86485

Reducing Black Carbon Emissions from Diesel Vehicles: Impacts, Control Strategies, and Cost-Benefit Analysis

THE WORLD BANK

The World Bank:
1818 H Street NW
Washington DC 20433
Telephone: 202-473-1000
Internet: www.worldbank.org

COVER PHOTOS: Paul Marcus/Shutterstock.com and Asian Development Bank via Flickr

Contents

Figures

Tables

Acronyms

ARI	Acute Respiratory Infection
BC	Black Carbon
BRT	Bus Rapid Transit
CARB	California Air Resources Board
CBA	Cost-Benefit Analysis
CEA	Cost-Effectiveness Analysis
CNG	Compressed Natural Gas
CO	Carbon Monoxide
DICE	Dynamic Integrated Climate-Economy
DOC	Diesel Oxidation Catalyst
DPF	Diesel Particulate Filter
EF	Emission Factor
EPA	Environmental Protection Agency
GDP	Gross Domestic Product
GIS	Geographic Information System
GNI	Gross National Income
GWP	Global Warming Potential
HC	Hydrocarbons
HHDT	Heavy Heavy-duty Truck
HK	Hong Kong
HRQL	Health-related Quality of Life
HVF	Heavy Vehicle Fee
ICCT	International Council for Clean Transportation
IPCC	Intergovernmental Panel on Climate Change
IRR	Internal Rate of Return
LDV	Light-duty Vehicles
LHDT	Light Heavy-duty Truck
LNG	Liquefied Natural Gas
LPD	Linear Population Density
MHDT	Medium Heavy-duty Truck
MJ	Megajoule
NMMAPS	National Morbidity, Mortality, and Air Pollution Study
NO_X	Nitrous Oxide
NPV	Net Present Value
NREL	National Renewable Energy Laboratory
OC	Organic Carbon
OECD	Organization for Economic Cooperation and Development
OEM	Original Equipment Manufacturer
OMB	Office of Management and Budget
PAF	Population Attributable Fraction
PFT	Partial Flow Technology
PM	Particulate Matter
PPP	Purchasing Price Parity
QALY	Quality-Adjusted Life Years
RF	Radiative Forcing
SCC	Social Cost of Carbon
SLCP	Short-lived Climate Pollutants
SOF	Soluble Organic Fraction
ULSD	Ultra-low-sulfur Diesel
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
VED	Vehicle Excise Duty
VKT	Vehicle Kilometers Travelled
VSL	Value of a Statistical Life
WHO	World Health Organization
WTP	Willingness to Pay

Acknowledgements

This study was undertaken by a team from the International Council for Clean Transportation (ICCT) led by Ray Minjares and David Vance Wagner, with contributions from Anil Baral, Sarah Chambliss, Sebastian Galarza, Francisco Posada, Ben Sharpe, Grace Wu, Kate Blumberg, Fanta Kamakaté, and Alan Lloyd. The ICCTs technical advisors included Aaron Cohen (Health Effects Institute), David Fahey (National Oceanic and Atmospheric Administration), Jan Fuglestvedt (Centre for International Climate and Environmental Research), James Hammitt (Harvard University), and Michael Walsh (International Consultant).

The study was commissioned by the World Bank, and led by Sameer Akbar with support from Ellen Baum, who worked closely with the team from ICCT. Michael Levitsy guided the study during its initiation phase. Management oversight was provided by Jane Ebinger and valuable guidance was provided by Karin Kemper during the course of this study.

The study report received insightful comments from technical peer reviewers. We would like to thank Masami Kojima, Kirk Hamilton, Todd Johnson, Andreas Kopp and Richard Hosier.

Executive Summary

A 2013 scientific assessment of black carbon emissions and impacts found that black carbon is second to carbon dioxide in terms of its climate forcing. Black carbon is 3,200 times more effective on a mass-equivalent basis than carbon dioxide in causing climate impacts within 20 years, and 900 times more effective within 100 years. Black carbon increases global and regional temperatures when emitted into the atmosphere, where individual particles directly absorb energy from the sun and radiate it back as heat. Black carbon also reduces the strong cooling effect of large, highly reflective surfaces such as glaciers and Arctic ice. High concentrations of black carbon in the atmosphere can change precipitation patterns and reduce the amount of radiation that reaches the Earth's surface, which affects local agriculture.

Acute and chronic exposures to particulate matter are associated with a range of diseases, including chronic bronchitis and asthma, as well as premature deaths from cardiopulmonary disease, lung cancer, and acute lower respiratory infections. In 2012 the International Agency for Research on Cancer re-categorized diesel engine exhaust as carcinogenic to humans based on evidence that sufficient exposure is associated with an increased risk of lung cancer. Particles containing black carbon are predominantly less than 100 nanometers in diameter, allowing them to penetrate deep into the lungs and deliver toxic components to the bloodstream.

The transportation sector accounted for approximately 19 percent of global black carbon emissions in the year 2000. Road transportation accounted for nine percent of global black carbon, with diesel engines responsible for nearly 99 percent of those emissions. In the near term, black carbon emissions from mobile engines are projected to decline as a consequence of policies implemented in the United States, Canada, Europe, and Japan. However, black carbon emissions are projected to increase in the next decade as vehicle activity increases, particularly in East and South Asia.

This report aims to inform efforts to control black carbon emissions from diesel-based transportation in developing countries. It presents a summary of emissions control approaches from developed countries, while recognizing that developing countries face a number of on-the-ground implementation challenges. A cost-benefit framework for economic analysis of diesel black carbon emissions control transport projects is also presented that factors in both climate and health benefits.

Historically, technical interventions to control diesel black carbon emissions in developed countries have successfully relied on fuel quality improvements and vehicle emissions standards. Standards in Japan, Europe, the United States, and some other parts of the developed world require new vehicles to be equipped with filters and ultra-low-sulfur diesel (ULSD). This enables the use of diesel particulate filters and the adoption of strict emissions standards (e.g., Euro 6/VI). Particulate filters and ULSD represent one of the commonly used control strategies for diesel particulate emissions and have significantly reduced black carbon emissions in developed countries. However refinery investment / upgrades or importation (which can be expensive) are key to the availability of ULSD. Other programs targeting the existing fleet and addressing transportation demand can also provide significant benefits. Such programs include vehicle scrappage and replacement, inspection and maintenance, and vehicle retrofitting. Complementary policies to limit growth in travel demand and long-term growth in emissions include fuel taxation, congestion charging, and logistics management, among other strategies.

Diesel emissions control interventions in developing countries must be sensitive to the strength of local governance and technical capacity. No developing country has adopted fuel and vehicle standards equivalent to Euro 6/VI. Among non-OECD countries, Brazil has adopted the most stringent fuel and vehicle standards, followed closely by Russia. Brazil is on track to adopt more stringent fuel quality standards by 2015; similarly tough vehicle standards are less certain. Significant progress has been made in China and India to move to Euro 4/IV standards for light- and heavy-duty vehicles; some major cities, including Delhi and Beijing, have taken steps to advance beyond the national requirements. Implementation of improved fuel and vehicle standards, which requires both government regulation and enforcement, may be difficult however in regions where governance is weak and technical capacity is limited. Emissions control strategies should be both sensitive to local needs and aim for maximum feasible reductions guided by best practices. Policy roadmaps can be a useful tool in providing greater predictability of interventions.

***In the cost-benefit analysis, a range of simulated diesel black carbon control interventions were found to produce clear net benefits*.** This study applies a new cost-benefit analysis methodology to four simulated diesel black carbon emissions control projects -- diesel retrofit in Istanbul, green freight (plus retrofit) in Sao Paulo, fuel and vehicle standards in Jakarta, and Compressed Natural Gas (CNG) buses in Cebu -- taking into account the additional climate benefits of black carbon reductions. For some projects, the net benefits (taking into account health and climate impacts) were positive only when assuming a large benefit from black carbon control on the climate in the near term (using 20 years Global Warming Potential or GWP) and a low social cost-of-carbon discount rate.

The analysis demonstrates that consideration of black carbon may make some projects viable that otherwise would not be considered worthwhile. While investments in many diesel emissions control projects can be justified without the consideration of black carbon, the inclusion of black carbon into a cost-benefit modeling framework was found to provide a more comprehensive assessment. In two of the four cases the health benefits of diesel emissions control alone were substantial enough to justify the interventions; the other two projects became viable only with the inclusion of climate benefits.

Further work is needed to fully test this methodological framework with real-world projects and to establish clearer guidelines for the incorporation of black carbon into cost-benefit analysis. Additional work is also needed to narrow the range of assumptions for the discount rate tied to the social cost of carbon as well as the Global Warming Potential. Importantly, a social cost of black carbon needs to be developed and alternative methodologies need to be explored whereby temperature response and damage functions (for climate impacts) are applied.

The analytical framework presented in this report can be refined and expanded to incorporate the multiple benefits of reducing diesel black carbon emissions, including for agricultural productivity and ecosystems. While this report focuses on quantifying just the health and climate benefits of transport interventions, it also serves to highlight the challenges that could be faced when undertaking more comprehensive evaluation of transport projects. Ultimately, this framework could also be expanded to include analysis of project types not considered here, such as projects dealing with modal shifts where human behavioral dynamics become very important compared to technological interventions.

1 Overview

1.1 Trends in Diesel Black Carbon Emissions

In the year 2000, an estimated 7.7 teragrams of black carbon (BC) emissions were released from all sources (Lamarque et al. 2010). OECD countries accounted for 1.2 teragrams, or approximately 16 percent of the global total. In the same year, surface transportation and international shipping activity accounted for approximately 1.47 teragrams per year (Figure 1-1), or 19 percent of global black carbon emissions. Road transportation accounted for nine percent of global black carbon, and diesel engines, primarily on-road heavy-duty vehicles and international marine vessels, was responsible for nearly 99 percent of black carbon from these sources (Uherek et al. 2010).

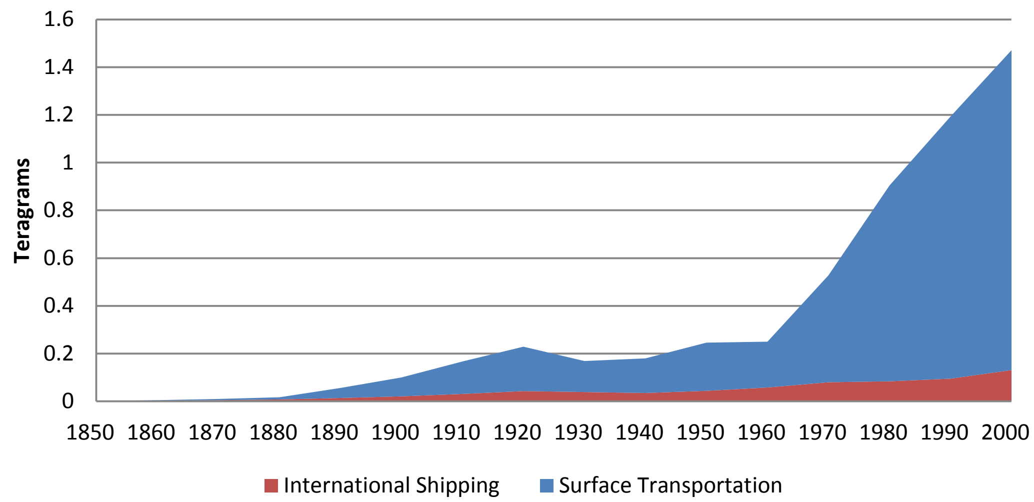

Figure 1-1: Historical Trends in Black Carbon Emissions from Surface Transportation (teragrams of black carbon per year).

Source: Vurren et al (2011). Surface transportation updated from Bond et al. (2007); international shipping in 2000 from Eyring et al. (2010) and scaled backward using CO_2 emission time series from Bugaug et al. (2009).

Projections of future black carbon emissions prepared for the IPCC Fifth Assessment Report project a general decline in global black carbon emissions from all sources including transportation, of 44-57 percent by 2100 (Vuuren et al. 2011). The decline is based on the assumption that countries will get richer over time, and as they get richer they will deal with their emission problems. However, black carbon emissions from surface transportation are not expected to decline in developing countries given the projected growth in vehicle activity and the slow rate of change in both fuel quality and in engine technology.

Vehicle activity is projected to expand dramatically through 2050 (see Figure 1-2). Light-duty vehicles (LDV) will remain the largest contributor, followed by heavy-duty trucks; both are projected to increase as developing countries grow economically. Emissions of black carbon are expected to grow as well in the absence of stringent emission controls in certain critical regions.

On a global basis, 10 countries or regions accounted for about 60 percent of black carbon emissions from on-road sources in 2010. As emissions decline in regions that require the best available control technologies, including Canada, the United States, the EU-27, and Japan, they will increase rapidly in regions where emissions controls lag. For example, India and China are projected to account for two-thirds of BC emissions from global on-road transportation by 2030 (Figure 1-3).

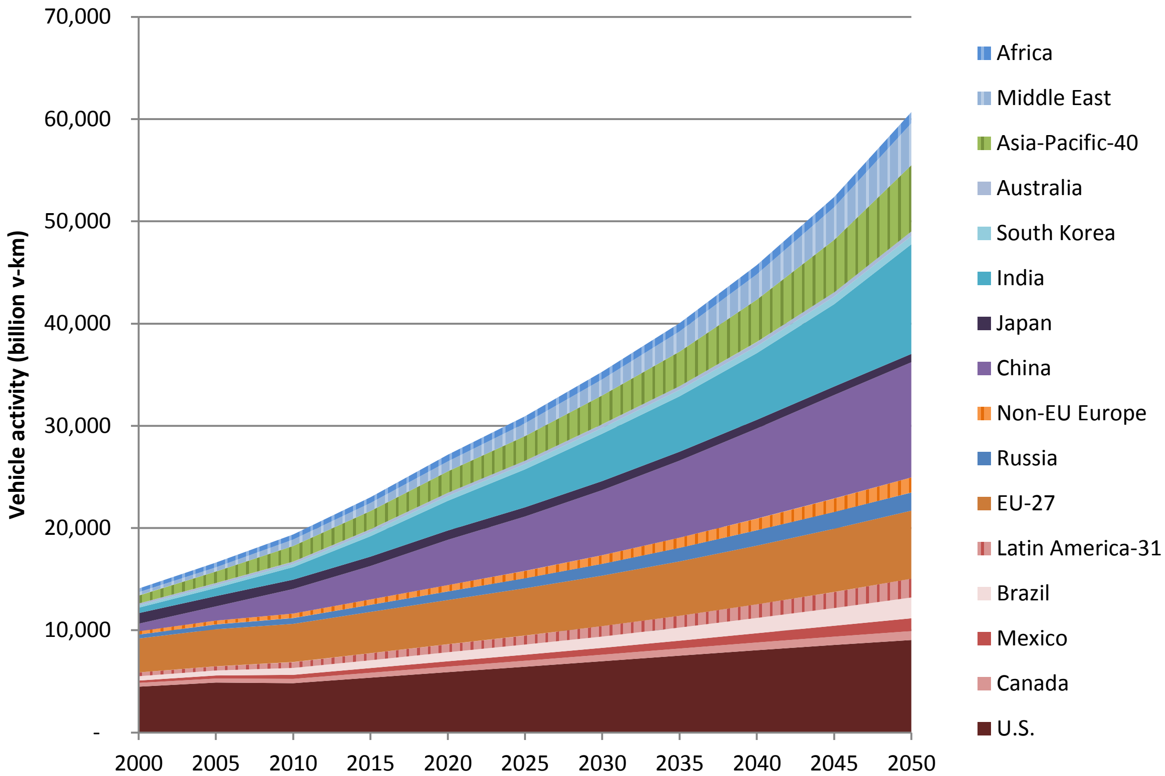

Figure 1-2: Vehicle Activity Forecast By Region, to 2050. Source: Data from Facanha et al. (2012).

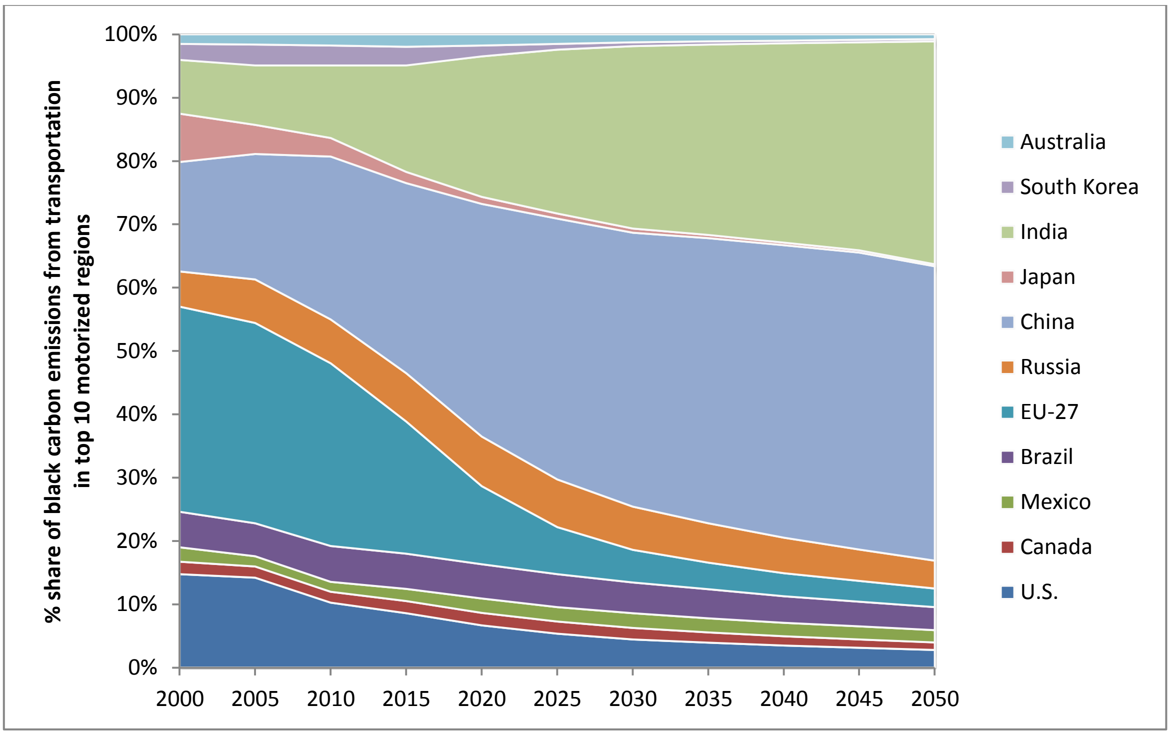

Figure 1-3: Global Black Carbon Emissions from Transportation by Region among Top 10 Motorized Regions, 2000-2050. Source: Data from Facanha et al. (2012).

1.2 Black Carbon as a Component of Particulate Matter

Fuel sulfur content has the biggest impact on emissions, affecting direct sulfate emissions, an important component of particulate matter (PM), and limiting the use of engine and after-treatment technologies designed to reduce emissions. Diesel generated from non-petroleum sources can also produce slight differences in emissions. Biodiesel, for example, can produce lower overall tailpipe PM emissions but slightly higher nitrogen oxide (NOx) emissions compared to conventional diesel fuel (because of the higher oxygen content of the fuel).

Despite the versatility and efficiency of diesel engines, diesel vehicles have historically had higher PM and NOx emissions. PM mass emissions are much lower in gasoline, natural gas, or liquefied petroleum gas spark ignition engines.

Particulate matter is generally divided into three groups, based on chemical and physical properties: (1) the solids fraction composed of elemental carbon and ash; (2) the soluble organic fraction, which is made up of organic material derived from engine lubricating oil and fuel; and (3) sulfate particulates that originate from the sulfur present in the fuel and lube oil.

Table 1-1: Black Carbon Fraction of PM2.5 Emissions for Gasoline and Diesel Vehicle Types, by European Emission Standard

Mode	Fuel	Uncontrolled	Euro I	Euro II	Euro III	Euro IV	Euro V	Euro VI
LDV	Gasoline	3%	23%	17%	17%	17%	17%	17%
	Diesel	47%	70%	80%	72%	69%	25%	25%
Bus	Gasoline	3%	23%	17%	17%	17%	17%	17%
	Diesel	50%	65%	65%	61%	83%	83%	7%
LHDT	Gasoline	3%	23%	17%	17%	17%	17%	17%
	Diesel	47%	70%	81%	72%	69%	23%	25%
MHDT	Gasoline	3%	23%	17%	17%	17%	17%	17%
	Diesel	46%	70%	80%	72%	68%	23%	25%
HHDT	Gasoline	3%	23%	17%	17%	17%	17%	17%
	Diesel	51%	65%	65%	61%	83%	83%	8%

Source: Based on Bond et al. (2007).

As seen in Table 1-1, in the absence of a filter the majority of the particulate mass consists of black carbon.[1] The fraction of engine particulate matter that is black carbon varies by fuel and engine type, measurement approach, duty cycle (the demand on the engine during the course of operation), and other variables. In California in 2006, black carbon (measured as elemental carbon) accounted for approximately 50 percent of $PM_{2.5}$ emissions from the statewide fleet of diesel vehicles (Chow, Watson, Lowenthal, Chen, and Motallebi 2010). When measured on a per-vehicle basis, black carbon emissions can account for 80 percent of $PM_{2.5}$ mass, particularly for diesel vehicles without the particulate filter that is required by Euro VI for vehicles with heavy-duty engines and by Euro 5 and 6 for vehicles with light-duty engines. Report Objectives and Structure

This study aims to inform the growing effort to control black carbon emissions (as part of particulate matter) from diesel vehicles while recognizing that developing countries face a number of on-the-

[1] Elemental carbon is also used in this report to refer to black carbon; technically speaking, they are measured differently, which can produce differences in emissions estimates from a particular source.

ground implementation challenges and lack an analytical framework to factor the economic benefits of diesel emissions controls into a project's cost-benefit analysis.

Chapter 2 summarizes the current state of knowledge about impacts of BC emissions on climate and health. Chapter 3 discusses lessons from OECD countries on the application of strategies to reduce BC emissions and their implications for developing countries. Chapter 4 illustrates the application of a cost-benefit analysis framework to simulated projects, with and without the inclusion of BC emissions in the analysis; and the final chapter (Chapter 5) presents the conclusions of the report and suggests next steps.

2 Diesel Black Carbon Impacts on Climate and Health

2.1 Climate

Black carbon is a strongly light-absorbing carbonaceous aerosol produced under conditions of incomplete fossil fuel and biomass combustion. These very small particles consist of a mixture of graphitic and amorphous carbon. When emitted, they persist on average in the atmosphere for more than a week until they come into contact with a surface or are washed out of the atmosphere by precipitation.

Three recently published assessments of the latest black carbon climate science have extensively reviewed and summarized this area of scientific literature (Bond et al. 2013; Shindell et al. 2011b; U.S. EPA 2012). Further discussion is included in Appendix A.

Black carbon affects the climate directly by absorbing solar radiation or changing surface albedo and indirectly by affecting cloud brightness, emissivity, and lifetimes. Bond et al. (2013) find that the best estimate of total radiative forcing in the industrial period, accounting for all forcing pathways, including interaction with clouds and on the cryosphere, is +1.1 Wm^{-2} with an uncertainty range of 0.17 Wm^{-2} to 2.1 Wm^{-2} . They conclude that, after CO_2, black carbon is the second most important human emission in the present-day atmosphere in terms of its climate forcing.

The particulate matter in diesel exhaust consists primarily of solid carbonaceous particles of black carbon and organic carbon, with the remaining mass composed of metals, ash, and semi-volatile organics and secondary particles such as sulfates and nitrates. Figure 2-1 shows the components of PM mass emissions from a typical heavy duty diesel engine without a particle filter. While CO_2 is the dominant source of positive forcing from diesel emissions, strong radiative forcing is exerted by other pollutants as well, including organic carbon, black carbon, aerosol precursors such as SO_2 and NO_2 (which produce light scattering sulfates), carbon monoxide, volatile organic compounds (VOCs), and nitrogen oxide (NOx). The aerosols organic carbon, sulfates, and nitrates strongly reflect light and cause negative radiative forcing over relatively short time scales, thus offsetting the positive forcing from black carbon. As a result, the net climate forcing from black carbon emissions is assessed after accounting for the impacts of co-emitted pollutants.

The transportation sector accounts for approximately 16 percent of the global annual mean anthropogenic radiative forcing of black carbon, the vast majority of which is from diesel-fueled vehicles (Koch et al. 2007). The net climate forcing from diesel combustion is highly contingent on the ratio of BC to cooling aerosols (U.S. EPA 2012; Ramana et al. 2010). This is illustrated in Figure 2-2. On balance, Euro II heavy-duty vehicles are a valuable target for reducing climate impacts of short-lived forcing agents. Euro II heavy-duty vehicles cause net positive radiative forcing when all emissions components are considered, even when uncertainty in their forcing is included. Positive forcing is caused by black carbon in the atmosphere as well as ozone production via NO_x. These effects may be offset by negative forcing from ammonia via NO_x, although the magnitude of the ammonia effect is less certain. For a pre-regulation, light-duty diesel vehicle, black carbon emissions are greater, and this leads to a greater contribution to overall radiative forcing. But since emissions of other pollutants, such as sulfates, are also large, their effects (and uncertainties) are magnified. This contributes to overall uncertainty in the net effect of this vehicle type. Higher fuel sulfur content leads to higher sulfate emissions that cause direct negative forcing in the atmosphere and indirect negative forcing via the production of clouds. Additional negative forcing is generated by ammonia production as well as black carbon interactions with liquid clouds. Further research is needed to reduce uncertainty in the cloud effects to validate whether emissions from pre-regulation vehicles indeed cause the positive forcing.

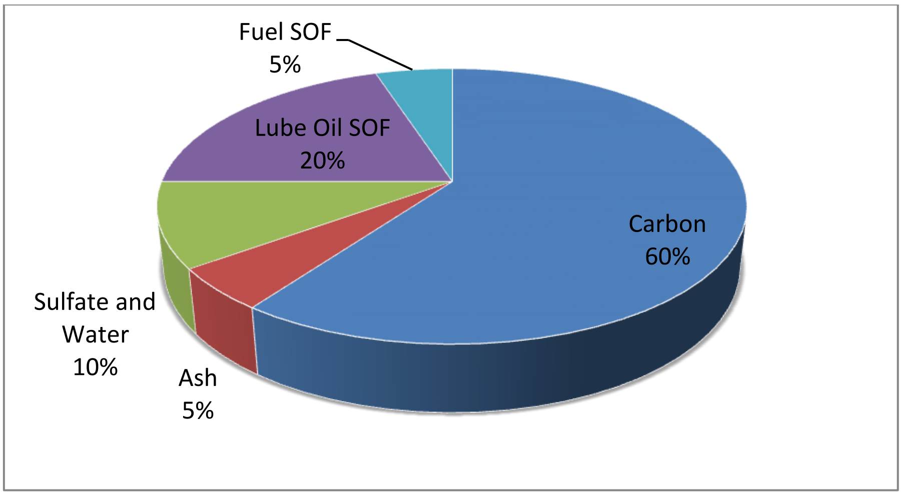

Figure 2-1. Composition of PM Mass Emissions from a Conventional Heavy-duty Diesel Engine without a Particle Filter.
Source: CARB 2011.

The ratio of black carbon to total $PM_{2.5}$ and other short-lived forcers varies among vehicles in relation to technology, fuel type, and maintenance practices, and can range from less than 10 percent to greater than 80 percent. According to one estimate, the elimination of black carbon along with light-reflecting organic carbon and sulfates from transportation sources across the globe would reduce radiative forcing of transportation-related black carbon by an estimated 50 percent (Koch et al. 2007).

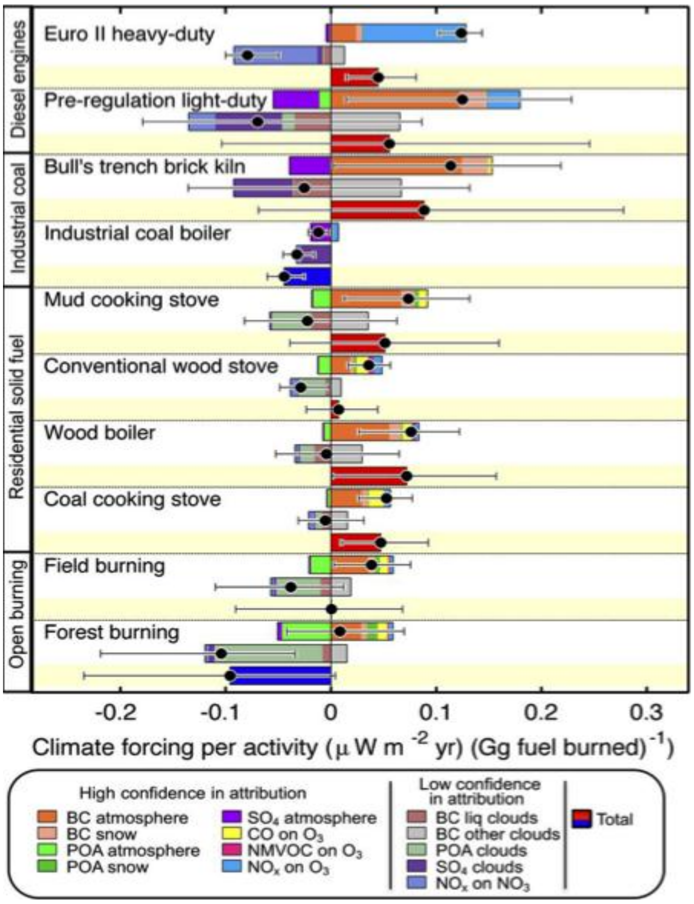

Figure 2-2. Climate Forcing of Black Carbon Emissions Sources, including Diesel Engines.
Source: Bond et al 2013.

A recent analysis that modeled the climate impacts of tight emissions control standards for gasoline and diesel-fueled transportation (equivalent to Euro 6/VI) demonstrated that the benefits of diesel emissions controls are affected by differences in regional climate and emissions. The reduction in sulfates necessary to implement tight standards could have a strong warming effect on its own, but this is mitigated in the Sahara and Arctic regions where bright underlying surfaces have a similar albedo to sulfates. In contrast, the warming effect may be magnified in regions where the underlying surface is much darker than sulfates (Shindell et al. 2011a). As a result, assessing the benefits of diesel controls may be more complicated once regional factors and the magnitude of baseline sulfate emissions are considered. Assuming adoption of tight emissions standards across the globe by 2030, however, negative forcing over all regions is expected relative to baseline 2030 emissions.

There is ongoing debate in the policy and scientific communities regarding the most justifiable metric for a climate policy that includes short-lived and long-lived forcing agents. There is also a movement toward a “two-basket” approach that uses time-scale appropriate metrics for SLCFs and LLCFs (Jackson 2009; Rypdal et al. 2005; Sarofim 2012; Smith et al. 2012). Bond et al. (2011) estimated 20-year and 100-year global average GWPs that include cryosphere effects and give detailed values for only energy-related sources of black carbon. Updated estimates published in Bond et al (2013) give GWP values of 3,200 on a 20-year time scale and 900 on a 100-year time scale. These values are used in this study.

Appendix A provides additional information on the climate impacts of black carbon emissions from diesel vehicles.

2.2 Health

Factors such as engine type and operating conditions, fuel, and engine technology influence the characteristics of diesel exhaust, causing significant variation in the mix of exhaust gases and the size and composition of the particulate matter (Mauderly et al. 1994). Vehicle emissions undergo further near-road and atmospheric chemical transformations that result in the formation of secondary particles and gases. The health impacts of diesel exhaust depend not only on the magnitude of the emissions but also the proximity of emissions to populations and the specific chemical components of the emissions (HEI 2010). While epidemiological studies estimating the impact of PM exposure on health have used PM mass, another measure of impact is exposure to ultrafine particles, which is characterized by particle number and size distribution.

2.2.1 Exposure to Diesel Exhaust

Diesel emissions are a major contributor to overall air pollution. Multivariate receptor models are used to estimate the contribution of various combustion sources, including motor vehicles, to ambient PM concentrations (HEI 2010). Application of these models in urban areas across the globe has found that motor vehicle emissions contribute 5-55 percent of $PM_{2.5}$ concentrations across U.S. cities; 53 percent in Barcelona, Spain; 2-12 percent in Beijing and 32-34 percent in Guangzhou, China; and 15-18 percent in Mumbai, India. When data in the literature is available to distinguish between diesel and gasoline vehicles, the highest published contribution of diesel vehicle exhaust to total ambient $PM_{2.5}$ was 50 percent in the city of Atlanta between 1998 and 2000 (HEI 2010).

Total population exposure to diesel exhaust may be underestimated by only considering its contribution to ambient pollution. A disproportionate share of exposure to traffic-related pollutants--up to 50 percent in the U.S.--takes place within vehicles or near roads, two microenvironments where exhaust concentrations are much higher (HEI 2010). High roadside concentrations are a major source of exposure in urban areas with heavy vehicle and foot traffic, and these high concentrations are a growing concern in developing countries undergoing rapid urbanization (Han 2006).

2.2.2 Health Effects of Diesel Exhaust

Emissions from diesel engines and other mobile sources are known to have serious health impacts. The International Agency for Research on Cancer added diesel exhaust to its list of known carcinogens in 2012 based on historical studies of occupational exposure (International Agency for Research on Cancer 2012). A comprehensive review of the health effects of traffic-related emissions was published by the Health Effects Institute in 2010; it concluded that the existing evidence base is suggestive of causal associations between traffic exposure and all-cause mortality, asthma onset and exacerbation in children, respiratory symptoms in adults, decreased lung function, and cardiovascular mortality and morbidity (HEI 2010). Among the key findings, the study concluded that a causal relationship exists between traffic exposure and exacerbation of asthma and that individuals living within 300-500 meters from a highway or a major road are within a high exposure zone for traffic emissions. The study did not identify a single pollutant surrogate for traffic-related emissions, which potentially increases the complexity and burden of undertaking source-specific health impact studies.

Two components of diesel exhaust, particulate matter and SO_2, are recognized by the U.S. EPA and the WHO as hazardous pollutants. Expert consensus, expressed in a scientific statement by the American Heart Association, is that exposure to $PM_{2.5}$ can trigger cardiovascular events and reduce life expectancy (Brook et al. 2010). There is less certainty about whether SO_2 directly causes health problems, but both the WHO and the EPA recommend SO_2 concentration limits based on the pollutant's association with changes in pulmonary function, childhood respiratory disease, and all-age mortality.

2.2.3 PM Components, including Black Carbon

Specific elemental components of particulate matter may contribute differently to PM toxicity (Laden et al. 2000; Franklin et al. 2008), but which components of PM are most toxic remains an active area of inquiry (Kennedy 2007; Bell 2012). There is evidence that, in the past, PM from mobile sources had a higher toxicity than from other sources (Laden et al. 2000).

In this study, the health impacts of black carbon are considered equivalent to PM health impacts, in line with the mainstream health impact assessment approaches of both the WHO and the EPA. Several ongoing studies are now investigating whether the high BC content of diesel PM may influence its toxicity. Recent reports by the EPA and the WHO recognize associations between BC and a continuum of cardiovascular and respiratory effects but conclude that it is not currently possible to distinguish the health impacts of BC from $PM_{2.5}$ (U.S. EPA 2012; Janssen et al. 2012). Toxicological studies suggest that BC in its pure form may not be toxic, but it may function as a carrier of toxins to the lungs, the body's major defense cells, and possibly the blood circulation system (Janssen et al. 2012). The higher share of black carbon in diesel exhaust would help to explain the toxicity of this emission source. Based on the evidence that black carbon is an efficient transporter of toxic compounds into the lungs, a strategy to reduce black carbon emissions from diesel vehicles (and, by implication, particulate matter) would produce health benefits.

Appendix B provides additional information on the health impacts of emissions from diesel vehicles.

3 Diesel Black Carbon Emissions Control Strategies

No country or jurisdiction directly regulates diesel black carbon. A wide range of strategies and policies have been implemented, however, to effectively control black carbon. Practices in OECD nations in particular provide lessons for developing countries to address diesel emissions. Many changes to both vehicles and fuels require government regulation, good governance, and local capacity in order to be successful. In regions with little experience in advanced or technology-based diesel BC emissions control, demonstrations and locally focused projects may be appropriate to build capacity and prove effectiveness before scaling-up solutions to the regional or national level.

A vehicle emissions reduction program often focuses on three areas: new vehicles, fuels, and the in-use fleet. In some countries it may make sense to start with the in-use fleet and transportation demand management. In certain cases, fiscal policies can be effective tools to complement mandatory regulatory requirements. The order or priority in approach should be dictated by the baseline technology, the rate of growth of the fleet, the feasibility of available options, the institutional capacity to support the intervention, and other local considerations. Successful strategies tend to take a holistic approach that integrates all maximum feasible and cost-effective emissions reduction strategies.

3.1 Emissions Control Technologies

The technological strategies used to control particulate matter can be classified as in-cylinder or after-treatment. In addition, reducing fuel and engine lubricant sulfur levels lowers sulfate PM emissions. This section looks at the various emissions control technologies and their impacts on reducing harmful pollutants.

3.1.1 In-Cylinder PM Control

Advanced engine calibration techniques and the implementation of electronic controls in new diesel engines have improved air-fuel mixtures and produced significant in-cylinder emissions reductions in both PM and NOx. Emissions of PM can be reduced by changes in engine design that adjust the mixture of fuel and air in the engine cylinder. The fuel injection system design should also be carefully matched to the air management system. The main goal is to control the local concentration of fuel and air inside the combustion chamber and to avoid the conditions that lead to PM formation.

3.1.2 PM After-treatment Devices

Diesel Oxidation Catalysts

A diesel oxidation catalyst (DOC) is a flow-through catalytic converter that oxidizes pollutants such as CO, HC, and the soluble organic fraction (SOF) of PM to carbon dioxide (CO_2) and water in the oxygen-rich diesel exhaust stream. A DOC is effective at oxidizing the SOF and gaseous HC but does not reduce the number or mass of exhaust soot particles. Reductions of 20-50 percent in overall PM emissions (mass basis) from DOCs have been reported (Chatterjee et al. 2008; NESCAUM 1999; Walker 2004). Performance of the DOC can be adversely affected, however, by diesel fuel sulfur content. PM emissions will increase if sulfur levels are above 150 parts per million, attributable to the increase in SO_2 emissions and conversion to SO_4. Although DOCs have been shown to reduce a portion of the total PM mass, they do not significantly reduce particle number or black carbon mass emissions because they do not collect or burn the soot portion of the exhaust (Chatterjee et al. 2008).

Partial Flow Technology

A partial flow technology (PFT) system or partial flow filter is a PM reduction device consisting of a DOC and a flow-through filter element. The soluble organic fraction portion of the exhaust is oxidized in the DOC as described in the previous section, and then some portion of the remaining soot is captured and combusted in the filter.

Although PFT systems are generally more effective than the DOC in lowering PM mass—reductions are typically cited as being greater than 50 percent (Chatterjee et al. 2008; Jacobs et al. 2006)—the performance and durability of these systems have yet to be fully characterized in the published literature.

Diesel Particulate Filters

A wall flow diesel particulate filter (DPF) is a filter designed to capture and eliminate solid accumulation mode carbonaceous particles from diesel vehicle engine exhaust. Figure 3-1 illustrates how exhaust gases are redirected and channeled through porous walls as they escape to the filter exit. After the PM is trapped in the filter, the next stage is to combust these carbonaceous particles. There are two basic methods for combusting the captured PM: passive regeneration and active regeneration.

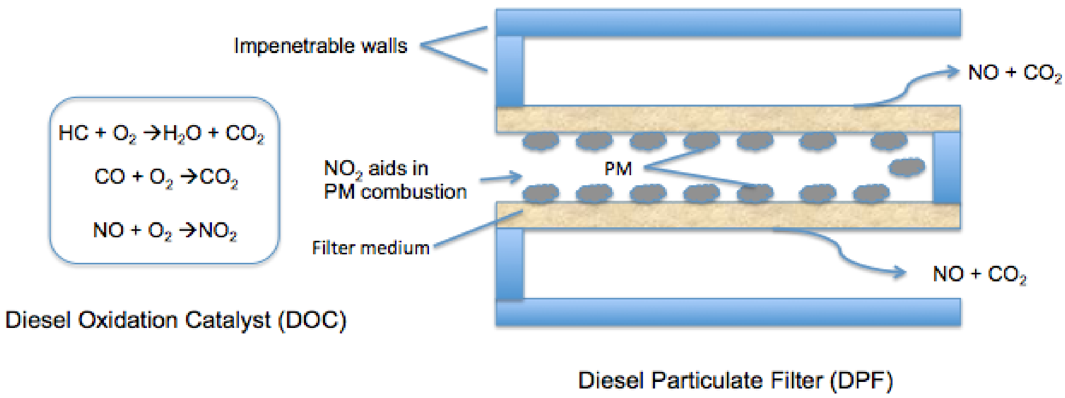

Figure 3-1: Diesel Particulate Filter Operation.

In passive regeneration, trapped PM is combusted at the high temperatures achieved during the normal operation of the vehicle—neither the vehicle operator nor the engine management system needs to induce the regeneration process. To facilitate combustion under normal operating temperatures (200-400ºC for most heavy-duty vehicles), nitrogen dioxide (NO_2) can be introduced. This oxidation can occur upstream of the filter with the aid of a diesel oxidation catalyst, and this design is referred to as a continuously regenerating DPF.

In active regeneration, sophisticated engine controls measure back pressure in the filter (which increases as PM accumulates). When pressure reaches a certain level, excess fuel is injected either into the combustion cylinder or directly into the exhaust. Typically, the fuel is injected directly into the exhaust upstream of the DOC. The fuel is then oxidized in the oxidation catalyst, causing an exothermic reaction that increases the temperature of the exhaust gas. The resultant heat causes the DPF temperatures to rise, initiating the oxidation of the PM. Actively regenerated wall flow DPFs increase fuel consumption to levels that are higher than any of the other PM control technologies listed here.

Of the three particulate control technologies, the wall flow DPF is the most efficient, with mass basis PM reductions typically cited between 85-95 percent (Kleeman, Schauer, and Cass 2000; Walker 2004). In addition to effectively filtering and combusting PM mass, particle number reductions can be on the order of 99.5 percent or higher when compared with direct engine emissions (Maus and Brück 2007; Vaaraslahti et al. 2006). The durability and long-term performance of DPF systems for a wide variety of heavy-duty applications have been well established, including for buses, municipal vehicles, long haulers, and construction equipment. Hundreds of thousands of DPFs have been installed on new vehicles as well as retrofitted in older ones.

In any DPF using a catalyst (either a DPF that has catalyzed filter media or one being used upstream in the form of a DOC), sulfur—in competing for space on the catalyst surface—has similar negative effects to those discussed with regard to DOCs. Operation of a passive DPF with higher-sulfur fuels can cause the filter to be overloaded with soot, leading to uncontrolled soot burning and permanent damage to both the filter and the vehicle. Actively regenerated DPFs, meanwhile, can experience increased back pressure under high sulfate levels, requiring more frequent regeneration. This translates into penalties in the form of higher fuel consumption and shorter maintenance intervals (Chatterjee et al. 2008).

3.2 New Vehicle Emissions and Fuel Quality Standards

Emissions standards, which limit the amount of a pollutant released by or evaporated from vehicles and engines during a predefined driving test, are a crucial element of all vehicle emissions control programs. Vehicle standards go hand in hand with fuel quality requirements (notably limits on sulfur emissions) that enable advanced emissions control technologies to be optimized. Compliance and enforcement measures are fundamental to a successful emissions control program because they ensure standards are met over a vehicle's useful life. The UNEP's Partnership for Clean Fuels and Vehicles (PCFV) program provides an updated map of diesel fuel sulfur levels globally as of June 2012 (PCFV, 2014). It shows that some cities even in developing countries have access to lower sulfur diesel than available nationwide.

The emissions limit standard values for diesel PM in the European Union (EU), the United States, the State of California (which has leeway from the U.S. government to set its own air quality standards), and Japan set global precedents for diesel emissions control. The emissions limits in these regions are based on the reductions that were achievable by the best available technologies considered at the time of policy adoption. Other aspects, such as cost-effectiveness and safety, are also taken into consideration. In addition, changes in engine technology and after-treatment devices have allowed for a steady tightening of emissions standards over time.

Although no jurisdiction has adopted emissions standards for diesel black carbon, emissions standards for $PM_{2.5}$, particularly the U.S. 2007 and European Euro 5/V and 6/VI standards, set limits that require particulate filters on diesel vehicles and are highly effective at reducing black carbon. In order to enable the effective use of particulate filters, however, the sulfur content of diesel fuel must be lowered through fuel quality standards. Desulfurization alone can moderately reduce total $PM_{2.5}$ vehicle emissions, but significant reductions in black carbon only become possible starting at 50 ppm (i.e., when the particulate filter is able to function properly).

Extensive programs have been carried out in the United States, the European Union, and Japan to understand the linkage between vehicle technology, fuel quality, and emission levels.[2] Fuel improvements can play an important part in reducing pollutant emissions; the adoption of better technologies, including after-treatment devices required to meet Euro 6/VI standards and beyond, can enable further reductions in emissions. Desulfurization alone, however, is not an effective black carbon emissions control strategy.

3.3 In-use Emissions Reduction Strategies

A comprehensive vehicle emissions reduction strategy includes programs to reduce emissions from vehicles already on the road. A wide range of strategies is available to address the diverse nature of legacy fleets in terms of vehicle ages, types, and locations.

[2] The Auto/Oil Air Quality Improvement Research Program (AQIRP) established in the United States in 1989 included major oil companies, automakers, and four associate members. A test program called the European Programme on Emissions, Fuels, and Engine Technologies (EPEFE) was initiated by the European Commission and joined by the auto and oil industries. The Japan Clean Air Program (JCAP) was formed by the Petroleum Energy Center as a joint research program of the auto and oil industries and supported by the Ministry of Economy, Trade, and Industry.

Mandatory periodic inspections, remote sensing, and spotter programs are important tools to help regulators ensure that private consumers maintain their vehicles. For captive fleets—such as urban buses and municipal service vehicles—strong centralized service facilities, regular maintenance training, and, if necessary, additional maintenance-related regulations (such as required daily visual inspections of emissions control equipment prior to operation) are desirable.

The following is a summary of various in-use strategies.

3.3.1 Scrappage and Replacement of Older Vehicles

Scrappage programs are designed to eliminate high-emitting vehicles from the fleet. These typically target older vehicles meeting less stringent emissions standards and operating with degraded pollution control equipment. Scrappage programs can be designed to accomplish the dual goals of reducing emissions and stimulating economic growth; for example, programs can be structured so that the owner of a scrapped vehicle must simultaneously purchase a new or newer vehicle with more stringent emissions standards.

3.3.2 Retrofits of Older Vehicles

Retrofits are best applied toward vehicles with some useful life and for which scrappage is not cost-effective. Retrofits most appropriate to diesel black carbon control include the installation of diesel particulate filters where low-sulfur diesel fuel is available. Many diesel emissions reduction programs in the United States, such as the EPA's National Clean Diesel Campaign, also fund vehicle "repowers"— the replacement of a vehicle's engine with a newer one manufactured to more strict emissions control standards or one burning a cleaner fuel (e.g., compressed natural gas).

3.3.3 Compliance Programs

Compliance programs are implemented by governments to ensure that vehicles throughout their useful life meet the emissions limits for which they are certified. These compliance programs are not effective, however, at controlling emissions of legacy vehicles already on the road. Furthermore, compliance programs can only feasibly be implemented in countries with a sizable domestic vehicle manufacturing industry and sufficient technical capacity to conduct in-use emissions testing—a particular challenge for heavy-duty vehicles. Developed countries and regions such as the United States, Japan, and the European Union have mature compliance programs; some developing countries, including China, have begun developing heavy-duty vehicle compliance programs for their domestic fleets.

3.3.4 Inspection and Maintenance Programs

Inspection and maintenance (I/M) programs require vehicle owners to regularly certify the emissions performance of their vehicles at an authorized emissions test facility. Vehicles whose emissions exceed a certain threshold are generally required to undergo repair or maintenance. Experience suggests that I/M programs must be carefully planned and implemented in order to both gain public support and be cost-effective (Hausker 2004).

3.3.5 Spotter Programs

A "spotter" program is an initiative that trains citizens to report vehicles visibly emitting smoke from their tailpipes. Based on the reported license plate information of the smoky vehicle, regulators identify the vehicle owner and require him or her to seek repairs. Hong Kong has one of the most developed spotter programs in the world, and it has resulted in thousands of high-emitting vehicles being tested and repaired each year (Cheng 2010).

3.3.6 Fleet Maintenance

Maintenance is especially important on vehicles that must meet more stringent standards requiring advanced pollution control technologies (such as diesel particulate filters), as these vehicles must be carefully monitored to ensure proper functionality and to prevent engine damage.

3.3.7 Prevention of Vehicle Overloading

Vehicles are certified to meet emissions limits up to a specified vehicle weight or maximum loading. Overloading of heavy-duty vehicles can significantly increase the share of high emitting vehicles in the fleet (Cai and Wang 2004). Programs to reduce overloading (e.g., through mandatory weigh stations along freight truck routes, with heavy fines for violations) will thus reduce emissions.

3.3.8 Complementary or Enabling Strategies

Most of the in-use emissions control strategies described here target direct reductions of vehicle emissions through engine, vehicle, or fuel modification. Complementary measures include programs that seek to reduce emissions through limits on vehicle operation in certain regions, fiscal disincentives for operating high-emissions vehicles, programs and policies to reduce emissions through behavioral changes, and other indirect, non-vehicle-specific programs such as road improvements and intermodal shifting. Examples include the creation of low-emissions zones in cities, levying pollution taxes on certain types or ages of vehicles, and anti-idling regulations.

3.4 Fiscal Policies for Reducing Diesel Black Carbon

Fiscal policies have been used successfully in some regions to support fuel quality improvement and vehicle emissions reduction programs. Although there are no established best practices for the design of specific fiscal policies targeting black carbon, the following examples highlight some international experiences in the use of fiscal policies for general control of the environmental impacts of motor vehicles.

3.4.1 Fiscal Policies to Encourage Higher Quality Fuel Supply

There are several examples of countries or regions using differential taxation, tax incentives, and/or direct subsidies to refiners to accelerate the supply of lower sulfur fuels.

In Japan, the national government instituted direct tax incentives in two phases (from 1990-1992 and from 1993-1997) to subsidize refinery investments in reducing sulfur in diesel fuel, first to below 2,000 ppm and then to 500 ppm. Refineries had a choice of a 7-percent deduction in corporate taxes or a 30-percent accelerated depreciation on purchased equipment (Gallagher and Oliver 2005). The city of Tokyo subsequently (in 2001) initiated a two-year incentive program to subsidize up to 10 yen per liter for oil companies that supplied ≤ 50 ppm sulfur diesel fuel. These regulations quickly sparked negotiations at the national level between multiple ministries; the outcome of these negotiations was the nationwide availability of 50 ppm sulfur diesel by mid-2003, 21 months earlier than required by the national regulation. In 2005, 10 ppm near-zero sulfur fuel became available nationwide, two years ahead of schedule (Takahashi 2011). Since vehicular emissions are the dominant source of black carbon in Tokyo, these policies led to direct and significant reductions (80 percent) in mass concentration of black carbon between 2003-2010 (Kondo et al 2012).

To promote the supply of 50 ppm diesel fuel in Hong Kong, the government in July 2000 reduced by half the import duty for that fuel. Within two months, 50 ppm sulfur diesel became the main diesel fuel supplied at local filling stations (Hung 2004). A similar strategy was adopted to promote 10 ppm sulfur diesel, and the government continued to waive the concessionary duty for 10 ppm sulfur diesel fuel even after the 10 ppm sulfur limit was mandated in July 2010.

In the UK, the conversion of the diesel motor fuel market to 50 ppm diesel was achieved six years ahead of the EU schedule and well ahead of most other EU member states. This can largely be attributed to a series of 50 ppm diesel tax incentives, which began in fiscal year 1997 and ratcheted up each year until full market penetration of 50 ppm diesel was achieved in 2000 (HM Customs and Excise 2000). The UK's fuel tax incentive was accompanied by vehicle tax incentives. In 1998-1999, when a 2-pence tax differential in 50 ppm and conventional (> 50 ppm) diesel was introduced, the government also reduced the vehicle excise duty (VED) of £500 for heavy-duty diesel vehicles that had particulate traps and other pollutant abatement technologies installed (meeting a preexisting Reduced Pollution Certificate qualification). The following year, the VED reduction increased to £1,000. The incentives for cleaner fuel and cleaner vehicles worked together to promote a rapid shift to a cleaner diesel fleet in the UK and significantly reduced PM emissions (by 21 percent in 1999 alone) (Olivastri and M. Williamson 2000).

In Germany, the government focused on improving air quality from transportation in order to achieve health benefits. The German government in 2001 rolled out a series of fiscal measures for the early introduction of diesel and gasoline with ≤ 50-ppm sulfur content. The government also issued an extra tax of three pfennigs/liter on fuel with a higher-than-50 ppm sulfur level, beginning in November 2001, in order to create a financial disincentive to use that fuel. The government later extended the three pfennigs/liter extra tax on fuel with higher than 10-ppm sulfur content (beginning on January 1, 2003). As early as 2004, virtually all fuel sold in Germany contained ≤ 10-ppm sulfur, with only minimal and short-lived fuel price disruptions due to competition and gains in efficiency from refining technology. Selected fiscal policies to promote lower sulfur fuels are summarized in Table 3-1.

Table 3-1: Summary of Selected Fiscal Policy Support for Fuel Desulfurization

Policy Type	Region	Magnitude	Result
Tax Differentials at the Pump	Hong Kong	HK$ 0.89/L for 50-ppm difference; HK$ 0.56/L for 10-ppm	Became the first region to introduce 50-ppm sulfur diesel in Asia Exclusive availability of 10-ppm sulfur diesel by 2008
	United Kingdom	1-3 pence/L from 1997-1999	Rapid transition to full 50-ppm diesel market in 1999
	Germany	An extra 3 pfennigs/L tax on diesel greater than 50 ppm sulfur in 2001 Extended the 3 pfennigs/L extra tax for diesel with sulfur ≥10 ppm in 2003	Rapid shift to 50-ppm and 10-ppm sulfur diesel
Tax Incentive for Refiners	Japan (national)	7% deduction in corporate tax, or a 30% accelerated depreciation on equipment purchase	5,000-ppm→2,000-ppm(1992)→500-ppm (1997)
	United States	$0.05 per gallon of 15-ppm diesel for small refiners	Shift to 30-ppm average gasoline in 2006, 15- ppm diesel in 2009
Direct Government Subsidy to Refiners	Tokyo	10 yen/L	500-ppm → 50-ppm (2003) →10-ppm (2005) respectively 21 months and 2 years ahead of national regulatory schedule

3.4.2 Fiscal Policies to Discourage the Use of Higher Emitting Vehicles

Fiscal policies have also been used to discourage the use of higher emitting vehicles. Subsidies to vehicle owners to scrap older, high-emitting vehicles prior to the end of their useful life is one approach that has been used in both developed and developing nations and regions

California has one of the world's longest-running voluntary, subsidized vehicle scrappage programs. From 1998-2010 and with almost $700 million in funding, the Carl Moyer Program resulted in the cleanup of nearly 25,000 engines. Among the benefits was an estimated reduction of 6,000 tons of PM in emissions in the state. Funding through the Carl Moyer Program covers new purchases, fleet modernization (scrappage and replacement), repowers, and retrofits for both on-road and off-road vehicles.

China has, with mixed results, aggressively pursued vehicle scrappage to achieve conventional pollutant emissions reductions through the use of fiscal policies. The Ministry of Environmental Protection has an ambitious goal to retire approximately 10 million "yellow-label" vehicles (diesel vehicles not meeting the Euro 3/III standard and gasoline vehicles not meeting the Euro 1/I standard) by the end of 2015. Both national and local subsidies have been implemented to support this goal. National subsidies of 3,000-6,000 RMB ($460-$920) were insufficient when offered in 2009; the subsidies were raised to 6,000-18,000 RMB ($920-$2,800) in 2010 (MOF 2009; MOF and MOC 2010). It remains unclear, however, how successful the 2009-2010 subsidies program was and the program was not renewed.

Local-level subsidy programs to match or exceed national-level subsidies have proven to be much more successful in China. To date, Beijing has been the most successful city to encourage voluntary early retirement of yellow-label vehicles. Beginning in 2008, Beijing offered additional subsidies, graduated by vehicle age and ranging from 1,000-5,500 RMB ($150-$850) per vehicle, for the scrappage of yellow-label vehicles.

Subsidy levels have been revised twice and are now as high as 17,200 RMB ($2,650) per vehicle (Beijing EPB 2011). Beijing's voluntary scrappage program is further bolstered by a Low Emission Zone which prohibits yellow-label vehicles from entering the city center. Since Beijing's program does not require verification of vehicle retirement to qualify for subsidies, however, it is unclear how many of these vehicles were transferred outside of the city limits and continue to pollute other areas of the country.

Mexico City launched a program in 2001 to renew public transportation buses called "Programa de Sustitución de Microbuses por Autobuses Nuevos." In that year, 90% of microbuses were model year 1993 or older, so they were not fitted with any pollutant control systems. Emissions from buses were 11.7% of total emissions from the transport sector. The program gives owners of pre-1995 buses up to $7,700 USD as a down payment for the purchase of a replacement vehicle. In order to qualify for the fiscal bonus, vehicle owners provide all legal documents for the vehicle to demonstrate good standing and must prove the economic solvency of their business.

Furthermore, vehicle owners must submit their old vehicles to a certified scrapyard to be destroyed. Through this program and the introduction of diesel particle filters in some new vehicles, the Mexico City government aims to reduce bus NOx emissions by 10 percent and bus CO, PM_{10} and $PM_{2.5}$ emissions by 90 percent in the fleet of 13,000 buses (Gaurav et al., 2014).

A second type of fiscal policy used to discourage the use of high-emitting vehicles is a direct pollution tax. Both Germany and Switzerland levy differentiated taxes on heavy-duty trucks based on their certified emissions standard. In Switzerland, a performance-related heavy vehicle fee (HVF) is levied on all heavy-duty trucks traveling on public highways in the country. The tax rates by emissions standard are divided into three categories: the highest taxation rate per kilometer is levied on Euro 0, I, and II vehicles, the middle rate is levied on Euro III vehicles, and the lowest tax rate is levied on heavy vehicles certified to Euro IV or higher standards. Enforcement is achieved through the use of installed recording devices or electronic identification cards carried by drivers that contain information on the vehicle's distance traveled and certified emissions standard (Switzerland FDF 2011).

There are also a variety of global precedents for the use of fiscal policies to promote improvements in vehicle efficiency and/or CO_2 performance. When choosing a new vehicle, consumers typically have a much larger array of choices regarding vehicle efficiency than they do for conventional pollution performance; nevertheless, lessons learned in the establishment of fiscal policies for vehicle efficiency in OECD countries point to best practices for the design and implementation of effective fiscal policy in non-OECD countries.

3.5 Scaling up Diesel Black Carbon Emissions Reduction Programs

Some of the diesel black carbon emissions reduction strategies described in this report – such as fuel quality improvement programs, new vehicle emission standards, and I/M programs – apply to entire fleets or classes of vehicles within a region. Other strategies – such as scrappage, retrofit programs, and clean fuel vehicle replacements – are targeted interventions that apply to a small and specific group of vehicles only. In both cases, two key principles have been found to be important for supporting the scaling up of such programs to a broader regional or national level.

Pilot projects to demonstrate effectiveness and build capacity are critical to ensuring sustained, broader success. In Beijing, for example, a successful 2005-2007 urban bus diesel particulate filter retrofit pilot led to the retrofitting of 10,000 buses before the Olympics. The program also demonstrated the importance of low sulfur fuel. In Mexico City, a 2004 urban bus DPF retrofit pilot led to a shortened timeline for the city-wide ULSD rollout.

Government engagement and support at the national level can help in sharing lessons learned with other cities/regions in a country as well as in scaling-up local experiences to the national level. However, institutional capacity at the local level is necessary to enable scale-up. In China, for example, although municipal environmental protection bureaus are charged with I/M program implementation and oversight, the program structure, approved test procedures, and limit values are established at the provincial level based on guidance documents issued by the national government. The Ministry of Environmental Protection supports program development throughout the country by leading training programs, maintaining a national database of testing information, and conducting laboratory testing to confirm local-level results and improve local testing capacity.

3.6 Regional- and Country-level Road Maps for the OECD and Developing Countries

Policies in OECD countries, notably in the United States, Canada, Europe, and Japan, have yielded a dramatic decline in fleet-wide emissions of diesel particulate matter with concomitant reductions in black carbon. This success has been guided by two fundamental principles for policy design and implementation. First, vehicles and fuels are treated as a single system. Second, emissions standards are based on emissions levels and guided by best available control technology.

The United States and Europe have adopted two different strategies for diesel emissions control. The United States takes a fuel-neutral approach, whereby diesel, gasoline, and any other fuel type is required to achieve the same emissions performance. In contrast, the European approach sets different emissions standards for gasoline and diesel fuel. The Europeans also regulate both particle mass and the total number of particles emitted in vehicle exhaust. In contrast, the United States only regulates particle mass. Both Europe and the United States, however, establish different standards for light-duty and heavy-duty vehicles. The U.S. approach has been adopted by Canada and Mexico, while the European approach has been adopted by a larger number of countries (e.g., China, India, Brazil, Russia, Turkey, Thailand, Australia, Argentina, and South Africa).

In practice, both the United States and Europe have followed a policy road map that defines increasingly stringent emissions standards. Developing countries could also choose to follow a similar road map or leapfrog to the most stringent standards. All things being equal, the latter option could confer not only more rapid health and climate benefits but also more cost-effective emission reductions.

All black carbon control has historically been achieved through limits on particulate matter, although each emissions standard for particulate matter differs in its effectiveness at reducing black carbon. The most stringent emissions standards in the United States for diesel black carbon are the Tier II and U.S. 2007 standards for light-duty and heavy-duty vehicle particulate matter, respectively. The most stringent limits in Europe are the Euro 6 standards for light-duty diesel vehicles and the Euro VI standards for heavy-duty diesel vehicles.

Future reductions in diesel particulate matter and black carbon emissions in OECD countries will require the adoption of best-practice tailpipe emission standards outside of the European Union, the United States, Canada, Iceland, and Japan (see Tables 3-2 and 3-3). This entails alignment by important OECD countries such as Mexico, Chile, Turkey, New Zealand, and South Korea. Switzerland has adopted emissions standards harmonized with the European standards.

Table 3-2, 3-3, 3-4 and 3-5 outline broad policy road maps for non-OECD countries. Among these countries and regions, none has adopted emissions limits for diesel particulate matter equivalent to the best practices found in OECD countries. Brazil and Russia are the furthest along in policy adoption; China and India have made significant progress.

Table 3-2: Policy Road Map[3] for Light-duty Vehicle Diesel PM Emissions Standards in OECD[4] Countries

Region	Engine Mods	Euro I	Euro 2	Euro 3	Euro 4	Euro 5	Euro 6
Australia	1976	1996	2003		2007	2013	2018
Canada		1980	1984	1995		2004	2008
Chile			1994		2006	*2011/2013	2015
EU-28		1992	1996	2000	2005	2010	2015
Iceland		1992	1996	2000	2005	2010	2015
Israel				2002	2007	2009	2015
Japan	1986	1994	1997	2002	2005		2009
Mexico		1999	2001	2003			2017
New Zealand			2006		2008	2012	2018
South Korea	1990	1995		2005	2006	2011	2015
Switzerland		1992	1996	2000	2005	2010	2015
Turkey		2002			2009	2012	2015
United States		1980	1984	1995		2004	2008

[3] Euro-equivalent standards were identified for US, Canada, and Japan who do not follow the Euro standards.
[4] The blue colored values are from the policy road map developed by ICCT (Chambliss et al., 2013).

Table 3-3: Policy Road Map for Heavy-duty Diesel PM Emissions Standards in OECD[5] Countries

Region	Engine Mods	Euro I	Euro II	Euro III	Euro IV	Euro V	Euro VI
Australia	1976	1996		2003		2011	2018
Canada	1990	1992		1994			2007
Chile		1994	2006	*2006	*2012		*2015/2017
EU-28		1992	1997	2001	2006	2009	2014
Iceland		1992	1997	2001	2006	2009	2014
Israel						2009	2015
Japan	1988	1994	1997	2003	2005		2009
Mexico	1990	1993		1994			2017
New Zealand					2008	2011	2018
South Korea	1990	1995	1998	2002	2004	2009	2015
Switzerland		1992	1997	2001	2006	2009	2014
Turkey		2001			2007	2012	2015
United States	1990	1992		1994			2007

Table 3-4: Policy Road map[6] for Light-duty Vehicle Diesel PM Emission Standards[7] in Developing Countries

Region	Engine Mods	Euro I	Euro 2	Euro 3	Euro 4	Euro 5	Euro 6
Africa		2005	2015		2020	2025	
Brazil	1995	1999	2003	2007	2009	2014	2017
China	1994	2000	*2003/ 2005	*2006/ 2008	*2009/ 2013	*2014/2015	*2016/ 2018
India	1992	2000	*2001/2005	*2005/2010	*2010	2016	2020
Middle East		2002	2010		2017	2022	
Asia Pacific 40		2000	2010	2015		2020	2025
Non-EU Europe		2000	2005	2010		2015	2020
Latin America 31		2000	2005	2010		2017	2020
Russia	1990	1999	2006	2008	2013	2015	2020

Table 3-5: Policy Road Map for Heavy-duty Vehicle Diesel PM Emission Standards[8] in Developing Countries

Region	Engine Mods	Euro I	Euro II	Euro III	Euro IV	Euro V	Euro VI
Africa		2005			2020		
Brazil	1993	1996	2000	2006		2012	2016
China		*2000/2001	*2003/2004	*2006/2008	*2009/2013	2015	2018
India	1992	2000	*2003/ 2005	*2005/ 2010	*2010/	2016	2020
Middle East		2000			2017		
Asia Pacific 40		2005	2015		2020		2025
Non-EU Europe		2000	2010	2015	2017		2020
Latin America 31		2000	2010	2015	2020		2025
Russia			2006	2008	2013	2015	2020

[5] Ibid.

[6] Policy timelines for emerging countries with major vehicle markets (i.e., China, India, Brazil, Mexico, and Russia) were developed based on communication with local and international policy experts. Timelines account for proposed or planned regulations and the availability of low-sulfur diesel. The implementation schedule also accounts for necessary regulatory lead time and coordination between vehicle emission and fuel efficiency standards (Chambliss et al., 2013).

[7] The red color represents the emission standards adopted by the country and blue represents the year in which ICCT projects that new standards will be adopted (Chambliss et al., 2013).

[8] Ibid

Main Issues Affecting Diesel Emissions Policy in Developing Countries

Experience with advanced emissions control devices in vehicles is limited in developing countries, which raises concerns regarding the realistic implementation of stringent emissions control standards.

Increased up-front vehicle costs, particularly in diesel vehicles, may be a significant barrier to achieving fleets of advanced clean vehicles. The cost of taking a 4-cylinder 1.5L gasoline engine from no emissions controls to the most stringent proposed EU standard (Euro 6) is about $360, whereas the cost of taking a 4-cylinder 1.5L diesel engine from no emission controls to Euro 6 standard is about $1,400 -- and the cost for heavy-duty diesel vehicles is approximately 4-5 times higher.

Emissions from gasoline engines can be reduced to very low levels through precise air-fuel control and catalytic after-treatment. Controlling emissions of NOx and PM from diesel engines, however, is more complex and requires a combination of technologies for air management, fuel injection control, after-treatment, and system integration. These carry substantially higher costs relative to gasoline engine controls.

3.6.1 Maintenance Costs

Maintenance costs are expected to be low in developing countries. The difference between gasoline Euro 4 and Euro 6 passenger vehicles, for example, is limited to in-engine modifications that require no additional vehicle owner intervention and result in near zero additional maintenance costs.

Euro 6 diesel passenger vehicles differ from Euro 4 vehicles in the adoption of after-treatment emissions control systems which may require more active maintenance. Diesel particulate filter (DPF) systems on passenger vehicles require cleaning every 1,000 hours of vehicle operation; the costs are primarily labor-related. For a vehicle that operates 500 hours per year, DPF cleaning needs to happen every two years. In the United States, labor costs for DPF cleaning can reach $200. In comparison, labor costs in China were approximately four percent of U.S. labor costs in 2008,[9] which would lead to an estimate of $8 per cleaning or an annualized cost of $4 per year for DPF maintenance.

Heavy-duty commercial vehicles will face not only higher up-front costs from more stringent emissions standards but also higher maintenance costs. For example, a heavy-duty vehicle that is operated more frequently would require more regular DPF cleaning.

3.6.2 Supply of Cleaner Fuels

Investments in refineries to produce lower-sulfur fuels can present difficult decisions for policymakers and refinery operators. With regard to fuel supply, decision makers must evaluate the capacity of domestic and/or exporting refineries to supply fuels in line with existing and potential demand.

In the United States, the European Union, and Japan, it took 4-6 years to achieve the refinery upgrades needed to create the capacity to supply low sulfur 50 ppm fuel nationwide or continent-wide (Boyle et al. 2008).

To spur adoption of cleaner fuels and more stringent diesel emission standards, some countries could begin implementing tighter standards in sub-national regions (i.e., major metropolitan areas where pollution levels are particularly high). Refineries could then use this time to increase their production capacities to a national scale. This approach has been successful in Brazil, China, India, and the United States.

[9] http://www.bls.gov/ilc/china.htm#data_comparability.

Coordination is a simple and powerful mechanism for achieving rapid black carbon reductions, and the promulgation of emissions standards timed to coincide with the availability of low-sulfur fuel is absolutely critical. If fuel and emissions policies are not appropriately timed, countries will not be able to adequately reduce diesel black carbon emissions significantly.

3.6.3 Complementary Policies

Inspection and maintenance can identify high emitters and speed the retrofit or replacement of engines in line with the most stringent emissions standards. Other complementary policies include incentives for early scrappage and retrofitting. Vehicle labeling and restrictions, meanwhile, can motivate investments in cleaner engines. Forward-looking policies may encompass transportation planning and demand management to improve traffic flow and reduce both fuel consumption and black carbon emissions.

Developing countries face a variety of barriers to implementing diesel emissions control policies. Learning from the experiences of OECD countries can be a valuable tool in helping them build successful programs. Environmental agencies and experts from OECD countries have provided voluntary guidance to regulators in developing countries around the world (e.g., U.S. EPA, EMBARQ). In addition, coordinated training programs and travel exchanges can facilitate capacity building to ensure that lessons are learned and applied where they are needed. These steps can aid in the more rapid reduction of diesel black carbon emissions in developing countries.

4 Diesel Black Carbon Cost-Benefit Analysis

4.1 Introduction

In order to account for the climate impacts of black carbon from diesel transportation, a cost-benefit modeling framework was applied to simulated diesel emissions control projects. This helped to illustrate how the inclusion of black carbon may alter the evaluation of net benefits. The results presented here are intended to demonstrate the applicability of the modeling framework for a range of expected results–both health and climate–across a variety of project types and regions. The specific results presented in this report are not intended to provide recommendations to policymakers or decision makers, although the modeling framework could in the future be applied to real-world case studies.

A cost-benefit modeling framework was constructed and applied to four simulated projects (see Appendix C). Since the aim is to compare diverse projects at multiple scales in a variety of geographic locations, the cost-benefit modeling framework was devised to be general, flexible, and rapidly applicable given appropriate data. The modeling framework can evaluate a range of projects that involve technical interventions, including fuel switching, emissions control technologies, and emissions standards. It was not evaluated for projects that involve elements of behavioral dynamics, such as mode shift projects.

Both health[10] and climate impacts are monetized and compared against up-front and operational costs, assuming a start year of 2013 and an end year of 2035. An estimate of the net present value of each project is based on a 7-percent annual discount rate. An estimate of the internal rate of return for each project is also provided.

The health impacts quantified include annual premature mortality associated with cardiopulmonary disease and lung cancer among adults 30 years of age and older and acute lower respiratory infections in children under five years of age. Morbidities are not included here. Estimates of the value of statistical life are derived using a benefits transfer approach and based on a willingness-to-pay study (Viscusi 2004) conducted in the United States[11] that provides a central estimate of $6.7 million, adjusted to 2010 dollars. Annual premature deaths are distributed using a mortality lag function.

Climate impacts are valued based on an estimate of the social cost of carbon applied to net carbon-dioxide-equivalent emissions. Carbon dioxide, methane, black carbon, and organic carbon are converted to an annual carbon-dioxide-equivalent estimate using global warming potential values on 20-year and 100-year time horizons. Respectively, values for methane are 72 and 25 (Forster et al. 2007); values for organic carbon are -154 and -42 (Bond et al. 2011); and values for black carbon are 3,200 and 900 (Bond et al. 2013). Values for the social cost of carbon are $23 and $50 in 2012 dollars (Tol 2009) and represent three percent and one percent rate of time preferences, respectively. A 2.5-percent annual rise in carbon cost is applied (Yohe et al. 2007).

4.2 Projects Selected

The goal of project simulation was to identify a representative spectrum of potential project types within regions where diesel control projects are feasible The projects simulated and analyzed demonstrate the application of methods developed for this study. A logical follow-on would be to apply the tools developed for this study to a series of concrete, real-world case studies in order to rank projects by the net benefits they may produce.

[10] In this study, BC health impacts are considered equivalent to PM health impacts, in line with the mainstream health impact assessment approaches of both the WHO and the U.S. EPA.

[11] It is acknowledged that using a study from the US has limitations for interpreting the results for developing countries.

The following project simulations were chosen for cost-benefit analysis. The first three can be characterized as direct fleet interventions , in which a specific type of vehicle is targeted– bus replacement, diesel particulate filter retrofit, and green freight retrofit . The fourth project is fleet-wide in scope, looking broadly at the potential for programs affecting all vehicles in a given region – fuel quality and tailpipe emission standards.

Compressed Natural Gas *Bus Replacement* Program in Cebu, Philippines

Cebu is the second-largest city in the Philippines. The city is actively researching upgrades to its public transportation systems, including a new bus rapid transit (BRT) system. Consequently, the replacement of 300 Euro II non-BRT buses with new compressed natural gas (CNG) buses was evaluated.

Diesel Particulate Filter Retrofit Program in Istanbul, Turkey

Istanbul has experimented with diesel particulate filter (DPF) retrofitting pilot projects, but to date there has been no widespread adoption. Past experience means that data should be available on both the applicability to the current fleet and the challenges to implementation. Additional successful pilots combined with careful analysis could push wider-scale adoption. For this analysis, the benefits and costs of retrofitting an additional 300 Euro III buses with DPFs were evaluated.

Green Freight Retrofit Program in Sao Paulo, Brazil

In Brazil, trucks carry the largest share by volume of freight, and there is strong interest at the local and national levels in reducing emissions through retrofit programs. The World Bank has estimated that a basic retrofitting package aimed at reductions in fuel consumption may be appropriate for at least 250,000 trucks.[12] For this analysis, the impact of retrofitting 10,000 of these trucks with a combination of fuel efficiency improvement technologies and diesel particulate filters was evaluated.

Stringent *Fuel Quality and Tailpipe Emission Standards* in Jakarta, Indonesia

Indonesia currently has no refineries capable of producing ultra-low-sulfur diesel fuel, and has lagged behind in establishing stringent fuel quality regulations. With a diesel vehicle population of around seven million, refinery upgrades to reduce fuel sulfur levels could have a dramatic impact on total black carbon emissions in the country. For this analysis, the potential impact of refinery upgrades in order to supply ultra-low-sulfur diesel to Jakarta and enable the introduction of world-class vehicle tailpipe emissions standards for all vehicle classes were evaluated.

4.3 Results

Primary results for the project simulations are presented in this section. These results are based on assumptions for hypothetical project simulations. All projects were evaluated for costs and benefits for the period 2013-2035. Particular sensitivities are introduced and explored below.

4.3.1 Compressed Natural Gas Bus Replacement Program in Cebu, Philippines

In Cebu, a simulated scenario was modeled in which 300 existing Euro II diesel buses were replaced with Euro VI-equivalent CNG buses. All other operating parameters for the buses were assumed to remain identical. The buses were assumed to enter the fleet at a rate of 50 per year from 2013-2018. Each bus was assumed to operate for 20 years and then be retired. The analysis assumed compressed natural gas bus purchase costs of $172,500-207,000, with a cost for new CNG fueling stations of $5,600,000-6,720,000. Total annual operational costs, such as local fuel purchases and vehicle maintenance, were included and estimated to be $0.60-0.90 per km. The benefits included

[12] From a World Bank final draft report called "Mainstreaming Green Freight Trucks in Brazil" available online at http://siteresources.worldbank.org/BRAZILINPOREXTN/Resources/3817166-1323121030855/Green_Freight.pdf?resourceurlname=Green_Freight.pdf.

the monetized value of premature deaths avoided based on value of statistical life (VSL) estimates of \$643,800 (2013) and \$1,557,700 (2035). Other benefits included the value of avoided carbon-dioxide-equivalent emissions using a social cost of carbon of \$23-50 (2012). Other health benefits, such as reduced morbidities, were not included. See Appendix D for a more detailed description of the inputs.

The CNG buses have dramatically lower PM and BC emissions than the Euro II diesel buses they replace. Euro II diesel buses have no after-treatment control technology and representative PM emissions of 0.3 g/km or higher. The CNG buses, on the other hand, emit at Euro VI levels (i.e., 10 times lower levels). The cumulative reduction in black carbon through the year 2035 is estimated to be three metric tons. Furthermore, the CNG buses are modeled with tailpipe CO_2 emissions levels approximately 20 percent lower than equivalent diesel buses, yielding significant benefits from CO_2 emissions cuts. The primary pollutant for which the CNG buses yield poorer performance than the diesel buses is methane. As described in Appendix C, methane leakage from the supply and delivery of natural gas for the buses is modeled at 2 g/km. This results in a net increase in methane for the intervention scenario, which offsets the climate impacts of BC and CO_2 reductions.

Air quality modeling conducted for this project estimates a baseline annual average $PM_{2.5}$ concentration of 7.1 $\mu g/m^3$ in 2013, an intake fraction of 11.6 ppm, and a transportation attributable fraction of 11 percent to annual average $PM_{2.5}$ concentrations. By 2035, ambient $PM_{2.5}$ concentrations are projected to remain relatively flat at 7.2 $\mu g/m^3$, while the intake fraction will grow to 15.0 ppm and the transportation attributable fraction of ambient $PM_{2.5}$ will fall to four percent.

The relatively small scale of the intervention means that the total health benefits (approximately 22 cumulative premature mortalities avoided by 2035) are not large.[13] The overall health benefits from these avoided premature mortalities using the health methodology described in Appendices C and D are valued at \$9.5 million based on cumulative black carbon reductions of 87 metric tons.

Table 4-1 summarizes the range of climate benefits using various assumptions for GWP and social cost of carbon. Total benefits range from \$13.9-30.4 million. The addition of black carbon increases projected total benefits of the project by 12-57 percent, depending on assumptions of rate of time preference for the social cost of carbon and the time horizon for the GWP.

Assuming a 3-percent discount rate for the social cost of carbon, the project would not realize net benefits. Assuming a GWP20 for black carbon, the project would realize approximately \$7 million only if a 1-percent discount rate is assumed for the social cost of carbon. These results show that this project is sensitive to assumptions of discount rate and time horizon, and that bold assumptions for both would be necessary to find net benefits.

[13] One key assumption that bears on this result is that Euro 3/III standards are adopted nationwide in 2010 and that every decade there after the next stage in the European emissions standards is adopted (Euro 4/IV in 2020 and Euro 5/V in 2030). This results in an overall decline in baseline emissions and health impacts from transportation activity by 30 percent in the year 2025 from year 2010 levels. In the absence of this assumption, the baseline levels would be higher and the benefit of the CNG bus replacement would appear larger.

Table 4-1: Simulated Bus Fleet Intervention in Cebu (2013-35).

2035 Snapshot				
Global Warming Potential	GWP20		GWP100	
$/$CO_2$e discount rate	1%	3%	1%	3%
Cumulative mortalities avoided	22	22	22	22
Cumulative health benefits	$9,512,000	$9,512,000	$9,512,000	$9,512,000
Cumulative climate benefits	$20,908,000	$7,513,000	$12,330,000	$4,431,000
Total benefits	$30,420,000	$17,025,000	$21,842,000	$13,943,000
Total costs	$23,327,000	$23,327,000	$23,327,000	$23,327,000
Net present value	$7,093,000	($6,302,000)	($1,485,000)	($9,384,000)
Benefit-to-Cost ratio	1.3	0.7	0.9	0.6
Internal Rate of Return (IRR)	6%	4%	4%	3%
BC benefits/climate benefit	82%	82%	39%	39%
BC benefits/total benefits	57%	36%	22%	12%
$ spent per mortality avoided	$1,074,000	$1,074,000	$1,074,000	$1,074,000
Cumulative tons BC reduced	87	87	87	87
Cumulative tons CO_2e reduced	338,450	338,450	199,584	199,584
$ spent per metric ton CO_2e reduced	$69	$69	$117	$117

4.3.2 Diesel Particulate Filter Retrofit Program in Istanbul, Turkey

In the Istanbul simulation, 300 Euro III buses were retrofitted with diesel particulate filters and the deployment of the clean fleet followed the same timeline as in Cebu. The analysis assumed diesel bus retrofit costs at $5,000-8,000 per vehicle. Since Turkey now meets Euro V equivalent standards, the analysis assumed 50 ppm low sulfur fuel is already available in the baseline and does not assume additional costs for refining. No other capital costs were assumed.

Total annual operational costs, including local fuel purchases and vehicle maintenance, were included and estimated at $0.83-1.06 per km. This includes a small sacrifice in fuel economy that added to the fuel costs. The benefits include the monetized value of premature deaths avoided based on VSL estimates of $2,619,481 (2013) to $12,481,420 (2035). Other benefits include the value of avoided carbon-dioxide-equivalent emissions using a social cost of carbon between $23-50 (2012). Other health benefits, such as reduced morbidities, were not included. See Appendix D for a more detailed description of the inputs.

Air quality modeling for this project estimates a baseline annual average $PM_{2.5}$ concentration of 17.6 $\mu g/m^3$ in 2013, an intake fraction of 72 ppm, and a transportation contribution of 11 percent to annual average $PM_{2.5}$ concentrations. By 2035, total ambient $PM_{2.5}$ concentrations are projected to grow to 19.4 $\mu g/m^3$, while the intake fraction will grow to 107 ppm and the transportation attributable fraction of ambient $PM_{2.5}$ will grow to 13 percent.

The total black carbon cut as a result of the intervention (a cumulative 94 metric tons through 2035) is lower than in Cebu for several reasons. Notably, in the Cebu simulation, Euro II buses were replaced (versus cleaner Euro III buses in Istanbul). Fewer reductions are also likely because Turkey's base fleet is expected to improve more rapidly than the base fleet in the Philippines. The total climate benefits of the project are sensitive to the GWP time horizon, since under a GWP100 this project would realize climate costs while under a GWP20 it would realize climate benefits. Since the project results in net positive CO_2e emissions under GWP100 accounting, the cost per ton reduced is negative ($-65) and reflects the amount of spending that should be avoided to reduce one additional ton of emissions. Under GWP20 accounting, black carbon emissions reductions are weighted more heavily and the project results in a net reduction in CO_2 emissions at a cost of $31 per ton.

Regarding health impacts, the valuation in Turkey is actually fairly large: $129 million by 2035 for 47 avoided premature deaths. Istanbul was modeled with a relatively high urban population exposure to vehicle emissions (i.e., high intake fraction, or the fraction of emissions emitted that are inhaled by the population). In addition, Turkey's gross national income is the highest of any of the modeled regions, yielding the highest VSL estimates of all the projects analyzed.

Black carbon reductions are this project's only source of climate benefits. Additional co-pollutants were considered, but changes in these emissions did not produce climate benefits. The benefit of black carbon reductions is offset slightly by increases in fuel consumption, which leads to a climate penalty when estimating total benefits using a GWP100 value for all pollutants. See Table 4-2 for the full range of climate benefits.

The potential net benefits of this project range from $120.4-133.5 million for a benefit-to-cost ratio of between 28.3 and 31.2. The addition of black carbon to the project calculus would increase total benefits between 1-13 percent. A discount rate of three percent for the social cost of carbon would keep the additional benefit provided by black carbon below five percent. A decision to proceed with this project based on its net benefits does not appear sensitive to assumptions of GWP time horizon and the social cost of carbon discount rate.

Table 4-2: Simulated Diesel Retrofit in Istanbul (2013-35)

2035 Snapshot				
Global Warming Potential	GWP20		GWP100	
$/$CO_2$e discount rate	1%	3%	1%	3%
Cumulative mortalities avoided	47	47	47	47
Total health benefits	$129,022,000	$129,022,000	$129,022,000	$129,022,000
Total climate benefits	$8,908,000	$3,201,000	($4,199,000)	($1,509,000)
Total benefits	$137,930,000	$132,223,000	$124,823,000	$127,513,000
Total costs	$4,415,000	$4,415,000	$4,415,000	$4,415,000
Net present value	$133,515,000	$127,808,000	$120,408,000	$123,098,000
Benefit-to-Cost ratio	31.2	29.9	28.3	28.9
Internal Rate of Return (IRR)	781%	710%	590%	622%
BC benefits/climate benefits	209%	209%	125%	125%
BC benefits/total benefits	13%	5%	4%	1%
$ spent per mortality avoided	$94,000	$94,000	$94,000	$94,000
Cumulative tons BC reduced	94	94	94	94
Cumulative tons CO_2e reduced	144,191	144,191	(67,964)	(67,964)
$ spent per metric ton CO_2e reduced	$31	$31	($65)	($65)

4.3.3 Green Freight Retrofit Program in Sao Paulo, Brazil

A green freight project was modeled as a simulated retrofitting of 10,000 Euro III heavy-duty trucks each traveling 100,000 km per year in Sao Paulo to reduce fuel consumption by five percent. Retrofits to reduce fuel consumption can include a range of strategies, such as modest improvements to engine systems, reductions in drivetrain losses, reductions in aerodynamic drag, reductions in tire rolling resistance, and reductions in parasitic losses from auxiliary equipment. A diesel particulate filter was assumed to be included as a necessary component of the overall retrofit package. This analysis assumed a total retrofit cost for each Euro III heavy-duty truck of $10,000-12,000, which includes an upgrade to reduce fuel consumption and a retrofit to install a diesel particulate filter. Additional costs to supply low sulfur fuel were not accounted for because the necessary fuel is already provided in metropolitan areas throughout Brazil. This is in line with national policy to enable compliance with Euro V standards in metropolitan areas nationwide. No other capital costs were assumed.

Total annual operational costs, such as local fuel purchases and vehicle maintenance, were included. Baseline operating costs before the green freight retrofit were assumed to be $0.82-1.05 per km. After the retrofit to reduce fuel consumption and accounting for an offsetting fuel consumption increase for the diesel particulate filter, this cost falls to $0.80-1.02 per km. The benefits calculated included the monetized value of premature deaths avoided based on VSL estimates of $2,657,789 (2013) to $5,258,101 (2035). Other benefits included the value of avoided carbon-dioxide-equivalent emissions using a social cost of carbon of $23-50 (2012). Other health benefits, such as reduced morbidities, were not included. See Appendix D for a more detailed description of the inputs.

The fuel savings realized by the green freight retrofit in this simulation are large enough to pay for the cost of the diesel particulate filter installation and provide additional savings. Although the DPF offsets some of the fuel consumption improvement and adds to fuel costs, the net result is still significant long-term cost savings.

Air quality modeling for this project estimates a baseline annual average $PM_{2.5}$ concentration of 6.8 $\mu g/m^3$ in 2013, an intake fraction of 20.2 ppm, and a transportation attributable contribution of 35 percent to annual average $PM_{2.5}$ concentrations. By 2035, total ambient $PM_{2.5}$ concentrations are projected to remain relatively stable at 6.5 $\mu g/m^3$, while the intake fraction will grow to 29.7 ppm and the transportation attributable fraction of ambient $PM_{2.5}$ will fall to 21 percent.

Owing to the enormous size of the assumed retrofitted fleet, cumulative black carbon reductions, at 2,948 metric tons, are also significant. Total climate benefits for all the CO_2e reduced range from $263 million to $1.14 billion. The addition of black carbon to the project results in an increase of total benefits between 4-25 percent, depending on the assumptions for GWP time horizon and the discount rate for the social cost of carbon.

As for health benefits, the reduction of this many tons of diesel emissions in dense Sao Paulo avoids 735 premature deaths at a total valuation of $1.19 billion. More results are shown in Table 4-3.

The net present value of this project ranges from $1.57-2.44 billion. Since the project results in net savings from reductions in fuel consumption, negative costs produce a negative benefit-to-cost ratio.

In this project, the net present value under all assumptions for GWP time horizon and the discount rate for the social cost of carbon are strongly positive. The significant savings to vehicle operators in the form of reduced fuel consumption suggests this project model may provide a creative form of financing for diesel black carbon controls.

Table 4-3: Simulated Green Freight Retrofit in Sao Paulo (2013–35)

2035 Snapshot				
Global Warming Potential	GWP20		GWP100	
$/$CO_2$e discount rate	1%	3%	1%	3%
Cumulative mortalities avoided	735	735	735	735
Total health benefits	$1,185,598,000	$1,185,598,000	$1,185,598,000	$1,185,598,000
Total climate benefits	$1,137,917,000	$408,895,000	$731,928,000	$263,009,000
Total benefits	$2,323,515,000	$1,594,493,000	$1,917,526,000	$1,448,607,000
Total costs	($120,035,000)	($120,035,000)	($120,035,000)	($120,035,000)
Net present value	$2,443,550,000	$1,714,528,000	$2,037,561,000	$1,568,642,000
Benefit-to-Cost ratio	(19.4)	(13.3)	(16.0)	(12.1)
Internal Rate of Return (IRR)	589%	420%	477%	369%
BC benefits/climate benefits	51%	51%	22%	22%
BC benefits/total benefits	25%	13%	9%	4%
$ spent per mortality avoided	($163,000)	($163,000)	($163,000)	($163,000)
Cumulative tons BC reduced	2,948	2,948	2,948	2,948
Cumulative tons CO_2e reduced	18,419,784	18,419,784	11,847,928	11,847,928
$ spent per metric ton CO_2e reduced	($7)	($7)	($10)	($10)

4.3.4 Stringent Fuel Quality and Tailpipe Emissions Standards in Jakarta, Indonesia

Unlike the other scenarios, which are discrete interventions affecting a narrow class of vehicles, the intervention in Jakarta captures a simulated fleet-wide upgrade to vehicles and fuels operating in the entire metropolitan area. In the baseline scenario, it was assumed that sulfur levels in Jakarta's gasoline and diesel fuel reach and remain at 150 ppm and 350 ppm, respectively, and that the tailpipe emissions standard for all vehicle classes never progresses beyond Euro 3/III. The intervention scenario was intentionally chosen as an aggressive one to highlight the magnitude of emissions reductions that might be achieved through progressively more stringent fuel and vehicle standards. In the intervention scenario, it was assumed that Jakarta's gasoline and diesel reach 50 ppm sulfur by 2015 and 10 ppm by 2020 in order to enable progressively more stringent tailpipe standards for all vehicles. It was further assumed that Jakarta introduces Euro 4/IV, 5/V, and 6/VI in 2015, 2020, and 2025, respectively. A consistent population of Euro 0 vehicles represents the gross emitters in the fleet. (In this example no high-emitter reduction program was assumed.)

In this project simulation, the additional costs associated with the intervention are modeled for refineries (increased capital and operating costs associated with fuel desulfurization) as well as vehicles (additional technology costs associated with meeting more stringent emissions standards). The total costs for implementing this project scenario are approximately $19.4 billion through till 2035.

This analysis assumes capital costs to upgrade refining capabilities to produce ultra-low-sulfur diesel fuel by 2020. The incremental cost per liter to produce 50 ppm (down from 350 ppm) sulfur diesel fuel is assumed to be $0.03-0.06. The incremental cost per liter to produce 10 ppm sulfur diesel fuel is assumed to be $0.03-0.07.

Benefits calculated included the monetized value of premature deaths avoided based on VSL estimates of $637,471 (2013) to $1,261,155 (2035). Other benefits included the value of avoided carbon-dioxide-equivalent emissions using a social cost of carbon of $23-50 (2012). Other health benefits, such as reduced morbidities, were not included. See Appendix D for a more detailed description of the inputs.

Air quality modeling for this project estimates a baseline annual average $PM_{2.5}$ concentration of 32.9 μg/m^3 in 2013, an intake fraction of 76.3 ppm (based on a regional average of cities), and a transportation attributable contribution of 77 percent to annual average $PM_{2.5}$ concentrations. By 2035, total ambient $PM_{2.5}$ concentrations are projected to grow upto 37.5 μg/m^3, while the intake fraction will grow to 98.7 ppm and the transportation attributable fraction of ambient $PM_{2.5}$ will stay nearly flat at 76 percent.

The introduction of increasingly stringent tailpipe emissions standards upto Euro 6/VI results in a sharp decline in total emissions, including emissions of $PM_{2.5}$ and black carbon. By 2035, the intervention is expected to result in a cumulative reduction of 213,000 metric tons of BC. Total BC emissions rise even in the intervention scenario after 2035 as a consequence of the growing vehicle population.

Jakarta's densely populated urban area has the highest exposure to vehicle emissions (intake fraction) of any of the cities modeled for this exercise. The intake fraction of 134 ppm in 2012 grows consistently to nearly 200 ppm by 2050. This high intake fraction combined with sweeping emissions reductions yields an estimate of more than 43,557 avoided premature deaths by 2035. Despite Indonesia's relatively low gross national income, the total health benefits from these avoided premature deaths are still significant and are estimated to be in excess of $15.9 billion. The full results are shown in Table 4-4.

The addition of black carbon to the project assessment increases total benefits 21-75 percent, depending on assumptions for GWP time horizon and the discount rate for the social cost of carbon. Black carbon accounts for all the climate benefits realized from this intervention. Under all assumptions, the project provides a greater share of benefits relative to costs, as well as net benefits between $0.7-39 billion. With consideration of health impacts alone, the project's costs outweigh the benefits; however, the additional consideration of the climate impacts of black carbon causes the benefits to outweigh the costs and the project to appear more justifiable.

Table 4-4: Simulated Fleet-wide Desulfurization and Emissions Standards in Jakarta (2013–35)

2035 Snapshot				
Global Warming Potential	GWP20		GWP100	
$/$CO_2$e discount rate	1%	3%	1%	3%
Cumulative mortalities avoided	43,557	43,557	43,557	43,557
Total health benefits	$15,967,869,000	$15,967,869,000	$15,967,869,000	$15,967,869,000
Total climate benefits	$42,317,786,000	$14,515,182,000	$11,912,785,000	$4,086,172,000
Total benefits	$58,285,655,000	$30,483,051,000	$27,880,654,000	$20,054,041,000
Total costs	$19,393,220,000	$19,393,220,000	$19,393,220,000	$19,393,220,000
Net present value	$38,892,435,000	$11,089,831,000	$8,487,434,000	$660,821,000
Benefit-to-Cost ratio	3.0	1.6	1.4	1.0
Internal Rate of Return (IRR)	26%	13%	9%	6%
BC benefits/climate benefits	103%	103%	103%	103%
BC benefits/total benefits	75%	49%	44%	21%
$ spent per mortality avoided	$445,000	$445,000	$445,000	$445,000
Cumulative tons BC reduced	213,028	213,028	213,028	213,028
Cumulative tons CO_2e reduced	661,540,520	661,540,520	186,229,994	186,229,994
$ spent per metric ton CO_2e reduced	$29	$29	$104	$104

4.4 Summary

A cost-benefit modeling framework for diesel black carbon emissions control projects was developed and applied to simulated interventions in Cebu, Philippines; Istanbul, Turkey; Sao Paulo, Brazil: and Jakarta, Indonesia. Respectively, these represent a targeted fleet intervention to replace 300 diesel buses with buses powered by compressed natural gas; a targeted retrofit of 300 diesel vehicles to install diesel particulate filters; a green freight retrofit to reduce fuel consumption and install diesel particulate filters in 10,000 heavy-duty vehicles; and a city-wide desulfurization of vehicle fuels and an upgrade of emissions controls in new vehicles. All four projects were assumed to begin in 2013 and the costs and benefits were evaluated through 2035.

This assessment of simulated diesel black carbon interventions shows that health benefits alone are enough to justify the Istanbul and Sao Paulo interventions. The addition of climate benefits, particularly reductions in black carbon emissions, provides additional weight to the justification for each project. Table 4-5 shows how the benefit-to-cost ratio changes with the addition of black carbon into the project analysis.

The Jakarta simulation resulted in an estimate of net benefits that were sensitive to the inclusion of black carbon. In the absence of black carbon, the project would not be worthwhile on the basis of health benefits alone. In the Cebu case, even the incorporation of black carbon climate benefits under most assumptions would fail to justify this project on economic grounds. However, bold assumptions for the GWP time horizon and the discount rate for the social cost of carbon, a GWP of 3200 and one percent, respectively, did produce net benefits and would provide some justification for pursuing the project in Cebu.

Table 4-5: Change in Cost-Benefit Ratio of Simulated Projects after Accounting for Black Carbon

Global Warming Potential	GWP20		GWP100		
Social Cost of Carbon Discount Rate	1%	3%	1%	3%	
Jakarta, Indonesia, Fuel and Vehicle Standards	0.8	0.8	0.8	0.8	Before BC
	3.0	1.6	1.4	1.0	After BC
Istanbul, Turkey, Diesel Retrofit	27.0	28.4	27.1	28.5	
	31.2	29.9	28.3	28.9	
Cebu, Philippines, CNG Bus Replacement	0.6	0.5	0.7	0.5	
	1.3	0.7	0.9	0.6	
Sao Paulo, Brazil, Green Freight + Retrofit Project*	0.0	0.0	0.0	0.0	
	0.0	0.0	0.0	0.0	

* Project incurred zero net costs.

Given the uncertainties of the input data, these results are purely illustrative and demonstrate the potential change in net benefits to diesel emissions control projects when black carbon is included in the cost-benefit analysis. Results should not be interpreted as directly comparable or as necessarily indicative of results from similar active projects. Future work may better characterize such projects through the collection of data to represent all the characteristics of each intervention. Further investigation of the ranges of benefits and costs for similar projects in different regions (or different projects in the same region) is an important area for continued research.

5 Conclusions and Next Steps

It has been widely recognized for some time that fine PM emissions from diesel engines are harmful for human health. Emerging concerns regarding health impacts due to a changing climate system has brought attention to the climate impacts of PM emissions, particularly the light-absorbing black carbon fraction. The impact of BC on climate is only now being better understood with many questions still unanswered (Bond et al, 2013). However, it is accepted that reducing emissions from diesel engines provides an opportunity to deliver 'co-benefits' for climate and health.

A number of countries, particularly within the OECD, have adopted policies to reduce diesel emissions to protect public health and these actions have also reduced black carbon and delivered climate co-benefits. With strong growth in diesel vehicle population globally, driven by fast rising demand in developing countries like India and China, it is important to share the experiences, lessons, and approaches for reducing diesel emissions from OECD countries.

This report summarizes a wide range of black carbon emissions control strategies in OECD countries in North America, East Asia, and Europe -- from technology (e.g., diesel particulate filters and refinery upgrades) to policy measures (e.g., vehicle emission standards, fuel quality standards, scrappage programs, and tax incentives).These strategies provide a menu of options that could be considered by developing countries and tailored to the national context. A key characteristic (and lesson) of the programs implemented in the OECD is the value of national policy road maps for cleaner vehicles and fuels that provide a clear and predictable course for technology and fuel adoption by private and public stakeholders. That being said, more work is needed to demonstrate the effectiveness of these strategies in decoupling growth in diesel consumption from BC emissions.

This report looks at how to integrate black carbon emission reduction considerations in cost-benefit assessment and applies an analytic framework to four simulated projects to illustrate the associated opportunities and challenges at a project level. It shows the economic analysis is very sensitive assumptions about the GWP of BC and associated time horizon (20 and 100 years) as well as the discount rate. While capturing health and climate benefits can be very useful in such analysis, going forward additional work is needed to refine estimates of the social cost of black carbon and narrow assumptions on the discount rate and GWP time horizon.

The focus on BC as a short lived climate pollutant is fast growing. Looking ahead, global partnerships like the Climate and Clean Air Coalition[14] could support the exchange of knowledge and experience and help stakeholders better understand scientific uncertainties and the analytic, policy and technology options to address BC emissions.

[14] www.unep.org/ccac.

6 Bibliography

Beijing Environmental Protection Bureau (2011). *Municipal Government Issues New Policy Encouraging Scrappage of Older Vehicles* (in Chinese). Retrieved December 8, 2012, from http://www.bjepb.gov.cn/portal0/tab189/info6306.htm.

Bell, M. (2012). Assessment of the Health Impacts of Particulate Matter Characteristics (HEI Research Report No. 161). Health Effects Institute, Boston.

Bond, T. C., and Sun, H. (2005). Can Reducing Black Carbon Emissions Counteract Global Warming? *Environmental Science & Technology* 39 (16): 5921–26. doi:10.1021/es0480421.

Bond, T.C., Bhardwaj, E., Dong, R., Jogani, R., Jung, S., Roden, C., Streets, D. G., Fernandes, S., and Trautmann, N. (2007). Historical Emissions of Black and Organic Carbon Aerosol from Energy-related Combustion, 1850-2000. *Global Biogeochemical Cycles* 21 (2): GB2018. doi:10.1029/2006GB002840, with new emissions factors developed in collaboration with C. Liousse.

Bond, T. C., Zarzycki, C., Flanner, M. G., and Koch, D. M. (2011). Quantifying Immediate Radiative Forcing by Black Carbon and Organic Matter with the Specific Forcing Pulse. *Atmospheric Chemistry and Physics* 11 (4): 1505–25. doi:10.5194/acp-11-1505-2011.

Bond, T. C., Doherty, S. J., Fahey, D. W., Forster, P. M., Berntsen, T. K., DeAngelo, B. J., et al. (2013). Bounding the Role of Black Carbon in the Climate System: A Scientific Assessment. *Journal of Geophysical Research-Atmospheres*. doi:10.1002/jgrd.50171.

Boyle, G., Courtis, J., Huizenga, C., Rogers, J., Walsh, M., 2008. A Road Map for Cleaner Fuels and Vehicles in Asia (No. TA 6144 (REG)). Asian Development Bank.

Brook, R. D., Rajagopalan, S., Pope, C. A., III, Brook, J. R., Bhatnagar, A., Diez-Roux, A. V., et al. (2010). Particulate Matter Air Pollution and Cardiovascular Disease: An Update to the Scientific Statement from the American Heart Association. *Circulation*, *121*(21), 2331–2378. doi:10.1161/CIR.0b013e3181dbece1.

Buhaug, Ø., Corbett, J., Endresen, Ø., and Eyring, V. (2009). Prevention of Air Pollution from Ships. International Maritime Organization.

Cai, F., and Wang, Z. (2004). The Effects of Overloading on Fuel Consumption and Emission of Motor Vehicle. *Journal of Highway and Transportation Research and Development* 9. Retrieved from http://en.cnki.com.cn/Article_en/CJFDTOTAL-GLJK200409034.htm.

Chambliss, S., Miller, J., Facanha, C., Minjares, R., & Blumberg, K. (2013). The Impact of Stringent Fuel and Vehicle Standards on Premature Mortality and Emissions (pp. 1–96). Washington, DC: International Council on Clean Transportation. http://www.theicct.org/global-health-roadmap

Chatterjee, S., Walker, A., and Blakeman, P. (2008). Emission Control Options to Achieve Euro IV and Euro V on Heavy-Duty Diesel Engines. Society of Automotive Engineers.

Cheng, J. (2010). The Smog Squad of Hong Kong Has Truckers on the Run. *The Wall Street Journal* online edition. June 4.]; Retrieved from http://online.wsj.com/article/SB10001424052748704025304575284460465232720.html.

Chow, J., Watson, J., Lowenthal, D., Chen, L.-W., and Motallebi, N. (2010). Black and Organic Carbon Emission Inventories: Review and Application to California. *Journal of the Air and Waste Management Association* 60 (4): 497–507.

Eyring, V., Isaksen, I. S. A., Berntsen, T., Collins, W. J., Corbett, J. J., Endresen, O., Grainger, R. G., Moldanova, J., Schlager, H., and Stevenson, D. S. (2010). Transport Impacts on Atmosphere and Climate: Shipping. *Atmospheric Environment* 44 (37): 4735–71. doi:10.1016/j.atmosenv.2009.04.059.

Façanha, C., Blumberg, K., and Miller, J. (2012). Global Transportation Climate and Energy Roadmap. International Council on Clean Transportation. Retrieved from http://www.theicct.org/global-transportation-energy-and-climate-roadmap.

Forster, E. E., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R. A., Fahey, D. W., Haywood, J., Lean, J., Lowe, J. A., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schultz, M. G., and Van Dorland, R. (2007). Changes in

Atmospheric Constituents and in Radiative Forcing. In Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, U.K.: Cambridge University Press, p. 6.

Franklin, M., Koutrakis, P., and Schwartz, J. (2008). The Role of Particle Composition on the Association between PM2.5 and Mortality. *Epidemiology* 19 (5): 680–89.

Gallagher, K. S., and Oliver, H. H. (2005). Providing Low-Sulfur Fuels for Transportation Use: Policy Options and Financing Strategies in the Chinese Context. Conference Paper for the Kennedy School of Government, Harvard University.

Gaurav, B., Francisco, P., Fernandez, R., and Wagner, D. (2014). Best Practices in Reducing Emissions through Vehicle Replacement Programs. Washington, DC: International Council on Clean Transportation. In Press.

Han, X., & Naeher, L. P. (2006). A Review of Traffic-related Air Pollution Exposure Assessment Studies in the Developing World. *Environment International*, *32*(1), 106–120.

Hausker, K. (2004). Vehicle Inspection and Maintenance Programs: International Experience and Best Practices. Washington, D.C.: U.S. Agency for International Development.

Health Effects Institute (2010). Traffic-Related Air Pollution: A Critical Review of the Literature on Emissions, Exposure, and Health Effects. Traffic-Related Air Pollution: A Critical Review of the Literature on Emissions, Exposure, and Health Effects (HEI Special Report No. 17). Health Effects Institute, Boston, MA.

HM Customs and Excise, 2000. Using the Tax System to Encourage Cleaner Fuels: The Experience of Ultra-low Sulfur Diesel.

Hung, W., 2004. Taxation on Vehicle Fuels: Its Impacts on Switching to Cleaner Fuels. Energy Policy 34 (2006) 2566-2571.

International Agency for Research on Cancer. (2012). IARC: Diesel Engine Exhaust Carcinogenic. Lyon, France: World Health Organization. *Press Release, 12 June.*

Jackson, S. C. (2009). Parallel Pursuit of Near-term and Long-term Climate Mitigation. *Science* 326 (5952): 526–27. doi:10.1126/science.1177042.

Jacobs, T., Chatterjee, S., Conway, R., Walker, A., Kramer, J., and Mueller-Haas, K. (2006). Development of Partial Filter Technology for HDD Retrofit (No. 2006-01-0213). Warrendale, PA: SAE International. Retrieved from http://papers.sae.org/2006-01-0213/.

Janssen, N., Gerlofs-Nijland, M., Lanki, T., Salonen, R., Cassee, F., Hoek, G., Fischer, P., Brunekreef, B., and Krzyzanowski, M. (2012). Health Effects of Black Carbon. World Health Organization Regional Office for Europe, Bonn, Germany.

Kennedy, I. M. (2007). The Health Effects of Combustion-Generated Aerosols. *Proceedings of the Combustion Institute* 31 (2): 2757–70.

Kleeman, M., Schauer, J., and Cass, G. (2000). Size and Composition Distribution of Fine Particulate Matter Emitted from Motor Vehicles. *Environmental Science and Technology* 34 (7): 1132–42.

Kondo, Y., Ram, K., Takegawa, N., Sahu, L., Morino, Y., Liu, X., Ohara, T., 2012. Reduction of Black Carbon Aerosols in Tokyo: Comparison of Real-time Observations with Emission Estimates. *Atmospheric Environment* 54, 242ric.

Laden, F., Neas, L. M., Dockery, D. W., and Schwartz, J. (2000). Association of Fine Particulate Matter from Different Sources with Daily Mortality in Six U.S. Cities. *Environmental Health Perspectives* 108 (10): 941–47.

Lamarque, J.-F., Bond, T. C., Eyring, V., Granier, C., Heil, A., Klimont, Z., Lee, D., et al. Historical (1850–2000) Gridded Anthropogenic and Biomass Burning Emissions of Reactive Gases and Aerosols: Methodology and Application. *Atmospheric Chemistry and Physics*, *10*, no. 15 (2010):7017–7039. doi:10.5194/acp-10-7017-2010.

Mauderly, J., Snipes, M., Barr, E., Belinsky, S., Bond, J., Brooks, A., Chang, I., Cheng, Y., Gillett, N., Griffith, W., Henderson, R., Mitchell, C., Nikula, K., Thomassen, D., 1994. Pulmonary Toxicity of Inhaled Diesel Exhaust and Carbon Black in Chronically Exposed Rats (HEI Research Report 68 No. 68). Health Effects Institute, Boston, MA.

Maus, W., and Brück, R. (2007). Exhaust Gas After-treatment Systems for Commercial Vehicles–Technologies and Strategies for the Future. In ICPC Conference, Paper (Vol. 2). Retrieved from http://www.emitec.com/fileadmin/user_upload/Bibliothek/Vortraege/0704017%20AVL%20Powertrain%20Konferenz_final_end_englisch.pdf.

Ministry of Finance et al, (2009). Notice about Issuing of Implementation Methods for Vehicle Scrappage and Replacement (in Chinese). Retrieved September 2, 2012, from http://www.vecc-mep.org.cn/news/news_detail.jsp?newsid=33844.

Ministry of Finance and Ministry of Commerce (2010). Notice about Adjusting Subsidy Standards and Related Matters for Vehicle Scrappage and Replacement (in Chinese). Retrieved September 2, 2012, from http://www.vecc-mep.org.cn/news/news_detail.jsp?newsid=36388.

Northeast States for Coordinated Air Use Management (NESCAUM) (1999). Heavy-Duty Diesel Emission Reduction Project: Retrofit/Rebuild Component. Report prepared by NESCAUM for the U.S. Environmental Protection Agency. March. Retrieved December 11, 2012, from http://www.nescaum.org/documents/exsumhdd.pdf/view.

Olivastri, B. and M. Williamson. 2000. A Review of International Initiatives to Accelerate the Reduction of Sulfur in Diesel Fuel. Prepared for the Oil, Gas and Energy Branch, Air Pollution Prevention Directorate, Environment Canada. http://www.ec.gc.ca/Publications/A8F3253F-7D8B-4491-88F5-ADEDAE40900E/AReviewOfInternationalInDieselFuel.pdf.

Ramana, M. V., Ramanathan, V., Feng, Y., Yoon, S.-C., Kim, S.-W., Carmichael, G. R., and Schauer, J. J. (2010). Warming Influenced by the Ratio of Black Carbon to Sulphate and the Black-Carbon Source. *Nature Geoscience* 3 (8): 542–45.

Rypdal, K., Berntsen, T., Fuglestvedt, J. S., Aunan, K., Torvanger, A., Stordal, F., and Nygaard, L. P. (2005). Tropospheric Ozone and Aerosols in Climate Agreements: Scientific and Political Challenges. *Environmental Science and Policy* 8 (1): 29–43. doi:10.1016/j.envsci.2004.09.003.

Sarofim, M. C. (2012). The GTP of Methane: Modeling Analysis of Temperature Impacts of Methane and Carbon Dioxide Reductions. *Environmental Modeling and Assessment* 17 (3): 231–39. doi:10.1007/s10666-011-9287-x.

Shindell, D., Faluvegi, G., Walsh, M., Anenberg, S. C., Dingenen, R. V., Muller, N. Z., Austin, J., Koch, D., and Milly, G. (2011a). Climate, Health, Agricultural and Economic Impacts of Tighter Vehicle-Emission Standards. *Nature Climate Change* 1 (1): 59–66.

Shindell, D., Ramanathan, V., Raes, F., Cifuentes, L., and Oanh, K. N. T. (2011b). Integrated Assessment of Black Carbon and Tropospheric Ozone. Nairobi, Kenya: UN Environment Programme and World Meteorological Organization. Retrieved from http://www.unep.org/dewa/Portals/67/pdf/Black_Carbon.pdf.

Smith, S. M., Lowe, J. A., Bowerman, N. H. A., Gohar, L. K., Huntingford, C., and Allen, M. R. (2012). Equivalence of Greenhouse-Gas Emissions for Peak Temperature Limits. *Nature Climate Change* 2 (7): 535–38. doi:10.1038/nclimate1496.

Switzerland Federal Department of Finance (2011). *Performance-related Heavy Vehicle Fee: HVF – Overview*. Retrieved September 2, 2012, from http://www.ezv.admin.ch/zollinfo_firmen/steuern_abgaben/00379/index.html?lang=en.

Takahashi, E., Bureau of Environment, Tokyo Metropolitan Government. Presentation at Symposium on Regulations on Diesel Vehicles as a Measure against Climate Change:" The Diesel Vehicle Control in Tokyo". January 17, 2011.

Tol, R. S. J. (2009). The Economic Effects of Climate Change. *Journal of Economic Perspectives*, 23 (2): 29–51.

Uherek, E., Halenka, T., Borken-Kleefeld, J., Balkanski, Y., Berntsen, T., Borrego, C., Gauss, M., Hoor, P., Juda-Rezler, K., Lelieveld, J. Melas, D., Rypdal, K., and Schmid, S. (2010). Transport Impacts on Atmosphere and Climate: Land Transport. *Atmospheric Environment* 44 (37): 4772–4816.

PCFV. (2014). Diesel Fuel Sulphur Levels. *United Nations Environment Programme, Partnership for Clean Fuels and Vehicles*. Nairobi, Kenya. Retrieved March 20, 2014, from http://www.unep.org/transport/pcfv/PDF/Maps_Matrices/world/sulphur/MapWorldSulphur_Feb2014.pdf

U.S. EPA. (2012) Report to Congress on Black Carbon. United States Environmental Protection Agency. EPA-450/R-12-001. March, 2012. http://www.epa.gov/blackcarbon.

Vaaraslahti, K., Ristimäki, J., Virtanen, A., Keskinen, J., Giechaskiel, B., and Solla, A. (2006). Effect of Oxidation Catalysts on Diesel Soot Particles. *Environmental Science and Technology* 40 (15): 4776–81. doi:10.1021/es060615h.

Viscusi, W. K. (2004). The Value of Life: Estimates with Risks by Occupation and Industry. *Economic Inquiry* 42 (1): 29–48.

Vuuren, D. P., Edmonds, J., Kainuma, M., Riahi, K., Thomson, A., Hibbard, K., Hurtt, G. C., Kram, T., Krey V., Lamarque, J.-F., Masui, T., Meinshausen, M., Nakicenovic, N., Smith, S. J., and Rose, S. K. (2011). The Representative Concentration Pathways: An Overview. *Climatic Change* 109 (1–2): 5–31. doi:10.1007/s10584-011-0148-z.

Walker, A. P. (2004). Controlling Particulate Emissions from Diesel Vehicles. *Topics in Catalysis* 28 (1–4): 165–70. doi:10.1023/B:TOCA.0000024346.29600.0e.

Yohe, G.W., R.D. Lasco, Q.K. Ahmad, N.W. Arnell, S.J. Cohen, C. Hope, A.C. Janetos and R.T. Perez, 2007: Perspectives on Climate Change and Sustainability. Climate Change 2007: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden and C.E. Hanson, Eds., Cambridge University Press, Cambridge, UK, 811-841.

7 Appendix A: Diesel Climate Impacts

A.1. Mechanisms for Diesel Black Carbon Climate Impacts

Climate scientists measure and model the potential climate impacts of black carbon (BC) based on its radiative forcing (RF). Radiative forcing is a positive (increase) or negative (decrease) change in the energy from the sun that is absorbed at the top of the atmosphere predicted by radiative transfer models. Positive RF is associated with an increase in the energy of the earth-atmosphere system that causes an increase in surface temperature, shifts in precipitation patterns, and other effects at both regional and global scales that can vary with the timing and location of emissions.

Complex black carbon–cloud interactions are generally described as having indirect or semi-direct effects on climate--most, if not all, with a large range of uncertainty (Bauer et al. 2010; Chen et al. 2010; Jacobson 2010; Koch et al. 2011). A likely range for the magnitude of the indirect effects of black carbon was estimated between -0.4 and 0.4 watts per meter squared (Wm^{-2}) (Shindell et al. 2012). In a multi-model study, diesel BC generated relatively smaller cloud impacts ($< \pm$ 0.06 Wm-2) compared to biofuel BC (-0.11 Wm^{-2}) (Koch et al. 2011).

Aerosol species co-emitted with BC include sulfates, nitrates, and organic carbon (OC) that primarily reflect light and contribute negative radiative forcing that offsets the positive forcing of black carbon. Still, the radiative forcing of these species per unit mass of emissions is weaker than that of black carbon in diesel combustion. In the road subsector in the year 2000, the direct RF of BC is about 32 Wm^{-2}, the direct OC RF is -2.4 Wm^{-2}, and the direct SO_4 RF is -9.4 Wm^{-2} (Balkanski et al. 2010). The lowest OC:BC mass emissions ratio required to produce a neutral effect on top-of-atmosphere direct forcing is 10.7:1 for any region (Bond, Zarzycki, Flanner, and Koch 2011). Any lower ratio results in positive direct forcing. However, important processes, particularly changes in cloud formation that tend toward cooling, have not been included here. The non-BC aerosol indirect effect uncertainty differs by sector and source. In general, the uncertainty for aerosol indirect effects is greater than for direct effects, resulting from a combination of minimal observational data, complexity in modeling impacts of various cloud types, and model experimental design inconsistencies that lead to poor comparability. As such, the general rule of thumb for non-BC aerosol climate forcing is that confidence in a source's RF increases as the quantity of co-emitted aerosols decreases (Bauer et al. 2010; Kopp and Mauzerall 2010).

Modeling experiments have demonstrated that the climate impacts of black carbon and other non-CO_2[15] emissions will peak between 2000-2050, contributing to a global temperature increase of 0.1°K under projected emissions growth scenarios (Olivié et al. 2012). Cars, light trucks, heavy-duty trucks, buses, and coaches are the vehicle types that will contribute the most to the near-term climate forcing of transportation-related black carbon (Borken-Kleefeld, Berntsen, and Fuglestvedt 2010).

The radiative forcing of vehicle emissions can vary based on the potential of each pollutant to absorb short- and long-wave radiation instantaneously and to persist in the atmosphere. For example, the instantaneous radiative forcing of one kilogram of black carbon is up to a million times stronger than an equivalent mass of carbon dioxide (Bond and Sun 2005). The lifetime of black carbon is also very different from that of CO_2. On average, black carbon persists for a little more than a week in the atmosphere; hence it is called short-lived. In contrast, the increased levels of carbon dioxide can persist for centuries, and so it is called long-lived. Short-lived pollutants contribute to near-term impacts that occur over years and

[15] Non-CO_2 pollutants including in this modeling are ozone, methane, organic carbon, sulfate, CFC-12, HFC-134a, and contrails.

decades after emissions take place, while long-lived pollutants contribute to impacts that occur over centuries and millennia.

The transportation sector is an important source of black carbon and other short-lived pollutants. About 19 percent of all BC emissions were attributable to transportation in the year 2000 (Lamarque et al. 2010; U.S. EPA 2012). Approximately nine percent of global black carbon emissions were attributable to the on-road sector alone, of which 99 percent were from diesel engines (Uherek et al. 2010). Black carbon emissions were the most potent contributor to temperature change per unit mass emitted of any pollutant from the transportation sector from 1990-1999 (Fuglestvedt et al. 2010). Assuming constant year-2000 emissions and measuring impacts based on radiative forcing, on-road transportation would be the sector responsible for the largest positive radiative-forcing-related consequences between 2000-2020, and the second-largest between 2000-2100 taking into account all emissions and their effects from all sectors (Unger et al. 2010).[16]

A.2. Net Climate Forcing of Diesel Aerosol Emissions

Diesel engines are a significant source of light-absorbing aerosols. A recent study examined all direct, indirect, and semi-direct effects of aerosol emissions by BC-rich sectors and found that the transportation sector exerts strong positive forcing on both regional and global scales (Bauer and Menon 2012). Among the important sources of BC, the aerosol emissions from diesel engines exert a net positive RF—with direct effects being strongly positive and indirect effects being negative or slightly positive (Jacobson 2002; Koch et al. 2007; Bauer et al. 2010; Bahadur et al. 2011; Bauer and Menon 2012; U.S. EPA 2012). The uncertainties regarding the net warming of mobile diesel emissions of black carbon are among the smallest of all global sources of the pollutant (Bond et al. 2013).

The climate impact of diesel vehicles has been compared to that of gasoline vehicles, taking into account the full suite of emissions from each vehicle type and the radiative forcing of each component over short and long time periods (Tanaka et al. 2012). Diesel vehicles emit less carbon dioxide per kilometer driven, and so the European Union has implemented certain taxes that motivate consumers to choose diesel vehicles. However, this can result in an unexpected climate impact when diesel vehicles are sold without tight controls on emissions of black carbon (Minjares et al. 2012). By using a simple climate model, researchers have shown that a new diesel passenger vehicle sold without a particle filter (Euro 3 and 4) causes greater warming than a similar gasoline car for up to a decade after a one-year pulse of emissions. This is attributable to emissions of both black carbon and nitrogen oxides (NOx), which cause warming through the production of ozone. After a decade, the temperature impacts of black carbon and ozone (via NOx and hydrocarbons) have dissipated, and the diesel vehicle has a smaller warming effect, owing to the lower emissions of carbon dioxide that persist in the atmosphere well beyond the ten-year period. Diesel vehicles that capture black carbon and meet Euro 6 standards are essentially equivalent to gasoline vehicles in terms of their near-term temperature consequences.

A.3. Metrics for Short-lived Pollutants

Metrics provide a tool that expresses the climate response to an emission source in a way that is directly comparable to emissions from other sources. Developing such measures is challenging since a metric must embody the relationships contained in complex climate models, including the variable behavior of emissions species in their environment and the response of the environment (also subject to variation) to changes in their ambient concentration.

[16] Assuming constant emissions up to the time horizons and measuring climate impacts as RF at that time. These impacts are equivalent to integrated RF up to this endpoint for a single pulse of emissions.

The policy community's focus on black carbon has raised questions about the appropriate application of metrics to short-lived climate forcers (U.S. EPA 2012). Metrics are unitless weighting factors that can translate emissions of any pollutant into an estimated impact on a carbon-dioxide-equivalent basis.

Time horizon is one source of confusion in the application of metrics. For example, the global warming potential (GWP) for black carbon is nearly four times larger when evaluated on a 20-year time horizon compared to a 100-year time horizon (Bond et al. 2011). Differences in values can also depend on the type of metric used. This has opened up broader interest in metrics generally and how they should be applied to short-lived forcing agents (Amaas, Peter, Fuglestvedt, and Berntsen 2012).

The GWP has proved useful when applied to long-lived greenhouse gases regulated under the Kyoto Protocol (e.g., carbon dioxide, methane, and nitrous oxide). These are atmospheric gases with long lifetimes that have wide and even distributions across the globe. Based on these features, the GWP assumes that any emission at any time and place will cause radiative forcing equivalent to that of any other emission at any future time or location. It does not imply an equivalent temperature response except in certain idealized situations. Table A-1 summarizes published estimates for the GWP of black carbon over time.

Table A- 1: . Estimates of the Global Warming Potential of Black Carbon

Study	**GWP20**			**GWP100**		
	Direct Effect	Cryosphere Effect	**Total**	Direct Effect	Cryosphere Effect	**Total**
Bond and Sun (2005) [C]	2200	-	**2200**	680	-	**680**
Rypdal et al. (2009) [B,C]	2940	310	**3250**	830	90	**920**
Fuglestedt et al. (2010)						
From Berntsen et al. (2006) [B]	1780	-	**1780**	510	-	**510**
From Schulz et al. (2006)	1600	-	**1600**	460	-	**460**
From Naik et al. (2007) [B]	3870	-	**3870**	1110	-	**1110**
From Koch et al. (2007)[B]	1580	-	**1580**	450	-	**450**
Reddy and Boucher (2007) [A]	1690	990	**2680**	480	281	**761**
Bond et al (2011) [D]						
All sources	2600	300	**2900**	740	90	**830**
Energy-related	2400	400	**2800**	690	100	**790**
Bond et al. (2013)	2100		**3200**	590		**900**

Note: This table is for illustrative purposes only. Underlying each study is a climate forcing and emissions dataset that may not necessarily be comparable.

[A] GWP20 values derived using absolute GWP values taken from Forster et al. (2007), p.211.

[B] Taken from the average of regional GWP values.

[C] Accounts for some internal mixing of black carbon particles.

A.4. References

Amaas, B., Peter, G. P., Fuglestvedt, J., and Berntsen, T. (2012). How to Compare Different Short-lived Climate Forcers: A Review of Emission Metrics. Oslo, Norway: Center for International Climate and Environmental Research.

Bahadur, R., Feng, Y., Russell, L. M., and Ramanathan, V. (2011). Impact of California's Air Pollution Laws on Black Carbon and Their Implications for Direct Radiative Forcing. *Atmospheric Environment* 45 (5): 1162–67.

Balkanski, Y., Myhre, G., Gauss, M., Rädel, G., Highwood, E. J., and Shine, K.P. (2010). Direct Radiative Effect of Aerosols Emitted by Transport: From Road, Shipping, and Aviation. *Atmospheric Chemistry and Physics* 10 (10): 4477–89.

Bauer, S. E., and Menon, S. (2012). Aerosol Direct, Indirect, Semidirect, and Surface Albedo Effects from Sector Contributions Based on the IPCC AR5 Emissions for Pre-industrial and Present-Day Conditions. *Journal of Geophysical Research: Atmospheres* 117: D01206.

Bauer, S. E., Menon, S., Koch, D., Bond, T. C., and Tsigaridis, K. (2010). A Global Modeling Study on Carbonaceous Aerosol Microphysical Characteristics and Radiative Effects. *Atmospheric Chemistry and Physics* 10 (15): 7439–56.

Berntsen, T. K., Fuglestvedt, J. S., Myhre, G., Stordal, F., & Berglen, T. F. (2006). Abatement of Greenhouse Gases: Does Location Matter? *Climatic Change*, *74*(4), 377–411. doi:10.1007/s10584-006-0433-4.

Bond, T. C., and Sun, H. (2005). Can Reducing Black Carbon Emissions Counteract Global Warming? *Environmental Science & Technology* 39 (16): 5921–26. doi:10.1021/es0480421.

Bond, T. C., Zarzycki, C., Flanner, M. G., and Koch, D. M. (2011). Quantifying Immediate Radiative Forcing by Black Carbon and Organic Matter with the Specific Forcing Pulse. *Atmospheric Chemistry and Physics* 11 (4): 1505–25. doi:10.5194/acp-11-1505-2011.

Bond, T. C., Doherty, S. J., & Fahey, D. W. (2013). Bounding the Role of Black Carbon in the Climate System: A Scientific Assessment. *Journal of Geophysical Research*. doi:10.1002/jgrd.50171

Borken-Kleefeld, J., Berntsen, T., and Fuglestvedt, J. (2010). Specific Climate Impact of Passenger and Freight Transport. *Environmental Science and Technology* 44 (15): 5700–5706. doi:10.1021/es9039693.

Chen, W.-T., Lee, Y. H., Adams, P. J., Nenes, A., and Seinfeld, J. H. (2010). Will Black Carbon Mitigation Dampen Aerosol Indirect Forcing? *Geophysical Research Letters* 37 (9): L09801.

Fuglestvedt, J. S., Shine, K. P., Berntsen, T., Cook, J., Lee, D. S., Stenke, A., Skeie, R. B., Velders, G. J. M., and Waitz, I. A. (2010). Transport Impacts on Atmosphere and Climate: Metrics. *Atmospheric Environment* 44 (37): 4648–77. doi:10.1016/j.atmosenv.2009.04.044.

Jacobson, M. Z. (2002). Control of Fossil-Fuel Particulate Black Carbon and Organic Matter, Possibly the Most Effective Method of Slowing Global Warming. *Journal of Geophysical Research: Atmospheres* 107: D19-ACH 16-1–22.

Jacobson, M. Z. (2010). Short-term Effects of Controlling Fossil-Fuel Soot, Biofuel Soot and Gases, and Methane on Climate, Arctic Ice, and Air Pollution Health. *Journal of Geophysical Research: Atmospheres* 115: D14209.

Koch, D., Balkanski, Y., Bauer, S. E., Easter, R. C., Ferrachat, S., Ghan, S. J., Hoose, C., Iversen, T., Kirkevåg, A., Kristjansson, J. E., Liu, X., Lohmann, U., Menon, S., Quaas, J., Schulz, M., Seland, Ø., Takemura, T., and Yan, N. (2011). Soot Microphysical Effects on Liquid Clouds, a Multi-Model Investigation. *Atmospheric Chemistry and Physics* 11 (3): 1051–64.

Koch, D., Bond, T. C., Streets, D., Unger, N., and van der Werf, G. R. (2007). Global Impacts of Aerosols from Particular Source Regions and Sectors. *Journal of Geophysical Research: Atmospheres* 112: D02205.

Kopp, R. E., and Mauzerall, D. L. (2010). Assessing the Climatic Benefits of Black Carbon Mitigation. *Proceedings of the National Academy of Sciences* 107 (26): 11703–8.

Minjares, R., Blumberg, K., & Posada Sanchez, F. (2012). Alignment of Policies to Maximize the Climate Benefits of Diesel Vehicles through Control of Particulate Matter and Black Carbon Emissions. *Energy Policy*, 1–8. doi:10.1016/j.enpol.2012.09.053

Naik, V., Mauzerall, D. L., Horowitz, L. W., Schwarzkopf, M. D., Ramaswamy, V., and Oppenheimer, M. (2007). On the Sensitivity of Radiative Forcing from Biomass Burning Aerosols and Ozone to Emission Location. *Geophysical Research Letters* 34 (3): L03818.

Olivié, D. J. L., Cariolle, D., Teyssèdre, H., Salas, D., Voldoire, A., Clark, H., Saint-Martin, D., Michou, M., Karcher, F., Balkanski, Y., Gauss, M., Dessens, O., Koffi, B., and Sausen, R. (2012). Modeling the Climate Impact of Road Transport, Maritime Shipping and Aviation over the Period 1860–2100 with an AOGCM. *Atmospheric Chemistry and Physics* 12 (3): 1449–80. doi:10.5194/acp-12-1449-2012.

Reddy, M. S., and Boucher, O. (2007). Climate Impact of Black Carbon Emitted from Energy Consumption in the World's Regions. *Geophysical Research Letters* 34 (11): L11802. doi:10.1029/2006GL028904.

Rypdal, K., Rive, N., Berntsen, T. K., Klimont, Z., Mideksa, T. K., Myhre, G., and Skeie, R. B. (2009). Costs and Global Impacts of Black Carbon Abatement Strategies. *Tellus B* 61 (4): 625–41.

Schulz, M., Textor, C., Kinne, S., Balkanski, Y., Bauer, S., Berntsen, T. K., et al. (2006). Radiative Forcing by Aerosols as derived from the AeroCom Present-day and Pre-industrial Simulations. *Atmospheric Chemistry and Physics*, *6*(12), 5246.

Shindell, D., Kuylenstierna, J. C. I., Vignati, E., van Dingenen, R., Amann, M., Klimont, Z., et al. (2012). Simultaneously Mitigating Near-Term Climate Change and Improving Human Health and Food Security. *Science*, *335*(6065), 183–189. doi:10.1126/science.1210026

Tanaka, K., Berntsen, T., Fuglestvedt, J. S., and Rypdal, K. (2012). Climate Effects of Emission Standards: The Case for Gasoline and Diesel Cars. *Environmental Science and Technology* 46 (9): 5205–13.

Uherek, E., Halenka, T., Borken-Kleefeld, J., Balkanski, Y., Berntsen, T., Borrego, C., Gauss, M., Hoor, P., Juda-Rezler, K., Lelieveld, J. Melas, D., Rypdal, K., and Schmid, S. (2010).

Unger, N., Bond, T. C., Wang, J. S., Koch, D. M., Menon, S., Shindell, D. T., and Bauer, S. (2010). Attribution of Climate Forcing to Economic Sectors. *Proceedings of the National Academy of Sciences* 107 (8): 3382–87.

U.S. EPA. (2012) Report to Congress on Black Carbon. United States Environmental Protection Agency. EPA-450/R-12-001. March, 2012. http://www.epa.gov/blackcarbon.

8 Appendix B: Diesel Health Impacts

B.1. The Health Impacts of Diesel Gaseous Species

Oxides of sulfur and nitrogen, emitted concurrently with particulate matter, have been linked to hospitalizations for cardiac and respiratory illnesses, including asthma, and cardiopulmonary mortality (Wong et al. 2002; HEI 2012; Chen et al. 2008). Long-term exposure to SO_2 and NO_X, and especially to NO_2, is associated with mortality from all causes—and notably from cardiopulmonary mortality and lung cancer (Pope et al. 2002; Krewski et al. 2009; Cao et al. 2011). Sulfate particles, a byproduct of SO_2, have also been shown to increase the risk of mortality (Thurston et al. 1989; Krewski et al. 2009). Nitrogen oxides have additionally been associated with acute airway inflammation and impaired lung function (Strak et al. 2012), although clinical exposure to NO_2 alone was not found to alter vascular or lung function (Langrish et al. 2010). NO_2, along with fine particulate matter ($PM_{2.5}$) and black carbon (BC), has been associated as well with increased risk of stroke (Wellenius et al. 2012).

Ozone, while not directly emitted in exhaust, is produced by photochemical reactions with nitrogen oxide emissions from diesel vehicles and other precursors, including volatile organic compounds emitted from natural and anthropogenic sources. A major U.S. study found that ozone concentrations did not increase the risk of death from cardiovascular disease when discounting the effect of $PM_{2.5}$ concentrations, but they did increase the risk of death from respiratory causes (Jerret et al. 2009). Elevated ozone concentrations have been shown to increase rates of hospitalization for respiratory illnesses and childhood asthma (Friedman et al. 2001; Wong et al. 2002; Lee et al. 2006).

Researchers performing epidemiological studies have had difficulty disentangling the differential effects of PM from gaseous co-pollutants (Smith et al. 2009). One recent attempt found that, in a clinical setting, lung impairment caused by diesel exhaust was more extensive than that attributable to other emissions sources and that the mixture of gases and particles had a stronger effect than the two components in isolation. This suggests that the interaction between particles and gases may be important in explaining the overall health impacts of diesel exhaust (Mills et al. 2011).

B.2. Potential Benefits of Diesel Emissions Control Policies

A number of observational studies have estimated the reduction in adverse health consequences in response to improvements in air quality. An analysis done for 23 European cities estimates that reducing annual $PM_{2.5}$ concentrations to 15 micrograms per cubic meter ($\mu g/m^3$) would prevent 16,926 premature deaths, including 11,612 cardiopulmonary and 1,901 lung cancer deaths, and result in an increase in life expectancy from as little as one month to as much as two years (Boldo et al. 2006). A U.S. study estimated that a 10 $\mu g/m^3$ drop in $PM_{2.5}$ would translate to a six-month increase in average life expectancy (Pope et al. 2009).

The ongoing National Morbidity, Mortality, and Air Pollution Study (NMMAPS) examines the effectiveness of pollution control strategies in the United States. One analysis of NMMAPS data found that, between 1987-2000, air quality improved in the 100 largest counties in the nation as a result of environmental regulations, including national diesel-powered truck and bus emissions standards, national Tier-1 motor vehicle emissions standards, and the national low-emissions vehicle program. There has also been a trend showing a decline in the short-term risk estimates. This suggests that these policies have improved public health (Dominici et al. 2007).

Several studies internationally have quantified the health effects of specific transportation policy interventions. One such study tracked the mortality rates from various categories of heart and lung disease in the years leading up to the 2006 promulgation of tighter restrictions on diesel emissions in Tokyo and compared them to what took place in the years following the initiative (Yorifuji et al. 2011). Accounting for other time-series trends, researchers found that the mortality rate for cerebrovascular disease was reduced by 8.5 percent. In Hong Kong, where fuel sulfur levels were limited to 500 parts per million in 1990, the reduction in both PM and SO_2 emissions likely prevented deaths from respiratory and cardiovascular disease and increased the average life expectancy for city residents by about one month (Hedley et al. 2002). A single project implementing a cleaner (compressed natural gas) mass-transit system in the city of Dhaka, Bangladesh, is projected to save more than 850 lives as a result of particulate matter reductions alone (ABT Associates 2011). A major global-scale health impact assessment estimated that, by imposing tighter global standards on vehicle emissions and mandating the necessary improvements in fuel quality, regulatory agencies could prevent 200,000 deaths per year by 2030 (Shindell 2011).

Strategies for control of diesel PM and BC can reduce not only particle mass but also particle number and composition (which also affect public health). Modern after-treatment controls on diesel engines have produced large changes in the composition of diesel exhaust in ways that substantially benefit public health. Emissions from U.S. 2007-compliant diesel engines fitted with a diesel oxidation catalyst (that controls organic compounds) and a catalyzed diesel particle filter (that controls elemental carbon) showed not only a reduction in the mass of black carbon emitted, but also a reduced concentration of toxic polycyclic aromatic hydrocarbons and a 71-99 percent reduction in an array of unregulated particles and air toxics that constitute a minor portion of diesel emissions (Khalek et al. 2011; Shibata et al. 2010). A recent experiment exposing rats to the exhaust of U.S. 2007-compliant diesel engines found no genotoxic effects associated with exposure for up to three months (McDonald et al. 2012).

B.3. References

Abt Associates (2011). Quantifying the Health Benefits of the Greater Dhaka Sustainable Urban Transport Corridor Project.

Boldo, E., Medina, S., Le Tertre, A., Hurley, F., Mücke, H.-G., Ballester, F., and Aguilera, I. (2006). Apheis: Health Impact Assessment of Long-term Exposure to PM2.5 in 23 European Cities. *European Journal of Epidemiology* 21 (6): 449–58.

Cao, J., Yang, C., Li, J., Chen, R., Chen, B., Gu, D., and Kan, H. (2011). Association between Long-term Exposure to Outdoor Air Pollution and Mortality in China: A Cohort Study. *Journal of Hazardous Materials* 186 (2–3): 1594–1600.

Chen, G., Song, G., Jiang, L., Zhang, Y., Zhao, N., Chen, B., and Kan, H. (2008). Short-term Effects of Ambient Gaseous Pollutants and Particulate Matter on Daily Mortality in Shanghai, China. *Journal of Occupational Health* 50 (1): 41–47.

Dominici, F., Peng, R. D., Zeger, S. L., White, R. H., and Samet, J. M. (2007). Particulate Air Pollution and Mortality in the United States: Did the Risks Change from 1987 to 2000? *American Journal of Epidemiology* 166 (8): 880–88.

Friedman, M. S., Powell, K. E., Hutwagner, L., Graham, L. M., and Teague, W. G. (2001). Impact of Changes in Transportation and Commuting Behaviors during the 1996 Summer Olympic Games in Atlanta on Air Quality and Childhood Asthma. *Journal of the American Medical Association* 285 (7): 897–905.

Hedley, A. J., Wong, C.-M., Thach, T. Q., Ma, S., Lam, T.-H., and Anderson, H. R. (2002). Cardiorespiratory and All-Cause Mortality after Restrictions on Sulphur Content of Fuel in Hong Kong: An Intervention Study. *The Lancet* 360 (9346): 1646–52.

HEI, 2012. Effects of Short-Term Exposure to Air Pollution on Hospital Admissions of Young Children for Acute Lower Respiratory Infections in Ho Chi Minh City, Vietnam (HEI Research Report 169 No. 169). Health Effects Institute, Boston, MA.

Jerrett, M., Burnett, R. T., Pope, C. A. (III), Ito, K., Thurston, G., Krewski, D., Shi, Y., Calle, E., and Thun, M. (2009). Long-term Ozone Exposure and Mortality. *New England Journal of Medicine* 360 (11): 1085–95.

Khalek, I. A., Bougher, T. L., Merritt, P. M., and Zielinska, B. (2011). Regulated and Unregulated Emissions from Highway Heavy-Duty Diesel Engines Complying with U.S. Environmental Protection Agency 2007 Emissions Standards. *Journal of the Air and Waste Management Association* 61 (4): 427–42.

Krewski, D., Jerrett, M., Burnett, R., Ma, R., Hughes, E., Shi, Y., Turner, M., Pope, C. A. (III), Thurston, G., Calle, E. E., Thun, M. J., Beckerman, B., DeLuca, P., Finkelstein, N., Ito, K., Moore, D., Newbold, K., Ramsay, T., Ross, Z., Shin, H., and Tempalski, B. (2009). Extended Follow-up and Spatial Analysis of the American Cancer Society Study Linking Particulate Air Pollution and Mortality (HEI Research Report No. 140). Health Effects Institute, Boston.

Langrish, J. P., Lundbäck, M., Barath, S., Söderberg, S., Mills, N. L., Newby, D. E., et al. (2010). Exposure to Nitrogen Dioxide is not associated with Vascular Dysfunction in Man. *Inhalation Toxicology*, *22*(3), 192–198. doi:10.3109/08958370903144105.

Lee, S. L., Wong, W. H. S., and Lau, Y. L. (2006). Association between Air Pollution and Asthma Admission among Children in Hong Kong. *Clinical and Experimental Allergy* 36 (9): 1138–46.

McDonald, J., Doyle-Eisele, M., Gigliotti, A., Miller, R., Seilkop, S., Mauderly, J., Seagrave, J., Chow, J., and Zielinska, B. (2012). Advanced Collaborative Emissions Study (ACES) Subchronic Exposure Results: Biologic Responses in Rats and Mice and Assessment of Genotoxicity—Part 1. Biologic Responses in Rats and Mice to Subchronic Inhalation of Diesel Exhaust from U.S. 2007-Compliant Engines: Report on 1-, 3-, and 12-Month Exposures in the ACES Bioassay (HEI Research Report No. 166), Health Effects Institute, Boston.

Mills, N. L., Miller, M. R., Lucking, A. J., Beveridge, J., Flint, L., Boere, A. J. F., Fokkens, P. H., Boon, N. A., Sandstrom, T., Blomberg, A., Duffin, R., Donaldson, K., Hadoke, P. W. F., Cassee, F. R., and Newby, D. E. (2011). Combustion-Derived Nanoparticulate Induces the Adverse Vascular Effects of Diesel Exhaust Inhalation. *European Heart Journal* 32 (21): 2660–71.

Pope, C. A. (III), Burnett, R. T., Thun, M. J., Calle, E. E., Krewski, D., Ito, K., and Thurston, G. D. (2002). Lung Cancer, Cardiopulmonary Mortality, and Long-term Exposure to Fine Particulate Air Pollution. *Journal of the American Medical Association* 287 (9): 1132–41.

Pope, C. A. (III), Ezzati, M., and Dockery, D. (2009). Fine-Particulate Air Pollution and Life Expectancy in the United States. *New England Journal of Medicine* 360 (4): 376–86.

Shibata, K., Yanagisawa, N., Tashiro, Y., Mukunashi, T., Onodera, T., and Sakamoto, K. (2010). Reduction in the Emissions and Toxicity of Polycyclic Aromatic Hydrocarbons from a Heavy-Duty Diesel Engine with the Latest Aftertreatment Devices. *Journal of Health Science* 56 (1): 31–40.

Shindell, D., and Faluvegi, G. (2009). Climate Response to Regional Radiative Forcing during the Twentieth Century. *Nature Geoscience* 2 (4): 294–300.

Shindell, D., Faluvegi, G., Walsh, M., Anenberg, S. C., Van Dingenen, R., Muller, N. Z., et al. (2011). Climate, Health, Agricultural, and Economic Impacts of Tighter Vehicle-emission Standards. *Nature Climate Change*, *1*(1), 59–66. doi:10.1038/nclimate1066.

Smith, K. R., Jerrett, M., Anderson, H. R., Burnett, R. T., Stone, V., Derwent, R., Atkinson, R. W., Cohen, A., Shonkoff, S. B., Krewski, D., Pope, C. A. (III), Thun, M. J., and Thurston, G. (2009). Public Health Benefits of Strategies to Reduce Greenhouse-Gas Emissions: Health Implications of Short-lived Greenhouse Pollutants. *The Lancet* 374 (9707): 2091–2103.

Strak, M., Janssen, N. A., Godri, K. J., Gosens, I., Mudway, I. S., Cassee, F. R., Lebret, E., Kelly, F. J., Harrison, R. M., Brunekreef, B., Steenhof, M., and Hoek, G. (2012). Respiratory Health Effects of Airborne Particulate Matter: The Role of Particle Size, Composition, and Oxidative Potential—The RAPTES Project. *Environmental Health Perspectives* 120 (8): 1183–89.

Thurston, G. D., Ito, K., Lippmann, M., and Hayes, C. (1989). Reexamination of London, England, Mortality in Relation to Exposure to Acidic Aerosols during 1963-1972 Winters. *Environmental Health Perspectives* 79: 73–82.

Wellenius, G. A., Burger, M. R., Coull, B. A., Schwartz, J., Suh, H. H., Koutrakis, P., Schlaug, G., Gold, D. R., and Mittleman, M. A. (2012). Ambient Air Pollution and the Risk of Acute Ischemic Stroke. *Archives of Internal Medicine* 172 (3): 229–34.

Wong, C.-M., Atkinson, R. W., Anderson, H. R., Hedley, A. J., Ma, S., Chau, P. Y.-K., and Lam, T.-H. (2002). A Tale of Two Cities: Effects of Air Pollution on Hospital Admissions in Hong Kong and London Compared. *Environmental Health Perspectives* 110 (1): 67–77.

Yorifuji, T., Kawachi, I., Kaneda, M., Takao, S., Kashima, S., and Doi, H. (2011). Diesel Vehicle Emission and Death Rates in Tokyo, Japan: A Natural Experiment. *Science of the Total Environment* 409 (19): 3620–27.

9 Appendix C: Cost-Benefit Project Modeling Framework

The goal of the cost-benefit project modeling exercise is to develop and test a model that can account for both climate and health impacts and intervention costs. The model should have the capability to monetize social and environmental costs and should take a cost-benefit approach that is consistent with international best practices.

In the course of designing such a model, a balance must be struck between flexibility and specificity. A model that is too inflexible could not be applied to any project in any region. This is an important prerequisite when the projects to be evaluated are not well known beforehand. A model that is too flexible and general will not capture the specific local conditions that can strongly influence the estimate of costs and impacts.

The model used in this study has been designed such that the resulting analysis of specific projects in specific countries can be treated as examples necessary to demonstrate the cost and mitigation potential of an intervention.

The cost-benefit modeling framework consists of four modules:

- Emissions
- Technology Costs
- Impacts (Climate and Health)
- Economic Valuation

The final cost-benefit analysis compares the results of the technology cost, emissions (fuel consumption only), and impacts models. The four modules and their input parameters are described below.

C.1. Emissions

The emissions module estimates total current and future emissions from transportation-related sources for a single city or region under both baseline and intervention scenarios. The model was developed to estimate fleet-wide fuel consumption and emissions of carbon dioxide, particulate matter (PM_{10}, $PM_{2.5}$), black carbon, organic carbon, and methane (natural gas intervention only) from multiple on-road vehicle sources.

The emissions model is a basic, bottom-up emissions model using a standard approach in which annual emissions of each species are estimated for each individual vehicle type *a*, fuel type *b*, and emission standard *c* according to:

$$E_{a,b,c} = \Sigma(Pop_{a,b,c} \times EF_{a,b,c} \times VKT_a)$$

Where Pop is vehicle population, EF is emissions factor (corrected for fuel sulfur level), and VKT is annual vehicle kilometers traveled.

Table C-1 summarizes the primary inputs to the emissions module. In most cases, default values in the model may be used in lieu of known local inputs. For example, the model is built using the COPERT European emissions model factors as defaults. All emissions factors are presented in terms of grams emitted per kilometer traveled. For compressed natural gas (CNG) bus projects, methane leakage is currently estimated using a leakage emissions factor of 1.99 g/km, consistent with that identified for New Delhi (Reynolds and Kandlikar 2008).

Table C- 1: Inputs to the Fleet Emissions Module

- Historical and projected vehicle stock by vehicle type and fuel
- Tailpipe standard implementation dates by vehicle type
- Survival curves by vehicle type
- Annual per vehicle VKT by vehicle type
- Baseline emissions factors by vehicle type, fuel, and certified emissions standard
- Fuel consumption rates by vehicle type, fuel, and certified emissions standard
- Fuel sulfur levels by year
- High-emitter rates and profiles by vehicle type, fuel, and emissions standard

Inputs Shared with the Cost Module

- Number of vehicles affected by the intervention
- Activity (e.g., VKT) of vehicles affected
- Certified emissions standard of vehicles affected
- Specific technologies likely to be introduced as part of the intervention
- Useful life of vehicles affected by the intervention

To reduce data collection, the model can use a smaller number of inputs for fleets not directly affected by the intervention by using an average emissions factor for a fleet of one type of vehicle over time (e.g., trucks in a program designed to target buses). In contrast, a higher number of inputs is required to model emissions from the intervention fleet in order to estimate emissions from each vehicle type by age, fuel type, emissions standard, and so forth.

Recognizing that data on vehicle population may be unavailable in some regions, economic data (e.g., GDP per capita or GDP growth rate) may be used to estimate and project vehicle populations.

C.2. Technology Costs

The technology cost module estimates the total additional cost of the intervention over time compared with costs under a business-as-usual scenario. The module can estimate a range of project types, including fleet-specific fuel switching, high-emitter evaluation, refinery upgrading, and more.

For fleet intervention projects, the template for this module is the publicly available BestBus model, developed by MJ Bradley and Associates for Duke University. The model is a cost-of-ownership model that evaluates costs over the life cycle of the fleet and was designed to assess fleet operations in the United States. While the model was originally designed to estimate costs of bus fleets, one can simplify and expand the model to cover additional categories of vehicles.

The cost model was developed to evaluate the total cost of ownership over the useful life of a vehicle. Elements of total cost included in the model are bus purchase; purchase/installation of required fueling infrastructure; purchase/installation of required depot modifications and special tools; annual operator labor; annual bus maintenance and fuel costs; annual maintenance and operating cost of required fueling infrastructure, depot modifications, and special tools; and periodic bus overhaul costs. The analysis does not include full overhead for management functions such as road supervision, procurement, and so forth. In practice, these costs included in the model are simplified into three main inputs: initial capital costs, maintenance/operating costs, and fuel costs.

The model was designed to be dynamic, so that major assumptions about costs can be changed as new information becomes available and to facilitate “what if” and sensitivity analyses. The model includes default assumptions for many cost elements but also allows users to input their own values for every major assumption, if location-specific information is available.

The model was set up to evaluate the differential costs and emissions of the following baseline bus and retrofit options:

- Baseline Diesel (typical 1998-2001 diesel engine operated on either standard diesel fuel or a “baseline” biodiesel fuel blend)
- DPF Retrofit (baseline diesel retrofit with a diesel particulate filter, and the following propulsion technology options for new buses)
- CNG (natural gas engine compliant with 2007 or 2010 EPA emission standards and compressed natural gas fuel system)
- Clean Diesel (diesel engine compliant with 2007 EPA emission standards)

Some vehicle technologies require significant investments in new fueling infrastructure, depot modifications, and special tools. In the model, the cost of these depot investments is amortized over the entire useful life of the investment, which in many cases is longer than the useful life of the buses. The fueling infrastructure costs for diesel are zero since the model assumes that diesel fueling stations already exist and that no modifications are necessary. Default purchase and lifetime costs for the different bus technologies are shown in Table C-2.

Table C- 2: Default Costs for Bus/Truck Technology Intervention

Capital Costs (fleet intervention)		
Vehicle Type	**New Purchase Price**	
	low	high
baseline diesel	$150,000	$180,000
clean diesel	$158,000	$188,000
retrofit	$5,000	$8,000
compressed natural gas – original equipment manufacturer	$172,500	$207,000
CNG - conversion	$15,000	$30,000
liquefied natural gas - OEM	$172,500	$207,000
“green freight” retrofit	$10,000	$12,000
Fueling Infrastructure		
Fuel Type	Cost Per Fueling Station	
	low	**high**
diesel	$0	$0
CNG	$5,600,000	$6,720,000
LNG	$5,600,000	$6,720,000
Operational Costs (fleet intervention)		
Vehicle Type	Lifetime Average Cost Per Km	
	low	**High**
baseline diesel	$0.32	$0.55
clean diesel	$0.32	$0.55
retrofit diesel	$0.32	$0.55
CNG - OEM	$0.35	$0.60
CNG - conversion	$0.35	$0.60
LNG - OEM	$0.35	$0.60
“green freight” retrofit	$0.32	$0.55

Another fundamental input to the cost model is the cost of capital (i.e., capital discount rate, inflation rate, and so forth).

For large scale, fleet-wide emissions standards and fuel quality improvement interventions, the model estimates the incremental costs associated with additional emissions control technologies for vehicles as well as refinery capital upgrades and operations costs for fuel desulfurization. Default vehicle technology cost estimates are presented in Table C-3; these numbers are ICCT estimates of cumulative emissions control technology costs over no-control engines across a range of developed and developing countries.

Table C- 3: Cumulative Costs for Vehicle Technology Improvement by Emissions Standard

	Large Buses	Minibuses	Taxis	Private Cars	Motorcycles	Light Trucks	Heavy Trucks
Fuel	**Diesel**	**Diesel**	**Gasoline**	**Gasoline**	**Gasoline**	**Diesel**	**Diesel**
Euro I	$446	$446	$239	$239	$29	$446	$446
Euro II	$588	$588	$293	$293	$35	$588	$588
Euro III	$1,893	$1,707	$359	$359	$51	$1,707	$1,893
Euro IV	$3,298	$2,945	$432	$432	$51	$2,945	$3,298
Euro V	$4,898	$4,262	$474	$474	$54	$4,262	$4,898
Euro VI	$6,782	$5,502	$489	$489	$100	$5,502	$6,782

Costs for fuel desulfurization are computed based on estimates of refinery upgrade costs in cents/liter per 100 ppm sulfur reduction. Defaults values in the model are $0.01-0.02 per liter per 100 ppm sulfur reduction for low and high cost estimates (adapted from Hart Energy and MathPro 2012).

C.3. Impacts

From total emissions data, the impacts model estimates total climate impacts as well as total effects on human health in both the baseline and intervention scenarios.

Climate impacts are reported as carbon-dioxide-equivalent emissions (CO_2e). For simplicity and to maintain consistency with standard greenhouse emissions inventory accounting, a GWP is applied in this study. Table C-4 gives the pollutants accounted for in the climate impacts model and the conversion factors used to estimate total CO_2e.

Table C- 4: Emissions in Climate Impacts Model and Metric Conversion Values to Estimate CO_2e

	GWP20	GWP100
Black Carbon	3,200	900
Organic Carbon	-154	-42
Methane	72	25
Carbon Dioxide	1	1

Sources: Black carbon global warming potential (GWP) from Bond et al. 2013; organic carbon GWP from Bond et al. 2011; and methane and carbon dioxide GWP from Forster et al. 2007.

The health model developed by ICCT estimates health impacts in urban areas based on ambient concentrations of primary PM_{10} and $PM_{2.5}$. The health benefits of a policy intervention are determined based on the way in which ambient particulate concentrations diminish. Any analysis that yields a direct comparison of ambient particulate concentrations in intervention versus nonintervention (baseline) scenarios will allow for estimates of corresponding improvements in public health.

Emissions inventories provide information necessary to estimate population-level exposure and health consequences. This is based on a conversion of total annual emissions of fine particulate matter ($PM_{2.5}$) to annual average ambient concentrations. Long-term exposure to $PM_{2.5}$ is known to increase the risk of death from lung cancer, heart disease, and childhood respiratory infections.[17] Other pollutants can also be included, but for simplicity the current modeling evaluates only the potential premature mortality caused by exposure to primary $PM_{2.5}$. This will result in an underestimate of health effects but can still provide a useful indication of the magnitude and potential for benefits from new vehicle emissions control policies.

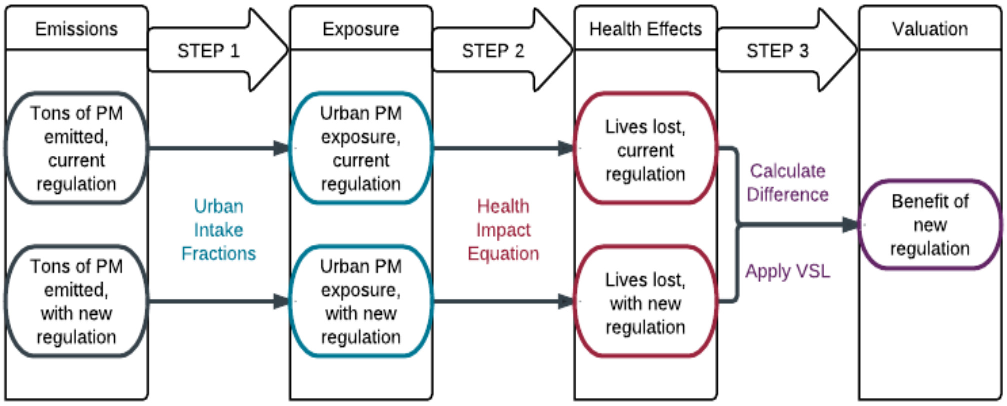

Figure C- 1: Conceptual Framework for Valuing the Health Benefits of an Emissions Control Strategy

The change in the contribution of vehicle emissions to outdoor ambient air pollution caused by a new policy is related directly to changes in health outcomes. Since policymakers are interested in comparing the a cost of policy against its benefits, the total lives saved can be converted through a formula to U.S. dollars or any other national currency to estimate the net benefits of the policy; costs can then be measured directly against the benefits to estimate U.S. dollars per life saved as a measure of cost-effectiveness. The following provides a detailed description of these methods, assuming the presence of at least two emissions inventories representing annual average emissions of $PM_{2.5}$ concentrations under business-as-usual and new policy scenarios.

Converting Emissions to Concentrations

Primary $PM_{2.5}$ emissions from the transportation sector are ground-based and mixed thoroughly in the urban atmosphere, which makes them well-suited to a health modeling approach that uses intake fractions as a measure of exposure. This can facilitate a rapid estimate of the population-level health burden that avoids the development and application of air quality dispersion models.

The concept of intake fraction was formally defined in Bennett et al. (2002) as a way to quantify the emissions-to-intake relationship for any given chemical compound from a particular source. To avoid the need for GIS-based analysis of pollutant concentrations and the resulting exposure in every model run, a health model has been developed in collaboration with researchers at the University of California, Berkeley, that utilizes a set of

[17] Based on a review of the evidence by the World Health Organization to estimate health impacts from all outdoor sources of air pollution.

pre-calculated, spatially derived exposure metrics called intake fractions for a global set of 3,646 cities (Apte et al. 2012). Through a series of equations, these metrics relate on-road emissions within an urban area to urban pollutant concentrations. A dimensionless parameter, the intake fraction of an emissions source can take any value from 0 to 1. An intake fraction of 20 ppm, for example, means that, of a million grams of a compound emitted, 20 grams are inhaled. Expressed mathematically (Apte et al. 2012),

Equation (1) $$iF = \frac{Population\ intake}{Total\ emissions} = \frac{\int_{T_1}^{\infty}\left(\sum_{i=1}^{P}\left(C_i(t) \times Q_i(t)\right)\right)}{\int_{T_1}^{T_2} E(t)dt}$$

where iF is the source-specific intake fraction,

T_1 and T_2 are the starting and ending times of an emissions process,

P is the number of people exposed,

$Q_i(t)$ is the volumetric breathing rate (m^3/s) for individual i at time t,

$C_i(t)$ is the incremental concentration (g/m^3) at time t in individual i's breathing zone that is attributable to the emissions process, and

E(t) is the emission rate from the process (g/s) at time t.

An intake fraction is an extrinsic property of an emissions source; it is context-specific, depending on the causes of dispersal and the availability of a nearby population to inhale the emissions (Bennett et al. 2002). Intake fractions can be generalized for a type of source in a given area.

In a specific application of the intake fraction concept, Apte et al. (2012) estimated the intra-urban intake fractions from spatially distributed, ground-level emissions sources (vehicles driven within the city). These intake fractions are valid for conserved, nonreactive emissions, which is a reasonable model for $PM_{2.5}$, carbon monoxide, and benzene within urban areas, since their decay rate is much lower than the rate at which they leave the city air compartment (wind speed over city length, u/L).

The intake fraction can be used to model annual average urban concentration from emissions because it incorporates a number of city-specific parameters into a single value. However, this application of the intake fraction requires certain assumptions. First, it applies to widely-dispersed and well-mixed ground-level emissions sources only. Transportation-related emissions are therefore a good candidate for this approach. Second, only conserved pollutants can be modeled. In this exercise, exposure to $PM_{2.5}$ is modeled while other pollutants are not considered. Since exposure to $PM_{2.5}$ contributes to the bulk of health impacts caused by transportation emissions, this limitation serves to produce underestimates of total health impacts. Finally, the intake fraction applies in urban areas, so impacts in rural areas are not considered here.

The equation to relate city-specific emissions to concentrations is as follows:

Equation (2) $$C = \frac{iF \times E}{P \times BR}$$

where C is the annual concentration of a conserved, nonreactive pollutant,

iF is the city-specific intake fraction for on-road emissions,

E is the total on-road emissions within the city,

P is the city population, and

BR is the annual individual breathing rate (assumed to be 5292.5 m^3/yr) (U.S. EPA, 2009).

A health modeling approach that uses an intake fraction from each city in the year 2000 to calculate the pollutant concentrations arising from vehicle emissions in that city is used. Because an intake fraction changes with a change in either city area or city population, it can be projected into future years using equation 3.

Equation (3) $$iF_t = iF_{t-1} \times \left(\frac{P_t}{P_{t-1}}\right)^m$$

where iF_t is the intake fraction in year t,

iF_{t-1} is the intake fraction in a previous model year,

P_t is the city population in year t,

P_{t-1} is the city population in a previous model year, and

m is a region-specific coefficient determined by linear population density.

Equation 3 is based on the observation that linear population density, (LPD), or the population of a city divided by the square root of the city's area, tends to remain constant over time. This metric can be used to predict change in city size based on population growth (Marshall 2007). LPD values were derived individually for each region using a linear regression.

The benefits of the intake fraction approach include the ability to calculate results quickly following changes in emissions inventories. This approach also has its limitations: concentration values are less precise than values produced from a spatially explicit model. This approach to estimating intake fractions can only be applied for conserved, nonreactive pollutants; it does not accommodate secondary pollutants like ozone and secondary particulate matter.

Exposure and Health Impacts

The intake fraction approach can be applied to estimate exposure and, using population and urban density, ambient urban concentrations for any conserved compounds emitted from on-road vehicles. Because national emissions inventories are calculated on an annual basis, ambient concentration estimates are annual averages. No assumptions are made regarding ambient concentrations at a finer temporal scale (i.e., assuming higher seasonal or 24-hour averages). For the health analysis conducted in this study, the compound of most interest is primary $PM_{2.5}$ since $PM_{2.5}$ concentrations are most strongly associated with adverse health effects from long-term exposure. Krewski et al. (2009) provide dose-response curves that can be incorporated into a methodology developed by the World Health Organization for calculating the health impacts of outdoor air pollution based on ambient annual $PM_{2.5}$ (Cohen et al. 2004; Ostro 2004). From these methods, the health modeling can calculate three endpoints: premature mortality in adults over the age of 30 from lung cancer and from a range of cardiopulmonary diseases and mortality in children under the age of five from acute respiratory infection (ARI) (Cohen et al. 2004). No morbidity health conditions, such as chronic bronchitis, are evaluated here. This follows the WHO precedent of only evaluating mortality. Limited work has been done to extrapolate exposure-response relationships for morbidity outcomes to the international scale, and broadly available and consistent underlying incidence data for morbidity effects is needed before the work can proceed.

PAF Methods

Health impact is quantified by calculating the Population Attributable Fraction (PAF), or the fraction of deaths from a discrete cause within the specified population that are attributable to a particular risk factor (Ezzati et al. 2004). This approach compares the current risks of mortality faced by the urban population due to long-term exposure to $PM_{2.5}$ with the lower risk they would face at some minimum exposure level (termed the counterfactual). Health effects studies have failed to find a minimum exposure threshold below which there is no increased risk of premature mortality (Ostro 2004). In any case, because of the ubiquity of low-level concentrations of non-anthropogenic $PM_{2.5}$, a counterfactual of zero may be unrealistic. The health modeling assumes a plausible minimum exposure level of four micrograms per cubic meter (μg/m^3) for $PM_{2.5}$ and 10 μg/m^3 for PM_{10}, as suggested by Ostro (2004).

Dose-Response Functions

Each of the three disease categories considered in the model—lung cancer, cardiopulmonary disease, and ARI—has a dose-response function that predicts the increase in risk relative to the counterfactual, expressed as the relative risk (RR). For acute respiratory infection, RR is calculated based on a commonly used linear dose-response function, in which a percentage increase in risk is directly proportional to an increase in ambient PM levels, as shown in Equation 4:

Equation (4) $RR = \exp[\beta(C_{obs} - C_{ME})]$

where β is an empirically determined coefficient specific to a disease category,

C_{obs} is the observed pollutant concentration, and

C_{ME} is the minimum exposure level (counterfactual).

Some U.S. and European models perform calculations of $PM_{2.5}$-related premature mortality from lung cancer or heart disease using a linear dose-response function from empirically derived relationships gathered from the American Cancer Society study (Krewski et al. 2009). With global modeling, however, observed concentrations may lie beyond the range of those evaluated in the American Cancer Society study. Extrapolation of this data to high concentrations may produce unrealistic values. In the WHO Comparative Quantification of Health Risks report, the authors addressed this issue by reviewing a large body of epidemiological studies of fine particulate effects with broader ranges of exposure values and chose to use a function that showed decreased response at higher concentrations (Cohen et al. 2004). This takes the form shown in Equation 5:

Equation (5) $RR = \left[\frac{C_{obs}+1}{C_{ME}+1}\right]^{\beta}$

where β is an empirically determined coefficient specific to a disease category,

C_{obs} is the observed pollutant concentration, and

C_{ME} is the minimum exposure level (counterfactual).

This function is used in the model for cardiopulmonary disease and lung cancer. Recent epidemiological analysis affirms the choice of these curves for cardiopulmonary effects (Pope et. al. 2011).

The PAF is calculated from the RR following Equation 6:

Equation (6) $PAF = \frac{RR-1}{RR}$

where PAF is the fraction of deaths in a given disease category attributable to $PM_{2.5}$ exposure, and

RR is the risk ratio for that disease category.

The PAF is then used for the final calculation of $PM_{2.5}$ health impacts as shown in Equation 7:

Equation (7) $E = PAF \times B \times P$

where

E is the number of premature mortalities in a given disease category associated with $PM_{2.5}$ exposure,

PAF for a given disease category is calculated as in Equation 6,

B is the baseline annual mortality rate for the given disease, and

P is the at-risk population.

The final values from Equation 7 require the use of data from sources external to this model. Baseline mortality rates (B) are made available from the WHO for 2004, 2008, 2015, and 2030. Mortality rates are interpolated for model years in which these statistics are not available. The at-risk population (P) is calculated as all urban residents within the relevant age category for a given disease. The populations of the cities included in the model are based on nationally reported data from 2000 and estimated for future model years based on national population growth rates. The percentage of the population within the relevant age categories is taken from UN population statistics (UNDESA 2012).

Effects Specific to On-Road Vehicle Emissions

The WHO methods described thus far enable estimates of premature mortality based on the total concentration of $PM_{2.5}$ within a city originating from vehicle emissions and other sources. Quantifying the specific impacts of on-road vehicle emissions requires some modification to the calculation. In the case of ARI this is relatively straightforward, as the linear dose-response function in Equation 4 is not dependent on background concentrations. The incremental change in risk from the addition of on-road vehicle emissions to ambient concentrations is calculated as shown in Equation 8:

Equation (8) $RR_T = \exp[\beta(\Delta C_T)]$

where

RR_T is the risk ratio specific to on-road transportation,

β is the empirically determined coefficient specific to a disease category, and

ΔC_T is the change in concentration due to on-road emissions.

RR_T can be used in Equations 6 and 7 to determine the number of premature mortalities associated with on-road transportation emissions.

Nonlinear dose-response functions complicate this calculation for lung cancer and cardiopulmonary disease because health benefits depend on total background concentration of fine particulates. If one assumes that an elimination of transportation-related emissions is the only source of change, referred to in the health model as the first-to-go (FTG) assumption, then the new risk ratio RR_{ftg} is calculated as shown in Equation 9:

Equation (9) $RR_{ftg} = \left[\frac{C_{obs} - \Delta C_T + 1}{C_{ME} + 1}\right]^{\beta}$

where C_{obs} is the total urban $PM_{2.5}$ concentration,

ΔC_T is the change in concentration due to on-road emissions,

C_{ME} is the minimum exposure level (counterfactual), and

β is the coefficient for the disease category.

If RR_{ftg} were used directly in Equation 6 to determine a new PAF_{ftg}, the PAF_{ftg} value would refer to a fraction of the baseline mortality occurring at the hypothetical concentration C_{obs}-ΔC_T rather than the observed concentration C_{obs}. Applying PAF_{ftg} in Equation 7 to estimate health endpoints would require a new value for baseline mortality rate B to reflect the change. A modification of Equation 6 that allows an estimate of the change in impact using the available observed mortality data is shown in Equation 10:

Equation (10) $$\Delta_{ftg}PAF = \left[\frac{RR_{ftg}}{RR} - 1\right]$$

The Δ_{ftg}PAF can then be used in Equation 7 to determine the change in the number of premature mortalities if transportation emissions were eliminated under FTG assumptions. Because of the decreasing impact of 1 µg/m^3 of $PM_{2.5}$ at high concentrations, the first-to-go change in PAF, Δ_{ftg}, is a low-end estimate.

At the other extreme, one could assume that all other anthropogenic sources of local air pollutants are eliminated, making on-road transportation emissions last-to-go (LTG). In this case, the effects of transportation-related fine particulate concentration C_T could be calculated as shown in Equation 11:

Equation (11) $$RR_{ltg} = \left[\frac{C_T + C_{ME} + 1}{C_{ME} + 1}\right]^{\beta}$$

where ΔC_T is the change in concentration due to on-road emissions,

C_{ME} is the minimum exposure level (counterfactual), and

β is the coefficient for the disease category.

In this case, the new PAF is calculated as shown in Equation 12:

Equation (12) $$\Delta_{ltg}PAF_{ltg} = \frac{RR_{ltg}}{RR}$$

where RR_{ltg} is the risk ratio calculated in Equation 11, and

RR is the risk ratio for all air pollution, calculated as in Equation 5.

PAF_{ltg} can then be used in Equation 7 to determine the change in premature mortalities under the LTG assumption. This produces a high-end impact estimate. One would expect premature mortalities to fall somewhere between the extremes of the FTG and LTG assumptions. To reflect that, the study reports premature mortalities attributable to on-road transportation emissions as the mean of the two calculations, with bars denoting the upper and lower extremes.

Table C- 5: Inputs to the Impacts Model

- emissions estimates under intervention and nonintervention scenarios
- value of a statistical life (VSL) in project country
- gross national income (GNI) per capita
- inflation rate
- discount rate
- life expectancy
- population size (historical and projected)
- average age of population
- baseline mortality parameters
- ambient $PM_{2.5}$ levels (baseline and intervention scenarios)
- intake fraction
- metric conversion factors for CO_2e

C.4. Economic Valuation

The cost-benefit model monetizes the carbon-equivalent emissions of black carbon (and co-pollutants of primary $PM_{2.5}$) as well as the health impacts caused by exposure to emissions of $PM_{2.5}$. The model can produce results in terms of net benefits as well as cost-effectiveness per life saved or carbon-dioxide-equivalent emissions reduced.

Health Valuation

To perform a cost-benefit analysis, the value of a statistical life (VSL) is estimated for the region in question using willingness-to-pay studies conducted in the United States and applying a benefit transfer to convert these results to other regions. VSLs are bounded by estimates of projected future income and consumption. Analysis is conducted and aggregated over long time horizons, based on health impacts distributed with a mortality lag adjustment.

VSL is the tool that enables the monetization of health benefits. It "reflects the aggregation of individuals' willingness to pay for fatal risk reduction and therefore the economic value to society to reduce the statistical incidence of premature death in the population by one."[18] It is not meant to represent the value of any individual person's life but to provide a way for quantifying societal benefits of a regulation or program that could result in reduced premature mortality. Methods adopted generally follow those used in two recent cost-benefit analyses of potential health benefits resulting from transportation-relevant policies and programs in developing countries (ICF International 2009; Abt Associates 2011). Three health valuation approaches are generally considered by analysts: cost of illness, monetized quality-adjusted life years (QALYs), and willingness to pay (WTP) (Hammitt 2000). WTP, which represents one's willingness to pay for statistical reduction in risk of mortality or illness/morbidity from a particular activity or exposure by capturing the trade-off between income and health improvements, remains the most widely adopted method for cost-benefit analysis.

In addition to using the mean value of VSL calculated from WTP studies conducted in the target country, a benefits transfer approach using the most widely cited base VSL derived from a U.S. hedonic wage study (Viscusi 2004) is applied. Its model allows for the option to apply the median VSL from a meta-analysis of WTP studies across the world adjusted for U.S. income at time of study (Viscusi and Aldy 2003; see Table C-6). Since VSL is inherently constrained by one's ability to pay (personal income), one can apply the following equation to adjust for income growth and inflation between the year for which the base VSL is reported and 2010 U.S. dollars for both national and U.S. VSLs (Robinson 2008):[19]

Equation (13) $$VSL_{t+n} = VSL_t(1+i)(1+g)^e$$

where

VSL_t and VSL_{t+n} represent the VSLs at time t (the year base VSLs are reported) and $t+n$ (the year for which monetary values are reported for the current study), respectively;

i is the inflation rate in real dollars between years t and $t+n$[20]; g is the gross national income (GNI) purchasing power parity (PPP) per capita growth rate between t and $t+n$; and

e is the income elasticity for VSL using the recommended value of 0.47.

[18] He and Wang (2010).

[19] All reported values are expressed in 2011 U.S. dollars, unless otherwise indicated.

[20] Calculated using the U.S. Consumer Price Index (CPI) for inflation between base VSL year and current year (http://www.bls.gov/data/inflation_calculator.htm).

The resultant VSLs in 2010 dollars are approximately \$6.7 million (Viscusi 2004) and \$9.7 million (Viscusi and Aldy 2003).

Table C- 6: Input Data and Sources for VSL Estimates and Benefit Transfers

Variable	Value	Year	Source
VSL (1)	6.3M U.S.$	2007	Viscusi (2004).
VSL (2)	6.7M U.S.$	2000	Viscusi and Aldy (2003); Meta-analysis of global WTP (revealed and stated preference) studies.
VSL (3)	----		National WTP study.
Gross National Income (GNI) per Capita Purchasing Price Parity (PPP)	International $	2010	World Bank International Comparison Program Database.
Income Elasticity for U.S.	0.47		Viscusi and Aldy (2003).
Income Elasticity for Benefit Transfer to Target Countries	1		World Bank staff.
Life Expectancy at Birth			World Health Organization Global Health Observatory Data Repository.
Life expectancy at Average Age			World Health Organization Global Health Observatory Data Repository.
Expenditures per Capita			World Bank International Comparison Program Database.
Discount Rate	7%		EPA Clean Air Act Retrospective; U.S. Office of Management and Budget Circulator on Regulatory Impact Analysis Guidelines.
GNI per Capita Annual Compound Growth Rate			World Bank International Comparison Program Database.

Benefit Transfer

Empirical data are the gold standard by which a VSL for any country should be derived. Frequently, empirical data are not available, and estimates must be made based on studies conducted in other countries. The method for deriving VSL for a country from data collected elsewhere is called benefit transfer. The benefit transfer approach attempts to adjust VSL using all available information that explains differences between countries. To date, no reliable evidence other than income has been used in benefit transfer. It is, however, plausible to adjust for differences using other information as well, including type of fatality (e.g., workplace injury versus air pollution injury) or the age of affected individuals.

Benefit transfer using income assumes that VSL differences between countries are entirely attributable to income differences and that this relationship can be described by empirically derived income elasticity (Hammitt and Robinson 2011), as in equation 14:

Equation (14)

$$VSL_b = VSL_a \left(\frac{GNI_b}{GNI_a} \right)^e$$

where

VSL_a and VSL_b represent the VSLs for country *a* (the United States) and *b* (the target country);

GNI_a and GNI_b are the gross national incomes (GNI) per capita for the same countries *a* and *b*; and

e is the income elasticity.

For all calculations requiring income estimates, GNI per capita at purchasing power parity (PPP) in current international dollars was selected, since this is the measure favored by the World Bank as the most accurate representation of well-being in monetary terms for international comparisons.[21] Growth rates reflect an average from 1995-2009 for the country of analysis.

Many of the U.S. WTP studies found income elasticities smaller than 1.0, but several similar studies conducted in developing countries revealed elasticities greater than 1.0 and closer to 2.0 (ICF International 2009; Hammitt and Robinson 2011). Based on guidance from World Bank staff, this cost-benefit analysis uses an income elasticity of 1.0.

Mortality Lag Adjustments

Manifestations of negative health consequences (including mortalities) from increased emissions for a given year are distributed among subsequent years. By the same logic, only a fraction of health benefits from reduced emissions experienced in any given year are observed the same year. This is an important consideration in health valuation because of the nature of social and consumer discounting. A premature mortality avoided today is worth more than a premature mortality avoided in the future. As such, this study follows the recommendation of the U.S. EPA's Advisory Council on Clean Air Compliance Analysis to use a 20-year mortality lag structure with the following distribution: 30 percent of Year 1's estimated health benefits can be ascribed to Year 1, 50 percent equally allocated in Years 2-5, and 20 percent equally allocated in Years 6-20 (U.S. EPA Science Advisory Board 2004).

Bounded VSLs Using Discounted Total Income and Consumption

An economic reality check is often recommended for benefit transfer estimates of VSLs, especially for low-income countries and when using elasticities greater than 1.0. According to economic welfare theory, the VSL cannot realistically fall below the present value of future lifetime income or expenditure. Extrapolated VSLs that are less than these values should be replaced by estimates of future lifetime income. To do so, one determines the average age of the population by taking the median age of life expectancy at birth. This age is then used in the determination of life expectancy at the average age (Hammitt and Robinson 2011; Viscusi 2004). Assuming constant income and consumption, the net present value (NPV) is calculated as follows in equation 15:

Equation (15)

$$NPV = P\left(\frac{1-(1+r)^{-n}}{r}\right)$$

where

P is the principal value (GNI-PPP per capita or expenditures per capita);

r is the discount rate; and

n is the number of years.

While the range often reported for discount rates is 3-7 percent, a discount rate of seven percent in developing economies is applied since the social rate of time preference often results in a discount rate closer to and often in excess of seven percent (Hammitt and Robinson 2011).

VSL Estimates with Projected Income Growth

[21] Acquired from the World Bank International Comparison Program Database.

To adjust the VSL for projected increases in income during the time horizon of the study (from base year *t* to future year *t+n*), equation 16 (Abt Associates 2011) can be applied:

Equation (16) $$GNI_{b,t+n} = GNI_{b,t}(1+g_a)^{ne}$$

Where

$GNI_{b,t}$ is the calculated or observed income of country *b* in year t;

$GNI_{b,t+n}$ is the projected income of country *b* after *n* years;

g_a is the annual GNI-PPP per capita growth rate[22];

n is the number of years since year *t*; and

e is the income elasticity for the VSL (Table H-5). This elasticity is the same used to transfer benefits between the U.S. and study countries, assuming individual preferences remain constant during the time horizon considered.

Sensitivity Analyses and Uncertainties

In this framework, the median value for each country uses parameters that are the most widely accepted and empirically supported for developing countries: elasticity = 1.0, VSL = $6.7 million (in 2007 U.S. dollars), and discount rate = 7 percent. All other permutations of elasticities (1.5, 2.0), U.S. and target-country-derived VSLs, and discount rates (3 percent) can also be presented as a range, demonstrating both the sensitivity to and uncertainty involved in these parameters. Future work may wish to evaluate the sensitivity of estimates to the present value of lost earnings as an alternative benefits measure.

Final Calculations for Monetizing Health and Climate Benefits

Annual benefits are calculated by multiplying the mortality-lag-adjusted number of deaths avoided each year with the VSL estimated from income projections for the same year. Annual health benefits are reported as the net present value in 2011 U.S. dollars using a 7-percent discount rate. The cumulative benefit is simply the sum of annual benefits over the time horizon of the study. Net benefit, as a measure of economic "reasonableness," is the difference between cumulative benefit and total cost of implementing the policy (Wilhelmine et al. 2006).

Cost-effectiveness is an additional calculation option given in the model and represents the cost of implementing the policy per premature mortality avoided (dollars/life saved)—either cumulative or annual cost and mortality avoided—and is used in parallel to identify the most efficient and "least burdensome" of reasonable policy options.

Climate benefits are calculated by multiplying emissions of CO_2e in each year with a corresponding social cost of carbon value estimated in that year. Values applied for the social cost of carbon are given in Appendix D. Based on Yohe et al. (2007), an annual inflation rate of 2.5 percent is applied to estimate the social cost of carbon (SCC) in future years. It may be appropriate to adjust the SCC for real income growth but not for inflation (unless all other quantities are adjusted - in particular the mortality values). In addition, if nominal rather than real dollars are used, it may be appropriate to use a nominal rather than a real discount rate. These issues should be explored in future research.

[22] Annual compound growth rates estimated from 2000-2009 using a World Bank database applying the following equation: $r = \left(\frac{x_2}{x_1}\right)^{\frac{1}{t_2 - t_1}}$ $r = \left(\frac{x_2}{x_1}\right)^{\frac{1}{t_2 - t_1}}$ $r = \left(\frac{x_2}{x_1}\right)^{\frac{1}{t_2 - t_1}}$ $r = \left(\frac{x_2}{x_1}\right)^{\frac{1}{t_2 - t_1}}$ where r is the growth rate, x_1 and x_2 are population or income values and t_1 and t_2 are years for which x_1 and x_2 are reported.

Table C- 7: Inputs to the Cost-Benefit Analysis Model

- VSL for U.S. (or target country) income
- VSL year
- GNI PPP for the U.S. (or target country)
- Income growth elasticity for the U.S. (or target country)
- GNI per capita for target country
- GNI per capita PPP for target country
- Life expectancy at birth for target country
- Life expectancy at average age in target country for current year
- Expenditures per capita (actual individual in current international $ at PPP)
- Social cost of a metric ton of carbon-dioxide-equivalent in current and future years

C.5. References

Abt Associates (2011). Quantifying the Health Benefits of the Greater Dhaka Sustainable Urban Transport Corridor Project.

Apte, J. S., Bombrun, E., Marshall, J. D., & Nazaroff, W. W. (2012). Global Intra-urban Intake Fractions for Primary Air Pollutants from Vehicles and Other Distributed Sources. *Environmental Science and Technology*, *46*(6), 3415–3423. doi:doi: 10.1021/es204021h.

Bennett, D.H., McKone, T.E., Evans, J.S., Nazaroff, W.W., Margni, M.D., Jolliet, O., Smith, K.R., 2002. Defining Intake Fraction. Environmental Science and Technology 36, 206A–211A.

Bond, T. C., Zarzycki, C., Flanner, M. G., and Koch, D. M. (2011). Quantifying Immediate Radiative Forcing by Black Carbon and Organic Matter with the Specific Forcing Pulse. *Atmospheric Chemistry and Physics* 11 (4): 1505–25. doi:10.5194/acp-11-1505-2011.

Bond, T. C., Doherty, S. J., & Fahey, D. W. (2013). Bounding the Role of Black Carbon in the Climate System: A Scientific Assessment. *Journal of Geophysical Research*. doi:10.1002/jgrd.50171.

Cohen, A., Anderson, H., Ostro, B., Dev Pandey, K., Kryzanowski, M., Kunzli, N., et al. (2004). Urban Air Pollution. In M. Ezzati, A. D. Lopez, A. Rogers, & C. J. L. Murray (Eds.), *Comparative Quantification of Health Risks* (Vol. 2, p. 1353). Geneva: World Health Organization.

Ezzati, M., Lopez, A. D., Rodgers, A., & Murray, C. J. (2004). *Comparative Quantification of Health Risks: Global and Regional Burden of Disease Attributable to Selected Major Risk Factors*. OMS.

Forster, E. E., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R. A., Fahey, D. W., Haywood, J., Lean, J., Lowe, J. A., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schultz, M. G., and Van Dorland, R. (2007). Changes in Atmospheric Constituents and in Radiative Forcing. In Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, U.K.: Cambridge University Press, p. 6.

Hammitt, J.K., 2000). Valuing Mortality Risk: Theory and Practice. *Environmental Science and Technology* 34 (8): 1396–1400.

Hammitt, J. K., and Robinson, L. A. (2011). The Income Elasticity of the Value per Statistical Life: Transferring Estimates between High and Low Income Populations. *Journal of Benefit-Cost Analysis* 2 (1): Article 1.

Hart Energy and MathPro (2012). Draft Technical and Economic Analysis of the Transition to Ultra-Low Sulfur Fuels in Brazil, China, India, and Mexico. Phase 2 Report: Results of the Refinery Modeling Analysis and Additional Technical Material. International Council on Clean Transportation.

He, J. and Wang, H. (2010). The Value of Statistical Life. World Bank. http://elibrary.worldbank.org/content/workingpaper/10.1596/1813-9450-5421.

ICF International (2009). Final Report: Sub-Saharan Africa Fuel Refinery Project. Report prepared for the World Bank and the African Refiners Association.

Krewski, D., Jerrett, M., Burnett, R., Ma, R., Hughes, E., Shi, Y., Turner, M., Pope, C. A. (III), Thurston, G., Calle, E. E., Thun, M. J., Beckerman, B., DeLuca, P., Finkelstein, N., Ito, K., Moore, D., Newbold, K., Ramsay, T., Ross, Z., Shin, H., and Tempalski, B. (2009). Extended Follow-up and Spatial Analysis of the American Cancer Society Study Linking Particulate Air Pollution and Mortality (HEI Research Report No. 140). Health Effects Institute, Boston.

Marshall, J. D. Urban Land Area and Population Growth: A New Scaling Relationship for Metropolitan Expansion. Urban Stud. 2007, 44, 1889−1904.

Ostro B. (2004) Outdoor Air Pollution: Assessing the Environmental Burden of Disease at National and Local Levels. Geneva, World Health Organization (WHO Environmental Burden of Disease Series, No. 5).

Pope, C. A. (III), Burnett, R. T., Turner, M. C., Cohen, A., Krewski, D., Jerrett, M., Gapstur, S. M., and Thun, M. J. (2011). Lung Cancer and Cardiovascular Disease Mortality Associated with Ambient Air

Pollution and Cigarette Smoke: Shape of the Exposure-Response Relationships. *Environmental Health Perspectives* 119 (11): 1616–21.

Reynolds, C., and Kandlikar, M. (2008). Climate Impacts of Air Quality Policy: Switching to a Natural Gas-Fueled Public Transportation System in New Delhi. *Environmental Science and Technology* 42 (16): 5860–65.

Robinson, L. A. (2008). Valuing Mortality Risk Reductions in Homeland Security Regulatory Analyses. Report prepared for U.S. Customs and Border Protection, Department of Homeland Security, June.

U.S. Environmental Protection Agency, Science Advisory Board (2004). Advisory Council on Clean Air Compliance Analysis: Response to Agency Request on Cessation Lag.

US Environmental Protection Agency (2009). Exposure Factors Handbook: 2009 Update (External Review Draft); EPA/600/R-09/052A; Washington, DC, 2009; http://cfpub.epa.gov/ncea/cfm/recordisplay. cfm?deid=209866.

UNDESA (2012). Monitoring and Global Population Trends: Population Division. United Nations Department of Economic and Social Affairs. http://www.un.org/en/development/desa/population/.

Viscusi, W. K. (2004). The Value of Life: Estimates with Risks by Occupation and Industry. *Economic Inquiry* 42 (1): 29–48.

Viscusi, W. K., and Aldy, J. E. (2003). The Value of a Statistical Life: A Critical Review of Market Estimates throughout the World. *Journal of Risk and Uncertainty* 27 (1): 5–76.

Wilhelmine Miller, Robinson, L., Lawrence, R.S (2006) Committee to Evaluate Measures of Health Benefits for Environmental, Health and Safety Regulation (2006). *Valuing Health for Regulatory Cost-Effectiveness Analysis*. The National Academies Press.

Yohe, G.W., R.D. Lasco, Q.K. Ahmad, N.W. Arnell, S.J. Cohen, C. Hope, A.C. Janetos and R.T. Perez, 2007: Perspectives on Climate Change and Sustainability. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden and C.E. Hanson, Eds., Cambridge University Press, Cambridge, UK, 811-841.

10 Appendix D: Monetization of Costs and Benefits

D.1. Methods for Assessing Policy Effectiveness

Policies to reduce diesel black carbon may require large capital investments in refinery upgrades, natural gas refueling infrastructure, vehicle upgrades, retrofits, and other measures. These would be expected to produce certain benefits, including a reduction in global temperature change as well as declines in transportation-related premature mortality. Methods to compare the expected benefits of policies against their costs are useful to the policy community for evaluating effectiveness. This information can play an important role in clarifying whether a policy should be adopted or rejected.

Two commonly applied analytical tools for evaluating costs and benefits are Cost-Benefit Analysis (CBA) and Cost-Effectiveness Analysis (CEA; see Figure E-1). Both of these are applicable to an assessment of diesel particulate matter controls.

CBA monetizes the benefits of a policy (health improvements or other associated outcomes) and compares this to the estimated cost of implementing that policy. Of the two frameworks, its analytical approach is more consistent with economic utility theory, as both costs and benefits are aggregated in such a way as to reflect *net* individual and social well-being. Because the outcome is net benefits, the decision criteria are that a policy is “reasonable” if benefits exceed costs, and the “least burdensome” or preferred choice among many policies is the one with the largest net benefits (Wilhelmine et al. 2006). CBA has historically been the preferred analytical tool of regulatory agencies in the United States, including the Environmental Protection Agency. in addition, monetizing health benefits allows policymakers to incorporate nonhuman health benefits (e.g., ecosystem health, infrastructure damage). The most salient disadvantages of CBA are the ethical reservations about monetizing all benefits and the failure to account for distributional effects, with a related qualm being the limitations in metrics that lead to undervaluation of human health impacts. Nonetheless, CBA remains a critical part of the decision-making process since regulatory adoption is often constrained by budgetary allocation.

In CEA**,** the cost required to implement a policy is presented per unit of “benefit” as a ratio—for example, of total cost of an air pollution abatement technology per disease or mortality avoided (dollars per case of chronic bronchitis or per quality-adjusted life year). CEA also shares a theoretical consistency with CBA, as it uses opportunity costs as a basis for evaluation--though one must be careful not to interpret CEA as representing economic efficiency in terms of social welfare since a smallest dollar cost does not necessarily generate lowest welfare loss (Krupnick 2004). Additionally, the CEA decision criteria only enables a relative comparison of policies to identify the least burdensome policy. Reasonableness (that is, whether a policy is worthwhile) can only be established if regulators apply a CEA threshold for policy action (Wilhelmine et al. 2006), in which case CEA and CBA would be highly comparable, with the added benefit that CEA enables a better assessment of changes stemming from incremental changes in policy. CEA guidelines have primarily been developed for health care policies and medical expenditures and not for regulatory purposes. Because the benefits measure is often a health index and non-monetary in nature, CEA is more palatable to those who wish to avoid monetizing health and safety impacts.

Both frameworks are founded in economic efficiency and considered “complementary measures of outcomes” by the Institute of Medicine and the National Academy of Sciences (Wilhelmine et al. 2006). Only a handful of published studies exist, however, that examine whether CBA and CEA result in systematic differences in recommendations or decision outcomes, and these studies are restricted to analyses of health services and medical treatments. Since the most widely accepted and valid CEA studies rely on health state indices, known broadly as quality-adjusted life years (QALYs), and CBA studies rely primarily on willingness-to-pay measures (WTP), the choice between CBA and CEA must necessarily consider the trade-offs between WTP and QALYs (Krupnick 2004). It is also possible, however, to quantify benefits simply as number of lives saved or cases of illness avoided. Keeping best practices in mind, the major differences between CEA and CBA are (1) the choice between a monetary or health index/effectiveness measure; and (2) the social equity of welfare gains.

Willingness to Pay (WTP) versus Quality-Adjusted Life Years (QALYs) in Comparing CBA and CEA

The criteria for assessing health valuation measures can generally be thought of as either policy-focused or technical in nature (Krupnick 2004). QALYs are integrated index measures for the health-related quality of life (HRQL) impacts of different conditions and incorporate both mortality and morbidity into one measure. Unlike other single-dimension measures of public health, such as mortality, QALYs also consider trade-offs between different health states and longevity. WTP[23] measures one's willingness to pay for a statistical reduction in risk of mortality or illness/morbidity from a particular activity or exposure by capturing the trade-off between income (which can be used to derive utility from other goods) and health improvements. For mortality, WTP is also known as the value of a statistical life (VSL), which is the total amount a society would be willing to pay to avoid the risk associated with causing one "statistical" death. For example, if stated preference studies yielded a WTP of $5 to reduce risk of dying by 1/10,000, then the VSL would be $50,000.

The Policy Approach

Much of the controversy over the choice of CBA or CEA arises from ethical and equity issues between QALYs and WTP. Because WTP measures are inherently constrained by one's *ability* to pay (income level), there is strong concern that CBA has a bias toward programs and policies that benefit the prosperous over the poor. The income independence of QALYs assumes that health preferences and consumption are or can be separate. However, QALYs can be affected by the distribution of income as well (life expectancy differences between those who have higher or lower incomes) (Hammitt 2000; Wilhelmine et al. 2006). For example, the utility consequences of a health impairment could depend on one's ability to mitigate it, which might depend on one's economic circumstances. Some question whether the income distribution effect is actually smaller in CEA. Another equity concern with QALYs is that they inherently value healthier and younger individuals more. Like income disparities, this can be corrected for, but income adjustments do not correct for discrimination between nations with different mean incomes.

Because QALYs have gained general acceptance among health professionals and academics and WTP/VSLs are preferred by environmental economists, palatability of and familiarity with a certain outcome (net benefits as measured using VSLs or cost per benefit) are also of importance. Besides professional and academic biases for certain approaches, cultural differences between countries may make the choice of valuation a case-by-case consideration. Some policymakers may consider it unethical to "value" human life, even if this concern is largely the result of misunderstanding what exactly is being valued in VSL studies (Hammitt 2002). Even under the CEA approach, a decision must still be made regarding what cost per QALY is acceptable or excessive, and this requires some determination regarding how much should be spent to save each life.

[23] WTP can also be defined as the rate of substitution between income/wealth and whatever good is being valued. It is supposed to represent the individual's preference for more income versus more of the good being valued.

The Technical Approach

Since CBA and CEA are *economic* valuations and are tools to judge economic efficiency, health and environmental economists have often considered whether an approach is "utility based," that is, consistent with affected individuals' preferences within the subfield of welfare economics (Hammitt 2002). Between stated and revealed preference measures, stated preference (contingent) is beginning to gain more credibility since the market behavior revealed in wage-risk studies has been criticized as being too dissimilar from other associated health risks being considered. For QALYs, there are a number of methods for arriving at an index for a health condition, and both the standard gamble method and the trade-off methods are consistent with assessing utility and risk, conditional on individual preferences being consistent with QALYs (Hammitt 2002). Because QALYs and the stated preference WTP studies rely on similar survey methodologies, they share similar limitations, although WTP surveys are often viewed as of higher methodological quality with respect to defining realistic, plausible valuation questions (Krupnick 2004).

Another consideration is the parsimony of an approach or the estimation of a measure. Simplicity and transparency are desirable qualities in any model but must be balanced by building in enough of the mechanisms that best capture reality. Bulky models with many inputs can increase the level of uncertainty or lead to over-fitting of the model. It may be best to avoid too many corrections or adjustments that unnecessarily fine-tune for every consideration, especially if they have ambiguous directional impacts on VSL (e.g., health status, income, age distribution, discount rates, time lag, life span, cultural differences), as they may introduce more error into VSL estimates. For example, to monetize QALYs for purposes of comparing environmental damages, one must apply WTP techniques of annualizing VSL that are not widely practiced among regulatory agencies and not recommended by the OMB (Krupnick 2004; Wilhelmine et al. 2006).

Outcomes of sensitivity tests to inputs with large ranges of uncertainty can also be considered in choosing a health measure since it is often better to use metrics that are more robust against assumptions. For example, many studies have found that VSL is fairly invariant to discount rate (U.S. EPA 2011; Muller et al. 2011), whereas the EPA's Clean Air Act retrospective and prospective analyses showed strong sensitivity of VSL to probable income elasticities (U.S. EPA 2011). QALYs and WTP can both in theory provide a comprehensive picture since each measure can include several health statuses. One drawback of the WTP is that it has been elicited for only a very small number of morbidities. Still, WTP can include more than just health (i.e., to aggregate analyses for gross economic damages as done in many environmental economics studies; see Muller et al. 2011). Additionally, it is firmly agreed upon that WTPs better capture qualitative aspects of risk and preferences for both acute and chronic health effects (which QALYs, by definition, cannot). This could be a disadvantage of WTP as well, for estimates may be overly influenced by these concerns.

Finally, there are practical considerations of data availability. Thanks to the predominance of VSLs available from WTP wage-risk studies conducted using blue-collar employee data, many CBA studies have methodologies for which more and better information is readily available (ICF International 2009). If CEA is used in lieu of CBA, however, a threshold value would have to be identified in order to establish reasonableness. CEA ratios would otherwise need to be compared to adopted policies and regulations considered successful in order to determine whether or not the cost justifies the life years gained.

A key and summary consideration arising from these comparisons between WTP and QALYs is whether one is seeking to undertake a largely *economic* valuation or a *public health* valuation. Another emergent difference is a health versus utility effectiveness--that is, QALYs that appropriately integrate across all health effects can better measure, in aggregate, the public health impacts resulting from a regulation while WTP is preferred for assessing social welfare (not restricted to health). In either case, it may be more valuable to regulators to see some quantification of each effect separately rather than in the aggregate so as to understand the component parts that make up the whole.

D.2. Determining the Value of a Statistical Life

VSLs used in regulatory analyses are predominantly derived from meta-analyses of several WTP studies and are often a combination of contingent (stated preference, surveyed responses for each health endpoint) and revealed preference (wage-risk or market) studies. The mean or best value of these individual studies is used to model a statistical distribution, from which the mean meta-analysis VSL is derived. Certain studies have questioned the dataset and statistical methods used in such meta-analyses (ICF International 2009).

Oftentimes, CBA studies incorporate a range of available VSLs that use different correction factors for non-risk variables in both stated and revealed preference studies. Since VSLs are income dependent, income elasticities must be applied to VSL estimates derived from studies conducted in countries with different average incomes. ICF International's CBA for a Sub-Saharan African refinery project used a VSL derived from a meta-analysis of 16 WTP studies conducted across a range of developed and developing countries, and applied a range of income elasticities. An ICCT CBA study examining low-sulfur fuel in China derived a VSL from three contingent WTP studies conducted in mainland China and applied a 1.42 elasticity for mortalities (Blumberg et al. 2006; see Table D-3 for VSLs and elasticities of discussed studies). Hammitt and Robinson (2011) recommend using higher elasticities when extrapolating VSLs for low-income countries from studies conducted in high-income countries.

There are a number of population characteristics that should be considered in VSL estimates. Generally, the choice is between use of VSL from the WTP studies in countries with income levels most similar to the study population in question and WTP meta-analysis studies from the United States and other high-income countries, which have the advantages of completeness and superior sample size. Other adjustments with associated ranges of uncertainty include choice of discount rate (2-7 percent, though usually 3 percent) and application of an age distribution to account for age differentiation in the valuation of risk.

In cost-effectiveness analysis, the ratio of costs to benefits can be estimated with few additional steps. This representation can take the form of dollars per premature deaths avoided, or whichever benefit is of primary interest. A determination of policy effectiveness then requires a comparison against the cost-effectiveness of policies that have been implemented and a determination as to the reasonability of pursuing the policy in question.

One limitation of cost-effectiveness analysis is that multiple benefits cannot be integrated into a single benefit value. In the case of ultralow-sulfur diesel fuel, premature deaths avoided and climate impacts avoided via black carbon reduction must be presented as separate cost-effectiveness estimates and evaluated against projects independently.

In cost-benefit analysis, multiple benefits can be integrated to produce a single estimate of net benefits. This requires weighting factors specific to each benefit type, then aggregating the economic value of these benefits to society. Finally, a ratio of benefits to costs is presented as an estimate of net benefits.

D.3. Methods for Monetizing Climate Benefits from Black Carbon Emissions Reductions

An estimate of the social cost of carbon, also called the marginal damage cost, is necessary to monetize the benefit to society of a reduction in black carbon emissions. The marginal damage cost is defined as the net present value of the incremental damage attributable to a small increase in black carbon emissions. This value represents a price that would need to be placed on black carbon in order to internalize the externalities caused by its emission and to restore the efficiency of the market (Tol 2009)

The marginal damage cost requires a series of modeling steps. These includes projections of future emissions, simulations of atmospheric chemistry and climate impacts, and modeling of economic damages, taking into account their variation in space and time. Baron et al. (2009) used the Dynamic Integrated Climate-Economy (DICE) model to estimate the net benefits of a reduction in black carbon emissions from cooking stoves. Based on an estimated reduction of 1,502 gigagrams of black carbon, or 1,502,000 metric tons, the authors estimated a climate benefit equal to $375 million. This values a metric ton of black carbon at $249, equivalent to a social cost of carbon of less than one dollar per metric ton, assuming a 100-year global warming potential value for black carbon of 790. That value seems to be far lower than what can be reasonably expected.

No other estimates of the marginal damage cost for black carbon were found in the literature; however a relatively large number of estimates for carbon dioxide have been made available.

Comprehensive studies informed by climate modeling have been conducted to estimate the present cost of carbon and to project it for future years. The World Resources Institute provided a summary of recent attempts to value the social cost of carbon (Greenspan Bell, and Callan 2011).

The United Kingdom has undertaken several reviews of the social cost of carbon, beginning with a 2006 report that provided a central estimate of $85 per ton of carbon dioxide (in 2000 dollars) (Stern 2007). More recently, the U.K. government re-estimated the social cost of carbon in terms of both a traded price for emissions that fall under the EU Emissions Trading Scheme and a non-traded price for emissions that do not (U.K. Department of Energy and Climate Change 2009). These are informed by binding targets across Europe and by the U.K. government. The central value for one traded metric ton of CO_2e was estimated at $34.10 ($19.50 to $42.30) in 2009 dollars, assuming an exchange rate of 1.625 dollars per pound. The central value for non-traded carbon dioxide was estimated at.$83.00 ($41.00 to $124.00) per metric ton of CO_2e. The 2009 total estimated damage cost per ton of carbon dioxide was given as 29 British pounds (or $51 in 2012 dollars). Values were estimated out to 2050 and were projected to rise over time.

In the United States, the Obama administration in 2009 established an interagency working group to set a social cost of carbon to be used in analyses of the regulatory impact of proposed regulations under a cost-benefit analysis framework. The working group recommended a range of values from $5-$35 (in 2007 dollars) per ton of carbon dioxide, with variations based on assumptions of a discount factor of 2.5-5 percent. Non-CO_2 greenhouse gases were not included in this analysis. The working group proposed a central estimate of $21, assuming a discount factor of three percent. That is equivalent to $23 in 2012 dollars.

None of the estimates presented by either the United States or the United Kingdom include an analysis of the climate impacts of black carbon. In the case of the United States, the social cost of carbon estimate was made only for carbon dioxide. The United Kingdom went one step further and included all regulated greenhouse gases, and so its valuation captures a broader set of climate effects.

The value given to offsets of carbon in trading markets has been lower than these estimates. On September 17, 2012, the Intercontinental Exchange reported a value of $9.70 (7.40

Euros) per metric ton of CO_2e for a December 2012 contract on EU allowances.[24] On September 14, 2012, the exchange reported a value of $15.65 per allowance of one metric tons of CO_2e on a December 2013 contract for California carbon allowances.[25] These values, subject to fluctuation based on market conditions, represent the price a regulated entity is willing to pay to offset one metric ton of carbon to comply with limits on emissions. This is affected by the overall emissions target set by regulation, the types of projects from which offsets are being generated, and the challenges faced by regulated entities in reducing their own emissions. Market prices do not represent the amount society is willing to pay to reduce the climate impacts of these emissions.

Yohe et al. (2007) summarized existing estimates of the social cost of carbon. Early calculations by the Intergovernmental Panel on Climate Change estimated the social cost at between $5 and $125 per metric ton in 1990 dollars. Later estimates by Clarkson and Deyes (2002) put a central estimate at $105 per ton in 2000 dollars, while Pearce (2003) arrived at an estimate ranging between $4 and $9 per ton of carbon, assuming a 3-percent discount rate. Tol (2005) reviewed more than 100 calculations in the literature and found that, among peer-reviewed estimates, the central value was $43 per metric ton with a standard deviation of $83. The author concluded that the social cost of carbon is unlikely to exceed $50 per ton. In contrast, Downing et al. (2005) concluded that $50 per ton of carbon represents a lower benchmark for reducing the threat of dangerous climate change.

Tol (2009) also conducted a more recent review of attempts to estimate the social cost of carbon. More than 200 estimates were identified in the literature. The variations in estimates had numerous causes, the most important of which was the rate at which future benefits and costs were discounted. Other factors included differences in projections of future emissions, warming rates, economic scenarios, and the treatment of uncertainty. Tol (2009) fitted these estimates to a Fisher-Tippett distribution using the mode and standard deviation of each study. Table D-1 gives these results. One metric ton of carbon was converted to carbon dioxide using a ratio of 12/44. Dollars in 1995 were converted to 2012 dollars using a factor of 1.52.

Table D- 1: Social Cost of Carbon (2012 U.S. dollars per metric ton of carbon dioxide).
Source: Tol 2009.

	Pure Rate of Time Preference		
	0%	**1%**	**3%**
Mean	61	50	21
Standard Deviation	64	61	25
33rd percentile	28	19	8
67th percentile	72	59	23
95th percentile	202	170	85

For a discount rate of three percent, the mean estimate of the social cost of carbon (dollars per ton of CO_2) is $21, which is in line with the U.S. Interagency Working Group's estimate of $23 (in 2012 dollars), assuming the same discount rate. Current market prices for carbon offsets are slightly lower.

For a discount rate of one percent , the social cost of carbon estimate is $50, which is in line with the U.K. government's estimate of $51 in 2012 dollars. Tol (2005) traces uncertainty in the estimate of the social cost to large differences in the choice of discount rate; Downing et al. (2005) reinforce the notion that differences in discount rates are as much to blame for variability in the social cost as uncertainty in climate impact models.

[24] https://www.theice.com/emissions.jhtml.
[25] http://online.wsj.com/article/SB10000872396390443779404577643592149738280.html.

Table D-2 gives estimates of the social cost of black carbon. This takes mean estimates of the social cost of carbon in Table D-1 above and converts them to black carbon using global warming potential weighting factors drawn from Bond et al. (2011). This approach is necessary in light of the absence of direct estimates of the social cost of black carbon.

Table D- 2: Social Cost of Black Carbon (2012 U.S. dollars per metric ton of black carbon). Source: Global warming potential values from Bond et al. 2011.

	Pure Rate of Time Preference	
	1%	**3%**
GWP-20	$140,000	$58,800
GWP-100	$39,500	$16,590

Yohe et al. (2007) suggest that the social cost of carbon should grow at an annual rate of 2.4 percent and that, for short-lived pollutants like methane, the rate should grow 50 percent faster. These features should be taken into consideration when conducting a cost-benefit analysis.

Uncertainties

This approach to deriving a social cost of black carbon is subject to some methodological uncertainty. The GWP is a weighting function that weights all years between zero and the time horizon as equal, then zero for years thereafter. This “rectangular discounting” is fundamentally different from the standard discounting used in health impact analysis. This also neglects the long term effects of CO_2 (or the "persistent tail" of the response.)

The limitations of the rectangular discounting of the GWP have been known for some time (Hammitt et al. 1996). Future improvements can be made by studies that directly estimate a social cost of black carbon, which is preferred so as to limit methodological uncertainty.

An alternative approach would apply a weighting factor that is based on temperature response and a damage function. Such estimates have been published for non-CO_2 greenhouse gases (Anthoff et al. 2011) but not for black carbon. Uncertainties in the damage function can negate the benefit of using this alternative metric, so care should be taken in how it is applied.

Table D- 3: Value of a Statistical Life Estimates Taken from the Literature

Study/Analysis	Values	Elasticity	Original Source	Notes
U.S. EPA (2012) (BenMAP)	6.3M (2000 U.S. dollars)		Mean of distribution of VSLs from 26 studies (5 contingent, 20 wage); methodology and studies “mirror” Viscusi (1992).	Applied equally across ages (which yields a higher VSL).
	5.5M (2000 U.S. dollars)		Mrozek and Taylor (2002), using $200 for each additional 1/ 10,000 chance of an accidental death occurring within one year (2000 U.S. dollars).	From meta-analysis of 33 studies that supposedly better controls for unobserved determinants of wages (in hedonic wage studies).
	5.5M (2000 U.S. dollars)		Viscusi and Aldy (2003) who suggest income elasticity of 0.5-0.6.	Median VSL of 6.7 million (2000 U.S. dollars).
U.S. EPA (1999) (Clean Air Act Analysis)	4.8M (1990 U.S. dollars)	0.4 (0.08 to 1)	Mean estimated from Weibull distribution using 26 studies (5 contingent, 21 wage).	Standard deviation of 3.2 million U.S. dollars. Applied equally across ages (to differentiate by age violates federal codes).
ICF International (2009)	6.3M (2007 U.S. dollars)	1.5 (1-2)	Viscusi (2004) hedonic wage study.	From evaluation of literature including WTP studies for 17 ‘low income’ countries, but settled on highest-quality U.S. study.
Muller et al. (2011)	6M (2-10M, 1990 U.S. dollars)		Mean of 28 studies reviewed and given in U.S. EPA (1999). Lower value from Mrozek and Taylor (2002), a meta-analysis of revealed-preference methods; upper value from Viscusi and Moore (1989).	

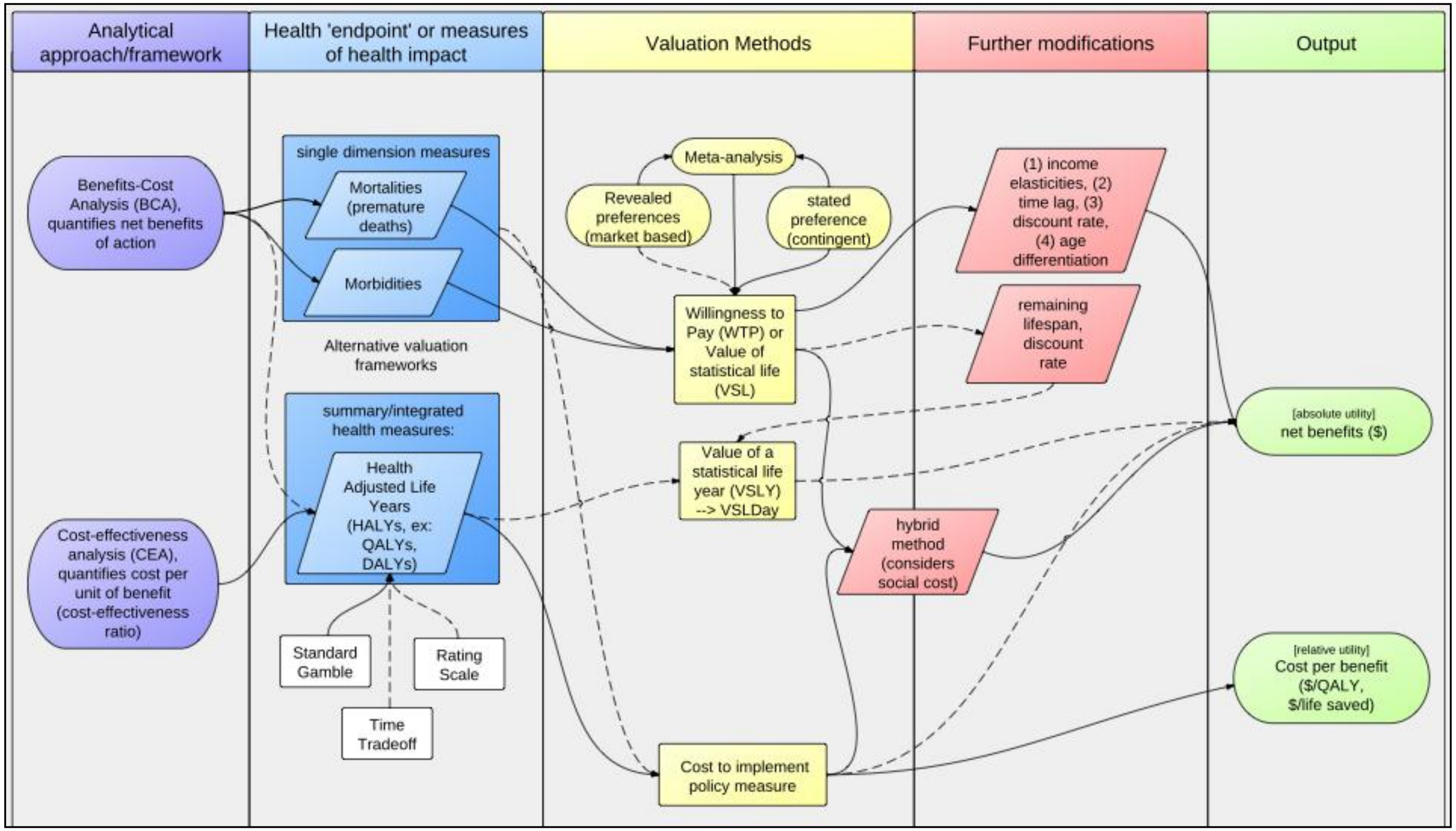

Figure D- 1: Cost-Benefit and Cost-Effectiveness Frameworks for Evaluation of Policies. Aspects of each analytical framework and available measures for health endpoint valuations are shown. Dotted lines show practices that are particularly problematic and, according to environmental and health economists, not defensible.

D.4. References

Anthoff, David & Rose, Steven K. & Tol, Richard S. J. & Waldhoff, Stephanie (2011) "The Marginal Damage Costs of Different Greenhouse Gases: An Application of FUND," Paper WP380, Economic and Social Research Institute (ESRI).

Baron, R. E., Montgomery, W. D., and Tuladhar, S. D. (2009). An Analysis of Black Carbon Mitigation as a Response to Climate Change. Copenhagen, Denmark: Copenhagen Consensus Center. Retrieved from http://ftc_dev.planck.mocsystems.com/uploads/tx_templavoila/AP_Black_Carbon_Baron_Montgomery_Tuladhar_v.4.0.pdf.

Blumberg, K., He, K., Zhou,Y., Liu, H., and Yamaguchi, N. (2006). Costs and Benefits of Reduced Sulfur Fuels in China. International Council on Clean Transportation.

Bond, T. C., Zarzycki, C., Flanner, M. G., and Koch, D. M. (2011). Quantifying Immediate Radiative Forcing by Black Carbon and Organic Matter with the Specific Forcing Pulse. *Atmospheric Chemistry and Physics* 11 (4): 1505–25. doi:10.5194/acp-11-1505-2011

Clarkson, R. and K. Deyes (2002), Estimating the Social Cost of Carbon Emissions, The Public Enquiry Unit - HM Treasury, London, Working Paper 140.

Downing, T.E., D. Anthoff, R. Butterfield, M. Ceronsky, M.J. Grubb, J. Guo, C.J. Hepburn, C.W. Hope, A. Hunt, A. Li, A. Markandya, S. Moss, A. Nyong, R.S.J. Tol, and P. Watkiss (2005), Social Cost of Carbon: A Closer Look at Uncertainty, Department of Environment, Food and Rural Affairs, London.

Greenspan Bell, R., & Callan, D. (2011). *The Social Cost of Carbon in U.S. Climate Policy, in Plain English* (pp. 1–16). Washington, DC: World Resources Institute. Retrieved from http://pdf.wri.org/more_than_meets_the_eye_social_cost_of_carbon.pdf.

Hammitt, J. K., Jain, A. K., Adams, J. L., & Wuebbles, D. (1996). A Welfare-based Index for Assessing Environmental Effects of Greenhouse-gas Emissions. *Nature*, *381*(6580), 301–303.

Hammitt, J. K., and Robinson, L. A. (2011). The Income Elasticity of the Value per Statistical Life: Transferring Estimates between High and Low Income Populations. *Journal of Benefit-Cost Analysis* 2 (1): Article 1.

ICF International (2009). Final Report: Sub-Saharan Africa Fuel Refinery Project. Report prepared for the World Bank and the African Refiners Association.

Krupnick, A. J. (2004). *Valuing Health Outcomes* (pp. 1–107). Resources for the Future. Retrieved from http://www.rff.org/RFF/Documents/RFF-RPT-ValuingHealthOutcomes.pdf.

Mrozek, J. R., & Taylor, L. O. (2002). What Determines the Value of Life? A Meta-analysis. *Journal of Policy Analysis and Management*, *21*(2), 253-270.

Muller, N. Z., Mendelsohn, R., & Nordhaus, W. (2011). Environmental Accounting for Pollution in the United States Economy. *American Economic Review*, *101*(5), 1649–1675. doi:10.1257/aer.101.5.1649.

Pearce, D.W. (2003). "The Social Cost of Carbon and its Policy Implications." Oxford Review of Economic Policy, 19, (3), 1-32.

Stern, N. (2007). Stern Review: The Economics of Climate Change. Cambridge, U.K.: Cambridge University Press. Retrieved from http://www.hm-treasury.gov.uk/sternreview_index.htm.

Tol, R.S.J. (2005). "The Marginal Damage Costs of Carbon Dioxide Emissions: An Assessment of the Uncertainties." Energy Policy, 33, 2064-2074.

Tol, R. S. J. (2009). The Economic Effects of Climate Change. *Journal of Economic Perspectives*, 23 (2): 29–51.

U.K, Department of Energy and Climate Change (2009). Carbon Valuation in UK Policy Appraisal: A Revised Approach.

U.S. Environmental Protection Agency (2011). Benefits and Costs of the Clean Air Act from 1990 to 2020.

U.S. EPA. (2012) Report to Congress on Black Carbon. United States Environmental Protection Agency. EPA-450/R-12-001. March, 2012. http://www.epa.gov/blackcarbon.

U.S. Environmental Protection Agency, Office of Mobile Sources, Engine Programs and Compliance Division (1999). Regulatory Impact Analysis—Control of Air Pollution from New Motor Vehicles: Tier 2 Motor Vehicle Emissions Standards and Gasoline Sulfur Control Requirements. EPA420-R-99-023.

Viscusi, W. K. (2004). The Value of Life: Estimates with Risks by Occupation and Industry. *Economic Inquiry* 42 (1): 29–48.

Viscusi, W. K., and Aldy, J. E. (2003). The Value of a Statistical Life: A Critical Review of Market Estimates throughout the World. *Journal of Risk and Uncertainty* 27 (1): 5–76.

Viscusi, W. K., & Moore, M. J. (1989). Rates of Time Preference and Valuations of the Duration of Life. *Journal of Public Economics*, *38*(3), 297-317.

Wilhelmine Miller, Robinson, L., Lawrence, R.S (2006) Committee to Evaluate Measures of Health Benefits for Environmental, Health and Safety Regulation (2006). *Valuing Health for Regulatory Cost-Effectiveness Analysis*. The National Academies Press.

Yohe, G.W., R.D. Lasco, Q.K. Ahmad, N.W. Arnell, S.J. Cohen, C. Hope, A.C. Janetos and R.T. Perez, 2007: Perspectives on climate change and sustainability. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden and C.E. Hanson, Eds., Cambridge University Press, Cambridge, UK, 811-841.

THE WORLD BANK