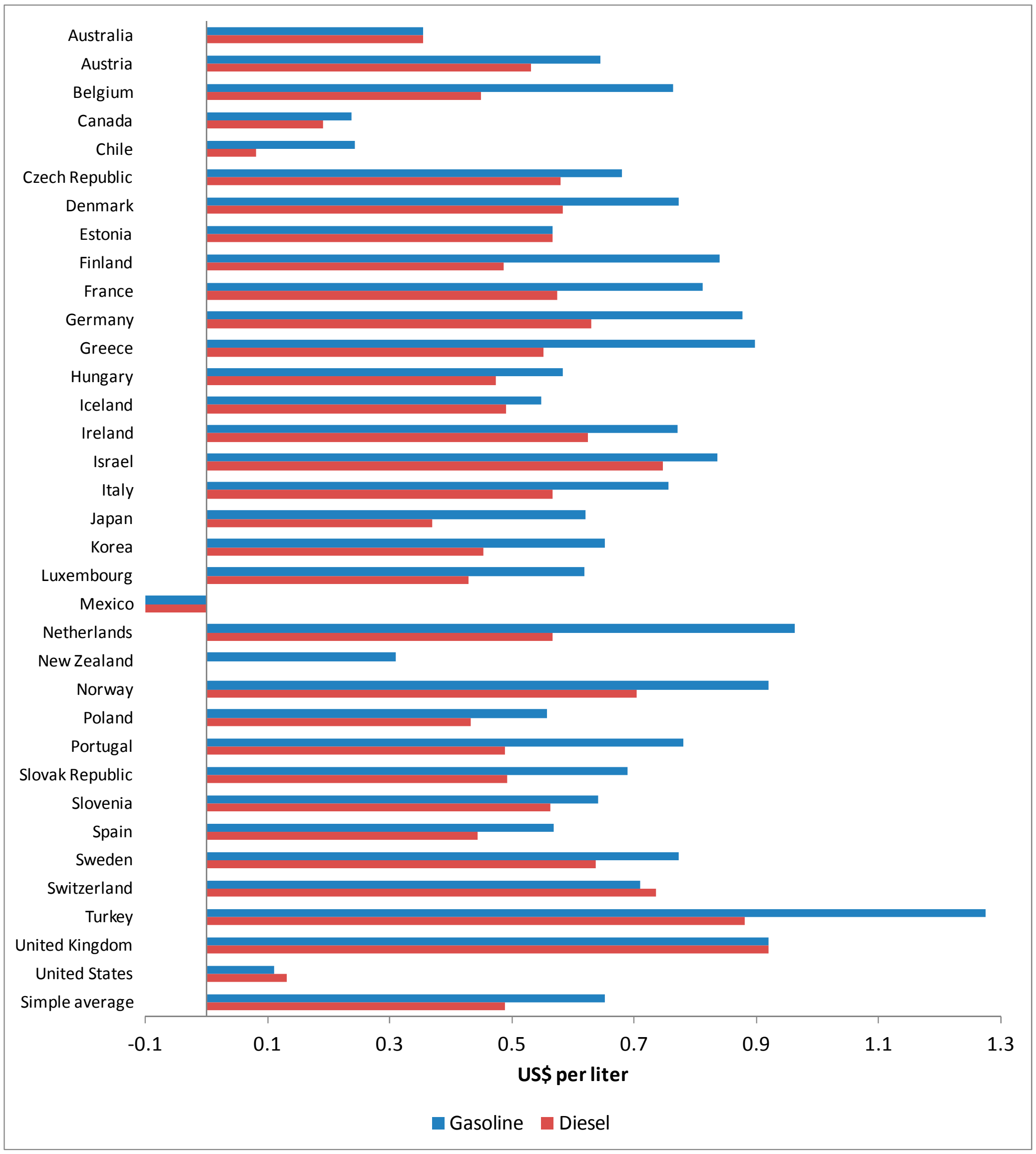


Source: OECD (2013).

These revenues mainly reflect three excise taxes: taxes on fuel, vehicle ownership, and (residential) electricity consumption.

Although fuel taxes promote all possibilities for reducing fuel use (better fuel efficiency, less driving), the main issue here is whether tax levels reasonably reflect environmental damages. This seems unlikely (for all countries at least), given huge disparities in tax rates (Figure 1.14). In 2010, gasoline taxes varied from (the equivalent of) over US 80 cents/liter in Finland, France, Germany, Greece, Israel, Norway, Turkey, and United Kingdom, to 11 cents/liter in the United States, and –13 cents/liter in Mexico. In most countries, diesel fuel is tax-favored relative to gasoline, though it is not obvious that trucks (the primary consumer of diesel) contribute less than cars to pollution, congestion, and so on.

Figure 1.14. Excise Tax Rates on Motor Fuels, 2010 (or thereabouts)



Source: OECD (2013).

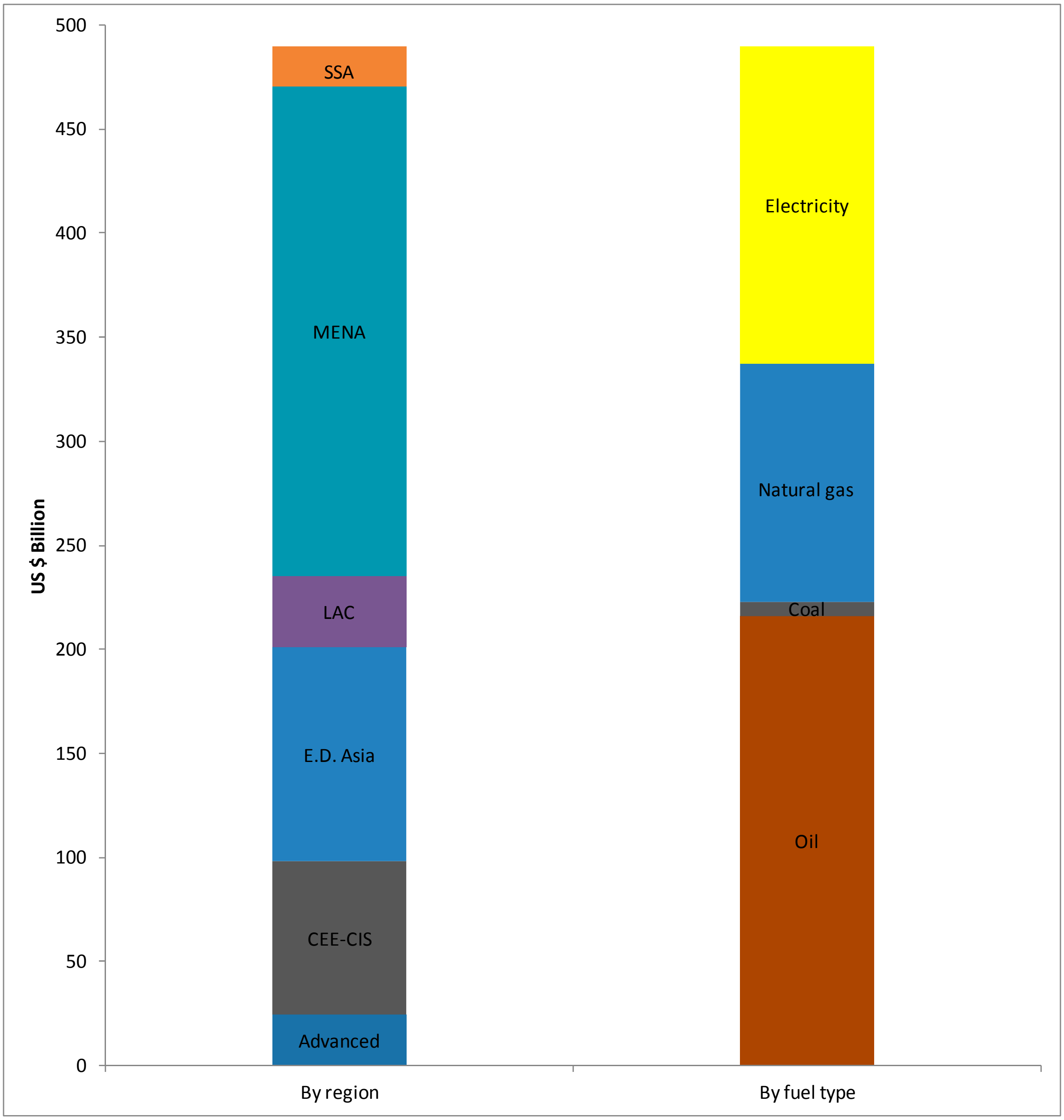
Other taxes underlying Figure 1.13 are not well targeted from an environmental perspective (see Chapter 1 of the main report). Vehicle taxes do not encourage vehicle owners to drive less, and (despite often varying with emissions classes) do not exploit all opportunities for raising fuel efficiency. Simple taxes on electricity consumption do not encourage cleaner power generation fuels, nor use of emissions control technologies. Although there is some momentum for carbon pricing (ECOFYS, 2013), presently over 90 percent of global CO_2 emissions are not covered by formal pricing programs (Parry, de Mooij, and Keen, 2012) and CO_2 prices (currently equivalent to about US $7 per ton of CO_2 in the EU Emissions Trading System) are typically a small fraction of estimated environmental damages (Chapter 3).

Moreover, many countries heavily subsidize (rather than tax) energy use. Estimated subsidies for fossil fuel use (measured by the gap between world fuel prices and domestic market prices) were $490 billion worldwide in 2011, with the Middle East and North African countries accounting for 48 percent of these subsidies (Figure 1.15). Notably 44 percent of these subsidies were for petroleum products, 23 percent for natural gas, 31 percent for electricity consumption, but only 1 percent for coal (the most polluting fuel)—so just eliminating subsidies, without introducing coal taxes, could have perverse fuel switching effects.[7]

Nonetheless, the overall picture is one of ample opportunities to rationalize energy prices by eliminating fossil fuel subsidies and shifting some of the burden of broader taxes onto fossil fuel products. Even in countries with high energy taxes, there is scope for restructuring them (e.g., shifting taxes off electricity and onto coal) to improve their effectiveness, and for better aligning tax rates to environmental damages. How to gauge appropriate tax levels for this purpose is the main contribution of this volume.

[7] Renewables are also subsidized, to the tune of $66 billion in 2010, according to IEA (2011), Figure 14.13.

Figure 1.15. Subsidies for Fossil Fuel Energy by Region and Fuel Type, 2011



Source: Clements and others (2013).
Notes: CEE-CIS is Central and Eastern Europe and Commonwealth of Independent States; LAC is Latin America and Caribbean; SSA is Sub-Saharan Africa; MENA is Middle East and North Africa; and ED Asia is Emerging and Developing Asia.

CHAPTER 2. VALUING POLLUTION DAMAGES FROM FUEL USE

This chapter begins (Section A) with a brief review of literature on valuing climate change damages from CO_2 emissions. However, the heart of the chapter (Section B) is about measuring the most important damage from local air pollution, human mortality risk.

A. Valuing CO_2 Damages

The future climate change damages from a ton of CO_2 emissions are the same, regardless of the fuel combustion process or where emissions are released. In principle therefore, each ton should be priced at the same level in different countries. If charges are imposed on fuel suppliers, the appropriate charge per unit of fuel is the CO_2 damage times the CO_2 emissions factor (i.e., CO_2 emissions released per unit of fuel combustion). The first component is discussed here, and emissions factor in Section B.

There are two economic approaches to assessing appropriate CO_2 emissions prices—the benefit/cost and cost-effectiveness approaches.[8]

Benefit-Cost Approach

The first approach assesses damages from the future global climate change caused by additional emissions, using 'integrated assessment models' which incorporate:

- links between current emissions and the future global time-path of atmospheric GHG concentrations;
- impacts of changes in that time-path on global temperature and other climate variables in future years;
- worldwide (monetized) damages at different points in time from those climate changes (e.g., agricultural impacts, costs of seal level protection, health impacts from altered climate and possible spread of vector-borne diseases, ecological impacts); and
- discounting of damages at different future dates to the present, to obtain a single summary statistic, or damage per ton of CO_2, known as the 'social cost of carbon' (SCC).

Although there are many uncertainties surrounding all these relationships, damage values are especially sensitive to discounting (CO_2 emissions have very long-range impacts

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[8] For more extensive discussions see, for example, Clarke and others (2007) ; Bosetti and others (2012) ; Griffiths and others (2012); NRC (2009), Ch. 5; and US IAWG (2010, 2013).

because they reside in the atmosphere for many decades and the climate adjusts gradually over time to higher atmospheric concentrations) and the treatment of extreme risks.

On discounting, one view is that the future benefits of emissions mitigation policies should be discounted using market interest rates (usually around 3–5 percent in advanced countries) because this is the standard way to evaluate the future benefits of any private (and many public) investments. Studies using market discount rates typically estimate the SCC in the order of about \$10 to \$50/ton of CO_2, for example, US IAWG (2013) put the SCC at \$35/ton (in their central case for year 2010 in 2010\$).

Others argue that for ethical reasons below market rates should be used to evaluate policies where the benefits accrue to future generations (as opposed to the current generation), to avoid discriminating against people who are not born yet. Under this approach, SCC estimates are much higher, for example, Stern (2006) put the SCC at \$85 per ton (in year 2004\$), with the difference (compared with earlier studies) largely reflecting different discount rates (Nordhaus, 2007).[9]

These SCC estimates often include a component for 'catastrophic' risks by postulating probabilities (based on judgment, given that the risks are unknown) that future climate change may result in very large world GDP losses. But the appropriate way to treat these risks remains very contentious (Pindyck, 2013): some (though not all) studies (e.g., Weitzman, 2009) suggest they warrant dramatically higher CO_2 prices.[10]

Typically, SCC estimates in the benefit/cost approach rise at around 1.5–2.5 percent a year in real terms, primarily reflecting the growth rate in output potentially affected by climate change.

Cost-effectiveness approach

Rather than explicitly valuing environmental damages (the approach taken elsewhere in this volume), the cost-effectiveness approach assesses least-cost pricing paths for CO_2 emissions that are broadly consistent with long-term climate stabilization goals.

For this purpose, numerous climate/economy models have been developed, with particular detail on the global energy sector, and linkages between emissions, atmospheric GHG concentrations, and future climate outcomes. Projecting future emissions prices needed to meet long-range climate targets is inherently imprecise however, given considerable

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[9] The discount rate in this approach is still positive (usually about 1–2 percent) to reflect the higher per capita consumption of future generations.

[10] Damage estimates are highly sensitive to the shape of the probability distribution for catastrophic risks, which is uncertain.

uncertainty over future emissions baselines (which depend, for example, on future population, per capita income, the energy-intensity of GDP, and the fuel mix) and the emissions impact of pricing (which depends, for example, on the future costs of low emission fuels and technologies).

A review of modeling exercises by Bosetti and others (2012) suggests that a (global) CO_2 price, starting at $20/per ton (in current $) in 2020 (and rising at 3–5 percent a year), would be roughly in line with keeping mean projected warming to about 3.5°C (a target that many view as far too risky), while a starting price of roughly twice this level might be needed to keep projected warming to 3°C. However, keeping projected warming to 2°C (the goal identified in the 2009 Copenhagen Accord) might now be beyond reach, as it would require the future development, and global deployment, of technologies that on net remove GHGs from the atmosphere, to help return future GHG concentrations back down to current levels.

Illustrative Value Used Here

Summing up, there is huge uncertainty, and controversy, over the appropriate CO_2 emissions price—and country governments may have their own perspectives on this. A value of $35/ton (based on US IAWG, 2013) is used here *for illustrative purposes* (this chosen value should not be construed as a recommendation for one SCC value over another or one climate stabilization target over another). The implications of alternative values for corrective taxes are easily inferred from the results and the accompanying spreadsheets.

A note on equity

A key principle in the UN Framework Convention on Climate Change is that developing countries have 'common but differentiated responsibilities' meaning (given their relatively low income and small contribution to historical GHG accumulations) that they should bear a disproportionately lower burden of mitigation costs than wealthier nations. This implies either their receiving compensation or their imposing lower emissions prices than others, or perhaps no price at all. The latter need not hinder international mitigation efforts, at least for the vast majority of low-income countries whose emissions constitute a tiny fraction of the global total (Gillingham and Keen, 2012).

B. Valuing Local Air Pollution Damages

Although local air pollution causes a variety of other harmful environmental effects, the focus here is premature (human) mortality which is, by far, the most important category in previous damage assessments (Chapter 1).

Valuing the pollution-mortality impacts from fuel combustion involves the following:

- determining how much pollution is inhaled by exposed populations, both in the country where emissions are released and (for emissions released from tall smokestacks) in countries where pollution may be transported to;

- assessing how this pollution exposure affects mortality risks, accounting for factors (like the age and health of the population) affecting vulnerability to pollution-related illness;

- monetizing the health effects;

- expressing the resulting damages per unit of fuels.

The focus here is damages from an incremental amount of pollution (as this is relevant for setting efficient fuel taxes) rather than damages from the total amount of pollution.

For a very limited number of countries, previous studies have estimated local air pollution damages, and there are major ongoing modeling efforts at the global level.[11] This volume is the first attempt to provide an assessment of fossil fuel emissions damages across a broad range of countries, using a consistent methodology.[12]

Although, insofar as possible, key country-specific factors determining environmental damages are captured, not all potentially significant factors (most notably cross-country differences in meteorological conditions affecting pollution formation) are feasible to include. The corrective tax estimates below may also become outdated as evidence and data evolves. Nonetheless, some broad sense of how missing factors may affect the results is given by comparing them (for selected countries) to those from a computational model of regional air quality. And accompanying spreadsheets, indicating corrective taxes by fuel product and country, are easy to update.

The discussion proceeds as follows.

Subsections (i) to (iii) address, respectively, the first three steps in the above bulleted list, and Section (iv) summarizes the resulting cross-country estimates of local pollution damages.

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[11] For example, NRC (2009) and Muller and Mendelsohn (2012) estimate pollution damages for the United States; ExternE (2005) for Europe; SEPA/WB (2007) for China; Cropper and others (2012) for India.
At a global level, the Global Burden of Disease project (see below) estimates regional mortality rates from air pollution, though not the health effects from emissions released in individual countries, which is needed to infer corrective fuel taxes. The Climate and Clean Air Coalition (CCAC) is developing a sophisticated modeling system to quantify air pollution damages which presently covers four countries, but will eventually apply to many more. As this work progresses, it will provide useful information for refining corrective tax estimates (e.g., accounting for cross-country differences in meteorological factors).

[12] Methodological consistency implies that differences in estimated damages across countries reflect real factors rather than different estimation procedures.

Section (v) compares the results with those from the computational model. The final subsection discusses procedures for converting emissions damages into corrective fuel taxes, the results of which are presented in Chapter 2 of the main report.

(i) Estimating Population Exposure to Pollution

As noted in Chapter 1, the main cause of mortality risk from pollution is $PM_{2.5}$, particulates (with diameter up to 2.5 micrometers) that are small enough to permeate the lungs and bloodstream. $PM_{2.5}$ can be emitted directly (as a primary pollutant) from fuel combustion, but is also produced (as a secondary pollutant) from chemical reactions in the atmosphere involving primary pollutants, most importantly sulfur dioxide (SO_2), but also nitrogen oxides (NO_x).

'Intake fractions' are used here to estimate how much pollution from stationary and mobile emissions sources in different countries is inhaled by exposed populations (see Box 2.1 for some technical details). Specifically, these fractions (as used here) indicate grams of $PM_{2.5}$ inhaled per ton of direct $PM_{2.5}$, SO_2 , and NO_x. Intake fractions are a powerful concept and are being used increasingly in pollution damage assessment,[13] not least because they circumvent the need to develop data and computationally intensive air quality models.

Intake fractions depend on three main factors:

- the height at which emissions are released. The most important distinction here is between emissions from tall smokestacks (e.g., at power plants), which are more likely to be dispersed (without harm) but are also transported considerable distances, and emissions released at ground level (e.g., from cars and residential heating) which tend to stay locally concentrated.

- the size of the population exposed to the pollution. For smokestack emissions, people living as far away as 2,000 km or more from a plant can still intake some of the pollution (Zhou and others, 2006). Even if a plant were to be located away from an urban center, its emissions could therefore still cause significant health damages elsewhere. Long-distance transportation of pollution also raises thorny issues about how one country should account for cross-border environmental damages when setting its own fuel taxes.

- meteorological conditions (most notably wind speed and direction), topography (e.g., proximity to mountain barriers that may block pollution dispersion), and ambient ammonia concentrations (which catalyze atmospheric reactions of SO_2 and NO_x).

[13] See, for example, Apte and others (2012); Bennett and others (2002); Cropper and others (2012); Humbert and others (2011); Levy, Wolff, and Evans (2002); and Zhou and others (2006).

For long-distance pollution, a strength of the approach here is that it uses highly disaggregated data on population density (in different countries) up to 2,000 km away from emission sources. The estimates of population exposure may therefore be considerably more accurate than in other studies using much more spatially aggregated population data, or that only consider people living within shorter distances to the plant.

A weakness is that the intake fraction approach cannot easily account for cross-country differences in meteorological (and related) conditions, not least because emissions are transported across multiple climate zones and wind patterns. However, studies suggest that population is usually, by far, the more important factor (Zhou and others, 2006).

The estimation of population exposure for coal plants, other stationary sources, and mobile emissions sources are discussed in turn below.

Box 2.1. Intake Fractions: Some Technicalities

The intake fraction (*iF*), for a primary pollutant, at a particular location, is given by the formula (e.g., Levy, Wolff, and Evans, 2002):

$$iF \equiv \frac{\sum_{i=1}^{N} P_i \times \Delta C_i \times BR}{Q}$$

P_i is the population residing in a region, indexed by i, defined by its distance from the emissions source, where the region could be in the country where emissions are released or in some other country or some combination of both. ΔC_i is the change in the ambient concentration of pollution ($PM_{2.5}$), perhaps defined by the daily average change in pollution per cubic meter, caused by emissions from the source—ΔC_i is influenced by meteorology and other factors. BR is the average breathing rate, that is, the rate at which a given amount of ambient pollution is inhaled by the average person, for example in Zhou and others (2006) the breathing rate is 20 cubic meters per day.

The numerator in the above expression is therefore the total amount of pollution taken in by potentially exposed populations (per day). In the denominator, Q is the emissions rate of the primary pollutant (tons per day). The intake fraction is defined as average pollution inhaled per unit of emissions released, and is usually expressed as grams of $PM_{2.5}$ inhaled per ton of primary emissions.

Exposure to coal plant emissions

Although intake fractions have been extensively estimated for emissions emitted at ground level for many different regions (see below), estimates are much more limited (due to the complexities involved in modeling long-distance pollution transport) for emissions released from tall smokestacks.

The approach here uses a widely cited study by Zhou and others (2006) who followed a two-step statistical procedure. They started by simulating, using a sophisticated model of regional air quality, how emissions are transported to different regions, and mapped the result to data on regional population density, to estimate intake fractions for a variety of primary pollutants from 29 coal plants in China.[14] For the average coal plant they estimated, for example, that about 5 grams of $PM_{2.5}$ ends up being inhaled, for each ton of SO_2 emitted. Zhou and others (2006) then use statistical techniques to obtain a set of coefficients indicating what fraction of an average plant's emissions are inhaled by an average person residing within 100 km, 100–500 km, 500–1,000 km, and 1,000–3,300 km away from the emissions source.

These coefficients can be combined with data on the number of people living within the four distance classifications from the plant to extrapolate intake fractions for a coal plant in any country, without the need to develop a sophisticated model of regional air quality. To keep the calculations tractable, the last distance category is truncated here (without much loss of accuracy) at 2,000 km.[15]

For extrapolation purposes, the Carbon Monitoring for Action (CARMA) database[16] is used to determine the geographical location of about 2,400 coal plants in about 110 different countries for year 2009 (this data covers about 75 percent of the total produced electricity by coal power plants worldwide).

And from the LandScan data, population counts by grid cell (for 2010) are obtained for each of these 110 countries, as well as countries without coal plants, but where people are still at risk of inhaling cross-border emissions.[17] This population data is extremely fine—each grid cell is 1 km square or less.

Mapping these two datasets provides—for all coal plants in the sample—an extremely accurate estimate of the population living at the four distance classifications from each plant. Multiplying populations in these distance categories by the corresponding coefficient from Zhou and others (2006) for a particular pollutant, and then adding over the four distance categories, gives the estimated intake fraction for that pollutant for each coal plant.

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[14] They used the California Puff (CALPUFF) air quality model, calibrated to Chinese data on regional emissions sources and pollution concentrations. This model is recommended by the US Environmental Protection Agency (EPA 2013) for estimating long-distance pollution transport (for documentation, see www.src.com/calpuff/calpuff1.htm).

[15] Zhou and others (2006) estimate the inhalation rate (for SO_2-induced $PM_{2.5}$) for someone living within 100 km of a coal plant is about 8 times that for someone living 100-500 km away, 43 times that for someone living 500–1,000 km away, and 86 times that for someone living 1,000-3,300 km away. However, taking into account the average number of people living at different distance classifications, they find that people living within 100 km, 100-500 km, 500-1,00 km and 1,000-3,300 km of the plant inhale 53, 27, 6, and 14 percent respectively of the total pollution intake.

[16] See http://carma.org.

[17] The LandScan data is compiled by Oak Ridge National Laboratory (see www.ornl.gov/sci/landscan).

Finally, the national average intake fraction for the pollutant is obtained for each country by taking a weighted sum of intake fractions for individual plants in that country, where the weights are each plant's share in total coal use.[18]

There are some caveats to the intake fraction approach (as applied to long-distance, but not ground level, pollution).

Most notably, adjustments are not made for meteorological or topographical conditions or local ammonia concentrations. The last factor is relevant because SO_2 and NO_x form $PM_{2.5}$ by reacting with ammonia—in fact when SO_2 and NO_x are reduced substantially, this can 'free up' ammonia for the remaining emissions to react with and makes them more likely to form fine particulates. To the extent that all these factors vary (across a radius of 2,000 km) for the average coal plant in another country, relative to these conditions for the average coal plant in China, the estimates here overstate or understate intake fractions for other countries. This issue is discussed further in section (v) when results (for selected countries) are compared with those from a computational air quality model (that does account for meteorology, ammonia concentrations, and related factors).

Second, intake fractions may also depend on the precise height of the smokestack from which pollutants are emitted, with emissions from tallest smokestacks having greatest propensity to dissipate before they are inhaled. Again data is not available on global variation in the height of smokestacks at power plants to adjust for this. However, intake fractions do not appear to vary much with differences in smokestack height (e.g., Humbert and others, 2011).

Third, mortality risks to people living within proximity to two or more power plants are taken as additive (or in other words, the intake fraction for one coal plant is the same, whether or not some of the people inhaling its pollution are also exposed to pollution from other plants). For the most part this seems reasonable, aside perhaps for countries where air pollution is especially severe (and people's ability to inhale pollution starts to become saturated), but even then (see Box 1.3 of the main report) there may not be much relevance for corrective fuel tax estimates.

Exposure to other stationary source emissions and vehicle emissions

Due to lack of data—particularly in regard to geographical location—it is not feasible to estimate population exposure to emissions for other uses of coal (e.g., metals smelting) and (for purposes of calculating the impact of coal tax reform) environmental damages and corrective taxes for these other uses are taken to be the same as for power plant coal use.

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[18] Total coal use is the sum of coal used by plants in the sample for that country. Coal use is inferred from the plant's CO_2 emissions, as there is a proportional relation between fuel use and CO_2 emissions, and emissions at the plant level is reported in the CARMA database.

Essentially the same procedure (and data sources) as outlined above is used to estimate average population exposure to approximately 2,000 natural gas plants in 101 countries. This is possible because natural gas produces the same three primary pollutants as coal so the Zhou and others (2006) coefficients can be applied again.[19]

Intake fractions for each primary pollutant tend to be greater for natural gas than for coal as (on average) gas plants are located closer to population centers, but the differences are not very large. Nonetheless, the local pollution effects of natural gas combustion are far less severe than for coal because natural gas produces only very minimal amounts of SO_2 and primary $PM_{2.5}$.

With respect to natural gas use in homes (primarily for space heating), population exposure to (outdoor) pollution is far more localized, given that emissions are released (and stay) close to the ground. The same applies for vehicle emissions.

For both cases, estimates from Humbert and others (2011) and Apte and others (2012) are combined. Humbert and others (2011) report a global average intake fraction for ground-level sources of SO_2, NO_x, and direct $PM_{2.5}$. And Apte and others (2012) estimate, but only for direct $PM_{2.5}$, intake fractions for 3,646 urban centers across the world, accounting for local population density and meteorology.[20] Here the city-level intake fractions for direct $PM_{2.5}$ (or a simple average of them for countries with more than one city in the data of Apte and others, 2012) are extrapolated to the country level, by weighting them by the fraction of the population living in the relevant urban area.[21] Intake fractions for SO_2 and NO_x by country are then inferred from the Humbert and others (2011) estimates, scaling them by the ratio of the intake fraction for $PM_{2.5}$ for that country to the global average intake fraction for $PM_{2.5}$ from Apte and others (2012).[22]

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[19] Some characteristics for natural gas plants (e.g., the rate and temperature at which emissions are released) may differ somewhat from those for coal but, most likely, this causes little bias in the intake fraction estimates for natural gas emissions.

[20] Although Apte and others (2012) mainly focus on vehicle emissions, their results apply broadly to any ground-level source of these emissions.

[21] The urbanization rate by country for 2010 was obtained from World Bank (2013). Intake fractions for ground-level emissions in rural areas are not available on a country-level basis, though they will be considerably smaller than those for urban areas. Although it makes little difference for the results, an estimate of rural intake fractions is included based on a single global-level estimate reported in Humbert and others (2011). This is then weighted by the rural population share for each country and added to the intake fraction estimates for urban areas.

[22] Motor vehicles also produce volatile organic compounds (VOCs) whose primary effect is to form ozone. Ozone damages are ignored here however, given the much weaker link between ozone and mortality compared with fine particulates.

(ii) From Pollution Exposure to Mortality Risk

This section discusses the two steps needed to assess how (additional) pollution exposure increases mortality risk in different countries. First is to establish the baseline rate of mortality for illnesses potentially aggravated by pollution. Second is to multiply these baseline mortality rates by estimates of the increased likelihood of mortality with extra pollution relative to mortality without extra pollution, and then aggregate over illnesses.

Much of the discussion relies on work by the Global Burden of Disease (GBD) project, which provides the most comprehensive assessment to date of mortality and loss of health due to (pollution-related and other) diseases, injuries, and risk factors for all regions of the world.[23]

Baseline mortality rates

The increased mortality risk due to extra pollution inhaled by a population of given size will depend on the age and health of the population. Seniors, for example, are generally more susceptible to pollution-induced illness than younger adults. Health status also matters—if someone already is suffering from a heart or lung condition that is potentially aggravated by inhaling pollution, they are more vulnerable than a healthy person. And if people are more likely to die prematurely from other causes (e.g., traffic accidents, non-pollution-related illness), they are, by definition, less likely to live long enough to die from pollution-related illness.

The role of these factors can be summarized by calculating an age-weighted mortality rate for illnesses potentially worsened by pollution. The focus is on the four adult diseases—lung cancer, chronic obstructive pulmonary disease, ischemic heart disease (from reduced blood supply) and stroke—all of whose prevalence is increased when people intake pollution.

Annual mortality rates from these four illnesses were estimated for each country, taking into account the age structure of the population, as follows. GBD data provides mortality rates for the four diseases for 12 different age classifications at the regional level, with the world divided into 21 regions (Appendix for Chapter 2 of the Supplement lists the countries within each region). These age classes are for people 25 and older (mortality risks for those under 25 are assumed to be zero—see below). Age-weighted mortality rates by disease at the country level are then obtained using the share of the country's population in each age class.[24]

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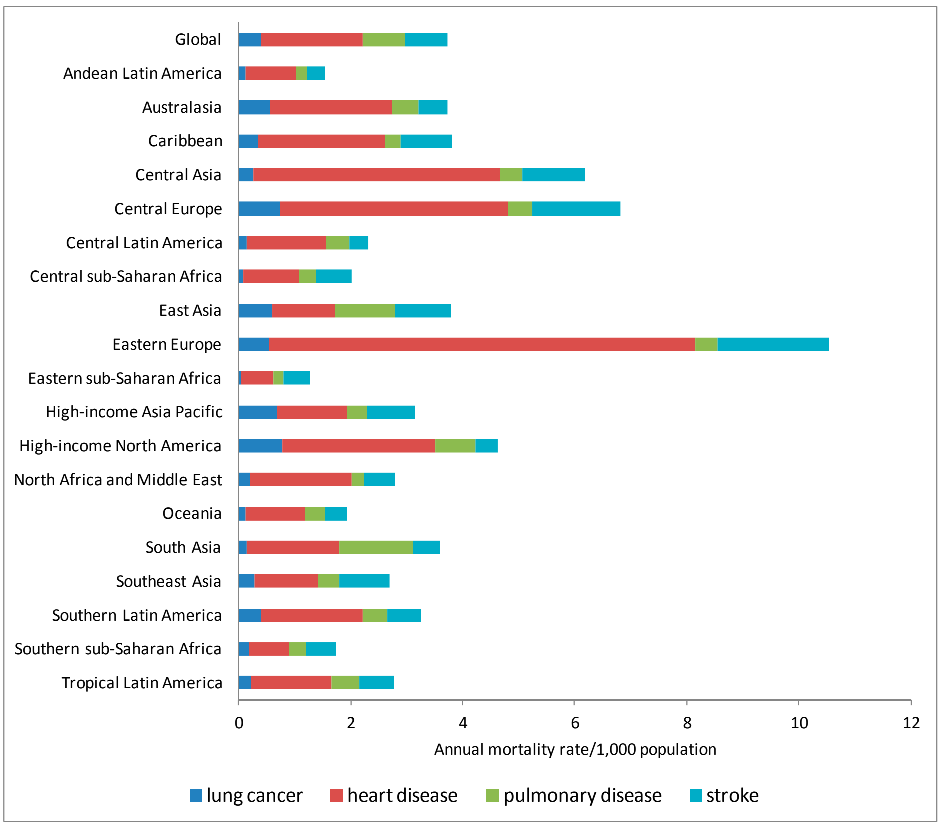
[23] See www.who.int/healthinfo/global_burden_disease/about/en.

[24] The mortality data was obtained from http://ghdx.healthmetricsandevaluation.org/global-burden-disease-study-2010-gbd-2010-data-downloads. The GBD project will provide country-level estimates of mortality rates at later date. The population share data, by country, needed for calculations here and below is from http://unstats.un.org/unsd/demographic/products/dyb/dyb2.htm.

Figure 3.1 shows the results for the regional country groupings. At a global level, the total mortality rate for diseases potentially worsened by pollution is 3.7 deaths per 1,000 people per year (most of these deaths, roughly 89 percent on average, would still occur with no pollution). Eastern Europe has the highest mortality rate, 10.6 deaths per 1,000 people, in part due to the high prevalence of alcohol and smoking related illness. The lowest mortality rate is 1.3 deaths per 1,000 people in Western Sub-Saharan Africa (where people are more prone to die from other causes rather than surviving long enough to suffer pollution-related illness).

Figure 2.1. Baseline Mortality Rates for Illnesses Whose Prevalence is Aggravated by Pollution, Selected Regions, 2010



Sources: See text.

Notes: Figure shows number of people (aged 25 and over) per 1,000 of the population dying from diseases whose incidence can be increased by outdoor air pollution. Only a minor portion should be attributed to pollution—the rest would occur anyway, even if there were no pollution.

For all regions, heart disease is the biggest source of mortality—at a global level it accounts for almost half of total deaths from the four diseases, with pulmonary disease and stroke accounting about 20 percent each, and lung cancer about 10 percent. These shares vary somewhat by region—in Eastern Europe, for example, heart disease accounts for 72 percent of total deaths.

Pollution damages estimated here are understated in the sense that premature deaths to those under 25, most notably form infant mortality, are excluded. One reason for omitting these deaths is that the valuation of mortality risk for infants is even more unsettled and contentious than that for adults (see Box 2.3 below).[25]

Enhanced Mortality from Air Pollution

A limited number of studies for the United States have estimated the relation between pollution concentrations and increased mortality for pollution-related diseases—so-called 'concentration response' functions.[26]

For example, Pope and others (2002) tracked the health status of a large cohort of adults in 61 US cities over a long time period to attribute health outcomes to $PM_{2.5}$ concentrations as opposed to other factors (e.g., age, gender, income, dietary habits, smoking prevalence). Their bottom line estimate was that each 10 microgram/cubic meter increase in $PM_{2.5}$ concentrations increases annual mortality risks from all pollution-related illness in the United States by 6.0 percent. Up till recently, the concentration-response functions underlying Pope and others (2002) were used in regulatory assessments by the United States Environmental Protection Agency (US EPA). However, based on more recent evidence[27] US EPA now assumes a 10 microgram/cubic meter increase in $PM_{2.5}$ concentrations raises all pollution-related mortality risks by 10.6 percent (US EPA 2011, Ch. 5).

An important question is whether these findings—which are based on evidence for the United States where $PM_{2.5}$ concentrations vary geographically by about 5–30 micrograms/cubic meter—apply to other regions. The assumptions used here are based on a best statistical fit, for each of the four pollution-related illnesses, of various model runs for different regions and different types of studies in Burnett and others (2013).[28] The resulting coefficients imply that each 10 microgram/cubic meter increase in $PM_{2.5}$ concentrations increases the risk of all pollution-related mortality (averaged worldwide) by 9.8 percent. Although Burnett and others (2013) provide a state-of-the-art review of the limited number of studies, much more research is needed to improve understanding of the complex relation

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[25] Given that only pollution inhaled by people age 25 and over potentially has heath effects in the assessment, the intake fractions for each country are first multiplied by the share of people age 25 and over in the total population for that country, before applying the pollution/health relationships discussed below.

[26] To apply these relations to intake fractions the latter is first divided by the breathing rate to convert it from pollution inhaled to pollution exposure (see Box 2.1).

[27] See especially Krewski and others (2009), Lepeule and others (2012), and IEC (2006).

[28] Burnett and others (2013) bring together evidence from studies on mortality risks and exposure to ambient air pollution, emissions from solid cooking fuel, second-hand tobacco smoke, and active smoking. In the latter three cases, exposures were converted into estimated annual $PM_{2.5}$ exposure equivalents.

between pollution and mortality (especially for chronic illness), and in the meantime the health impacts calculated here should be viewed very cautiously.

A further caveat is that evidence suggests additional pollution exposure may, paradoxically, have significantly weaker impacts on mortality risk in regions where pollution concentrations are already very high, as the human body becomes progressively 'saturated' with pollution (e.g., Burnett and others, 2013; Goodkind and others, 2012; HEI 2013; WHO 2004). In other words, while the concentration response function appears to be approximately linear in pollution concentrations up to some point (in which case an extra microgram/cubic meter of $PM_{2.5}$ concentrations has the same impact on mortality rates regardless of the initial pollution concentration level), eventually it may flatten out (in which case an extra microgram/cubic meter of $PM_{2.5}$ concentrations has a diminishing impact on elevating mortality rates, the higher the initial $PM_{2.5}$ concentration). However, as discussed in Box 1.3 of Chapter 1 (main report), the corrective fuel tax calculations here abstract from this complication, on the assumption that if efficient taxes were implemented, this would have a large enough impact on emissions to lower pollution concentrations into the region where the concentration response function is approximately linear.[29]

(iii) Valuing Mortality Risks

Health risk valuation is highly controversial. Many people are uncomfortable with the idea of assigning values to the lives saved from policy interventions. Nonetheless, policymakers should still consider methodologies that have been developed for this exact purpose, despite the implication—unpalatable to many—that people with lower income are willing to sacrifice a smaller amount of their consumption to reduce health risks than people with a much higher income.

In reality, people are constantly trading off money and mortality risk in a variety of decisions on a daily basis (e.g., when deciding whether to pay extra for a safer vehicle or to accept a higher-paying but riskier job like cleaning skyscraper windows). Economic studies attempt to measure these trade-offs, and a consistent finding across a broad range of countries is that mortality risk values generally rise with per-capita income (e.g., OECD 2012).

Below, methodological approaches for valuing mortality risks—or more precisely, the value per premature death avoided—are discussed, along with empirical evidence, and what this might imply for different countries. Although not all governments will endorse this approach, the implications for corrective fuel taxes of alternative risk values are easily inferred from the results and accompanying spreadsheets.

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[29] This seems reasonable, based on Figure 1 in Burnett and others (2013), where saturation effects (at least for strokes) become especially pronounced only when $PM_{2.5}$ concentrations approach 100 micrograms/cubic meter. At least on a nationwide average basis, $PM_{2.5}$ concentrations are well below this level for typical countries (see Figure 1.9) and this is at current (rather than efficient) fuel tax levels.

Methodological approaches

There are two distinct approaches to assessing people's 'willingness to pay' to reduce mortality risk—a different approach, based on valuing losses in human capital, is discussed in Box 2.2, though it is generally less preferred by economists.

The 'revealed preference' approach uses observed market behavior to assess mortality risk values, most usually by inferring people's willingness to accept lower wages in return for a job with lower fatality risk (given other characteristics of jobs and workers). In contrast, the 'stated preference' approach relies on responses to (web-based or other) questionnaires, most usually 'contingent valuation' studies where people are asked direct questions about their money and risk tradeoffs.

A potential drawback of revealed preference studies based on labor market data is that they focus on relatively healthy, average-age workers and on (immediate) accidental death in the workplace. Risks from pollution-related mortality—which primarily affects seniors and results from longer-term risk exposure—might be valued somewhat differently.

Stated preference studies can avoid these problems through choice of a sample that is more representative of the at-risk populations and questions about specific hazards (e.g., cancer) posed by air pollution. The main concern about stated preference studies is that they are hypothetical—whether survey respondents would actually behave the way they say they would when confronted with risk/money trade-offs in the marketplace is unclear, leaving open the question of how accurately they describe people's actual tradeoffs.

Both approaches focus on the costs to the individual (and grief to family members) from mortality risk and omit broader costs borne by third parties, such as medical costs. However, these broader costs may be small relative to the value of mortality risks to individuals: for example, when avoided medical costs later in the lifecycle (due to premature mortality) are subtracted from higher short-term treatment costs, the net medical burden may be relatively modest.

Box 2.2. The Human Capital Approach

The human capital approach to valuing mortality risk does not (unlike willingness to pay approaches) measure people's own valuation of these risks—instead it focuses on measuring productivity losses from premature mortality. Traditionally, this approach has been applied to lost years of working-age life, with a person's annual productivity proxied by market wages or per capita GDP, and productivity losses across future years discounted back to the present.

However, the human capital approach may undervalue the full economic cost of premature mortality in several respects. For example, the value of lost non-work time (i.e., time in retirement and leisure time while working age) is often excluded. And people's valuation of pain and suffering prior to death are also excluded, as is grief to surviving family members. For these reasons, economists generally prefer willingness to pay approaches.[1]

[1] For comparison, in SEPA/WB (2007) the costs of air and water pollution in China are about twice as high using the willingness to pay measures of mortality risk compared with the human capital measure.

Empirical Evidence

The starting value for mortality risk valuation used here, and its extrapolation to other countries, is based on a widely peer reviewed study by OECD (2012). This extrapolation accounts for differences in per capita income across countries but not, for reasons discussed in Box 2.3, for other factors (e.g., age).

Starting value for mortality risk reduction. In OECD (2012), the central case recommendation is to value mortality risks in OECD countries (as a group) at $3 million per life saved, in year 2005 US$.

This figure (which is updated below) was obtained from an extensive statistical analysis using several hundred stated preference studies applied to environmental, health, and traffic risks in a variety of countries (mostly the United States, China, Canada, United Kingdom, and France). Stated preference studies were used because they have been conducted in numerous countries, while revealed preference studies have mainly been confined to the United States (where there is ample labor market data). Stated preference studies tend to produce lower valuations than revealed preference studies and in this regard, the pollution damage estimates might be understated here.[30]

[30] Exactly why most stated preference studies imply lower mortality risk valuations than revealed preference studies remains something of a puzzle.

Box 2.3. Other Determinants (Beyond Income) of Mortality Risk Valuation

OECD (2012) discusses several (non-income-related) factors that might cause mortality valuation to differ across countries, but in each case concludes that available evidence is not sufficiently conclusive to make recommended adjustments.

As regards population characteristics, conceivably the average age of the at-risk population matters, but whether on balance this has a positive or negative effect on mortality risk valuation is unclear. On the one hand, older individuals should have lower willingness to pay to reduce mortality risk given that they have fewer years of life left. Offsetting this, however, is that they might be wealthier and therefore have higher willingness to pay to increase expected longevity by a given amount (compared with younger people). Some studies suggest there is little or no net effect of age on people's valuation of mortality risk, while others suggest a modest decrease at older ages (e.g., Krupnick, 2007; Chestnut, Rowe, and Breffle, 2004; Alberini and others, 2004; Hammitt, 2007). Two expert panels in the United States have recommended against age-related adjustments to mortality valuation (Cropper and Morgan, 2007; US NAS, 2008) and the US Environmental Protection Agency has, for now, abandoned analyses with these adjustments.

Even more unsettled is the appropriate value to apply to child mortality—not least because children have not been the subject of revealed and stated preference studies. As noted above, child mortality is excluded from the damage estimates here.

Evidence on whether healthier populations are willing to pay more to extend longevity than less healthy populations is similarly inconclusive (e.g., Krupnick and others, 2000). Unhealthy people may gain less enjoyment from living longer, but if they also gain less enjoyment from consumption, they may be willing to give up more consumption to prolong life. People in different countries may also have different preferences for trade-offs between consumption goods and mortality risks (e.g., due to cultural factors), but again there is no solid evidence on which to base an adjustment. Definitive evidence is also lacking on whether pollution-related risks (e.g., elevated cancer risk) are valued differently from accident risks (e.g., risk of immediate death in a car accident).

Income adjustment. The value for mortality risk (per life) for individual countries (denoted $V_{country}$) is extrapolated from that for the OECD as a whole (denoted V_{OECD}), using the formula:

$$(3.1) \quad V_{country} = V_{OECD}\left(\frac{I_{country}}{I_{OECD}}\right)^{\varepsilon}$$

Here, $I_{country}$ and I_{OECD} denote real income per capita in a particular country, and that for the OECD, respectively. Relative per capita income is appropriately measured using purchasing power parity (PPP) rather than market exchange rates, as the former (which takes the local price level into account) more accurately reflects people's ability to pay out of their income for (local) products or risk reductions (this data is taken from IMF 2013 and World Bank 2013).

ε measures how mortality risk values vary with income: specifically, it is the percent change in the mortality value per one percent change in real per capita income. Based on OECD (2012), the illustrative calculations here assume ε is 0.8.[31]

The \$3 million mortality value for the OECD is updated to 2010 for inflation (using the average Consumer Price Index for the OECD) and real income (using equation (3.1) and the ratio of per capita income in the OECD in 2010 to that in 2005) to give V_{OECD} = \$3.7 million. This figure is then extrapolated to other countries, using (3.1) and their relative per-capita income for 2010.

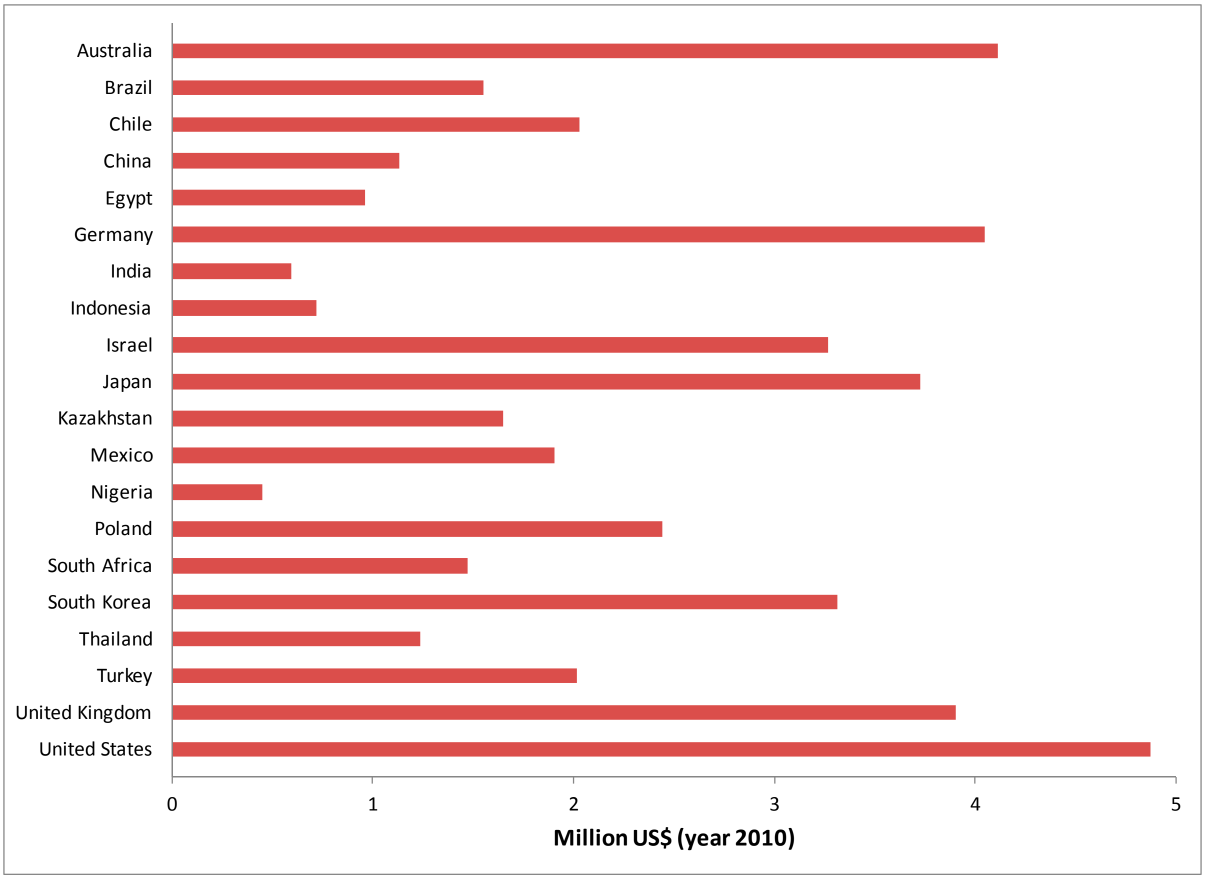
One tricky issue is how to value mortality risks for people across the border in other countries. To keep the approach tractable, the same mortality risk value for these people is used as for people in the country where emissions are released. An alternative (and perhaps more appealing) approach would be to use a weighted average mortality risk valuation, where each affected country's risk valuation is weighted by its share of deaths in total deaths caused by the source country's emissions. If a source country has high per capita income relative to neighboring countries, this approach would imply somewhat lower pollution damage estimates than obtained here and vice versa for emissions from countries with relatively low regional income. However the differences in emissions damages estimated from the two approaches may not be that large: for example, if 40 percent of the affected population resides in other countries and mortality risks for these countries are 50 percent lower than the source country for the emissions, then emissions damages will be 20 percent lower (a notable though not dramatic amount) compared with the approach taken here.

Implied Mortality Risk Valuations

Figure 2.2 indicates the implied mortality risk values for 20 selected countries. Mortality values per death are highest in the United States at \$4.9 million. They are just above \$4 million in Australia and Germany; between \$3 and \$4 million in Israel, Japan, South Korea, and United Kingdom; between \$2 and \$3 million in Chile, Poland and Turkey; between \$1 and \$2 million in Brazil, China, Egypt, Kazakhstan, Mexico, South Africa, and Thailand; and below \$1 million in India, Indonesia, and Nigeria. To re-emphasize, these values are purely illustrative—as shown below, if mortality values in all countries were set at the OECD average, the corrective tax estimates for relatively low income countries would increase considerably.

[31] Alan Krupnick, a leading expert on the issue, recommended a value of $\varepsilon = 0.5$, which would significantly increase pollution damages for relatively low-income countries.

Figure 2.2. Value of Mortality Risk, Selected Countries, 2010



Source. See text.
Note. Figure shows the value assigned to a premature death caused by pollution.

The illustrated mortality values here differ quite a bit from values used at various points in different government studies. However, as shown by the examples in Table 2.1, there appears to be no systemic pattern to these differences—the values for the United States, Canada, and Germany used here are much lower than in government studies for these countries, but the converse applies in other cases. At any rate, the purpose here is not to pass judgment on government practices but simply to obtain, for illustrative purposes, a consistently estimated set of cross-country mortality risk values.

Table 2.1. Examples of Mortality Risk Valuations used in Previous Government Studies

Country	Type of mortality risk	Year of study	government value from study: 000 units of local currency	government value from study: 000 of US$ in 2010	value used here relative to government value
Australia	General	2007	AUD 3,500	2,511	1.64
Austria	Transport	2009	EUR 2,837	3,382	1.27
Canada	Transport	2006	CAD 6,110	5,354	0.78
Denmark	Transport	2012	DKK 16,070	1,769	2.43
France	Transport	2010	EUR 1,360	1,503	2.50
Germany	Pollution	2009	EUR 1,000 - 3,000	1,203 - 3,608	1.12 -3.37
New Zealand	Transport	2009	NZD 3,500	2,179	1.54
Sweden	Transport	2010	SEK 23739 - 31,331	2,558 - 3,377	1.24-1.64
United Kingdom	Transport	2000	GBP 1,145	2,111	1.85
United States	Pollution	2006	USD 7,400	8,007	0.61

Sources: Collected from various government websites and personal communications with government officials.

(iv) Results

Here selected estimates of local air pollution damages per ton of emissions are discussed (damages per unit of fuel are discussed in Chapter 2 of the main report). Appendix for Chapter 2 of the Supplement provides the full set of estimates by emissions, emissions source, and country.

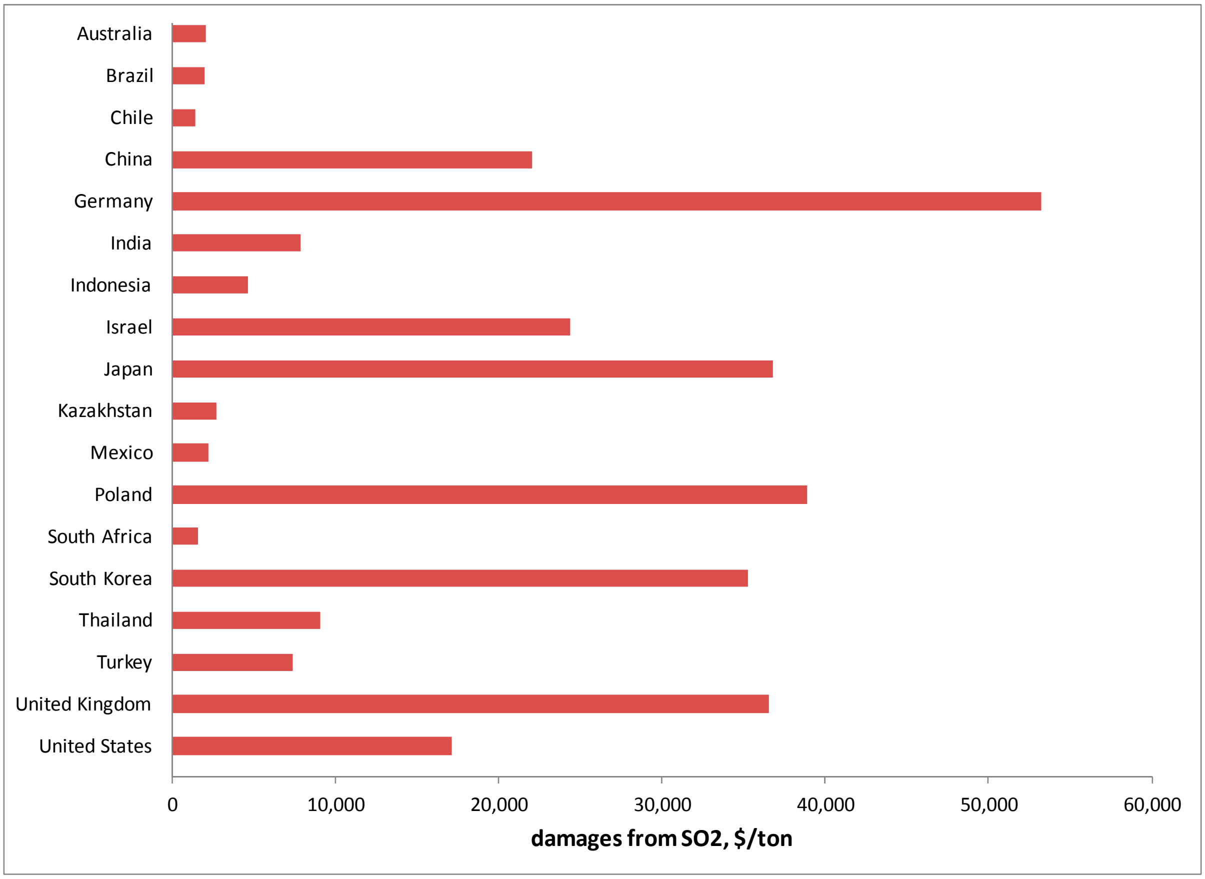
Figure 2.3 shows estimated damages per (metric) ton for SO_2 from coal plants for selected countries. The range of damage estimates is striking.

The United States, with a damage of about $17,000/ton (in year 2010$), is an intermediate case. Damages estimates are a lot higher (about $35,000–39,000/ton) in Japan, Poland, South Korea, and the United Kingdom, and higher still (about $53,000/ton) in Germany, reflecting much higher population exposure to power plant emissions (this more than outweighs any influence of lower mortality values for these countries).

On the other hand, due to a combination of lower population exposure and lower mortality risk values, Australia, Brazil, Chile, Kazakhstan, Mexico, and South Africa have dramatically lower damage values (about $1,500-3,000/ton). For example, premature mortalities per ton of emissions in Australia are just 15 percent of those for the United States.

Damages for China are about $22,000/ton: although the illustrated value of mortality risk value for China is only 23 percent of that for the United States, this is more than offset because average population exposure to emissions is six times as high.[32]

Figure 2.3. Damages from Coal Plant Sulfur Dioxide (SO_2) Emissions, Selected Countries, 2010



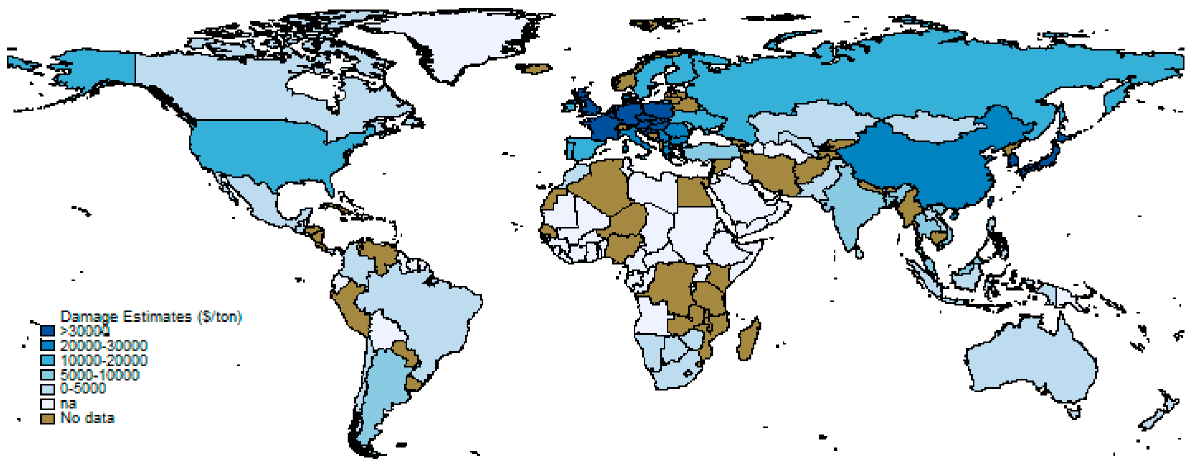
Sources: See text.

Figure 2.4 shows a heat map indicating SO_2 damages/ton from coal combustion for all countries, where damages are grouped into various bands. No color indicates a country that does not consume coal, while brown indicates a country where data to estimate population exposure is not available. Damages/ton are highest in European countries where both per

[32] The China findings seem to be broadly in line with a recent, and far more sophisticated, assessment of lives saved from an SO_2 control policy in Nielsen and Ho (2013). If all their estimated acute and chronic deaths avoided are attributed to the reductions in SO_2 emissions this would imply (see their page ix) about 25 lives saved per kiloton reduction in SO_2 emissions, though this is an overstatement as some of the deaths avoided are due to indirect reductions in other pollutants (and they are careful to emphasize large uncertainties associated with these estimates). The calculations here imply about 17 lives saved in China per kiloton reduction in SO_2 emissions.

capita income and population density are relatively high, while for countries in North and South America, Asia, and Oceania damages/ton generally take intermediate values (for Africa, many countries do not use coal, and for those that do data restrictions often preclude damage estimates).

Figure 2.4. Damages from Coal Plant Sulfur Dioxide (SO_2) Emissions, All Countries, 2010



Sources: See text.

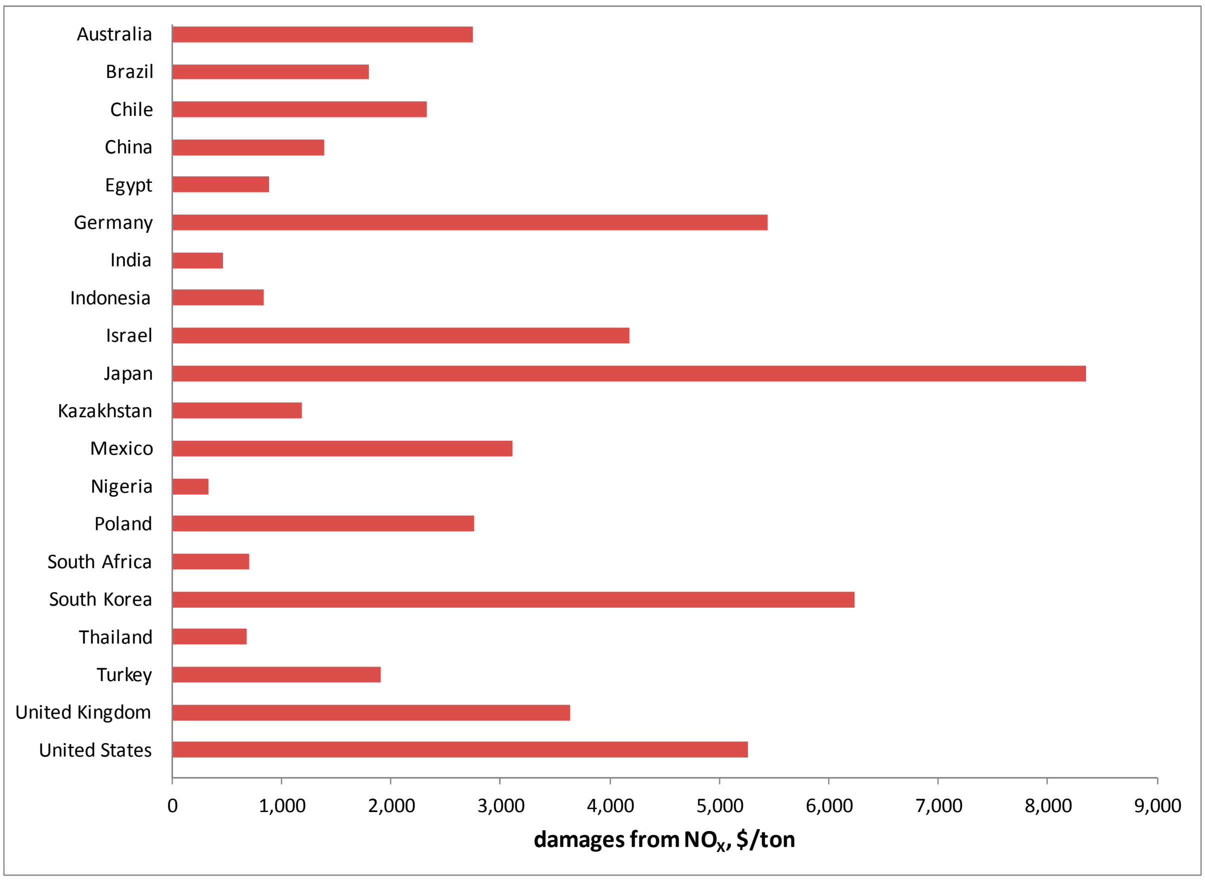
Notes. na refers to no coal use in a particular country.

As regards other power plant emissions, on a per ton basis damages from direct coal plant emissions of $PM_{2.5}$ are around 25 percent larger than for SO_2—this is a broadly consistent finding across countries, so the relative pattern of damages across countries is similar to that for SO_2 in the previous figures. Damages for NO_x (from coal plants) also follow a broadly similar pattern across countries as for SO_2 and $PM_{2.5}$, though in absolute terms damages/ton from NO_x are around 20 to 50 percent lower than for SO_2 (mainly because NO_x is less prone to forming fine particulates). As regards NO_x emissions from natural gas plants (essentially, the only source of local emissions from these plants), the damages/ton across countries are generally similar to the NO_x damages from coal plants (often damages from the former are higher, reflecting the tendency of gas plants to be located closer to population centers, but the differences are not especially pronounced).

Figure 2.5 shows the damages/ton from ground-level NO_x emissions (estimated for vehicles but also applied to home heating). Again, there are significant cross-country differences, for example, estimated damages exceed $5,000/ton in Germany, Japan, South Korea and the United States but are less than $1,000/ton in India, Indonesia, Nigeria, South Africa and Thailand.

The relative differences are, however, smaller than for power plant emissions. This is because ground-level emissions tend to remain locally concentrated and therefore the large average distance between cities, for example in the United States, or the coastal location of cities, for example in Australia, reduce population exposure to a much lesser extent than for power plant emissions. Consequently, damages/ton in the United States are much closer to those in Germany, and in Australia are closer to typical European countries, in Figure 2.5 compared with the relative damages for power plant emissions in Figure 2.3.

Figure 2.5. Damages from Ground-Level Nitrogen Oxide (NO_x) Emissions, Selected Countries, 2010



Sources: See Chapter 3.

(v) Robustness Checks

The above assumptions about how pollution exposure affects health are based on state-of-the-art evidence in the GBD—though this evidence is far from definitive—and Chapter 2 of the main report notes how corrective tax estimates vary with alternative values for mortality risk.

This subsection focuses on a couple of other issues relevant for tall smokestack emissions that are dispersed over great distances (rather than locally concentrated, ground-level emissions).

First is the reasonableness of the air pollution modeling (based on Zhou and others, 2006) implicitly underlying the intake fractions for China (from which intake fractions for other countries are extrapolated). Second is how and to what extent the failure to capture cross-country differences in meteorology and related factors might bias damage estimates from the intake fraction approach.

These issues are examined by comparing selected results from the intake fraction approach with those from the TM5-Fast Scenario Screening Tool (FASST).[33] This tool (described in the appendix for Chapter 2 of the supplement) provides a simplified representation of how pollution concentrations in different regional 'boxes' change in response to additional emissions, and links these changes to population exposure and health impacts. The parameters underlying the air quality component of the model are chosen so that it yields predictions consistent with those from a highly sophisticated model of regional air pollution formation developed by the UN Environment Program (UNEP 2011).

Unlike in the intake fraction approach, cross-country damage estimates from TM5-FASST capture regional differences in meteorology, ammonia concentrations, and other factors. On the other hand, the estimation of population exposure is for regions averaged over large areas—the world is divided into 51 of them—which will understate population exposure if (as seems likely) power plants are located in areas with higher population density than the regional average.[34] Insofar as possible, other inputs to TM5-FASST—particularly baseline mortality rates by region and disease, impacts of additional $PM_{2.5}$ exposure on mortality rates, and the local valuation of mortality risks—are chosen to be consistent with the intake fraction approach, to facilitate a cleaner comparison of results.

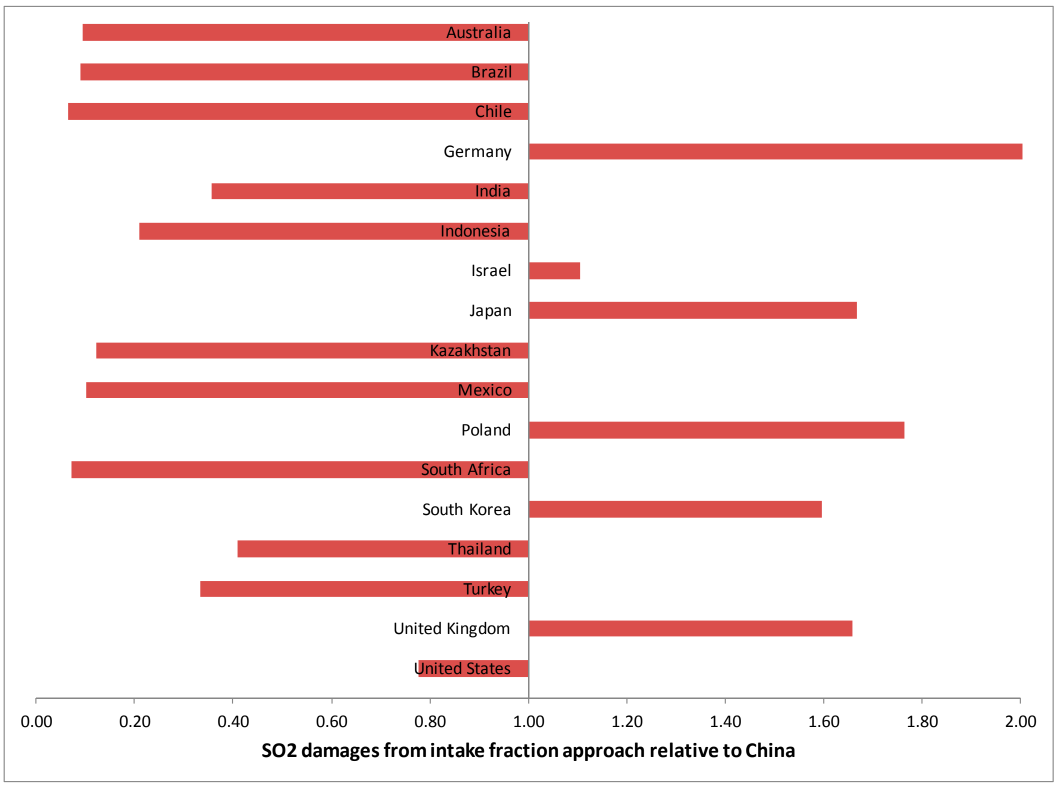
As regards the first issue, TM5-FASST estimates SO_2 damages/ton for China at about $12,000, or just over half of the damage estimate from the intake fraction approach. Some of this difference reflects, as just noted, differences in population exposure, but some also likely reflects differences in assumptions about the impact of emissions on air quality. Unfortunately, it is not really possible to make a definitive judgment about which air quality model (underlying the above intake fraction approach and the FASST-TM5 model) is the more realistic.

[33] Simulations using this tool were conducted for this report by Nicholas Muller.

[34] For example, one region is the whole of China, another the United States, and another combines Angola, Botswana, Malawi, Mozambique, Namibia, Zambia, Zimbabwe.

As regards the second issue, Figures 2.6 and 2.7 show SO_2 damages/ton for selected countries expressed relative to damages/ton for China, from the intake fraction approach and TM5-FASST, respectively. To the extent there are differences in relative damages for particular countries across the two figures, this suggests differences in meteorological factors between that country and China play a potentially significant role. For some cases, this does not appear to be a major concern—for example, the two approaches suggest damages/ton for Japan are 62–67 percent higher than for China, and 22–24 percent lower for the United States. But there are some exceptions, for example, relative damages for Israel, Poland, and the United Kingdom from the intake fraction approach are substantially higher than from TM5-FASST,[35] and vice versa for Thailand and Turkey. In short, meteorological factors can significantly alter damage estimates in certain cases, though both the sign and scale of these effects are very country specific.[36]

Figure 2.6. Estimated SO_2 Damages Relative to China from Intake Fraction Approach, 2010

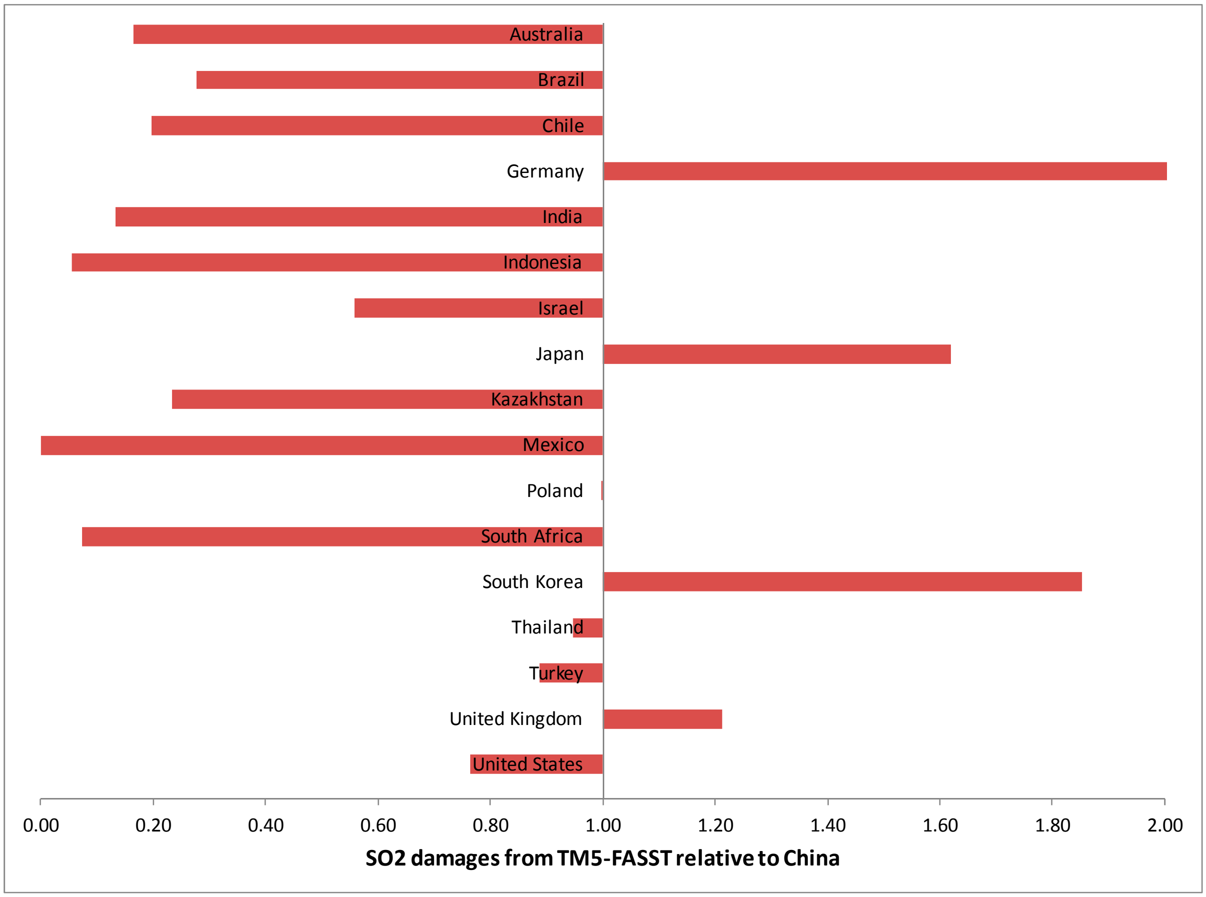


Sources: See text.

[35] A possible explanation is that accounting for winds blowing from west to east partially transports pollution away from these densely populated countries to less densely populated countries (like Scandinavia and Ukraine) which is taken into account in Figure 2.7 but not Figure 2.6.

[36] One further caveat is that, as discussed in Zhou and others (2006), they estimate a somewhat higher impact of NO_x on $PM_{2.5}$ concentrations at intermediate distances from power plants than in other studies. Although this implies higher damage values for NO_x, there is little effect on the corrective coal estimates in Chapter 2 (main report) as NO_x damages are still modest relative to damages from SO_2 and direct $PM_{2.5}$.

Figure 2.7. Estimated SO_2 Damages Relative to China from TM5-FASST Model, 2010



Sources: See text.

(vi) Expressing Damages per unit of Fuels

To assess efficient taxes on fuel use, the above damages expressed per ton of emissions need to be converted into damages per unit of fuels, or per unit of energy, using appropriate emissions factors. These factors relate the amount of emissions (e.g. SO_2) released into the atmosphere to combustion of a particular fuel (e.g., natural gas) in a particular activity (e.g., power generation). The Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) model, developed by the International Institute for Applied Systems Analysis (IIASA), was used to estimate these factors—see Box 2.4 for more details.[37]

[37] For information about the GAINS model, see http://gains.iiasa.ac.at and IIASA (2013). Fabian Wagner provided considerable help in calculating all the emissions factors.

Box 2.4. Emissions Factors from the GAINS Model

The GAINS model, estimates—by country—emission factors for carbon and local air emissions for different fossil fuels used in different sectors of the economy. These are reported in kilotons of pollutant per petajoule (heat content) of fuel input though they could be expressed in emissions per unit of weight or volume (by multiplying heat content per unit of weight/volume using the GAINS data). Two calculations are performed.

First, uncontrolled emission factors (denoted EF_U) are calculated from the basic properties of the fuel and combustion processes (e.g., Amann and others, 2011; Cofala and Syri, 1998a and b; Klimont and others, 2002). For example, as defined for SO_2 emissions, the emissions factor is calculated by:

$$EF_U = \frac{sc}{hv} \cdot (1 - sr)$$

Here *sc* is the sulfur content per unit of weight; *hv* is the heat value per unit of weight; and *sr* is the sulfur retention fraction (i.e., the portion of sulfur that is retained in ash rather than released into the atmosphere).

Second, various controlled emission factors (denoted EF_C) are calculated for emissions after application of an abatement technology denoted *t* (e.g., a particular type of scrubber, or hotter boiler that reduces NO_x emissions), from the following formula:

$$EF_C = EF_U \cdot (1 - re_t)$$

Here re_t represents the fraction of emissions that are abated (and that would otherwise be released into the atmosphere) due to technology *t.* Where specific regulations (technology mandates, emission rate standards) exist (and are enforced) GAINS calculates the controlled emissions factors based on the regulation. GAINS can be used to report an average across all emissions control technologies that might be applied to plants, taking into account the potential application rates of alternative control technologies *t*.

For mobile emissions sources, GAINS again calculates uncontrolled and controlled emissions factors for gasoline and diesel. In each case, the controlled factors are an average across technologies applied to existing (on-road) vehicles.

The GAINS data for all these calculations is quite detailed for some countries, while in other cases judgment is used to transfer estimates to countries where data is not directly available.

As regards coal, emissions factors (for CO_2, SO_2, NO_x and direct $PM_{2.5}$) are defined relative to energy or heat content in petajoules (PJ) rather than tons of coal, given significant variation in energy content across different types of coal. Where relevant, the factors represent a weighted average across different coal types—in these cases, a more refined pricing system (than the one estimated here) would vary charges according to the emissions intensity of the particular coal type.

One emissions factor is obtained for carbon (where opportunities for abating emissions at the point of combustion are presently very limited) and two factors—one uncontrolled and the other controlled—for the local air pollutants. These factors are used to estimate two different taxes, one for coal used by plants with no emissions controls and the other for (the average plant) with such controls (e.g., SO_2 scrubbers). In each case, the corrective tax is the product of the emissions factor for a pollutant and the damage per ton for that pollutant, and aggregated over all pollutants.

Similar procedures are used to obtain emissions factors and corrective taxes for natural gas used by power plants and households (though in the latter case only uncontrolled emissions are relevant). Damages and corrective taxes are again expressed per unit of energy, because emissions per unit of volume can vary significantly depending on gas pressure.

As regards mobile sources, CO_2 and NO_x emissions factors per PJ and also per liter were obtained for gasoline vehicles and similarly, along with SO_2 and $PM_{2.5}$, for diesel vehicles (the latter representing an average over light- and heavy-duty vehicles using diesel). GAINS provides both controlled and uncontrolled emissions factors for gasoline and diesel vehicles (averaged across both new and used vehicles on the road). The calculations here assume 90 percent of gasoline vehicles and 100 percent of diesel vehicles have control technologies in developed countries[38] and for developing countries application rates for control technologies are half of those for developed countries.

There are several interesting points to note about the emissions factors.

First, there is very little variation across countries in carbon emissions factors for a particular fuel. However, there is a lot of variation across fuel products—per PJ of energy, natural gas, gasoline, and motor diesel generate about 59 percent, 73 percent, and 78 percent of the carbon emissions generated per one PJ of coal.

Second, there is significant cross-country variation in uncontrolled and controlled SO_2 emission factors for coal (see Figure 2.8). For example, on average the SO_2 emissions/PJ for Japanese coal plants with no control technologies is only 30 percent of that for comparable US plants, while in Israel the emissions rate is about 70 percent greater than for the United States. Control technologies can dramatically reduce emissions however, for example, SO_2 emission rates at US coal plants with such technologies are 80 percent lower than for plants without these technologies, and 91 percent lower for German coal plants. Direct $PM_{2.5}$ emissions rates for uncontrolled plants follow a similar pattern to those for SO_2, but the control technologies have an even more dramatic impact on reducing pollution.

Third, NO_x emission rates from (uncontrolled) plants differ from rates for ground-level sources (depending, for example, on combustion temperature which can affect the amount of nitrogen and oxygen sucked in from the ambient air), but the differences are not large.

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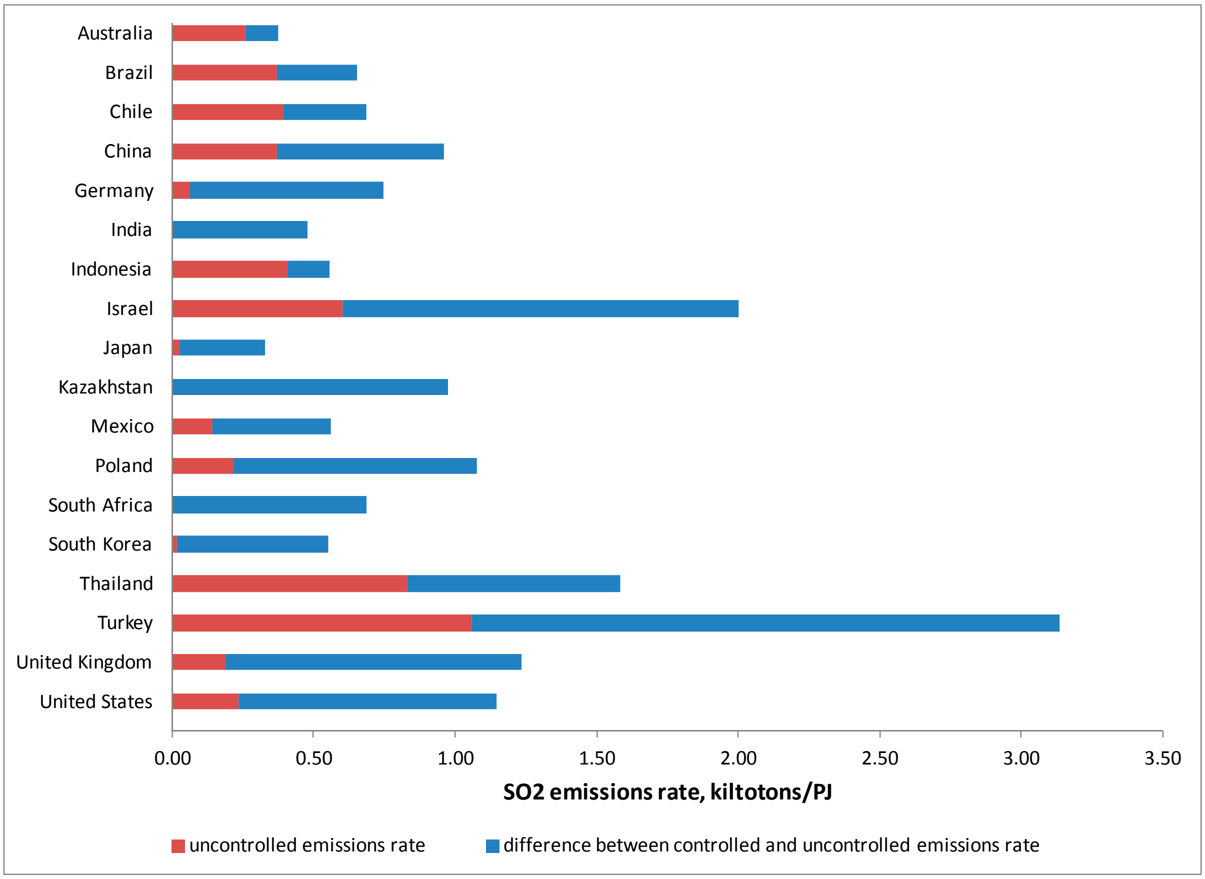
[38] This seems reasonable, for example, based on a comparison of the IIASA data with average emission rate data for the on-road, US light-duty vehicle fleet reported in BTS (2012), Table 4.43.

Figure 2.8. Average SO_2 Emission Rates at Coal Plants with and without Control Technologies, 2010



Sources: See text.

Notes: The uncontrolled emissions rate for a country is the sum of the two bars. For some countries (e.g., India, South Africa), there had been no adoption of sulfur control technologies in 2010.

C. Summary

An illustrated value for CO_2 damages is taken from a recent study, although these damages are much disputed.

To assess air pollution damages from power plant combustion of coal and natural gas, average population exposure to these emissions (which can be transported long distances) is estimated and combined with data on local mortality rates for pollution-related illness and evidence about how changes in pollution exposure affects these mortality rates. The most controversial step is monetizing these health effects which, for illustration (without making any recommendations) are inferred from a recent OECD study, though the implications of alternative assumptions are transparent from the results. Estimation of pollution from motor vehicle fuels and other ground-level sources are built up from studies about how much of these emissions are inhaled by people in different urban centers. All these pollution damage estimates are then combined with data on local emissions factors for different fuels to infer environmental damages from fuel use (though emission rates when control technologies are applied are very different from the uncontrolled emission rates for both power plants and vehicles).

Chapter 3. Measuring (Non-Pollution) Externalities from Motor Vehicles

This chapter consists of three sections focused on the three major, non-pollution-related, externalities from motor vehicles, namely traffic congestion, traffic accidents, and (to a much lesser extent) wear and tear on the road network (which is relevant for trucks). Other data and assumptions needed to implement the corrective (motor) fuel tax formulas from Chapter 1 of the main report are discussed in the appendix for Chapter 3 of the supplement.

A. Congestion Costs

Basically, what is needed here is the cost of reduced travel speeds for other road users due to an extra km of driving by one vehicle, averaged across different roads in a country and times of day. This cost estimate can then be used in the formula for corrective motor fuel taxes (equation (1.1) of the main report). As noted in Chapter 1 (main report), to most effectively manage congestion on the road network, countries should ideally transition to km-based taxes that vary with the prevailing degree of congestion on different roads. However, until these schemes are comprehensively implemented, it is appropriate to charge motorists for congestion costs through fuel taxes.

The congestion cost has two main components. First is the added travel delay (on average) to other road users, as defined a bit more technically in the appendix for Chapter 3 of the supplement, which (due to lack of direct data) needs to be extrapolated at the country level. Second (to convert delays into a monetary cost) is the *value of travel time* (VOT), which is related to local wage rates.

The discussion is organized as follows.

Subsection (i) begins by using a city level database (covering numerous countries) to establish statistical relationships between congestion delays and various transportation indicators. Subsection (ii) uses these results, and country-level data for those same indicators, to extrapolate congestion delays to the country level. Section (iii) discusses how delays are converted into congestion costs. Section (iv) discusses the results. And section (v) provides a quick check on the results by comparing them with cost estimates obtained from detailed country-level data (for the couple of countries where this data is readily available).

The focus is on the most important cost component (time lost to motorists). Box 3.1 discusses some broader costs that should, in principle, be factored into corrective fuel tax assessments, but which are beyond the scope here. For this reason, and other assumptions made below, the congestion cost estimates here are probably on the low side.

Box 3.1. Some Broader Costs of Congestion

One additional cost (beyond the pure time losses from travel delay) is the added fuel cost to other motorists, due to the possible deterioration in fuel efficiency experienced under congested conditions. The link between slower travel speeds and fuel consumption rates is complicated however (e.g., Greenwood and Bennett, 1996; Small and Gómez-Ibáñez 1998, section 3.2). Sometimes congestion slows traffic (without increasing stop and go), which could improve fuel efficiency over some range of (relatively fast) travel speeds. For the United States as a whole, Shrank and Lomax (2011), pp.5, put added fuel costs at about 5 percent of the total costs of congestion, suggesting that these costs may have only very modest implications for corrective fuel taxes.

Other, more subtle, costs of congestion may be more significant. For example, people may choose to set off earlier or later to avoid the peak of the rush hour, which may cause them to arrive earlier or later at their destination than they would otherwise prefer (perhaps because early arrival means they waste time waiting for an appointment, or late arrival runs the risk of penalties at work). Furthermore, congestion can result in day-to-day uncertainty over travel times, making it more difficult to plan out the day (e.g., scheduling appointments, dinner times, and day care pickups). Studies suggest that travel time variability alone might raise the overall costs of congestion by around 10 to 30 percent (e.g., Eliasson, 2006; Fosgerau and others, 2008; Peer and others, 2012).

(i) Travel Delays at the City Level

The starting point is the Millennium Cities Database for Sustainable Transport which provides detailed information on transportation in 100 cities (though 10 are discarded due to missing data).[39] These cities are listed in the appendix for Chapter 3 of the supplement.

The data is for 1995 and therefore rather dated, though it is the best available. Moreover, the age of the data need not be a problem as it is used for estimating statistical relationships between travel delay and transportation indictors which are then matched with recent, country-level data on those indicators, to provide up-to-date country-level estimates of travel delay. This approach is reasonable so long as the statistical relationships between delays and transportation indicators have not changed substantially since 1995.

The average road network speed in the database is the average speed of all motor vehicles (7days/24 hour average) on all classes of road in the metropolitan area.[40] This data provides

[39] The database was developed by the International Association of Public Transport (UITP) and the Institute for Sustainability and Technology (ISTP) in 2001.

recurrent congestion delays (occurring each day under normal driving conditions) but not (the average amount of) non-recurrent congestion (occurring from sporadic events like accidents, bad weather, and road-works). In this sense congestion costs are understated, perhaps significantly.[41]

As indicated in Table 3.1, across all cities the average travel speed is 34.2 km/hour, with speeds well above this average in North American cities (47.7 km/hour) and well below it in non-affluent Asian cities like Delhi (20.6 km/hour).

The speed data is used to infer average travel delays using assumptions about travel speeds that would occur in the absence of congestion.[42] As indicated in Table 3.1, these estimated average delays per vehicle km are lowest in North America (0.006 hours/km), and more than twice as large in Western and Eastern Europe, the Middle East, Africa, and affluent Asian cities like Tokyo. Delays per km are greater still in Latin American cities and non-affluent Asian cities.

Table 3.1. City-Level Travel Delays and other Characteristics, Averaged by Region, 1995

Region	Number of cities	Average speed, km/hour	Average delay, hours/km	Metropolitan GDP, 1995 US$ per capita	Annual km driven per car, 1000 km	Road capacity per car, km/car	Cars per capita
Africa	7	33.6	0.0159	2,500	11.8	33.2	0.1
Asian Affluent Cities	5	31.3	0.0164	34,800	12.2	16.3	0.22
Other Asian Cities	12	20.6	0.0342	4,200	10.5	20.0	0.09
Eastern Europe	5	31.3	0.0164	5,600	7.6	8.1	0.31
Western Europe	33	32.9	0.0144	31,900	11.3	12.4	0.41
Latin America	5	29.4	0.0195	5,400	10.1	16.0	0.19
North America	15	47.7	0.0058	27,900	18.5	17.3	0.57
Middle East	3	36.9	0.0153	7,700	14.9	12.7	0.19
Oceania	5	44.2	0.0074	19,800	12.9	22.4	0.58
All Cities	90	34.2	0.0158	21,000	12.4	16.6	0.34

[40] The speed data are calculated through traffic counts and assumptions about how speed varies with traffic volume.

[41] A study for Canada, for example, suggests that non-recurrent congestion costs could be as large as those for recurrent congestion (Transport Canada, 2006).

[42] These free flow speeds (which are not available in the data) are taken to be 57 km/hour (35 miles/hour) or 65 km/hour (40 miles/hour), according to whether cities have relatively high or relatively low road density per urban hectare (in fact for some cities, the observed travel speeds are pretty close to the free flow speeds). These assumptions are roughly in line with those in Parry and Small (2009).

Sources: Millennium Cities Database and (for average delay) authors' calculations.
Notes: Figures are simple averages across urban centers in different regions. The average road network speed is the average speed of all vehicles (7days/24 hour average) on all classes of road in the metropolitan area.

The first step is to use statistical regressions to obtain a relationship that can be used to predict average delays for countries as a whole, using some common indicators that are available both for (90 cities in) the Millennium Cities Database and in the country-level data discussed below. These variables include:

- metropolitan GDP per capita (an indicator of a city's level of economic development);
- annual car km (an indicator of traffic mobility);
- road length or capacity per car;
- cars in use per capita (this, and the previous variable, are indicators of traffic intensity, relating to transport infrastructure and supply).

Commonly used statistical techniques are used to estimate coefficients that show the contribution of each of these indicators in explaining average travel delays across cities, using functional forms that best fit the data. Further details, along with the statistical regression results, are provided in the appendix for Chapter 3 of the supplement.

Ideally (to improve statistical accuracy) additional variables would be included in these regressions. However as the purpose is for country-level extrapolations, only those indicators for which data is available at the country level can be used. Despite this limitation, a reasonably good statistical fit is still obtained.

(ii) Projecting Country-Level Delays

The estimated statistical relationships between the average delay and the four key indictors at the city level are now used to project the average delay for 150 countries (i.e., all countries for which this data is available), with the country-level indicators. For this purpose, GDP per capita is taken from World Bank (2013) and all other indicators from the World Road Statistics, published by International Road Federation (IRF 2009).[43] For a large number (81) of the countries, data on car kms travelled is missing. The appendix for Chapter 3 of the supplement describes how this data gap was filled, using supplementary statistical regressions.

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[43] The most recent data is for 2007, which is assumed to provide a reasonable approximation for delays in 2010.

Table 3.2 summarizes the key indicators by regions. At the country level (comparing with Table 3.1), per capita incomes are lower, annual km driven per car are higher, and road capacity per car is smaller, compared with the city level data.

Table 3.2. Country-Level Travel Delays and other Characteristics, Averaged by Region, 2007

Region	Number of countries	Predicted average delay, hours/km	Country GDP, 2007 US$ per capita	Annual km driven per car, 1000 km	Road capacity per car, km/car	Cars per capita
Africa	45	0.0046	2,300	36.3	1,395	0.03
Asia	33	0.0053	9,900	16.3	362	0.11
Europe	43	0.0025	26,900	9.4	65	0.35
Latin America	11	0.0049	5,100	21.9	185	0.09
North America	11	0.0048	12,100	19.5	103	0.15
Oceania	7	0.0028	11,900	18.1	290	0.2
All Countries	150	0.0041	12,400	21.0	551	0.16

Sources: IRF (2009) and authors estimates to fill in some of the data on annual km driven per car (see Appendix for Chapter 3 of the Supplement). Average delay is predicted using procedures described in the text.
Notes: Figures are simple averages across countries—therefore, for example, high average delays in Mexico substantially inflate the average delay figure for North American countries.

The estimated coefficients from the city-level analysis are used together with country-level variables to predict the average (nationwide) delay in the 150 countries. Since the city-level regression is based on 90 major cities, the average delay predicted represents the urban congestion level for each country, excluding the rural areas. To predict the average delay at the country level, the predicted urban average delay is scaled by the urban population ratio (on the assumption that rural congestion is negligible).

Comparing the results, shown in the second column of Table 3.3, with those from Table 3.2, the average vehicle delays at the country level are about one quarter to one-half of those at the city level. This makes sense—the city level data focuses only on delays in large cities (where congestion is especially severe), while the country-level estimates also account for driving in rural areas and medium to smaller cities. Nonetheless, it is important to bear in mind that average travel delays at the country-level are estimated with a fair amount of imprecision, especially for countries where there might be substantial errors in the measurement of transportation indicators.[44]

(iii) From Delays to Congestion Costs

This section explains how delays that one vehicle imposes on others are inferred from the above estimates and then monetized. Complications posed by other vehicles on the road (e.g., buses) are also discussed.

Inferring delays on others imposed by one vehicle

It turns out that, under a specification commonly used by transportation engineers for the relationship between travel speed/time and traffic volume, there is a very simple relationship between the average delays per km (estimated above) experienced by individual drivers, and the increased travel time that one extra vehicle implies for all other vehicles on the road.

In particular, when travel delay is a simple power function of traffic volume (relative to road capacity), with the exponent in this function denoted by β, then the extra delay one vehicle imposes on other vehicles is simply β, times the average delay per km (see the appendix for Chapter 3 of the supplement). Empirical studies suggest that β is roughly in the range of 2.5–5.0, with higher values in this range applicable to larger urban centers: here β is taken to be 4.[45]

Finally, delays to other passengers are obtained by multiplying delays to other vehicles by the vehicle occupancy rate, taken to be 1.6.[46]

Alternative assumptions about vehicle occupancy and the exponent β would have proportional effects on the congestion costs reported below (e.g., if $\beta = 5$ or average vehicle occupancy is 2, congestion costs would be 25 percent greater).

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[44] In a handful of cases (where the results looked especially questionable), average delay per km was extrapolated from another country. For example, for Bangladesh and Kazakhstan extrapolation was from India and Russia respectively (making an adjustment for differences in urbanization rates between countries).

[45] This assumption is consistent with the Bureau of Public Roads formula, which has been the traditional method for predicting vehicle speed as a function of the volume/capacity ratio. See Small and Verhoef (2007) pp. 69–83 and Small (1992) pp. 70–71, for further discussion.
Obviously the above approach is highly simplified—speed/volume relationships may vary a lot with the characteristics of specific roads (e.g., speed limits, frequency of stop lights and sharp bends) and across different times of day. But the above assumption seems to represents a reasonable rule of thumb for representing average travel conditions in urban areas.

[46] This is slightly higher than average vehicle occupancy rates for London, Los Angeles, and Washington DC calculated in Parry and Small (2009).

Value of Travel Time (VOT)

The discussion now turns to the VOT, which is needed to monetize congestion costs.

According to economic theory (e.g., Becker, 1965), on average people should organize their time such that they are indifferent between an extra hour at work and an extra hour of leisure (e.g., relaxing at home, looking after the children). Therefore, an extra leisure hour is commonly valued by the benefit to individuals of an extra hour of foregone work, namely the after tax hourly wage (i.e., the market wage after netting out personal income and employee payroll taxes, and consumption taxes paid when wages are spent).

As a first pass, people might also value an extra hour of travel time by the net of tax wage, which would suggest a VOT of around 50–70 percent of the market wage for a typical advanced country. More generally, the monetary cost of travel time might be lower (if people enjoy driving, for example, because they can listen to music) or higher (if people enjoy the workplace, for example, because of interaction with colleagues).

There is a large empirical literature estimating the VOT for personal travel using similar revealed and stated preference techniques to those discussed in Chapter 2. In this case, the former might involve, for example, estimating people's willingness to pay extra auto fuel and parking costs to save time over an alternative, slower travel mode, while the latter might involve directly asking people what tolls they might pay for a faster commute.

For the United States, Canada, France, and United Kingdom, literature reviews suggest a VOT of about half the market wage is a reasonable rule of thumb for general automobile travel (see Table 3.3). The VOT is somewhat higher for commuting (e.g., due to penalties for late arrival at work) than for leisure-related trips (e.g., shopping, taking the children to school, going to the gym)—16 percent higher according to Wardman (2001). Here the VOT is taken to be 60 percent of the market wage, given that most delays occur during the (commuter-dominated) peak period.

Table 3.3. Reviews of Empirical Literature on the Value of Travel Time (VOT)

Study	About the study	recommended VOT, % of market wage
Waters (1996)	Reviews 56 estimates from 14 countries	35-50
Wardman (1998)	Review of UK studies	52
Mackie et al. (2003)	Review of UK studies	51
US Department of Transportation (1997)	Review of US studies	50
Transport Canada (1994)	Review of US and Canadian studies	50
Commissariat General du Plan (2001)	Review of French studies	59

Notes: Summary findings for these reviews were taken from Small and Verhoef (2007), pp. 52–53. Studies take a weighted average over different trip types (usually at peak period) except for Waters (1996) who focused exclusively on commuter trips.

The VOT/market wage ratio is taken to be the same across all countries.[47] The wage data is from the International Labor Organization's Global Wage Database (ILO 2012) and is a nationwide measure for year 2010.[48]

Figure 3.1 shows the VOT for selected countries. Broadly speaking, the relative pattern of VOTs across countries is similar to that for the value of mortality risks in Figure 2.2 of Chapter 2.[49]

[47] There is plenty of evidence (at least from advanced countries) that the VOT increases approximately in proportion to income, which backs up this assumption (see, e.g., Small and Verhoef, 2007, pp. 52, and Abrantes and Wardman, 2011, who determine that a 10 percent increase in income increases the VOT by 9 percent). It might be argued that the VOT should be adjusted upwards in countries with relatively low vehicle ownership rates, where ownership is skewed towards higher wage groups. No adjustments are made however, partly because of data limitations. But also the issue is not clear cut—conceivably, higher income people (at least those living in more expensive housing closer to downtown areas) drive less under congested conditions than other motorists.

[48] There are data gaps for six countries in ILO (2012). For these cases, wages are proxied using GDP per capita from www.imf.org/external/pubs/ft/weo/2013/01/weodata/index.aspx. Ideally urban wage rates (adjusted downwards for differences compensating for higher living costs) would be used in preference to nationwide wages, but a comprehensive, international data set is not available.

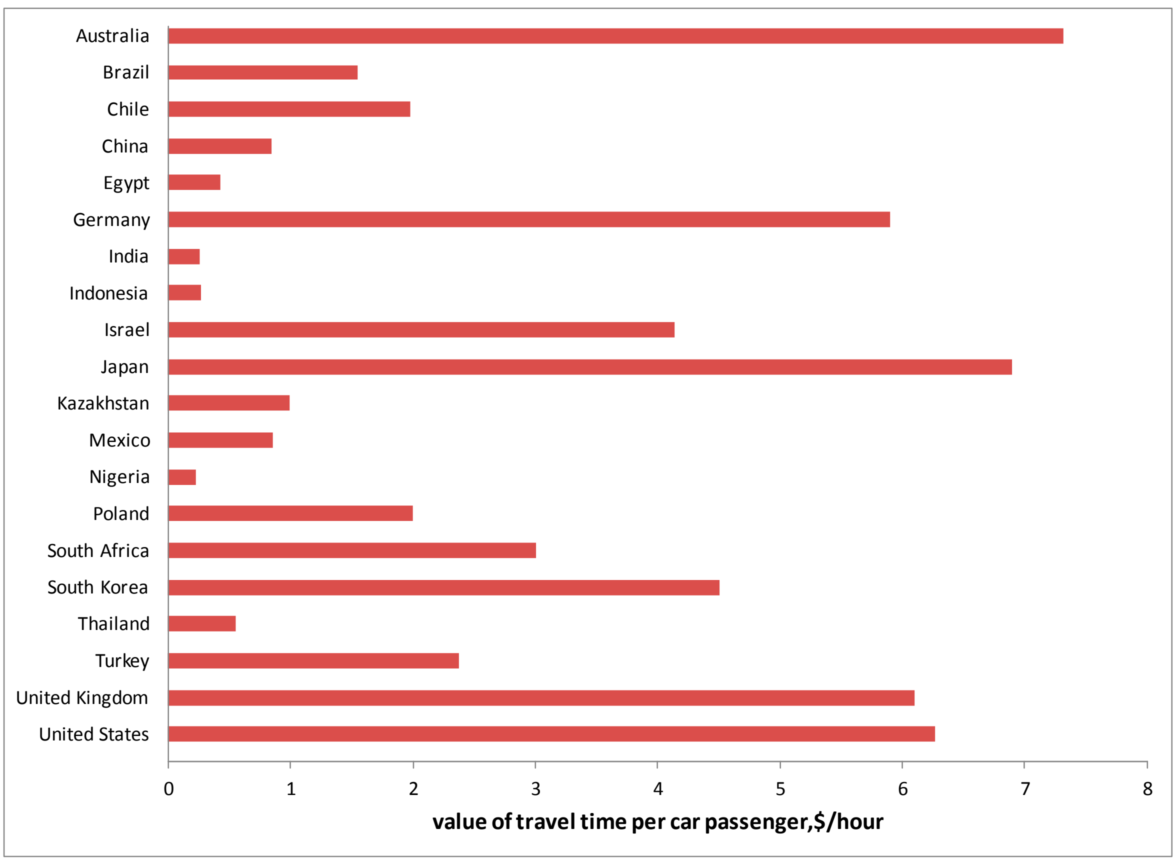
[49] There are some nuances. The relative differences between developed and developing countries are a bit more pronounced as here relative wages across countries are compared while Figure 2.2 compares relative income raised to the power 0.8. And there are some differences even among similar income countries. For example, the United States has a higher mortality valuation than Australia but (slightly) lower value of travel time, reflecting the depressing effect on US wages of relatively high labor force participation among migrants and secondary family workers, and relatively little influence of labor unions or labor market regulations on inflating wages.

Accounting for other vehicles

The estimates here assume that all vehicles on the road are cars, whereas in practice the vehicle fleet reflects a mixture of cars, buses, trucks, and two-wheelers. The appendix for Chapter 3 of the supplement discusses and applies a formula that shows the ratio of congestion costs (properly estimated accounting the mix of vehicles) relative to the congestion cost estimated here.

Basically, if trucks and/or two-wheelers account for a sizeable portion of the vehicle fleet (but buses do not) the estimates are not very different. However, if buses account for a significant portion of vehicle kms then the estimates here can substantially understate congestion costs (see the appendix for Chapter 3 of the supplement). The reason is that cars have significantly more impact on increasing travel times for other road users when a greater portion of vehicles on the road are carrying large numbers of passengers. An adjustment is not made here however, because data on the share of buses in (urban) vehicle kms is not available for many countries.[50]

Figure 3.1. Value of Travel Time, Selected Countries, 2010



Sources: See text.

[50] And in many cases the bus share is very low (e.g., about one percent or less of vehicle km travelled in Washington DC, Los Angeles, and London—see Parry and Small 2009).

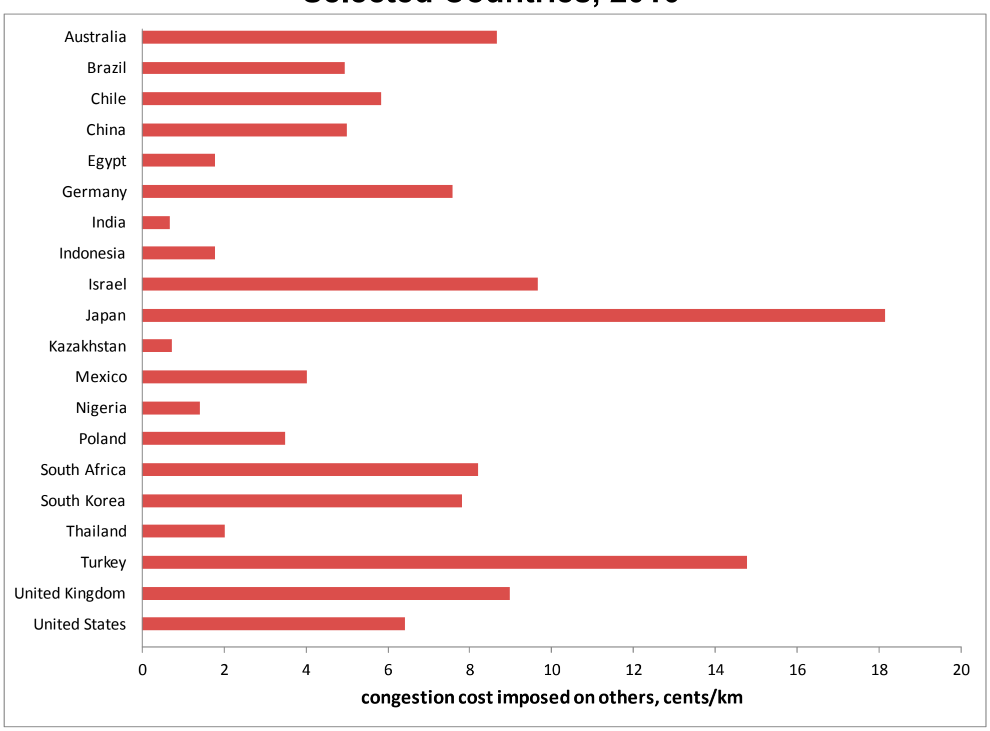
Finally, in the computation of corrective diesel fuel taxes, an extra truck km is assumed, based on the literature,[51] to contribute twice as much to congestion as an extra car km (trucks drive slower and take up more road space, though a partially offsetting factor is that they tend to be driven less intensively on congested roads).

(iv) Results

Figure 3.2 shows nationwide congestion costs (imposed on others) per extra car km, for twenty selected countries.

The congestion cost for the United Kingdom, for example, is 9.0 US cents/km. Australia, Germany, Israel, South Africa, and Korea all have a broadly similar congestion cost, while Turkey's is substantially higher, and Japan's higher still (although Japan has a relatively high VOT, most of the difference is due to its greater estimated travel delays). Congestion costs for the United States, where a smaller portion of nationwide driving occurs under congested conditions, are lower at 6.4 cents/km (though this US estimate seems on the high side relative to a potentially more accurate estimate discussed below). China's estimated congestion cost is 5.0 cents/km, less than the US (despite China having greater average travel delays) due to its much lower assumed VOT. Low VOTs also help explain the low congestion costs (less than 1 cent per km) in, for example, India and Kazakhstan.

Figure 3.2. Congestion Costs (Imposed on Others) per Car km, Selected Countries, 2010



Sources: See text.

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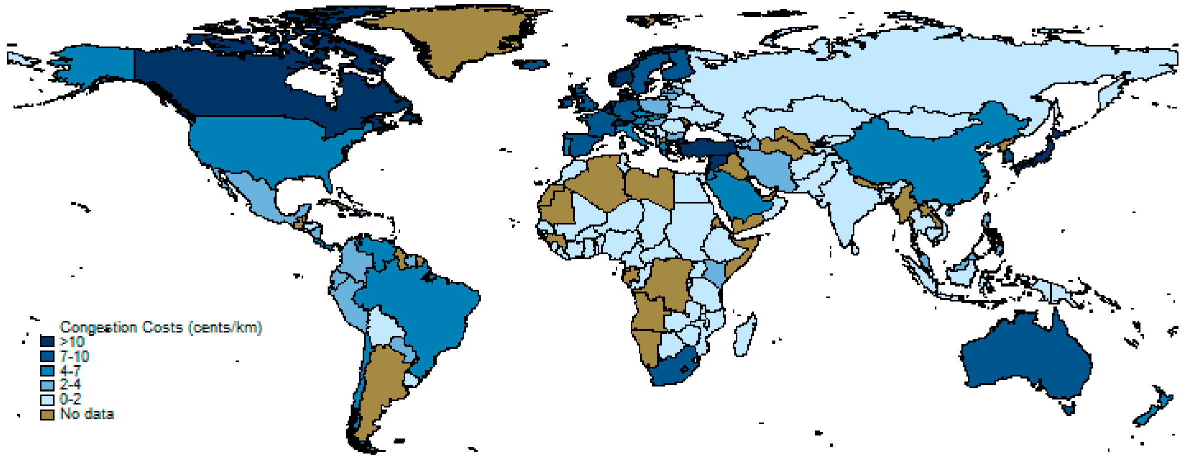
[51] See, for example, Lindsey (2010), pp. 363, TRB (2010), Parry and Small (2009).

Figure 3.3 shows ranges of estimated congestion costs for all countries (where data permits). Again these are relatively high in Western Europe (where a large portion of driving occurs under congested conditions and people have a high valuation of travel time) and (except for South Africa) relatively low in Africa (where the value of travel time is lowest). The United States, Latin America, and Australia, are intermediate cases.

Figure 3.3. Congestion Costs (Imposed on Others) per Car km, All Countries



Sources: See text.

(v) Robustness Checks

For the United Kingdom and the United States, detailed data on travel delays for road classes in different regions are available that can be combined into an alternative estimate of nationwide average delay, as a check on the above estimates (see the appendix for Chapter 3 of the supplement for estimation procedures and the data sources).

For the United Kingdom, the average delay per vehicle km obtained from this other data source is almost exactly (within 1 percent) of that estimated above, providing some re-assurance that the above approach (at least for the United Kingdom) might be reasonable. For the United States, the average delay using the alternative data is 59 percent of that estimated above, suggesting that the above estimate may be on the high side for that particular country.

The delay estimates from country-level data should be more reliable than the extrapolations presented above, though the former are surprisingly hard to come by (transportation authorities do not routinely collect this data). The above approach suffers from imprecision, given the rather limited number of indicators common to both the city and country level data, issues with the quality of both city- and country-level data sets, and the possibility that the underlying relationship between the average delay and city/country characteristics might have evolved over time with changes in infrastructure, technology, and traffic rules. However, it is hard to gauge the direction (let alone magnitude) of bias for individual countries. Moreover, broadly speaking the pattern of relative congestion costs across different countries estimated above seems plausible, even though the individual country estimates may not be especially accurate.

B. Accident Costs

The total societal costs from road traffic accidents can be substantial (and are often under-appreciated). Gauging the appropriate charge for accident risk to reflect in fuel taxes (as a complement to other measures like road safety investments) is difficult however, for two reasons.

First, conceptually it is a bit tricky to judge whether certain categories of costs should be viewed as 'internal' because individuals take them into account in their driving decisions, or external, that is, borne by others (only the latter warrant corrective taxes).

Second, although data is often available for road fatalities, it is usually not available for other accident costs (e.g., nonfatal injuries, medical and property damage costs), and even fatality data is not always broken down in a way that permits assessment of external costs.

The estimates here necessarily rely on some judgment calls, extrapolations (to fill in missing costs), and transfers of fatality breakdowns across similar countries. For these and other reasons, the accuracy of cost estimates can be questioned. But again, the estimates at least provide some plausible and transparent sense of (external) accident costs, they shed light on why these costs differ across countries, and they highlight the data needed to improve the future accuracy of cost assessments.

The discussion proceeds as follows.

Subsection (i) walks through some conceptual issues, attempting to categorize different accident costs into internal versus external risks. Section (ii) discusses the estimation of (external) costs. Subsection (iii) presents some results.

(i) Classifying Accident Risks—Some General Principles

Here the main societal costs of road accidents are discussed, which include personal costs of fatal and non-fatal injuries, medical costs, and property damages. Costs for trucks are also mentioned.[52]

[52] There are a range of other costs from traffic accidents such as those from traffic hold ups, police and fire services, insurance administration, and legal costs. However, these are beyond the scope here as, at least

Injuries

It is helpful to consider separately injury risks to pedestrians/cyclists, to vehicle occupants in accidents involving only a single vehicle, and to vehicle occupants in accidents involving multiple vehicles.

Pedestrian/cyclist injuries

It is normally assumed that motorists do not take into account injury risks they pose to pedestrians and cyclists when deciding how much to drive.[53]

Injury risk to occupants in single-vehicle collisions

As for accidents involving one vehicle, it is standard to view the injuries to occupants of such vehicles as risks that are taken into account: in other words, if individuals put themselves at greater risk (by getting in the car more often), this is not viewed as a basis for taxation to deter this behavior.[54] For similar reasons, injury risks to other occupants (e.g., family members) in single-vehicle collisions are generally viewed as internalized risks.

Injury risks to occupants in multi-vehicle collisions

Here the delineation between internal and external risks becomes murky.

The issue is how extra driving by one vehicle affects injury risks to occupants of other vehicles. All else the same, extra driving by one motorist implies more cars on the road and greater risks to others, because cars have less road space on average and are therefore more likely to collide—in this case, injury risks to other vehicle occupants would increase (approximately) in proportion to the amount of traffic.

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according to some studies, they appear to be modest relative to other costs (e.g., US FHWA, 2005; Parry, 2004, Table 2). That might seem surprising for traffic hold ups, as some accidents cause severe traffic disruptions, though these accidents constitute only a small share of total accidents. As for productivity losses, these are taken into account in the monetary values assigned to different types of injuries.

[53] See, for example, Newbery (1990), Parry (2004). Once on the road, drivers likely take care to lower the risks of hitting pedestrians/cyclists though, as this is reflected in observed injury data, it is taken into account in the corrective tax estimates.

[54] Motorists may lack an accurate sense of risks to themselves but, in the absence of evidence to the contrary, it seems reasonable to assume that the average motorist does not systematically understate or overstate these risks—and even if there were such evidence, information campaigns to better educate drivers might be a better response than corrective fuel taxes.

However, all else might not be the same: with more vehicles on the road, motorists may drive more carefully or be obliged to drive slower. This implies an offsetting reduction in accident frequency and in the average severity of injuries in a given accident (because vehicles collide at slower speeds). Although slightly slower driving may not do much to reduce injury risks to (unprotected) pedestrians, the effect may be more pronounced for other vehicle occupants (who have a greater degree of protection). Thus, what matters is the impact of additional driving on the 'severity-adjusted' injury risk to other drivers. However, available evidence on this is inconclusive.[55]

Here an intermediate assumption between two more extreme cases is considered. In one case, additional driving leads to a proportionate increase in injury risks to others (i.e., there is no offsetting decline in severity-adjusted accident risk due to slower/more careful driving). In the second case, extra driving has no effect on (severity-adjusted) injury risks to others (i.e., increased risk to others is completely offset by a decline in the average severity of injuries).

In the first case, it is assumed that half of injuries in multi-vehicle collisions are external based (approximately) on the logic that on average one vehicle is responsible for the collision and another is not and that those at fault take into account risks to occupants of their vehicle but not occupants of other vehicles (e.g., Parry 2004). In the second case, all injuries in multi-vehicle collisions are internal. Splitting the difference here implies that one-quarter of multi-vehicle collision injuries are treated as external.

Medical and property damage costs

Medical costs (associated with all traffic-related injuries) are largely borne by third parties (the government or insurance companies), though individuals typically bear some (minor) portion of these costs through, for example, co-payments and deductibles.

It is difficult to pin down how much property damage (primarily repairs/replacement costs for damaged vehicles) drivers take into account. In countries with comprehensive insurance systems, some costs are borne by third parties (insurance companies) but other costs are borne by drivers in the form of deductibles and possibly elevated future premiums following a crash.[56]

[55] For example, Edlin and Karaca-Mandic (2006) find that extra driving substantially increases average insurance costs per km driven, suggesting higher per km property damage costs (though how other costs, like fatality risk, change is not clear). On the other hand, studies by Lindberg (2001), Traynor (1994), Fridstrøm and others (1995) suggest that extra driving may have only limited effects (and possibly even a negative effect) on severity-adjusted accident risk.

[56] Premiums may also vary (usually very moderately) with an individual's (stated) annual driving, which also provides some (albeit very weak) link between extra driving and property damages (in the form of greater premiums) paid by drivers. And to the extent that insurance companies have some market power, this may also, in effect, tax motorists for risks of property damage.

Accident risks from heavy vehicles

As regards accident costs for trucks (needed to compute the corrective diesel fuel tax) at first glance, it might appear that trucks would impose much greater risks to other road users than cars, given their much greater weight. But a counteracting factor is that trucks are driven at slower speeds than cars and another is that truck drivers are professionals, which may further reduce their crash risk (e.g., because truck drivers are unlikely to drink and drive).[57] According to a detailed study by the United States Federal Highway Administration (US FHWA 1997, Table V-24), overall external accident costs per vehicle km are only slightly higher for heavy vehicles than for cars—therefore these costs are taken here to be the same for cars and trucks.[58]

(ii) External Cost Assessment

For most countries, IRF (2009) provides data on traffic fatalities (for year 2007 or close to it), based on local data (e.g., from police reports). And WHO (2013a) provides data on the breakdown of fatalities by pedestrians, cyclists, occupants of (motorized) two to three wheelers, occupants of four wheelers, and a miscellaneous category (e.g., bus riders). In cases where only total fatalities are reported, the breakdown is assumed to be the same as in another, similar country in the same region. The data used here likely underreports, perhaps substantially, road fatalities for many developing countries providing, yet another reason why the corrective fuel tax estimates presented later might be understated.[59]

The vehicle occupant data does not separate out deaths in multi-vehicle collisions from those in single-vehicle collisions. Based on a simple average across five country case studies (discussed in the appendix for Chapter 3 of the supplement), 57 percent of fatalities to occupants of two, three, and four wheelers are assumed to occur in multi-vehicle collisions. Of these (based on the above discussion) 25 percent represent external fatality risks, along with all of the pedestrian/cyclist fatalities. The same values by country are used to monetize these fatalities, as used in Chapter 3 for pollution deaths.[60]

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[57] In 2010, the crash frequency (per km driven) in the United States for light-duty vehicles was almost four times that for trucks (BTS, 2012, Tables 2.21 and 2.23).

[58] See also Parry and Small (2009).

[59] For example, fatalities in India are 114,444 for 2007 according to IRF (2009), but were estimated at 231,027 in 2010 by WHO (2013a). Using the WHO data would further strengthen the finding in Chapter 5 that current taxes fall short of their corrective levels in most cases.

[60] In principle, it might seem that a higher value should be used for traffic-related deaths, given that the average age of someone dying in a road accident is lower than for the average person dying from pollution-related illness (e.g., Small and Verhoef 2007, pp. 101). However for reasons discussed in Box 2.3 of Chapter 2, an adjustment is not made.

Data is not available, for most countries, on other components of external accident costs—various nonfatal injuries, medical costs and property damages. However, based on comprehensive estimates of these costs for the United States, United Kingdom, Sweden, Finland, and Chile a relationship between the ratio of these other external costs[61] to the external costs of fatalities was estimated, as a function of the share of external fatalities in total fatalities (see the appendix for Chapter 3 of the supplement)—in countries with a high incidence of pedestrian deaths, the relative size of other external costs tends to be smaller. The external cost ratio was then inferred for different countries, based on their share of external fatalities in total fatalities, and the external costs scaled up accordingly.

(iii) Results

Figures 3.4 and 3.5 show the external accident costs for selected countries, and for all countries, respectively, expressed, to facilitate comparison with congestion costs, per vehicle km of travel by car or truck (see the appendix for Chapter 3 of the supplement on measurement of vehicle km).

Higher income countries tend to have a lower incidence of injuries per km driven because, as countries develop, vehicle and road safety tends to improve, and the ratio of pedestrians/cyclists to motorists declines (e.g., Kopits and Cropper, 2008).[62] This is partially, but not entirely, offset by higher valuations of fatality and injury risk in higher income countries. Loosely speaking therefore, these figures show a pattern of lower external accident costs per km in higher income countries. For example, costs are below 4 cents/km in Australia, Western European countries, Japan, and the United States, and above 8 cents/km in various Latin American and African countries, India, Kazakhstan, and Russia.

Note also (comparing with Figures 3.2 and 3.3) that external accident costs can be of the same broad order of magnitude as congestion costs. In fact in half of the selected countries, accident costs are larger than congestion costs.

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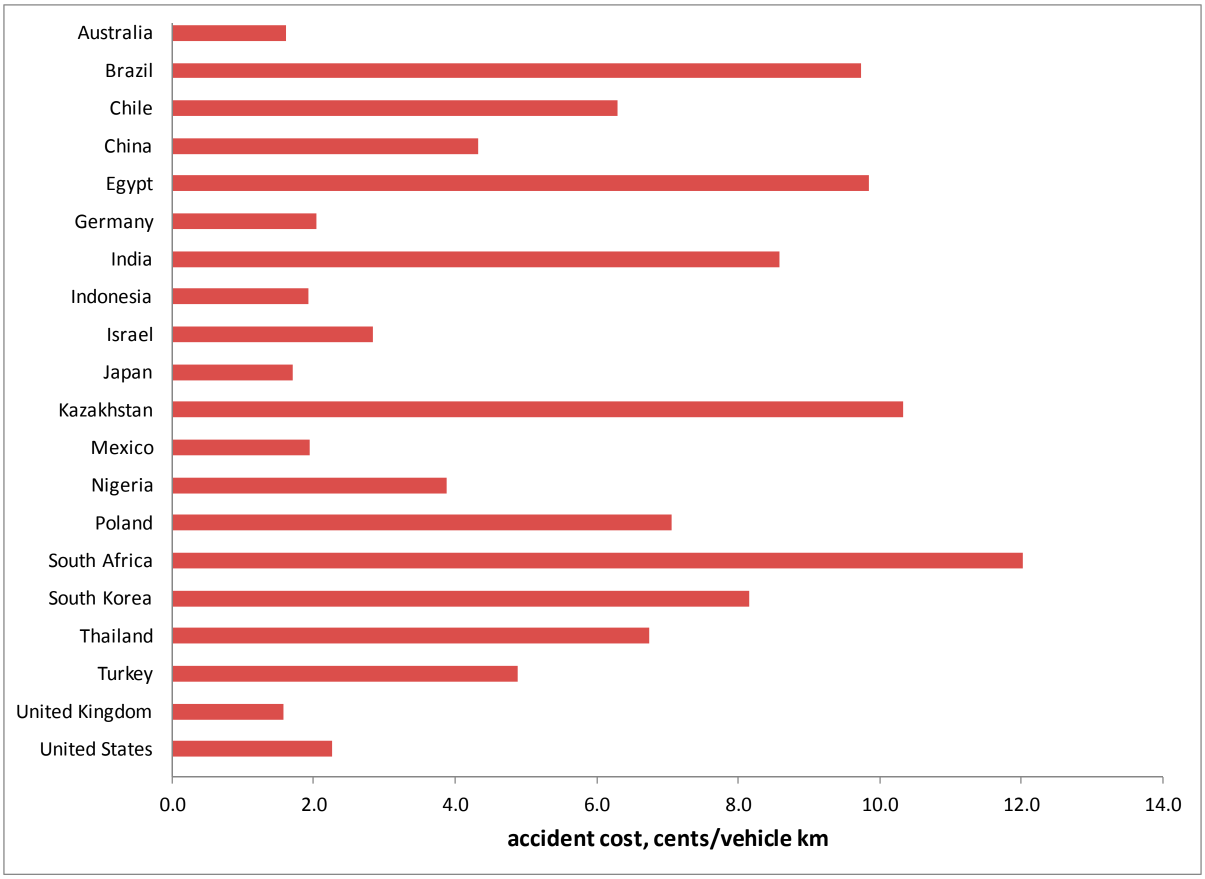
[61] Property damages account for 42 percent of other external costs, non-fatal injuries 38 percent, and medical costs 20 percent, based on a simple average across the studies.

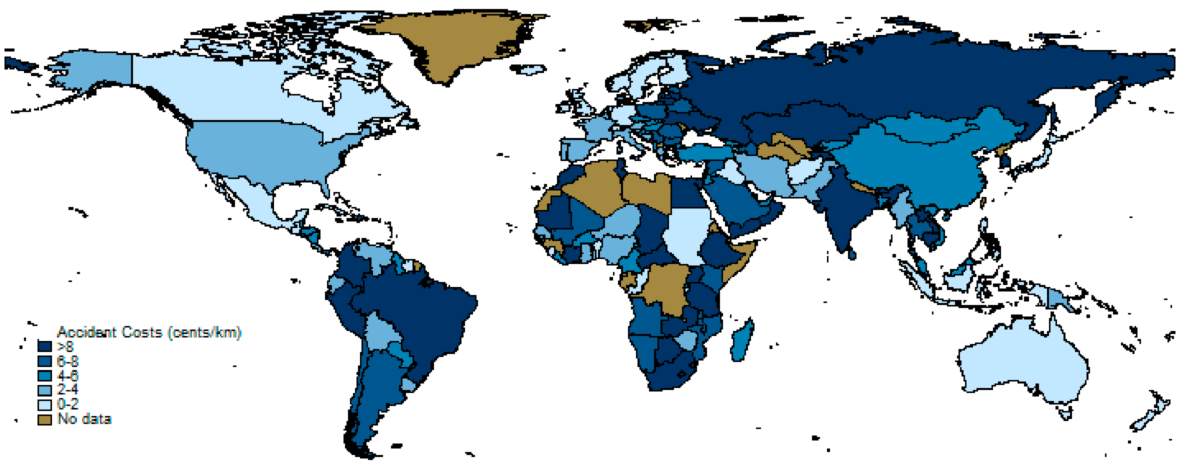
[62] In India, for example, there are 40 external deaths per billion vehicle km compared with 2 in the United States.

Figure 3.4. (External) Accident Costs per Vehicle km, Selected Countries, 2010



Sources: See text.
Notes: Figure shows (external) accident costs (reflecting fatal and non-fatal injuries, medical costs and property damages) expressed per km driven by cars or trucks.

Figure 3.5. (External) Accident Costs per km Driven, All Countries, 2010



Sources: See text.

C. Road Damage Costs

Vehicle use causes an additional adverse side effect through wear and tear on the road network. However, given that road damage is a rapidly rising function of a vehicle's axle weight, nearly all of the damage is attributable to heavy-duty vehicles—road damage costs make little difference for corrective fuel taxes for light-duty vehicles (e.g., US FHWA, 1997) and are ignored here.[63] Road damage costs also vary considerably across different classes of trucks, which would matter for the design of a finely tuned system of axle weight tolls. However, the concern here is with damage caused by trucks as a group, to factor into the corrective diesel fuel tax.

Road damage consists both of the pavement repair costs incurred by the government and increased operating costs for vehicles due to bumpier roads. However, if the government steps in to repair roads once they reach a pre-determined state of deterioration, then as a rough rule of thumb, the total external cost of road damage can be measured by average annual spending on maintaining the road network.[64]

A complication here is that road damage is jointly caused by vehicle traffic and weather (e.g., ice creating and exacerbating holes and cracks in the pavement which are further enlarged by vehicle traffic). Empirical work to provide a rough rule of thumb for apportioning damage to trucks versus weather is rather sparse. Depending on road strength (e.g., thickness), Paterson (1987), pp. 372, suggests that vehicles cause 40 to 90 percent of damage in warm, dry, or sub-humid climates; 20 to 80 percent in arid, freezing climates; and 10 to 60 percent in moist, freezing climates.[65] Here trucks are assumed to account for 50 percent of the damage in all countries.[66]

IRF (2009), Table 8.2, provides spending on road maintenance and capacity investments (aggregated over all levels of government) for 2007 (or thereabouts) for 74 countries. For other countries IRF (2009) provides total highway spending, but not the maintenance/investment decomposition—this is therefore inferred from a similar country in the same region, for which the breakdown is available. For the remaining 10 countries where no spending data is available at all, maintenance expenditure per truck km is taken to be the

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[63] Road damage increases approximately in proportion to the third power of a vehicle's axle weight (e.g., Small, Winston, and Evans, 1989).

[64] If roads are repaired more frequently, government resource costs are higher but there is less deterioration of vehicle operating costs, and vice versa. See Newbery (1988) for a more precise discussion.

[65] See also Newbery (1988) for similar findings.

[66] A more accurate calculation (but which would have little impact on overall corrective diesel fuel tax estimates) would involve classifying countries by climate zone and (better still) average road strength, and applying different assumptions about the fraction of damages attributable to trucks.

same as a similar country in the same region. Scaling by the share of trucks (as opposed to climate) in total damage gives the damage attributable to trucks by country.

D. Summary

Congestion costs are estimated using extrapolations of travel delays from a city-level database to the country level (in the absence of direct data on these delays) and travel time valuations from the literature. Accident costs are assessed by making assumptions about what portion of road fatalities in different countries reflects risks that motorists do not take into account, and making some upward adjustments to allow for other components of accident risk (property damage, medical costs, and non-fatal injuries). Both congestion and accident costs are sizeable (and likely understated)—in some cases, accident costs exceed congestion costs. Road damage is also estimated by attributing a portion of road maintenance expenditures to trucks, though these costs are modest in relative terms. There are a few extra steps needed to infer corrective motor fuel taxes, but for the details, the reader is referred to the appendix for Chapter 3 of the supplement .

All of the cost estimates are rudimentary and there will be ample scope for reforming them in future as data (e.g., on travel delays) becomes more widely available and analytical work helps to resolve some of the uncertainties (e.g., over the value of travel time in low-income countries, or the safety risks that one driver poses to other road users). For the meantime, however, these cost estimates enable a first-pass estimate of corrective motor fuel taxes that can be scrutinized and might usefully serve as a starting point for discussions about tax reform.

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APPENDICES

The following contains supplementary information for Chapters 2 and 3 of the supplement.

Appendix for Chapter 2 of the Supplement

This appendix provides information about regional classifications for the baseline data on pollution-related mortality rates; country-by-country estimates of air pollution damages from different emissions; and describes the TM5-FASST tool used to conduct additional simulations as a cross-check on the main pollution damage estimates.

(i) Regional Classifications for Mortality Rates

As discussed in Chapter 3, the baseline mortality rates are obtained by combining regional average mortality rates for four pollution-related illnesses by age group with country-level data on the age structure of the population. The regional mortality data is from Burnett and others (2013), with the countries grouped into regional classifications as shown in Table A1.

(ii) Air Pollution Damages by Emissions and Country

Table A2 summarizes, by country, estimated air pollution damages by the type of emissions, and the source of those emissions. Cases where emissions from a fuel are not applicable (because the fuel is not consumed) are denoted by #N/A in black, while cases where data unavailability prevents a damage estimate are denoted by #N/A in red (and the same for other tables in the appendix).

Details on the TM5-FASST Tool

The TM5-FASST tool estimates air pollution damages per ton for different emissions in several steps.

First, the baseline mortality rates for four pollution-related illnesses are calculated by region, according to the following formula:

(A1) $$RR(PM_{2.5}) = 1 + \alpha \cdot \Delta PM_{2.5}$$

Here RR denotes the risk of premature death for a particular illness relative to that in the baseline case (with current pollution concentration levels). RR-1 is therefore the proportionate change in the relative risk. $\Delta PM_{2.5}$ denotes the change in $PM_{2.5}$ concentrations relative to the initial situation. α is a parameter which is calibrated separately for each of the four pollution-related diseases to be consistent with state-of-the-art evidence in Burnett and others (2013).

The change in premature deaths for a change in $PM_{2.5}$ concentrations is given by:

(A2) $$(RR - 1) \times mortality\ rate \times population$$

where mortality rate refers to the baseline rate and population is the exposed population (consists of all ages greater than 25 years). Both population data and mortality rate data are provided by (IHME, 2013). Deaths are monetized using the same mortality values as used in Chapter 2 of the supplement.

Next, one ton of SO_2 emissions is added from a particular source and processed through an air quality model that links all emissions sources to $PM_{2.5}$ concentrations in different regions—the world is divided into 51 regions. The air quality model in the FASST tool is a simplified version of a far more sophisticated air quality model in UNEP (2011). The change in $PM_{2.5}$ concentrations in each region is then used to calculate changes in premature deaths based on the above equations, and the result is then monetized.

Averaged across the twenty countries, the damages from SO_2 are \$17,640/ton. Individual country estimates, relative to those from the intake fraction approach used here, are discussed in Chapter 2 of the supplement.

As a check on the above results, simulations were also run with an alternative specification for relative risk given by:

(A3) $$RR(PM_{2.5}) = 1 + \alpha(1 - e^{-\gamma \Delta PM_{2.5}^{\delta}})$$

where again parameters α, γ and δ are calibrated for each of the four diseases to be consistent with Burnett and others (2013). However, the results are only moderately affected. For example, the average damage from SO_2 across the twenty countries is 15 percent smaller than when the linear functional form is used.

Table A1. Country Classifications for Baseline, Pollution-Related Mortality Rates

Asia Pacific, High Income	**Australasia**	**Europe, Western**	**Latin America, Andean**	**Sub-Sah. Africa, Central**	**Sub-Sah. Africa, West**
Brunei	Australia	Andorra	Bolivia	Angola	Benin
Japan	New Zealand	Austria	Ecuador	Central African Republic	Burkina Faso
South Korea		Belgium	Peru	Dem. Republic of the Congo	Côte d’Ivoire
Singapore	**Oceania**	Switzerland		Congo	Cameroon
	Fiji	Cyprus	**Latin America, Central**	Gabon	Cape Verde
Asia, Central	Federated States of Microne	Germany	Colombia	Equatorial Guinea	Ghana
Armenia	Kiribati	Denmark	Costa Rica		Guinea
Azerbaijan	Marshall Islands	Spain	Guatemala	**Sub-Sah. Africa, East**	The Gambia
Georgia	Papua New Guinea	Finland	Honduras	Burundi	Guinea-Bissau
Kazakhstan	Solomon Islands	France	Mexico	Comoros	Liberia
Kyrgyzstan	Tonga	United Kingdom	Nicaragua	Djibouti	Mali
Mongolia	Vanuatu	Greece	Panama	Eritrea	Mauritania
Tajikistan	Samoa	Ireland	El Salvador	Ethiopia	Niger
Turkmenistan		Iceland	Venezuela	Kenya	Nigeria
Uzbekistan	**Europe, Central**	Israel		Madagascar	Senegal
	Albania	Italy	**Latin America, Southern**	Mozambique	Sierra Leone
Asia, East	Bulgaria	Luxembourg	Argentina	Mauritius	São Tomé and Príncipe
China	Bosnia and Herzegovina	Malta	Chile	Malawi	Chad
North Korea	Czech Republic	Netherlands	Uruguay	Rwanda	Togo
Taiwan	Croatia	Norway		Sudan	
	Hungary	Portugal	**Latin America, Tropical**	Somalia	
Asia, South	Macedonia	Sweden	Brazil	Seychelles	
Afghanistan	Montenegro		Paraguay	Tanzania	
Bangladesh	Poland	**North America, High Income**		Uganda	
Bhutan	Romania	Canada	**North Africa / Middle East**	Zambia	
India	Serbia	United States	United Arab Emirates		
Nepal	Slovakia		Bahrain	**Sub-Sah. Africa, Southern**	
Pakistan	Slovenia	**Caribbean**	Algeria	Botswana	
		Antigua and Barbuda	Egypt	Lesotho	
Asia, Southeast	**Europe, Eastern**	The Bahamas	Iran	Namibia	
Indonesia	Belarus	Belize	Iraq	Swaziland	
Cambodia	Estonia	Barbados	Jordan	South Africa	
Laos	Lithuania	Cuba	Kuwait	Zimbabwe	
Sri Lanka	Latvia	Dominica	Lebanon		
Maldives	Moldova	Dominican Republic	Libya		
Myanmar	Russia	Grenada	Morocco		
Malaysia	Ukraine	Guyana	Oman		
Philippines		Haiti	Palestine		
Thailand		Jamaica	Qatar		
Timor-Leste		Saint Lucia	Saudi Arabia		
Vietnam		Suriname	Syria		
		Trinidad and Tobago	Tunisia		
		St. Vincent/Grenadines	Turkey		
			Yemen		

Sources: Burnett and others (2013).

Table A2. Damages from Local Air Pollution, All Countries, $/ton of Emissions, 2010

Country	sulfur dioxide			nitrogen oxides			(direct) fine particulates		
	$ per ton			$ per ton			$ per ton		
	coal	natural gas	ground level	coal	natural gas	ground level	coal	natural gas	ground level
North America									
Canada	3,908	8,994	13,284	2,757	5,917	2,712	4,887	11,480	349,161
Mexico	2,240	3,599	7,704	1,797	2,179	1,575	2,700	4,416	203,680
United States	17,132	18,978	17,005	12,472	12,092	3,468	21,402	23,294	445,484
Central & South America									
Argentina	8,328	4,928	7,553	3,475	2,375	1,532	10,420	6,167	193,736
Barbados	#na	26,387	#na	#na	12,166	#na	#na	33,014	#na
Bolivia	#na	343	355	#na	237	73	#na	410	9,624
Brazil	2,004	4,293	5,013	1,492	2,401	1,021	2,626	5,258	130,726
Chile	1,409	1,989	7,057	1,029	1,185	1,434	1,730	2,482	182,276
Colombia	1,867	1,648	6,180	1,162	1,084	1,265	2,307	2,047	164,342
Costa Rica	#na	#na	2,316	#na	#na	477	#na	#na	63,036
Cuba	#na	4,447	3,627	#na	3,033	743	#na	5,293	96,420
Dominican Republic	3,007	#na	3,475	1,694	#na	714	3,713	#na	93,597
Ecuador	#na	748	721	#na	454	148	#na	915	19,505
El Salvador	#na	#na	696	#na	#na	143	#na	#na	18,949
Guatemala	882	#na	417	501	#na	87	1,076	#na	11,721
Honduras	#na	#na	936	#na	#na	194	#na	#na	26,190
Jamaica	#na	#na	1,617	#na	#na	336	#na	#na	45,210
Nicaragua	#na	560	265	#na	374	55	#na	665	7,338
Panama	1,560	#na	1,581	1,079	#na	324	2,031	#na	42,080
Paraguay	#na	#na	825	#na	#na	170	#na	#na	22,594
Peru	359	1,415	2,435	290	593	498	447	1,767	64,499
Saint Vincent/Grenadin	#na	#na	#na	#na	#na	#na	#na	#na	#na
Suriname	#na	#na	649	#na	#na	133	#na	#na	17,461
Trinidad and Tobago	#na	2,883	#na	#na	1,977	#na	#na	3,553	#na
Uruguay	#na	3,151	2,184	#na	2,164	443	#na	3,773	55,994
Venezuela	#na	2,027	4,000	#na	1,203	811	#na	2,575	102,381
Europe									
Albania	#na	#na	4,927	#na	#na	1,023	#na	#na	137,666
Austria	41,004	41,889	12,951	31,812	31,666	2,664	51,736	53,150	350,052
Belgium	53,017	51,863	10,883	34,613	34,243	2,201	64,698	63,189	276,234
Bosnia and Herzegovina	#na	#na	5,556	#na	#na	1,157	#na	#na	156,869
Bulgaria	23,980	#na	7,536	19,472	#na	1,545	28,991	#na	201,479
Croatia	35,046	35,676	10,533	28,197	27,410	2,179	44,610	45,720	290,953
Cyprus	#na	#na	2,232	#na	#na	458	#na	#na	59,950
Czech Republic	56,034	55,308	9,670	40,836	41,184	1,982	69,818	68,676	258,025
Denmark	26,136	26,025	6,276	20,048	19,993	1,277	34,589	34,627	162,816
Finland	14,814	16,035	10,786	12,152	12,711	2,198	17,739	19,320	281,719
France	33,555	37,779	15,908	24,511	27,670	3,239	41,725	46,003	414,075
Germany	53,192	56,125	20,082	35,624	36,603	4,115	65,936	69,514	535,454

Table A2. Damages from Local Air Pollution, All Countries, $/ton of Emissions, 2010 (continued)

Country	sulfur dioxide			nitrogen oxides			(direct) fine particulates		
	$ per ton			$ per ton			$ per ton		
	coal	natural gas	ground level	coal	natural gas	ground level	coal	natural gas	ground level
Greece	20,699	20,734	8,028	16,843	16,213	1,657	25,562	25,570	219,970
Hungary	41,057	40,925	11,070	30,712	30,608	2,275	51,744	51,840	298,250
Iceland	#na	#na	3,855	#na	#na	781	#na	#na	98,626
Ireland	12,897	18,828	4,991	10,468	14,585	1,030	16,217	22,833	136,535
Italy	26,627	31,596	13,346	20,905	22,958	2,744	33,654	40,278	360,129
Luxembourg	#na	86,775	#na	#na	65,283	#na	#na	105,443	#na
Macedonia	16,736	17,560	5,832	13,541	14,096	1,206	20,686	21,656	160,515
Malta	#na	#na	#na	#na	#na	#na	#na	#na	#na
Montenegro	21,031	#na	4,205	17,103	#na	867	26,405	#na	114,743
Netherlands	53,065	50,535	13,357	35,421	34,581	2,723	65,304	62,168	349,477
Norway	#na	17,667	35,210	#na	14,920	7,194	#na	23,495	928,330
Poland	38,887	35,828	9,468	28,429	27,749	1,955	49,082	45,043	259,582
Portugal	12,221	12,533	6,383	9,265	9,355	1,318	14,755	15,177	175,156
Romania	26,813	27,895	7,995	21,377	21,041	1,659	33,293	34,439	223,169
Serbia	24,142	24,194	6,728	18,319	18,274	1,393	30,381	30,841	186,463
Slovakia	42,444	46,050	7,275	32,616	33,770	1,508	53,469	58,463	202,158
Slovenia	52,466	52,388	10,936	39,744	39,419	2,273	67,044	66,807	307,217
Spain	16,871	19,270	19,055	13,364	14,498	3,897	20,852	23,980	504,326
Sweden	17,058	19,702	16,370	13,005	15,757	3,333	21,281	25,956	426,238
Switzerland	#na	46,015	11,919	#na	34,809	2,443	#na	57,827	317,909
Turkey	7,341	9,611	5,264	5,746	6,507	1,081	9,146	11,858	141,362
United Kingdom	36,577	40,069	12,325	22,857	27,378	2,518	45,415	48,658	324,687
Eurasia									
Armenia	#na	7,411	3,020	#na	5,584	622	#na	9,156	82,228
Azerbaijan	#na	8,462	3,498	#na	6,417	726	#na	10,520	97,516
Belarus	#na	26,576	15,038	#na	21,381	3,080	#na	33,671	400,285
Estonia	#na	28,605	8,435	#na	22,914	1,733	#na	34,958	226,999
Georgia	#na	6,049	2,762	#na	4,613	573	#na	7,525	77,102
Kazakhstan	2,668	6,107	3,104	2,225	5,306	644	3,184	7,588	86,461
Kyrgyzstan	1,934	#na	654	1,518	#na	137	2,328	#na	19,010
Latvia	23,252	28,935	10,572	19,784	23,459	2,174	29,743	36,413	285,607
Lithuania	#na	34,985	13,522	#na	27,769	2,782	#na	44,700	365,862
Russia	17,562	22,105	32,383	12,508	14,317	6,637	21,525	27,714	863,732
Tajikistan	#na	#na	418	#na	#na	88	#na	#na	12,393
Turkmenistan	#na	5,775	1,632	#na	4,770	340	#na	6,978	46,015
Ukraine	17,851	16,728	6,377	13,593	12,690	1,311	22,086	20,497	171,913
Uzbekistan	3,451	2,797	659	2,552	2,162	138	4,175	3,359	19,116
Middle East									
Bahrain	#na	7,161	2,451	#na	5,303	498	#na	8,563	63,360
Iran	#na	5,066	3,956	#na	3,694	813	#na	6,171	106,587
Iraq	#na	1,171	857	#na	877	176	#na	1,482	23,197

Table A2. Damages from Local Air Pollution, All Countries, \$/ton of Emissions, 2010 (continued)

Country	sulfur dioxide			nitrogen oxides			(direct) fine particulates		
	\$ per ton			\$ per ton			\$ per ton		
	coal	natural gas	ground level	coal	natural gas	ground level	coal	natural gas	ground level
Israel	24,369	24,926	11,652	15,717	15,759	2,364	29,482	30,226	299,185
Jordan	#na	2,429	1,113	#na	1,643	227	#na	2,975	29,144
Kuwait	#na	#na	9,771	#na	#na	1,976	#na	#na	247,625
Lebanon	#na	7,253	2,080	#na	4,753	423	#na	9,202	53,922
Oman	#na	7,088	3,095	#na	6,022	634	#na	8,028	82,631
Qatar	#na	16,731	7,246	#na	13,738	1,465	#na	19,600	183,468
Saudi Arabia	#na	4,895	4,651	#na	3,641	949	#na	6,018	121,849
Syria	#na	2,829	1,404	#na	1,864	291	#na	3,612	38,929
United Arab Emirates	#na	6,431	3,019	#na	4,845	615	#na	7,578	78,782
Africa									
Algeria	#na	3,442	1,834	#na	2,381	376	#na	4,242	49,099
Angola	#na	465	1,320	#na	312	273	#na	567	36,392
Benin	#na	#na	75	#na	#na	16	#na	#na	2,122
Botswana	1,007	656	680	798	556	140	1,238	879	18,629
Burkina Faso	#na	#na	68	#na	#na	14	#na	#na	2,027
Burundi	#na	#na	16	#na	#na	3	#na	#na	494
Cameroon	#na	312	419	#na	254	87	#na	391	11,732
Cape Verde	#na	#na	#na	#na	#na	#na	#na	#na	#na
Central African Republic	#na	#na	130	#na	#na	27	#na	#na	3,745
Comoros	#na	#na	#na	#na	#na	#na	#na	#na	#na
Congo (Brazzaville)	#na	66	87	#na	52	18	#na	77	2,378
Cote dIvoire (IvoryCoast	#na	312	289	#na	197	60	#na	391	8,096
Egypt	#na	5,288	1,912	#na	2,764	399	#na	6,460	54,506
Ethiopia	#na	#na	70	#na	#na	15	#na	#na	2,114
Gambia, The	#na	#na	73	#na	#na	15	#na	#na	2,017
Ghana	#na	270	117	#na	197	24	#na	344	3,273
Guinea-Bissau	#na	#na	81	#na	#na	17	#na	#na	2,315
Kenya	#na	234	90	#na	173	19	#na	289	2,683
Liberia	#na	#na	171	#na	#na	35	#na	#na	4,814
Libya	#na	2,470	1,296	#na	1,942	265	#na	2,952	34,272
Madagascar	#na	#na	81	#na	#na	17	#na	#na	2,371
Malawi	#na	148	38	#na	91	8	#na	#na	1,164
Mali	#na	#na	56	#na	#na	12	#N/A	#na	1,621
Mauritius	438	#na	#na	206	#na	#na	545	#na	#na
Morocco	1,540	1,762	1,563	930	1,085	324	1,901	2,167	43,251
Mozambique	#na	#na	44	#na	#na	9	#na	#na	1,303
Namibia	202	#na	281	167	#na	59	233	#na	8,111
Niger	#na	#na	28	#na	#na	6	#na	#na	844
Nigeria	#na	714	535	#na	425	111	#na	887	15,051
Rwanda	#na	#na	51	#na	#na	11	#na	#na	1,545

Table A2. Damages from Local Air Pollution, All Countries, $/ton of Emissions, 2010 (concluded)

Country	sulfur dioxide			nitrogen oxides			(direct) fine particulates		
	$ per ton			$ per ton			$ per ton		
	coal	natural gas	ground level	coal	natural gas	ground level	coal	natural gas	ground level
Sao Tome and Principe	#na	#na	#na	#na	#na	#na	#na	#na	#na
Senegal	134	#na	112	71	#na	23	164	#na	3,188
Seychelles	#na	#na	#na	#na	#na	#na	#na	#na	#na
Sierra Leone	#na	#na	68	#na	#na	14	#na	#na	1,959
South Africa	1,602	2,550	1,690	1,031	1,219	349	1,905	3,154	46,284
Sudan and South Suda	#na	207	100	#na	171	21	#na	239	2,934
Swaziland	#na	#na	#na	#na	#na	#na	#na	#na	#na
Tanzania	#na	175	116	#na	115	24	#na	221	3,429
Togo	#na	272	44	#na	187	9	#na	345	1,261
Tunisia	#na	3,925	1,834	#na	2,952	378	#na	4,758	49,730
Uganda	#na	#na	44	#na	#na	9	#na	#na	1,340
Zambia	#na	#na	84	#na	#na	18	#na	#na	2,430
Zimbabwe	51	#na	50	41	#na	10	65	#na	1,435
Asia & Oceania									
Afghanistan	#na	866	186	#na	642	39	#na	1,077	5,545
Australia	2,098	2,136	9,220	1,129	900	1,873	2,632	2,698	238,099
Bangladesh	6,057	6,131	1,757	4,082	3,757	371	7,181	7,430	51,932
Bhutan	#na	#na	#na	#na	#na	#na	#na	#na	#na
Brunei	#na	10,797	#na	#na	9,274	#na	#na	12,225	#na
Cambodia	#na	#na	486	#na	#na	103	#na	#na	14,655
China	22,045	25,577	4,422	15,530	16,605	920	27,609	32,238	124,441
Fiji	#na	#na	#na	#na	#na	#na	#na	#na	#na
Hong Kong	82,580	72,288	#na	53,207	49,085	#na	103,759	91,246	#na
India	7,833	6,837	1,093	5,683	4,762	230	9,773	8,549	32,075
Indonesia	4,617	5,627	2,159	2,492	2,699	449	5,636	6,936	60,669
Japan	36,786	47,176	31,548	24,230	24,772	6,405	44,381	57,309	812,178
Kiribati	#na	#na	#na	#na	#na	#na	#na	#na	#na
Korea, South	35,228	34,688	20,862	25,439	25,375	4,253	46,054	45,507	545,623
Malaysia	6,525	6,104	4,028	4,360	4,273	826	7,891	7,406	107,824
Maldives	#na	#na	#na	#na	#na	#na	#na	#na	#na
Mongolia	3,138	#na	2,253	2,736	#na	463	3,498	#na	60,870
New Zealand	1,568	1,296	2,508	479	396	510	1,981	1,637	65,153
Pakistan	2,254	2,902	630	1,698	2,075	132	2,942	3,663	18,290
Papua New Guinea	#na	#na	91	#na	#na	19	#na	#na	2,777
Philippines	3,372	4,426	1,393	1,969	2,246	290	4,053	5,377	39,237
Samoa	#na	#na	#na	#na	#na	#na	#na	#na	#na
Singapore	#na	21,698	42,652	#na	13,439	8,617	#na	27,223	1,077,044
Sri Lanka	4,262	#na	410	3,258	#na	87	5,068	#na	12,500
Taiwan	46,892	49,692	#na	35,615	36,445	#na	59,253	63,012	#na
Thailand	9,036	9,067	2,013	6,941	6,087	423	10,886	11,105	58,683
Vietnam	5,823	3,274	1,416	4,060	2,028	298	7,243	3,989	41,622

Sources: See Chapter 3.
Notes: The table shows estimates of the local pollution (health) damages per ton from each of three pollutants, according to whether emissions are released from coal combustion, natural gas combustion at power plants, or natural gas/motor fuel consumption at ground level. The color coding for #N/A is as follows: red and black indicate respectively cases where data is not available and the fuel is not used.

Appendix for Chapter 3 of the Supplement

This appendix consists of seven subsections. These discuss in turn technical details on the measurement of congestion costs; cities included in the city-level database to establish relationships between travel delays and transportation indicators; regression results using this data; the adjustment of congestion costs to account for the mix of vehicles on the road; two estimates of nationwide congestion costs that are based on country-level data; the quantification of non-fatality accident costs; and various miscellaneous data needs and procedures for assessing corrective fuel taxes.

(i) Measuring Congestion Costs: Some Technicalities

The total hourly costs (*TC*) of congestion to passengers in vehicles driving along a one-km lane segment of a highway can be expressed:

(A3.1) $$TC = V \cdot (T(V) - T^f) \cdot o \cdot VOT$$

Here *V* denotes traffic volume or flow—the number of cars that pass along the km-long stretch per hour (the implications of other vehicles on the road is discussed below). T^f is travel time per km when traffic is free-flowing and *T* (which exceeds T^f) is the actual travel time, an increasing function of the traffic volume (speeds fall with less road space between vehicles). *o* is vehicle occupancy (average number of passengers per vehicle). The total travel delay from congestion for all passengers is therefore $V \cdot (T - T^f) \cdot o$, where $T - T^f$ is the average delay per vehicle km. Multiplying total travel delay by the value of travel time (VOT) expresses delays as a monetary cost.

Dividing *TC* by traffic volume gives the average cost of congestion (*AC*) per vehicle km:

(A3.2) $$AC = (T - T^f) \cdot o \cdot VOT$$

This is the cost borne by individual motorists which (on average) they should take into account when deciding how much to drive.

Differentiating *TC* with respect to *V* gives the added congestion cost to all road users from an extra vehicle km:

(A3.3) $$\frac{dTC}{dV} = AC + \frac{dT}{dV} \cdot V \cdot o \cdot VOT$$

This includes the average cost (taken into account by the driver), as just described. It also includes the cost to occupants of other vehicles (which is not). The latter is the delay to other vehicles, $(dT/dV) \cdot V$, times the average number of people in other vehicles, times the VOT to express costs in money units.

Suppose (as discussed in Chapter 3 of the supplement) that travel delay can be approximated by a power function of traffic volume, that is:

(A3.4) $$T - T^f = \alpha V^\beta$$

where α and β are constants. α reflects factors like road capacity, while β reflects the rate at which additional traffic slows travel speeds. Differentiating this expression by V gives $dT/dV = \alpha\beta V^{\beta-1}$. And using (A4.4) gives:

(A3.5) $$\frac{dT}{dV} \cdot V = (T - T^f) \cdot \beta$$

That is, the delay to other vehicles is simply the product of average delay and the scalar β. As discussed in Chapter 3 of the supplement, empirical studies suggest a value for β of between about 2.5 and 5 for congested roads.

If speed data is available, average delay can be estimated using:

(A3.6) $$T = \frac{1}{S}\ ,\ T^f = \frac{1}{S^f}$$

where S and S^f are the actual and the free-flow travel speeds (km/hour).

(ii) Cities Represented in City-Level Database

Cities covered in the city-level database—which is used to obtain statistical relationships between travel delays and various transportation indicators—are listed in Table A3 (less ten cities that were dropped from the Millennium Cities Database due to incomplete data).

Table A3. Cities in the City-Level Database (Used In Extrapolating Congestion Costs)

Western Europe	**Eastern Europe**	**Middle East**	**Oceania**
Graz	Prague	Tel Aviv	Brisbane
Vienna	Budapest	Tehran	Melbourne
Brussels	Cracow	Riyadh	Perth
Copenhagen	Warsaw		Sydney
Helsinki	Moscow	**Africa**	Wellington
Lille		Cario	
Lyon	**North America**	Abijan	
Marseille	Calgary	Dakar	
Nantes	Montreal	Cape Town	
Paris	Ottwa	Johannesburg	
Berlin	Toronto	Tunis	
Frankfurt	Vancouver	Harare	
Hamburg	Atlanta		
Dusseldorf	Chicago	**Asian Affluent**	
Munich	Denver	Osaka	
Ruhr	Houston	Sapporo	
Stuttgart	Los Angeles	Tokyo	
Athens	New York	Hong Kong	
Milan	Phoenix	Singapore	
Bilogna	San Diego		
Rome	San Francisco	**Other Asian**	
Amsterdam	Washington	Mumbai	
Oslo		Chennai	
Barcelona	**Latin America**	Jakarta	
Madrid	Curitiba	Kula Lumpur	
Stockholm	Rio de Janeiro	Beijing	
Berne	San Paulo	Shanghai	
Geneva	Bogota	Guangzhou	
Zurich	Mexico City	Manila	
Glasgow		Seoul	
London		Taipei	
Manchester		Bangkok	
Newcastle		Ho Chi Minh City	

Sources: See Chapter 3 of the supplement.
Notes: Excludes 10 cities from the original database that were dropped due to insufficient data on transportation indicators for those cities.

(iii) Results from Statistical Methods Used to Relate Travel Delay to Travel Indicators

As discussed in Chapter 3 of the supplement, statistical regressions were used to estimate the contribution of various factors to explaining travel delays across the 90 cities in the data base. To obtain the best statistical fit (i.e., to reduce the 'noise' from outlying observations or extreme values), average delay and the four explanatory indicators are expressed in the natural logarithm form in the regression and second powers of these variables are included (both of these are standard statistical procedures). The regression results are presented in Table A4.

Table A4. Regression Results for City-Level Average Delay

Variables	Log average delay
log GDP per capita	0.061
	-0.409
log km driven per car	-5.308***
	-1.776
log road length per car	-0.796
	-1.08
log cars per capita	-1.038***
	-0.242
log GDP per capita^2	-0.0106
	-0.044
log km driven per car^2	-0.515**
	-0.196
log road length per car^2	-0.0414
	-0.11
log cars per capita^2	-0.100*
	-0.051
Constant	-21.23***
	-5.04
Observations	90
R-squared	0.659

Sources: See Chapter 3 of the supplement.
Notes: ***is significant at 1 the percent level, ** is significant at the 5 percent level and * is significant at the 10 percent level.

Interpreting these coefficients is less of a concern than the statistical fit (which is reasonably good) because the coefficients are used for prediction rather than for establishing causal relationships. In fact, the explanatory variables such as road length per car and cars per capita are likely to be simultaneously determined with traffic conditions such as average delay (the dependent variable) which confounds the interpretation of the coefficients.[67]

As noted in the text, kms driven per car is not available at the country level for 81 of countries. To fill in this gap, countries are grouped by region (Europe, Oceania, Africa, etc.) and statistical regressions are used to estimate a relationship for each regional grouping between kms travelled per car (for countries where this data is available) and four explanatory variables (available for all countries): per capita income, urban population density, vehicle ownership and road density (using data from IRF 2009 and World Bank 2013). Using this relationship, and the explanatory variables, kms driven per vehicle are then inferred for countries where direct data is missing.

The natural logarithm of km driven per car and the four explanatory indicators (of the 69 countries that have complete data) was taken, as well as including the second and third power of the log explanatory variables to add more flexibility. The regression results are shown in Table A5 though again, since the equation is used for prediction, the interpretation of the estimated coefficients is not especially of concern.

[67] For example, the negative sign for km per car suggests, perhaps, that extra traffic creates pressure or incentives for road investment (e.g., Duranton and Turner, 2011) or that bad traffic conditions discourage driving.

Table A5. Regression Results for kms Driven per Car

Variables	Log km driven per car
log GDP per capita	-5.545*
	-3.058
log cars per capita	2.596**
	-1.127
log road length per car	3.359
	-2.091
log road density	0.113
	-0.16
log GDP per capita^2	-1.373**
	-0.66
log cars per capita^2	1.470***
	-0.487
log road length per car^2	0.992
	-0.738
log road density^2	-0.145
	-0.093
log GDP per capita^3	-0.093**
	-0.044
log cars per capita^3	0.197***
	-0.063
log road length per car^3	0.104
	-0.083
log road density^3	-0.016
	-0.033
Constant	-4.988
	-4.568
Observations	69
R-squared	0.642

Sources: See Chapter 3 of the supplement.
Notes: ***is significant at 1 the percent level, ** is significant at the 5 percent level and * is significant at the 10 percent level. The above regression also includes geographical regional dummy variables.

(iv) Accounting for Delays to All Vehicle Occupants

This subsection presents illustrative calculations to show how congestion cost estimates change when the mix of cars, buses, trucks, and two-wheelers is taken into account (the formulas in subsection (i) above assume cars are the only vehicle).

Following from equation (A3.1) above, the total costs of travel delays to all road users, when different vehicle types are distinguished, is given by:

(A3.8) $$TC = (T - T^f) \cdot \sum_i V_i \cdot o_i \cdot VOT_i$$

Subscript i is used to denote a particular type of vehicle: i = car, bus, truck, or two-wheeler. To keep things simple, congestion is assumed to increase delay for all vehicles by the same absolute amount.

Differentiating (A4.8) with respect to V_{car}, and using the definition of AC from (A4.2), gives:

(A3.9) $$\frac{dTC}{dV_{car}} = AC + \frac{dT}{dV_{car}} \sum_i V_i \cdot o_i \cdot VOT_i$$

Comparing (A4.3) and (A4.9), the ratio of the cost imposed on other vehicle occupants, when the mix of vehicles on the road is taken into account, rather than assuming all vehicles are cars, is:

(A3.10) $$\frac{\sum_i (V_i/V) \cdot o_i \cdot VOT_i}{o_{car} \cdot VOT_{car}}$$

where V_i/V is the share of vehicle i in total kms driven by all vehicles.

For the calculations below, the occupancy of trucks and two-wheelers is taken to be one, while (based approximately on Parry and Small 2009 for US and UK cities) that for buses is taken to be 20. The VOT for two-wheelers and bus riders is taken to be the same as for car occupants. For (freight) travel by trucks, the VOT should include the employer wage (the market wage plus employer payroll taxes), as this reflects the per hour costs of labor time lost from congestion. Given the VOT for car travel is 60 percent of the market wage, this implies VOT_{truck}/VOT_{car} = 1.67.

The last column of Table A6 shows (based on A4.10) congestion costs with different scenarios for the vehicle fleet mix relative to congestion costs when cars are the only vehicles on the road.

If other vehicles consist only of trucks or two-wheelers, there is relatively little difference to the results: in Table A6, congestion costs are increased 1 percent when trucks account for 20 percent of the fleet and they are reduced 7 percent when two-wheelers account for 20 percent of the fleet (with, in either case, cars accounting for the other 80 percent of vehicles). However, when buses account for 10 percent of the fleet (i.e., every tenth vehicle on the road is a bus) congestion costs more than double, and when they account for 20 percent costs more than triple. This is because a car driver imposes considerably higher costs on others when the average number of vehicle occupants is higher, due to a significant share of (high-occupancy) buses on the roads. Congestion costs (and motor fuel taxes) may therefore be substantially understated in countries where buses account for a substantial share of vehicle traffic in urban centers.

Table A6. Ratio of Congestion Cost with Multiple Vehicles Relative to Costs when Cars are the only Vehicle

share of vehicle km by mode				congestion cost with multiple vehicles to cost with cars only
car	bus	truck	two wheeler	
1	0	0	0	1
0.8	0	0.2	0	1.01
0.8	0	0	0.2	0.93
0.8	0.2	0	0	3.30
0.5	0.1	0.1	0.3	2.04
0.9	0.1	0	0	2.15

Sources: See above.

(v) Assessment of Congestion Costs from Country-Level Data: United States and United Kingdom

Here the supplementary estimation of delays at the country level for the United States and United Kingdom, mentioned in Chapter 3 of the supplement, is explained. Estimated delays are for year 2008 (likely a close approximation to 2010) and costs are expressed in year US $2010.[68]

[68] Comparable country-level data for other cases is hard to come by. For Australia, an average delay estimate was computed using state-level delay data for 2010 from Association of Australian and New Zealand Road Transport and Traffic Authorities (Austroads, see http://algin.net/austroads/site/Index.asp?id=5) and vehicle km data from Australian Bureau of Statistics (2010). The result, which was 80 percent higher than the UK estimate below, seemed on the high side however (and is therefore not reported above). Some of the difference is due to different assumptions about free flow speeds but, most likely, the bulk of the difference reflects different data collection procedures across the respective transportation authorities (e.g., the UK data is more disaggregated and potentially more accurate), rather than real factors.

United States

For the United States, the Texas Transportation Institute (TTI) compiles high-quality data on travel delays for 449 urban centers categorized by population size into very large, large, medium, and small cities (Schrank and Lomax, 2011).

For the 101 largest cities, speed data is collected remotely by a private company for different times of the day for each link within the urban road network. For the other 348 (smaller) urban centers (which account for 15 percent of nationwide travel delays), speed is inferred from estimated speed/traffic volume relationships. Schrank and Lomax (2011) use traffic volume data from the Highway Performance Monitoring System, an inventory maintained by the Federal Highway Administration for all roadway segments in the United States.

The TTI for year 2008 was used to infer the nationwide congestion delay on others.[69] For each urban region in the TTI sample, total annual hours of delay to passengers in cars is divided by total annual vehicle km driven by cars to give the average hourly delay per car km. Delays at the regional level are weighted by the share of car kms in nationwide km and then aggregated to obtain a nationwide average measure of delay.

United Kingdom

For the United Kingdom, travel data for 2008 was used from UK Department for Transport (DFT), which compiles official statistics of the transport system in Great Britain. Since DFT does not provide annual hours of travel delays at the city level, travel delays were generated by comparing average vehicle speed during peak times with the free flow speed, both of which can be obtained from the DFT statistics.[70]

For each of the UK local authorities, the average travel time per km was calculated, and the free flow average travel time per km, using the average travel speed during morning peak (7am to 10am) and the free flow speed.

Annual car kms within each authority was then multiplied by the share of vehicle car km occurring during the morning peak period. The total annual hours of delay is then obtained by multiplying the extra travel time per km during morning peak time with twice the vehicle km driven during morning peak to account for the evening peak (4pm to 7 pm) which is assumed to carry the same traffic congestion.[71]

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[69] The data used is from http://mobility.tamu.edu/files/2012/01/complete-data.xls.

[70] The data used is from http://www.dft.gov.uk/statistics/tables, specifically data sets CGN0201, SPE0104, TRA8901, and TRA0307.

[71] According to DFT traffic distribution data (TRA0307), the shares of morning and evening peak kms in total kms driven are 0.21 and 0.22, respectively, which are very close.

Next, the total annual hours of delay is divided by total annual vehicle km driven by cars for each authority to get the average hourly delay per vehicle km, which is converted into passenger delays assuming a vehicle occupancy of 1.6. Average delay at the nationwide level is a weighted average of that at the authority level, with weights equal to the shares in nationwide car kms.

Delays to others per car km are about twice as high for the United Kingdom compared with the United States, which seems roughly plausible, given that a much greater share of nationwide driving occurs under congested conditions in the United Kingdom.

(vi) Estimating the Ratio of Other Accident Costs to Fatality Costs: Country Case Studies

As mentioned in Chapter 4, external accident costs for different countries are obtained by scaling up estimates of external fatality costs by an assumption about the ratio of other costs to external fatality costs. This ratio—which pivots off several country case studies—is attained as follows.

First, using data compiled by Herrnstadt and others (2013) for Finland, Sweden, United Kingdom, and United States and by Parry and Strand (2012) for Chile, comprehensive estimates of external accident costs were made for these five countries for 2010 or thereabouts. In these calculations, external fatalities are monetized using mortality values discussed above in Chapter 3. Other costs are valued using a combination of local data on the average (personal, medical, and property damage) cost associated with different injury (and no injury) classes and (in some cases) extrapolations of these costs from US data.[72]
85 percent of medical costs are taken to be external (borne by third parties) for all (fatal and non-fatal, internal and external) injuries and 50 percent of property damage costs (for all accidents) are external.

From these studies, five point estimates for the ratio of other external costs (medical, property damage, and non-fatal injury costs) to external fatality costs were obtained. This ratio tends to decline as the relative importance of pedestrian and other external deaths in total road deaths rises (the numerator in the ratio falls and the denominator rises). This ratio is 2.9 in the United States (where 23 percent of deaths are external) and only 0.16 in Chile (where 54 percent of deaths are external). A power function which best fits these five data points relating this cost ratio to the share of external fatalities in total fatalities was

[72] Detailed documentation of data sources and estimation procedures are provided in the above references. The breakdown of fatalities by driver, passenger of drivers, other vehicle occupant, pedestrian, and cyclist is available from the data sources and this breakdown is taken to be the same for the non-fatal injuries. Herrnstadt and others (2013) focused only on alcohol-related accidents—basically, their spreadsheet data was modified to include data for all traffic accidents.

estimated.[73] This relation was then used to extrapolate the other external cost/fatality external cost ratio for other countries depending on their share of external fatalities in total fatalities.

(vii) Miscellaneous Data and Procedures for Calculating Corrective Taxes

Here several remaining pieces of data and assumptions needed to implement the corrective fuel tax formula in Chapter 1 of the main report are discussed. These issues deal with the use of diesel fuel by both cars and trucks, the breakdown of fuel price responses, and conversion of road damage, accident, local pollution, and congestion costs into corresponding components of corrective fuel taxes. In the latter regard, fuel efficiency is needed to convert any congestion, accident, local pollution, or road damage cost expressed per vehicle km driven into a cost per liter of fuel. However, given the difficulty of accurately measuring fuel efficiency for most countries (see below), these costs are directly expressed per liter insofar as possible, to avoid the need for this data.

Diesel use by different vehicle types. External costs for cars are used to infer corrective taxes on gasoline. However diesel fuel is used by both cars and trucks and, given the practical difficulty of differentiating the diesel tax according to vehicle use, a weighted average of external costs for cars and trucks should be used in the corrective diesel tax formula, depending on their respective shares in diesel fuel consumption. The breakout of diesel fuel use by cars versus trucks is available for a limited number of countries and for other countries was taken from regional average figures.[74]

Breakdown of fuel price responses. An important piece of data is the fraction of the fuel demand response that comes from reduced driving (as opposed to the remaining fraction that comes from fuel efficiency improvements). For cars, this is taken to be 0.5 for all countries.[75] For diesel fuel used by trucks (where the high power requirements necessary to move freight limit opportunities for improving fuel efficiency through, for example, reducing vehicle size and weight) this portion is taken to be 0.6 (from Parry 2008).

Road damage. For road damage the above estimation procedures outlined in Chapter 3 of the supplement yield total external costs. These are divided by total diesel fuel consumption for

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[73] Specifically, the cost ratio is predicted by the equation $0.049x^{-2.56}$, where x is the share of external fatalities in total fatalities.

[74] The data source is from ICCT (2010). For example, cars account for about 11 percent of diesel fuel use at the global level, and 32 percent in the European Union.

[75] See Small and Van Dender (2006) and the review of other studies in Parry and Small (2005). In practice this portion will vary across countries, for example, it might be higher in countries with readily available alternatives to car use (which increases the responsiveness of driving to fuel prices) and in countries with binding fuel efficiency regulations (which reduces the responsiveness of fuel efficiency to fuel prices). But there is no international data on which country-specific assumptions can be based.

trucks to obtain costs per liter, which are then multiplied by 0.6 to account for the portion of the fuel response that comes from reduced km driven.

Accidents. The estimation procedures also yield total external costs for traffic accidents however expressing them per liter of fuel is a little more involved. This is because external costs per vehicle km are taken to be the same for cars and trucks, implying external costs per liter of fuel will be larger for cars than for trucks given that cars travel further on a liter of fuel, and larger for diesel fuel cars than gasoline cars (as the former have higher fuel efficiency). Truck fuel efficiency is taken to be one-third of that for gasoline cars (Parry 2008), and in turn diesel cars are assumed to be 20 percent more fuel efficient than their gasoline counterparts.

External accident costs per liter of gasoline can then be obtained by dividing total accident costs (for all vehicles) by a weighted sum of fuel use by gasoline cars, diesel cars, and trucks (fuel use data is discussed in the appendix for Chapter 2 of the supplement) where the weights are fuel efficiency of other vehicles relative to that for gasoline cars (1.2 and 0.33 respectively for diesel cars and trucks). In turn costs per liter for diesel cars and trucks are the external costs per liter for gasoline cars multiplied by the same weights. In applying the costs to the corrective fuel tax formula they are again multiplied by the portion (0.5 or 0.6) of the fuel price response that comes from reduced driving as opposed to fuel efficiency improvements.

Local pollution. As regards local pollution damages, these are estimated on a per liter basis. The scaling factor here however depends on how emissions are regulated (if at all). In countries like the United States where emissions are regulated on a per-km (or per-mile) basis and maintained (approximately) throughout the vehicle life, roughly speaking emissions vary only with km driven not fuel efficiency and therefore need to be multiplied by the driving fraction of the fuel price response.[76] In countries with little effective regulation of emissions no scaling factor should be applied (as emissions are proportional to fuel use). The calculations here apply a scaling factor of 0.5 for emissions in developed countries and no scaling factor for developing countries. More refined assumptions would not have much effect on the corrective fuel tax estimates given that local pollution costs are modest relative to congestion and accident costs (see Chapter 2 of the main report).

Congestion. As for congestion costs, these (necessarily) are estimated on a per-km basis and therefore need to be multiplied by fuel efficiency (see below) to express them in per liter terms (after scaling by the driving fraction of the fuel price response).

[76] In some cases, emissions standards are defined with respect to engine capacity (e.g., EU countries as well as other countries adopting EU standards). Here some fuel efficiency improvements (e.g., reducing vehicle weight) will affect emissions but others (e.g., more efficient engines) will not.

One further complication here is that driving on congested roads (which tends to be dominated by people commuting to work) is generally less sensitive to fuel prices than driving on uncongested roads. This reduces the congestion benefits from higher fuel taxes relative to the case where driving on all congested and uncongested roads alike falls in the same proportion to higher fuel taxes. Based on evidence of the relative price responsiveness of congested and non-congested driving, Parry and Small (2005) recommend scaling back congestion costs by a third in computing corrective fuel taxes—the same procedure is followed here.

Fuel efficiency. As regards fuel efficiency (of vehicles in use on the road), this could be obtained by dividing data on vehicle kms driven by fuel use. However, because the reliability of the km data varies across countries (it is generally less accurate for developing countries) instead fuel efficiency is based on a plausible assumption for different regions, and applied to all countries in the region. For example, based on estimates in Parry and Small (2005) for the United States and United Kingdom, fuel efficiency for gasoline vehicles is taken to be 10.5 km/liter (25 miles/gallon) in North America and 14.5 km/liter (35 miles/gallon) for higher income European countries and Japan.[77] Fuel efficiency for diesel cars and trucks is then inferred using the above ratios. Other assumptions would moderately affect the contribution of congestion costs to corrective taxes.[78]

Finally, to simplify the computation of corrective fuel taxes it is assumed that fuel efficiency in each country remains fixed at its current level, rather than increasing in response to higher fuel prices. This again leads to some understatement of the corrective fuel tax, as higher fuel efficiency implies greater reductions in driving, and hence congestion and so on, from an extra (tax-induced) liter reduction in fuel use (see equation 1.1 of the main report).[79]

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[77] These figures make some adjustment for recent increases in fuel efficiency. Other assumptions are: Central and South America and Eurasia 10.5 km/liter; lower income Europe and Asia 12.5 km/liter; and Middle East 8.5 km/liter.

[78] According to previous experience in the United States, a 10 percent increase in fuel prices increases fuel efficiency by about 2 percent (Small and Van Dender 2006).

[79] The understatement is not huge however. For example, based on assumptions here and the Appendix for Chapter 2 of the main report, even a 50 percent increase in gasoline prices would increase fuel efficiency by 12.5 percent.

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