



Report to Congress on Black Carbon

External Peer Review Draft

Department of the Interior, Environment, and Related Agencies
Appropriations Act, 2010

(This page intentionally left blank)

Report to Congress on Black Carbon
External Peer Review Draft

U.S. Environmental Protection Agency
Office of Air Quality Planning and Standards
Research Triangle Park, North Carolina

Office of Atmospheric Programs
Washington, D.C.

Office of Radiation and Indoor Air
Washington, D.C.

Office of Research and Development
Research Triangle Park, North Carolina

Office of Transportation and Air Quality
Washington, D.C.

(This page intentionally left blank)

Table of Contents

Section	Page Number
Executive Summary	EX-1
Chapter 1 Introduction	
1.1 Key Questions Addressed in this Assessment	1-2
1.2 Other Recent Assessments of BC	1-3
1.3 Organization of this Report	1-5
Chapter 2 Black Carbon Effects on Climate	
2.1 Summary of Key Messages	2-1
2.2 Introduction	2-3
2.3 Defining Black Carbon and Other Light-Absorbing PM	2-5
2.4 Key Attributes of BC and Comparisons to GHGs	2-9
2.5 The Role of Co-Emitted Pollutants and Atmospheric Processing	2-12
2.6 Global and Regional Climate Effects of Black Carbon	2-15
2.6.1 Global and Regional Radiative Forcing Effects of Black Carbon	2-16
2.6.2 Impact of BC Radiative Forcing on Temperature and Melting of Ice and Snow	2-33
2.6.3 Non-Radiative Forcing Impacts of BC (Surface Dimming and Precipitation)	2-36
2.6.4 BC Impacts in the Arctic	2-40
2.6.5 BC Impacts in the Himalayas	2-44
2.6.6 Summary of BC Impacts in Key Regions	2-45
2.7 Economic Value of BC Impacts on Climate	2-46
Chapter 3 Black Carbon Effects on Public Health and the Environment	
3.1 Summary of Key Messages	3-1
3.2 Introduction	3-1
3.3 Health Effects Associated with Exposure to PM _{2.5} , including BC	3-2
3.4 Non-Climate Welfare Effects of PM _{2.5} , including BC	3-5
3.5 Valuation Techniques for Assessing Air Pollutant Impacts of BC	3-7
Chapter 4 Emissions of Black Carbon	
4.1 Summary of Key Messages	4-1
4.2 Introduction	4-2
4.3 U.S. Black Carbon Emissions	4-2
4.4 Global Black Carbon Emissions	4-15
4.5 Long-Range Transport of Emissions	4-31
4.6 Historical Trends in Black Carbon Emissions	4-35
Chapter 5 Observational Data for Black Carbon	
5.1 Summary of Key Messages	5-1
5.2 Black Carbon and Other Light-Absorbing Carbon: Measurement Methods	5-2
5.3 Ambient Concentrations of Black Carbon	5-4
5.4 Trends in Ambient BC Concentrations	5-13

5.5 Remote Sensing Observations	5-19
5.6 Black Carbon Observations from Surface Snow, Ice Cores, and Sediments	5-23
5.7 Limitations and Gaps in Current Ambient Data and Monitoring Networks	5-33

Chapter 6 Mitigation Overview: Climate and Health Benefits of Reducing Black Carbon Emissions

6.1 Summary of Key Messages	6-1
6.2 Introduction	6-2
6.3 Effect of Existing Control Programs	6-3
6.4 Future Black Carbon Emissions	6-4
6.5 Climate Benefits of Reducing Black Carbon Emissions	6-9
6.6 Public Health and Welfare Benefits of Reducing Black Carbon Emissions	6-12
6.7 Key Factors to Consider in Pursuing BC Emissions Reductions	6-19
6.8 Overview of Main Mitigation Options	6-20

Chapter 7 Mitigation Approaches for Mobile Sources

7.1 Summary of Key Messages	7-1
7.2 Introduction	7-2
7.3 Emissions Trajectories for Mobile Sources	7-3
7.4 New Engine Standards in the United States	7-8
7.5 New Engine Standards Internationally	7-13
7.6 Mitigation Approaches for In-use Mobile Sources in the United States	7-15
7.7 Mitigation Approaches for In-use Mobile Engines Internationally	7-26

Chapter 8 Mitigation Approaches for Stationary Sources

8.1 Summary of Key Messages	8-1
8.2 Introduction	8-1
8.3 Emissions from Key Stationary Source Categories	8-2
8.4 Available Control Technologies for Stationary Sources	8-4
8.5 Cost-Effectiveness of PM Control Technologies	8-7
8.6 Mitigation Approaches Other than PM Control Technologies	8-8
8.7 Mitigation Approaches for Stationary Sources Internationally	8-9
8.8 Technical and Research Needs	8-13

Chapter 9 Mitigation Approaches for Residential Heating and Cooking

9.1 Summary of Key Messages	9-1
9.2 Introduction	9-1
9.3 Residential Wood Combustion in Developed Countries	9-3
9.4 Residential Cookstoves in Developing Countries	9-12

Chapter 10 Mitigation Approaches for Open Biomass Burning

10.1 Summary of Key Messages	10-1
10.2 Introduction	10-2
10.3 Emissions from Open Biomass Burning	10-2
10.4 Fire as a Resource Management Tool	10-5
10.5 Smoke Mitigation Technologies and Approaches in the United States	10-5
10.6 Mitigation Technologies and Approaches Globally	10-11

Chapter 11 Metrics for Comparing Black Carbon Impacts to Impacts of Other Climate Forcers

11.1	Summary of Key Messages	11-1
11.2	Introduction to Metrics	11-2
11.3	Metrics along the Cause and Effect Chain	11-2
11.4	Commonly-Used Metrics for GHGs	11-4
11.5	Applicability of Climate Metrics to Black Carbon	11-7
11.6	Using Metrics in the Context of Climate Policy Decisions	11-14

Chapter 12 Conclusions and Research Recommendations

12.1	Conclusions	12-1
12.2	High Priority Research Needs	12-3

Appendix 1 Ambient and Emissions Measurement of Black Carbon

Appendix 2 Black Carbon Emissions Inventory Methods and Comparisons

Appendix 3 Studies Estimating Global and Regional Health Benefits of Reductions in Black Carbon

Appendix 4 Efforts to Limit Diesel Fuel Sulfur Levels

Appendix 5 U.S. Emission Standards for Mobile Sources

Appendix 6 International Emission Standards for Heavy-Duty Vehicles

Bibliography

(This page intentionally left blank)

Executive Summary

Black carbon (BC) emissions affect the Earth's climate in a number of ways. The ability of BC to absorb light energy and its role in key atmospheric processes link it to a range of climate impacts, including increased temperatures, accelerated ice and snow melt, and disruptions to precipitation patterns. Mounting scientific evidence suggests that reducing current emissions of BC can provide near-term climate benefits, particularly for sensitive regions such as the Arctic. Because of the strong warming potential and short atmospheric lifetime of BC, BC mitigation offers an opportunity to address key climate effects and slow the rate of climate change. However, BC reductions cannot substitute for reductions in long-lived greenhouse gases (GHGs), which are essential for mitigating climate change in the long run.

Despite the number of climate-related studies on BC published in the last decade and solid data on emissions from key sources of BC such as mobile diesel engines, important uncertainties remain regarding both the magnitude of particular global and regional climate effects and the impact of emissions mixtures from other source categories. To advance efforts to understand the role of BC in climate change, on October 29, 2009, Congress established requirements for the U.S. Environmental Protection Agency (EPA) to conduct a BC study as part of *H.R. 2996: Department of the Interior, Environment, and Related Agencies Appropriations Act, 2010*. Specifically, the legislation stated that:

“Not later than 18 months after the date of enactment of this Act, the Administrator, in consultation with other Federal agencies, shall carry out and submit to Congress the results of a study on domestic and international black carbon emissions that shall include

- *an inventory of the major sources of black carbon,*
- *an assessment of the impacts of black carbon on global and regional climate,*
- *an assessment of potential metrics and approaches for quantifying the climatic effects of black carbon emissions (including its radiative forcing and warming effects) and comparing those effects to the effects of carbon dioxide and other greenhouse gases,*
- *an identification of the most cost-effective approaches to reduce black carbon emissions, and*
- *an analysis of the climatic effects and other environmental and public health benefits of those approaches.”*

To fulfill this charge, EPA has conducted an intensive effort to compile, assess, and summarize available scientific information on the current and future impacts of BC, and to evaluate the effectiveness of available BC mitigation approaches and technologies for protecting climate, public health, and the environment. The results are presented in this *Report to Congress on Black Carbon* (Report). In conjunction with other ongoing BC assessments, including work under the United Nations Environment Programme (UNEP), the Convention on Long Range Transboundary Air Pollution (CLRTAP), and the Arctic Council, this Report helps to clarify the potential benefits to climate of reducing BC emissions and the mitigation options that are available. The key messages of this Report can be summarized as follows.

- **Black carbon is the most strongly light-absorbing component of particulate matter (PM), and is formed by incomplete combustion of fossil fuels, biofuels, and biomass.**

BC can be defined specifically as the carbonaceous component of PM that absorbs all wavelengths of solar radiation, though it is commonly referred to as soot. Per unit of mass in the atmosphere, BC can absorb a million times more energy than carbon dioxide (CO₂). Current emissions of BC by source category in the United States and globally are shown in Figure A.

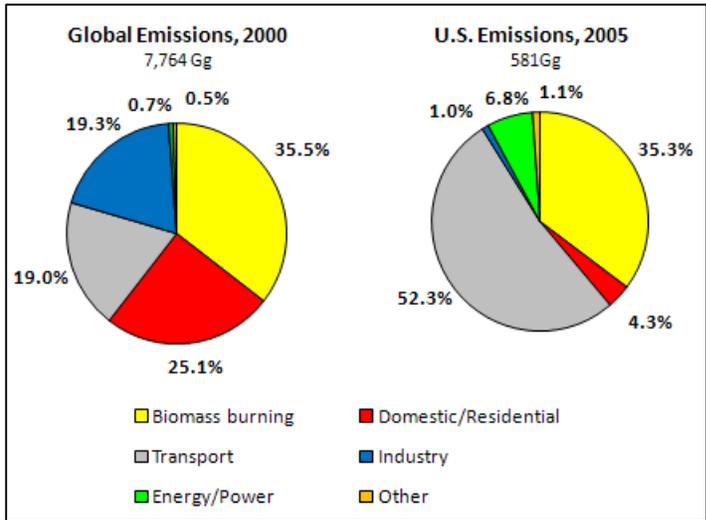


Figure A. BC Emissions by Major Source Category

- **The short atmospheric lifetime of BC and the mechanisms by which it affects climate distinguish it from long-lived GHGs like CO₂.**

BC has a short atmospheric residence time of days to weeks. This short lifetime, combined with the strong warming potential of BC, means that the climate benefits of reductions in current emissions of BC will be nearly immediate. In contrast, long-lived GHGs persist in the atmosphere for centuries. Therefore, reductions in GHG emissions will take longer to influence atmospheric concentrations and will have less impact on climate on a short timescale. However, since GHGs are by far the largest contributor to current and future climate change, deep reductions in these pollutants are essential over the long-term.

Emissions sources and ambient concentrations of BC vary geographically and temporally (Figure B), resulting in climate effects that are more regionally and seasonally dependent than the effects of long-lived, well-mixed GHGs. Likewise, mitigation actions for BC will produce different climate results depending on the region, season, and emission category.

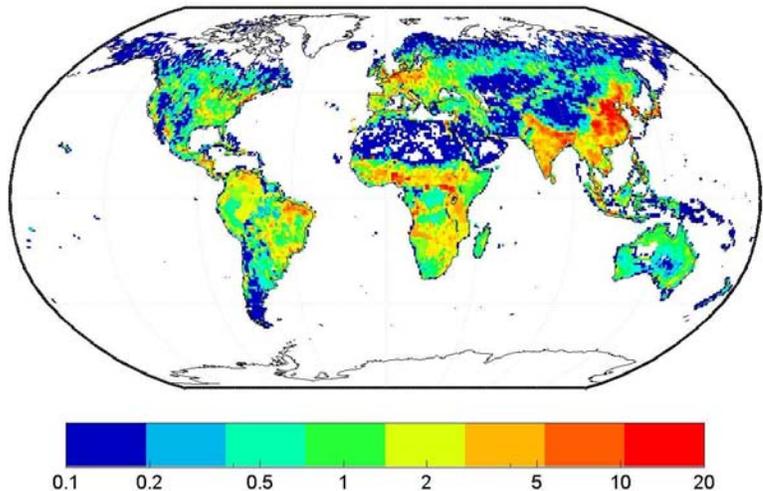


Figure B. BC Emissions, 2000, Gg (T. Bond)

BC influences climate through multiple mechanisms:

- *Direct effect:* BC absorbs both incoming and outgoing radiation of all wavelengths. In contrast, GHGs mainly trap outgoing infrared radiation from the earth’s surface.
- *Snow/ice albedo effect:* BC deposited on snow and ice darkens the surface and decreases reflectivity, thereby increasing absorption and accelerating melting. GHGs do not have this kind of impact.
- *Indirect effect:* BC also alters the properties of clouds, affecting cloud reflectivity, precipitation, and surface dimming. These impacts are associated with all ambient particles, but not GHGs.

➤ ***The direct and snow/ice albedo effects of BC are widely understood to lead to climate warming. The indirect effects of BC on climate via interaction with clouds are much more uncertain, but may partially offset the warming effects.***

The direct radiative forcing effect of BC is the best quantified and appears to be significant on both global and regional scales. In 2007, the Intergovernmental Panel on Climate Change (IPCC) provided a central estimate for global average direct forcing by BC of +0.34 (±0.25) Watts per square meter ($W m^{-2}$). Other studies have also estimated positive values, some exceeding $1 W m^{-2}$. In addition, according to the IPCC the BC snow/ice albedo effect contributes +0.1 (±0.1) $W m^{-2}$ to the global average forcing from BC. The geographical distribution of these effects is illustrated in the top two panels of Figure C. Based on the IPCC estimates, the direct and snow/ice albedo effects of BC together contribute more to warming than any GHG other than CO₂ and methane. The IPCC’s radiative forcing estimates for elevated concentrations of CO₂ and methane are +1.66 $W m^{-2}$ and +0.48 $W m^{-2}$, respectively.

All aerosols (including BC) affect climate indirectly by changing the reflectivity and lifetime of clouds. The net indirect effect of all aerosols is very uncertain but is thought to be a net cooling influence. The IPCC estimates that the globally averaged indirect forcing from all aerosols ranges between -0.10 $W m^{-2}$ and -1.80 $W m^{-2}$. BC has additional indirect effects—including changes to cloud stability and enhanced precipitation from colder clouds—that can lead to warming. The net climate influence of the indirect effects of BC is not yet clear. However, the warming due to the direct and snow/ice albedo effects very likely exceeds

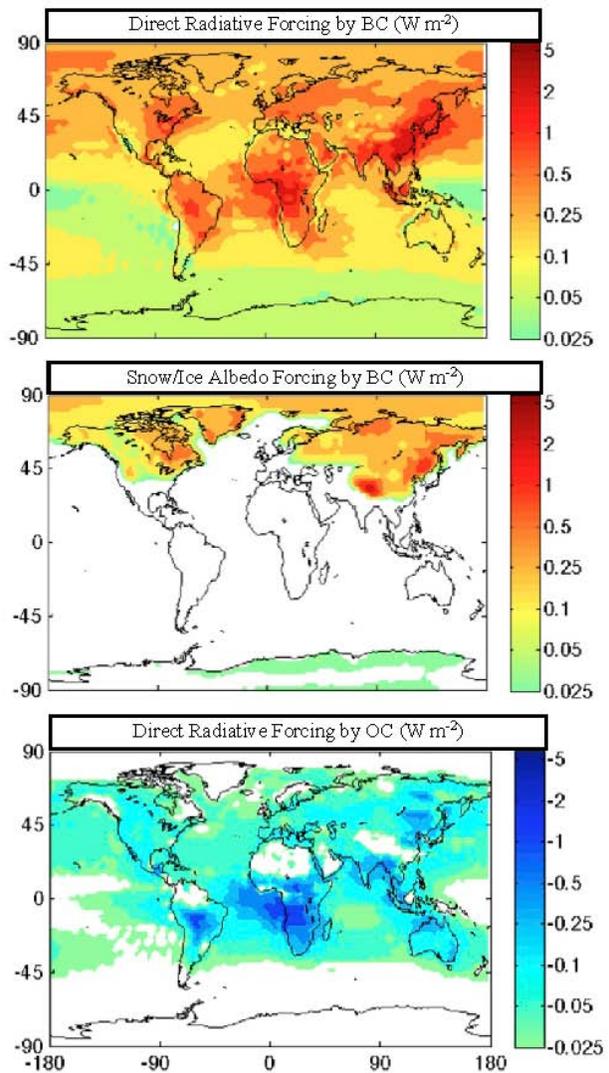


Figure C. Regional variability in radiative forcing for BC and OC, simulated with the Community Atmosphere Model.
(Source: Bond et al., 2011)

any cooling from indirect effects.

- ***The full effect of BC on climate must be assessed in the context of co-emitted pollutants. BC is always emitted with other particles and gases, and the composition of this emissions mixture determines the net impact on climate. Some combustion sources emit more BC than others relative to the amount of other constituents.***

The same combustion processes that produce BC also produce other pollutants, such as sulfur dioxide (SO₂), nitrogen oxides (NO_x), and OC. Some of these co-emitted pollutants result in “scattering” or reflecting particles (e.g. sulfate, nitrate, OC) which exert a cooling effect on climate. (The geographical variability in the direct cooling effects of OC is illustrated in Figure C above.) Some portion of co-emitted OC, notably brown carbon (BrC) mixtures, partially absorbs solar radiation. The net contribution of BrC to climate is presently uncertain.

The amount of BC relative to other constituents affects whether the source has a net warming or net cooling effect on the climate. This varies considerably among source categories. For example, the particles emitted by mobile diesel engines are about 75% BC, while particle emissions from biomass burning are dominated by OC.

Atmospheric processes that occur after BC is emitted, such as mixing, aging, and coating, can also affect the net influence on climate.

- ***Regional climate impacts of BC are highly variable, and the effects of BC on warming and melting are especially strong in sensitive regions such as the Arctic and the Himalayas.***

Studies have shown that BC has especially strong warming effects in the Arctic, contributing to earlier spring melting and sea ice decline. All particle mixtures reaching the Arctic are a concern, because over highly reflective surfaces such as ice and snow, even emissions mixtures that contain more reflective (cooling) aerosols can lead to warming if they are darker than the underlying surface. Studies have estimated atmospheric forcing from BC at 1.2 W m⁻² in the Arctic in springtime, with an additional +0.53 W m⁻² from snow albedo forcing. Where earlier melting of ice and snow reveals darker surfaces (e.g. ocean water or soils) underneath, the warming effect is magnified.

BC has also been shown to be a significant factor in the observed increases in melting rates of some glaciers and snowpack in parts of the Hindu Kush-Himalayan-Tibetan (HKHT) region (the “third pole”). Average radiative forcing of BC on snow and ice in the Tibetan plateau has been estimated at +1.5 Wm⁻², with instantaneous forcing in some places in the spring estimated as high as 20 W m⁻².

In the United States, deposition of BC on mountain glaciers and snow packs has been shown to reduce snow cover and overall snowpack and to contribute to earlier spring melting, which reduces the amount of meltwater available later in the year.

- ***BC contributes to surface dimming, the formation of Atmospheric Brown Clouds (ABCs), and changes in the pattern and intensity of precipitation***

The absorption and scattering of incoming solar radiation by BC and other particles causes surface dimming by reducing the amount of solar radiation reaching the Earth’s surface. In some regions, especially Asia, southern Africa, and the Amazon Basin, BC, BrC, sulfates, organics, dust and other components combine to form pollution clouds known as Atmospheric Brown Clouds (ABCs). ABCs have been linked to surface dimming and a decrease in vertical mixing, which exacerbates air pollution episodes. ABCs also contribute to changes in the pattern and intensity of rainfall, and to observed changes in monsoon circulation in South Asia. In general, regional changes in precipitation due to BC

and other aerosols are likely to be highly variable, with some regions seeing increases while others experience decreases.

➤ ***BC contributes to the adverse impacts on human health, ecosystems, and visibility associated with ambient fine particles (PM_{2.5}).***

Short-term and long-term exposures to PM_{2.5} are associated with a broad range of human health impacts, including respiratory and cardiovascular effects, as well as premature death. BC, OC, and other PM constituents all contribute to these adverse health effects. PM_{2.5} is also linked to adverse impacts on ecosystems, to visibility impairment, to reduced agricultural production in some parts of the world, and to materials soiling and damage. At present, there is insufficient information to differentiate the health effects of BC relative to other constituents of PM_{2.5}. However, emissions and ambient concentrations of BC and co-emitted pollutants are often highest in urban areas, where large numbers of people live. Controls applied to reduce BC will help reduce all of these harmful constituents.

➤ ***Currently, the majority of global BC emissions come from Asia, Latin America, and Africa; the United States accounts for approximately 8% of the global total. Emissions patterns and trends across regions, countries and sources vary significantly.***

Overall global BC emissions trends have shown net increases to present conditions. However, emissions of BC in North America and Europe have declined substantially since the early 1900s and are expected to decline further in the next several decades, due to pollution controls and fuel switching. Elsewhere, BC has been increasing, with most of the increase coming from developing countries in Asia, Latin America, and Africa. According to available estimates, these regions currently contribute more than 75% of total global BC emissions, with the majority of emissions coming from the residential sector (cookstoves) and open biomass burning. Current emissions from the United States, OECD Europe, the Middle East, and Japan come mainly from the transportation sector, particularly from mobile diesel engines. In the United States, nearly 50% of BC emissions came from mobile diesel engines in 2005.

➤ ***Available control technologies can provide cost-effective reductions in BC emissions from many key source categories, resulting in some near-term climate benefits, especially at the regional level.***

BC emissions reductions are generally achieved by applying technologies and strategies to improve combustion and/or control direct PM_{2.5} emissions from sources. Benefits in sensitive regions like the Arctic, or in regions of high emissions such as India and Asia, may include reductions in warming and melting (ice, snow, glaciers), and reversal of precipitation changes. BC reductions could help reduce the rate of warming soon after they are implemented. However, available studies also suggest that BC mitigation alone would be insufficient to change the long-term trajectory of global warming (which is driven by GHGs).

➤ ***These cost-effective mitigation strategies will also provide substantial public health co-benefits.***

Reductions in directly emitted PM_{2.5} can substantially reduce human exposure, providing large public health benefits that often exceed the costs of control. In the United States, the average public health benefits associated with reducing directly emitted PM_{2.5} are estimated to range from \$270,000 to \$1.1 million per ton PM_{2.5} in 2030. The cost of the controls necessary to achieve these reductions is generally far lower. For example, the costs of PM controls for new diesel engines are estimated to be less than \$13,000 per ton PM_{2.5}. Globally, the health benefits of mitigation strategies aimed at BC would be even larger, potentially averting hundreds of thousands of premature deaths each year.

- ***Considering the location and timing of emissions and accounting for co-emissions will improve the likelihood that mitigation strategies will be beneficial for both climate and public health.***

PM mitigation strategies that focus on sources known to emit large amounts of BC—especially those with a high ratio of BC to OC, like diesel emissions—will maximize climate co-benefits. The timing and location of the reductions are also very important. The largest climate benefits of BC-focused control strategies may come from reducing emissions affecting the Arctic, HKHT and other ice and snow-covered regions.

The effect of BC emission reductions on human health are a function of changing exposure and the size of the affected population. The largest health benefits from BC-focused control strategies will occur locally near the emissions source and where exposure affects a large population.

- ***The United States will achieve substantial BC emissions reductions by 2030, largely due to forthcoming controls on mobile diesel engines. Diesel retrofit programs for in-use mobile sources are also helping to reduce emissions. Other source categories, including stationary sources, residential wood combustion, and open biomass burning, have more limited mitigation potential due to smaller remaining emissions in these categories, or limits on the availability of effective BC control strategies.***

In the U.S., total **mobile source** BC emissions are projected to decline significantly by 2030 due to regulations already promulgated. BC emissions from mobile diesel engines (including on-road, nonroad, locomotive, and commercial marine engines) in the United States are being controlled through two primary mechanisms: (1) *emissions standards for new engines*, including requirements resulting in use of diesel particulate filters (DPFs) in conjunction with ultra low sulfur diesel fuel; and (2) retrofit programs for *in-use mobile diesel engines*, such as EPA's National Clean Diesel Campaign and the SmartWay Transport Partnership Program.

BC emissions from **stationary sources** in the United States have declined dramatically in the last century, with remaining emissions coming primarily from coal combustion (utilities, industrial/commercial boilers, other industrial processes) and stationary diesel engines. Available control technologies and strategies include use of cleaner fuels and direct PM_{2.5} reduction technologies such as fabric filters (baghouses), electrostatic precipitators (ESPs), and diesel particulate filters (DPFs).

Emissions of all pollutants from **residential wood combustion** (RWC) are currently being evaluated as part of EPA's ongoing review of emissions standards for residential wood heaters, including hydronic heaters, woodstoves, and furnaces. Mitigation options include providing alternatives to wood, replacing inefficient units or retrofitting existing units.

Open biomass burning, including both prescribed fires and wildfires, represents a potentially large but less certain portion of the U.S. BC inventory. These sources emit much larger amounts of OC compared to BC. The percent of land area affected by different types of burning is uncertain, as are emissions estimates. Appropriate mitigation measures depend on the timing and location of burning, resource management objectives, vegetation type, and available resources.

- ***Other developed countries have emissions patterns and control programs that are similar to the United States, though the timing of planned emissions reductions may vary. Developing countries have a higher concentration of emissions in the residential and industrial sectors, but the growth of the mobile source sector in these countries may lead to an increase in their overall BC emissions and a shift in the relative importance of specific BC emitting sources over the next several decades.***

For **mobile sources**, both new engine standards and retrofits of existing engines/vehicles may help reduce BC emissions in the future. While many other countries have already begun phasing in emissions

and fuel standards, BC emissions in this category in developing countries are expected to continue to increase. Emissions control requirements lag behind in some regions, as do on-the-ground deployment of DPFs and low sulfur fuels. Further or more rapid reductions in BC will depend on accelerated deployment of clean engines and fuels.

Emissions from **residential cookstoves** are both a large source of BC globally and a major threat to public health. Approximately 3 billion people worldwide cook their food or heat their homes by burning biomass or coal in rudimentary stoves or open fires, resulting in pollution exposures that lead to 2 million deaths each year. Significant expansion of current clean cookstove programs would be necessary to achieve large-scale climate and health benefits. A wide range of improved stove technologies is available, but the potential climate and health benefits vary substantially by technology and fuel. A number of factors point to much greater potential to achieve large-scale success in this sector today.

The largest sources of BC emissions from **stationary sources** internationally include brick kilns, coke ovens (largely from iron/steel production), and industrial boilers. Replacement or retrofit options already exist for many of these source categories.

Open biomass burning is the largest source of BC emissions globally. However, emissions of OC are approximately seven times higher than BC emissions from this sector, and more complete emissions inventory data are needed to characterize impacts of biomass burning and evaluate the effectiveness of mitigation measures at reducing BC. Successful implementation of mitigation approaches in world regions where biomass burning is widespread will require training in proper burning techniques and tools to ensure effective use of prescribed fire.

➤ ***The differences between lifetime and mechanisms of action for BC and long-lived GHGs hinder comparison of their relative climate impacts via the use of common metrics.***

Most metrics developed to express the climate impacts of CO₂ and other long-lived GHGs are ill-suited to short-lived climate forcers like BC. There is currently no single metric (for example, the global warming potential, or GWP) that is widely accepted by the science and research community for comparing the array of climate impacts from BC with GHGs. The lack of such a common metric is a clear impediment to direct comparisons among these pollutants. However, new metrics designed specifically for short-lived climate forcers like BC have recently been developed, and it may be possible to utilize these or other metrics, such as OC/BC ratios, to prioritize among source categories and mitigation options with regard to potential net climate effects.

➤ ***There are a number of high priority research topics that could help reduce key remaining uncertainties regarding the role of BC in climate change and public health.***

- Standardized definitions and improved instrumentation and measurement techniques for light-absorbing PM, coupled with expanded observations.
- Continued investigation of basic microphysical and atmospheric processes affecting BC and other aerosol species to facilitate improvements in modeling and monitoring of BC.
- Improving global, regional, and domestic emissions inventories with more laboratory and field data on activity levels, operating conditions, and technological configurations, coupled with better estimation techniques for current and future emissions.
- Focused investigations of the role of brown carbon (BrC).

- More detailed analysis of the climate and health benefits of controlling BC from sources of specific types or in specific locations.
- Refinement of climate metrics specific to BC and other short-lived climate forcers.

1

2

1. Introduction

3

4

Black carbon (BC) has recently received a great deal of attention among scientists and policymakers for its impacts on global and regional climate. Though substantial and immediate reductions in long-lived greenhouse gases (GHG) are essential for solving the problem of long-term climate change, BC offers a promising mitigation opportunity to address short-term effects and slow the rate of climate change. BC's high capacity for light absorption and its role in key atmospheric processes link it to a range of climate impacts, including increased temperatures, accelerated ice and snow melt, and disruptions in precipitation patterns. BC is also a constituent of ambient fine particles (PM_{2.5}) and is therefore associated with an array of respiratory and cardiovascular health impacts. This makes it ripe for "win-win" emissions reduction approaches that bring both climate and public health benefits.

13

14

Like many air pollutants, BC's atmospheric fate is affected by a number of complex physical and chemical processes that may enhance or attenuate BC's warming impacts. Some of these atmospheric processes are not yet completely understood, making it challenging to represent them accurately in climate models and to project future impacts. Furthermore, BC is always co-emitted with other pollutants, many of which have offsetting climate impacts. Thus BC must be studied in the context of the total emissions mixture coming from particular sources. In its 2007 Fourth Assessment Report, the Intergovernmental Panel on Climate Change (IPCC) noted that the climate effects of particles remained "the dominant uncertainty" in estimating climate impacts. Since that time, additional research has helped to reduce this uncertainty, through inventory improvements, advances in measurement technologies and methods, and increasing sophistication in the representation of particle atmospheric chemistry in climate models. Thus, though important uncertainties remain, substantial progress has been made in understanding the role of BC and other particles in climate processes. Recent work has clarified BC's climate effects and the emissions control approaches necessary to mitigate these impacts.

27

28

To further efforts to understand the role of BC in climate change, on October 29, 2009, the United States Congress established requirements for the U.S. Environmental Protection Agency (EPA) to conduct a BC study as part of *H.R. 2996: Department of the Interior, Environment, and Related Agencies Appropriations Act, 2010*. Specifically, the legislation stated that:

32

33

"Not later than 18 months after the date of enactment of this Act, the Administrator, in consultation with other Federal agencies, shall carry out and submit to Congress the results of a study on domestic and international black carbon emissions that shall include

36

- *an inventory of the major sources of black carbon,*
- *an assessment of the impacts of black carbon on global and regional climate,*

37

- 1 • *an assessment of potential metrics and approaches for quantifying the climatic effects of*
- 2 *black carbon emissions (including its radiative forcing and warming effects) and comparing*
- 3 *those effects to the effects of carbon dioxide and other greenhouse gases,*
- 4 • *an identification of the most cost-effective approaches to reduce black carbon emissions,*
- 5 *and*
- 6 • *an analysis of the climatic effects and other environmental and public health benefits of*
- 7 *those approaches.”*

8
9 To fulfill this charge, EPA has conducted an intensive effort to compile, assess, and summarize available
10 scientific information on the current and future impacts of BC, and to evaluate the effectiveness of
11 available BC mitigation approaches and technologies for protecting climate, public health, and the
12 environment. The results are presented in this *Report to Congress on Black Carbon*.

14 **1.1 Key Questions Addressed in this Assessment**

15
16 In evaluating the climate impacts and mitigation opportunities for BC, it is essential to recognize
17 from the outset that BC presents a different kind of climate challenge than CO₂ and other long-lived
18 GHGs. BC’s short atmospheric lifetime (days to weeks) and heterogeneous distribution around the
19 globe results in regionally concentrated climate impacts. Thus, the location of emissions releases is a
20 critical determinant of BC’s impacts, which is not the case for long-lived and more homogeneously
21 distributed GHGs like CO₂. The composition of the total emissions mixture is also key: since many co-
22 emitted pollutants such as sulfur dioxide, oxides of nitrogen, and most organic carbon particles tend to
23 produce a cooling influence on climate, the amount of BC relative to these other constituents being
24 emitted from a source is important. Furthermore, BC is linked to a whole variety of effects beyond the
25 warming attributable to GHGs. These include the darkening of ice and snow, which reduces reflectivity
26 and accelerates melting; changes in the formation and composition of clouds, which affect precipitation;
27 and impacts on human health.

28
29 These key characteristics of BC give rise to some important questions addressed in this Report,
30 including:

- 31 1. What is BC, and how does it lead to climate warming?
- 32 2. What is the net effect of atmospheric BC on global and regional temperature change in terms of
- 33 both magnitude and time scale?
- 34 3. What is known about the magnitude of BC’s effect on snow and ice, and its impacts on
- 35 precipitation?
- 36 4. What is known about BC’s contribution to PM_{2.5}-related human health impacts and other, non-
- 37 climate environmental impacts?
- 38 5. What kind of real-world BC data exists from monitoring networks and other observational
- 39 research?
- 40 6. How large are U.S. and international emissions of BC currently, which sectors are the main
- 41 contributors, and how are emissions projected to change in the future?
- 42

- 1 7. What technologies and approaches are available to address emissions from key sectors, and at
2 what cost?
- 3 8. What is the potential value of BC reductions as a component of a broader climate change
4 mitigation program, taking into account both co-pollutant emissions reductions and the public
5 health co-benefits?

6
7 In answering these questions, this Report focuses on synthesizing available scientific information
8 about BC from peer-reviewed studies and other technical assessments, describing current and future
9 emissions estimates, and summarizing information on available mitigation technologies and approaches,
10 including their costs and relative effectiveness. Given the number of recent studies and the limited time
11 available to the complete this Report, EPA did not seek to undertake extensive new analysis (such as
12 climate modeling of specific BC mitigation strategies), but instead relied on information available in the
13 literature. The report focuses on BC, where the bulk of scientific research is available, but acknowledges
14 the potentially important role played by other light-absorbing particles which are still subject to great
15 uncertainty. This Report also describes specific research and technical information needed to provide a
16 stronger foundation for future decision-making regarding appropriate and effective BC mitigation
17 policies.
18

19 **1.2 Other Recent Assessments of BC**

20
21 Numerous international and intergovernmental bodies, including the United Nations
22 Environment Programme (UNEP), the Convention on Long Range Transboundary Air Pollution (CLRTAP),
23 and the Arctic Council, have identified BC as a potentially important piece of the climate puzzle. Each of
24 these bodies has recently prepared an assessment of BC that included consideration of the impacts of
25 BC on climate, the potential benefits to climate of reducing BC emissions, and/or the mitigation
26 opportunities that appear most promising. These assessments have identified a number of additional
27 actions—from improvements in inventories to evaluation of specific mitigation opportunities—that
28 could be taken to help gather further information about BC and address emissions from key sectors.
29

30 In its draft *Integrated Assessment of Black Carbon and Tropospheric Ozone* (2011, in progress),
31 UNEP concludes that BC mitigation may offer near-term climate benefits. This study was designed to
32 assess the role of BC and ozone in climate and air quality, and recommend mitigation measures that
33 could be expected to provide benefits in both the climate and air quality arenas. Out of roughly two
34 thousand potential mitigation measures, the UNEP analysis has identified a small subset of measures as
35 providing the largest mitigation potential. The draft *Assessment* finds that full implementation of the
36 targeted measures (which included methane reductions for ozone mitigation, as well as BC reductions)
37 could greatly reduce global mean warming rates over the next few decades. Specifically, the analysis
38 suggests that warming anticipated to occur during the 2030s based on emissions projections could be
39 reduced by half through application of these BC and methane measures. In contrast, even a fairly
40 aggressive strategy to reduce CO₂ would do little to mitigate warming over the next 20-30 years. The
41 draft *Assessment* concludes that while CO₂ measures clearly are the key to mitigating long-term climate
42 change out to 2100, BC and methane measures could reduce warming and slow the rate of change in

1 the next two decades. The draft *Assessment* also recognizes the substantial benefits to air quality,
2 human health, and world food supplies that would result from reductions in BC and tropospheric ozone.

3
4 The CLRTAP Ad-hoc Expert Group on Black Carbon and the Arctic Council Task Force on Short-
5 Lived Climate Forcers focused mainly on identifying high-priority mitigation options and the need for
6 supporting information, such as national BC emissions inventories. These groups did not conduct
7 independent scientific assessments; rather, after a review of existing scientific literature, they concluded
8 that current evidence suggests BC plays an important role in near-term climate change. The CLRTAP Ad-
9 hoc Expert Group was co-chaired by the U.S. and Norway. In its final report presented to the
10 Convention's Executive Body in December 2010, the Expert Group highlighted key findings, including:

- 11 • There is general scientific consensus that mitigation of BC will lead to positive regional
12 impacts by reducing BC deposition in areas with snow and ice.
- 13 • There is virtual certainty that reducing primary PM will benefit public health.
- 14 • The Arctic, as well as alpine regions, may benefit more than other regions from reducing
15 emissions of BC.
- 16 • Climate processes unique to the Arctic have significant effects that extend globally, so action
17 must be taken in the very near term to reduce the rate of warming.
- 18 • Impacts on the Arctic and alpine areas will vary by country, but all countries will benefit
19 from local emission reductions of BC and other co-emitted pollutants.
20

21
22 The Expert Group concluded that because of the public health benefits of reducing BC, as well as
23 the location of the countries across the Convention regions in relation to the Arctic, the Executive Body
24 should consider taking additional measures to reduce BC. The report included information about key
25 sectors and emphasized the need to develop emission inventories, ambient monitoring and source
26 measurements in an effort to improve the understanding of adverse effects, efficacy of control
27 measures and the costs and benefits of abatement. Based in part on the findings of the Expert Group,
28 the CLRTAP Executive Body decided to include consideration of BC as a component of PM in their
29 ongoing process of revising the Gothenburg Protocol. This decision marks the first time an international
30 agreement has attempted to address the issue of short lived climate forcers in the context of air
31 pollution policy. Revisions to the Gothenburg Protocol are expected to be completed by the end of
32 2011.

33
34 The Arctic Council Task Force on Short-Lived Climate Forcers was formed following the issuance
35 of the Tromsø Declaration at the Arctic Council Ministerial in April 2009.¹ This declaration formally
36 noted the role that short-lived forcers may play in Arctic climate change, and recognized that reductions
37 of emissions of these compounds and their precursors have the potential to slow the rate of Arctic

¹ The Arctic Council comprises the eight member states with land above the Arctic Circle (Canada, Denmark including Greenland and the Faroe Islands, Finland, Iceland, Norway, Russian Federation, Sweden, and the U.S.), six permanent participants representing indigenous peoples resident in those member states, and a number of observers. The Council does not have legally-binding authority over its members, but rather promotes cooperation, coordination, and interaction regarding common Arctic issues.

1 snow, sea ice, and sheet ice melting in the near term. The Task Force, which is being co-chaired by the
2 U.S. and Norway, is charged with identifying existing and new measures to reduce emissions of short-
3 lived climate forcers (BC, ozone and methane) and recommending further immediate actions that can be
4 taken. The Task Force has focused its initial efforts on BC, and is working to prepare a menu of
5 mitigation options to present to the Arctic Council Ministerial in May of 2011. Its key findings include:

- 6
- 7 • Addressing short-lived climate forcers such as BC, methane and ozone offers a unique
8 opportunity to slow Arctic warming in the near term.
- 9 • Black carbon emitted both within and outside of the Arctic region contributes to Arctic
10 warming. Per unit of emissions, sources within Arctic Council nations generally have a
11 greater impact.
- 12 • Controlling black carbon sources also improves health and can therefore be considered a
13 no-regret measure.
- 14

15 Based on these findings, the Task Force recommends that Arctic Council nations take action to
16 reduce black carbon, and lays out a menu of specific mitigation options in key sectors such as land-based
17 transportation, residential wood combustion, agricultural and forest burning, and shipping. The Task
18 Force has also encouraged Arctic nations to develop and share domestic inventories of black carbon
19 emissions which can be used to further define—and refine—global inventories. The Task Force is
20 collaborating with the Arctic Monitoring and Assessment Programme (AMAP) working group, which is
21 preparing a report on the impact of black carbon on Arctic Climate, to conduct new climate modeling
22 scenarios to provide insights about the significance for the Arctic climate of black carbon emissions
23 sources from different sectors and different regions. AMAP is on schedule to finish this stage of its
24 analysis by May 2011.

25

26 These concurrent international assessments strongly suggest that reducing BC emissions will
27 slow the rate of warming and provide other near-term benefits to climate, as well as protecting public
28 health. The analyses conducted in support of these assessments provide useful information to clarify
29 BC's role in climate change, the impact of key emissions source categories, and the applicability of
30 different mitigation options. This *Report to Congress on Black Carbon* builds upon these efforts,
31 summarizing and incorporating their key findings as appropriate. Since all three of the efforts
32 mentioned above have been conducted by international bodies with a focus outside the U.S., readers
33 are encouraged to read their final reports and recommendations as an additional source of information.

34 **1.3 Organization of this Report**

35

36 This Report is organized into twelve chapters and five technical appendices. Each of the
37 chapters that follow this *Introduction* is described briefly below, along with the appendices.

38

39 **Chapter 2** describes how particles, including BC, absorb and scatter light, and identifies the
40 factors that influence the direction and magnitude of their influence on the Earth's climate. The chapter
41 defines "black carbon," describes how BC relates to other types of particles, and discusses how these

1 substances affect climate. Next, the chapter provides detailed information on the range of direct and
2 indirect impacts of BC on global and regional climate. It summarizes available estimates of BC's global
3 and regional radiative forcing and related temperature effects, snow and ice albedo effects, cloud
4 effects, and precipitation effects. The chapter also discusses approaches for valuing these climate
5 impacts.

6
7 **Chapter 3** outlines EPA's current scientific understanding of the health and non-climate
8 environmental effects of BC. This chapter discusses the large body of scientific evidence regarding the
9 adverse impacts of PM_{2.5} in general, and provides a summary of health research related to BC as a
10 component of the overall PM_{2.5} mix. It also describes BC's role in visibility impairment and ecological
11 effects, and briefly discusses the well-developed approaches for estimating and valuing public health
12 impacts of ambient PM_{2.5}.

13
14 **Chapter 4** provides a detailed look at BC emissions inventories. The chapter characterizes
15 current (2005) U.S. emissions of BC by source category, and provides detailed information regarding
16 emissions from sectors that are the most significant contributors to U.S. emissions, such as mobile
17 sources, open biomass burning, and stationary fossil fuel combustion. The chapter also provides an
18 overview of global and regional emissions inventories for BC, and contrasts these global inventories with
19 more refined regional inventories available for some areas, such as the U.S., China and India. Special
20 attention is paid to emissions near the Arctic. The chapter discusses the transport of emissions from
21 particular sources and regions, and describes historic emissions trends.

22
23 **Chapter 5** summarizes key findings from observational data on BC. This includes data from
24 ambient monitors, ice/snow cores, and remote sensing. The chapter describes the existing BC
25 monitoring networks, and summarizes available data regarding ambient levels in urban and rural areas,
26 both domestically and globally. The chapter also describes trends in ambient BC concentrations.

27
28 **Chapter 6** provides a broad overview of mitigation options, considers the potential climate and
29 human health benefits of BC emissions reductions. The chapter describes the findings of existing studies
30 on the global and regional benefits of BC mitigation, including specific strategies aimed at reducing
31 emissions from key sectors. The chapter acknowledges the large remaining uncertainties with regard to
32 evaluating the climate benefits of BC mitigation in some sectors, but notes that controls on BC and co-
33 emitted pollutants are generally associated with significant public health benefits, through reductions in
34 PM_{2.5} and its precursors. The chapter describes how the full range of benefits can be incorporated into
35 decisions regarding BC mitigation, and the impact of this incorporation on regional/sectoral emphasis
36 and control strategy choice.

37
38 **Chapters 7-10** describe existing control programs and technologies that have been
39 demonstrated to be effective in reducing BC emissions from sources categories of regional and/or global
40 importance. These include **Mobile Sources** (Chapter 7), **Stationary Sources** (Chapter 8), **Residential**
41 **Heating and Cooking** (Chapter 9), and **Open Biomass Burning** (Chapter 10). For each sector, the chapter
42 recaps current and projected emissions estimates (accounting for control programs currently in place

1 but not yet implemented), describes key control technologies and other mitigation strategies that can
2 help control BC emissions from specific source types, and provides available information regarding
3 control costs. Control options, costs, and known or potential barriers to mitigation are described
4 separately for U.S. domestic emissions and international emissions. In some cases, there are
5 considerable differences in mitigation approaches, cost, and feasibility between the U.S. and other
6 countries. Also, there are gaps in available information on these factors for many sectors.

7
8 **Chapter 11** considers how best to compare the effects of BC to the impacts of other climate
9 forcers, particularly CO₂. It evaluates the applicability of traditional metrics developed for CO₂ to BC,
10 and presents alternative metrics designed specifically for evaluating the climate impacts of short-lived
11 climate forcers like BC. It also considers how metrics can be used to prioritize emissions reductions
12 *among* BC emissions source categories, and describes some of the factors policymakers need to consider
13 in deciding which metrics to utilize to answer different policy questions.

14
15 The conclusion, **Chapter 12**, summarizes key findings of the report and identifies some
16 important research recommendations. This chapter draws together main points from the other
17 chapters of the report, highlighting main messages and identifying key gaps in current scientific
18 understanding. The chapter also provides a list of high-priority research needs for improving the current
19 scientific understanding of the impact of light-absorbing particles on climate, and for estimating the full
20 impact of mitigation approaches in different sectors and regions on both climate and public health.
21 These research needs may stimulate further work on BC by EPA and other organizations.

22
23 **Appendix 1** provides further details regarding alternative definitions of BC and other light-
24 absorbing particles, and the techniques and instruments used for ambient monitoring and measurement
25 of BC.

26
27 **Appendix 2** provides a detailed explanation of the methods that are used to compile U.S.
28 emissions inventories for BC. It also further explores the variety of global and non-U.S. regional
29 emissions inventories available and some of the key differences among those inventories.

30
31 **Appendix 3** summarizes the results of available studies which have estimated the public health
32 benefits that might accrue to alternative BC mitigation strategies, at either the global or regional level.

33
34 **Appendix 4** describes world-wide efforts to reduce the sulfur content of diesel fuels, which is an
35 important prerequisite to reducing BC emissions from mobile sources.

36
37 **Appendix 5** provides a full list of the emissions standards for different categories of mobile
38 sources in the United States, and the emissions limits set under those standards.

39
40 **Appendix 6** describes existing emissions standards for heavy-duty diesel vehicles internationally,
41 and the anticipated schedule for emissions reductions resulting from these standards.

(This page intentionally left blank)

2. Black Carbon Effects on Climate

2.1 Summary of Key Messages

- Black carbon (BC) is the most strongly light-absorbing component of particulate matter (PM), and is formed by incomplete combustion of fossil fuels, biofuels, and biomass.
 - BC can be defined specifically as the carbonaceous component of PM that absorbs all wavelengths of solar radiation. It is commonly referred to as “soot”. Per unit of mass in the atmosphere, BC can absorb a million times more energy than carbon dioxide (CO₂).
 - Other carbon-based PM may also be light-absorbing, particularly brown carbon (BrC), which are organic carbon (OC) compounds that absorb light within the visible and ultraviolet range of solar radiation. The net contribution of BrC to climate is presently uncertain.
- The full effect of BC on climate must be assessed in the context of co-emitted pollutants. BC is always emitted with other particles and gases, such as sulfur dioxide (SO₂), nitrogen oxides (NO_x), and OC. Some of these co-emitted pollutants result in aerosols (e.g. sulfate, nitrate) which exert a cooling effect on climate. The composition of this emissions mixture determines the net impact on climate.
- Atmospheric processes that occur after BC is emitted, such as mixing, aging, and coating, can also affect the net influence of BC on climate.
- The short atmospheric lifetime of BC and the mechanisms by which it affects climate distinguish it from long-lived GHGs like CO₂.
 - *BC has a short atmospheric residence time of days to weeks.* This short lifetime, combined with the strong warming potential of BC, means that the climate benefits of reductions in current emissions of BC will be nearly immediate. In contrast, long-lived GHGs persist in the atmosphere for centuries. Therefore, reductions in GHG emissions will take longer to influence atmospheric concentrations and will have less impact on climate on a short timescale. However, since GHGs are by far the largest contributor to current and future climate change, deep reductions in these pollutants are essential over the long-term.
 - *Emissions sources and ambient concentrations of BC vary geographically and temporally,* resulting in climate effects that are more regionally and seasonally dependent than the effects of long-lived, well-mixed GHGs. Likewise, mitigation actions for BC will produce different climate results depending on the region, season, and emission category.
 - *BC influences climate through multiple mechanisms:*
 - *Direct effect:* BC absorbs both incoming and outgoing radiation of all wavelengths. In contrast, GHGs mainly trap outgoing infrared radiation from the earth’s surface.

- 1 ▪ *Snow/ice albedo effect:* BC deposited on snow and ice darkens the surface and
2 decreases reflectivity (albedo), thereby increasing absorption and accelerating
3 melting. GHGs do not directly affect the Earth’s albedo.
- 4 ▪ *Indirect effect:* BC also alters the properties of clouds, affecting cloud reflectivity,
5 precipitation, and surface dimming. These impacts are associated with all ambient
6 particles, but not GHGs.
- 7 • The direct and snow/ice albedo effects of BC are widely understood to lead to climate warming. The
8 indirect effects of BC on climate via interaction with clouds are much more uncertain, but may partially
9 offset the warming effects.
- 10 ○ The direct radiative forcing effect of BC is the best quantified and appears to be significant
11 on both global and regional scales. In 2007, the Intergovernmental Panel on Climate Change
12 (IPCC) provided a central estimate for global average direct forcing by BC of +0.34 (±0.25)
13 Watts per square meter ($W m^{-2}$). Other studies have also estimated positive values, some
14 exceeding $1 W m^{-2}$. In addition, according to the IPCC the BC snow/ice albedo effect
15 contributes +0.1 (±0.1) $W m^{-2}$ to the global average forcing from BC. Based on the IPCC
16 estimates, the direct and snow/ice albedo effects of BC together contribute more to
17 warming than any GHG other than CO_2 and methane. The IPCC’s radiative forcing estimates
18 for elevated concentrations of CO_2 and methane are $+1.66 W m^{-2}$ and $+0.48 W m^{-2}$,
19 respectively.
- 20 ○ All aerosols (including BC) affect climate indirectly by changing the reflectivity and lifetime
21 of clouds. The net indirect effect of all aerosols is very uncertain but is thought to be a net
22 cooling influence. The IPCC estimates that the globally averaged indirect forcing from all
23 aerosols ranges between $-0.10 W m^{-2}$ and $-1.80 W m^{-2}$. BC has additional indirect effects—
24 including changes to cloud stability and enhanced precipitation from colder clouds—that
25 can lead to warming. The net climate influence of the indirect effects of BC is not yet clear.
26 However, the warming due to the direct and snow/ice albedo effects very likely exceeds any
27 cooling from indirect effects.
- 28 • Regional climate impacts of BC are highly variable, and the effects of BC on warming and melting are
29 especially strong in sensitive regions such as the Arctic and the Himalayas. Estimates of snow and
30 ice albedo forcing in key regions also exceed global averages.
- 31 ○ Studies have shown that BC has especially strong warming effects in the Arctic, contributing
32 to earlier spring melting and sea ice decline. All particle mixtures reaching the Arctic are a
33 concern, because over highly reflective surfaces such as ice and snow, even emissions
34 mixtures that contain more reflective (cooling) aerosols can lead to warming if they are
35 darker than the underlying surface. Studies have estimated atmospheric forcing for BC of
36 $+1.2 W m^{-2}$ in the Arctic in springtime, with an additional $+0.53 W m^{-2}$ from snow albedo
37 forcing. Where earlier melting of ice and snow reveals darker surfaces (e.g. ocean water or
38 soils) underneath, the warming effect is magnified.

- 1 ○ BC has also been shown to be a significant factor in the observed increases in melting rates
2 of some glaciers and snowpack in parts of the Hindu Kush-Himalayan-Tibetan (HKHT) region
3 (the “third pole”). Average radiative forcing of BC on snow and ice in the Tibetan plateau
4 has been estimated at $+1.5 \text{ W m}^{-2}$, with instantaneous forcing in some places in the spring
5 estimated as high as $+20 \text{ W m}^{-2}$.
- 6 ○ In the United States, deposition of BC on mountain glaciers and snow packs has been shown
7 to reduce snow cover and overall snowpack and to contribute to earlier spring melting,
8 which reduces the amount of meltwater available later in the year.
- 9 ● At the same time it contributes to net atmospheric warming, BC also contributes to surface dimming
10 (cooling), the formation of Atmospheric Brown Clouds (ABCs), and resultant changes in the pattern
11 and intensity of precipitation.
 - 12 ○ The absorption and scattering of incoming solar radiation by BC and other particles causes
13 surface dimming by reducing the amount of solar radiation reaching the Earth’s surface. In
14 some regions, especially Asia, southern Africa, and the Amazon Basin, BC, BrC, sulfates,
15 organics, dust and other components combine to form pollution clouds known as
16 Atmospheric Brown Clouds (ABCs). ABCs have been linked to surface dimming and a
17 decrease in vertical mixing, which exacerbates air pollution episodes. ABCs also contribute
18 to changes in the pattern and intensity of rainfall, and to observed changes in monsoon
19 circulation in South Asia. In general, regional changes in precipitation due to BC and other
20 aerosols are likely to be highly variable, with some regions seeing increases while others
21 experience decreases.
- 22 ● The climatic impacts of BC are less certain than those of the well-mixed GHGs. Because of this, and
23 because of the wide range of climate impacts associated with BC, it is currently difficult to assess
24 and assign a monetary value to the full range of climate damages attributable to BC emissions.

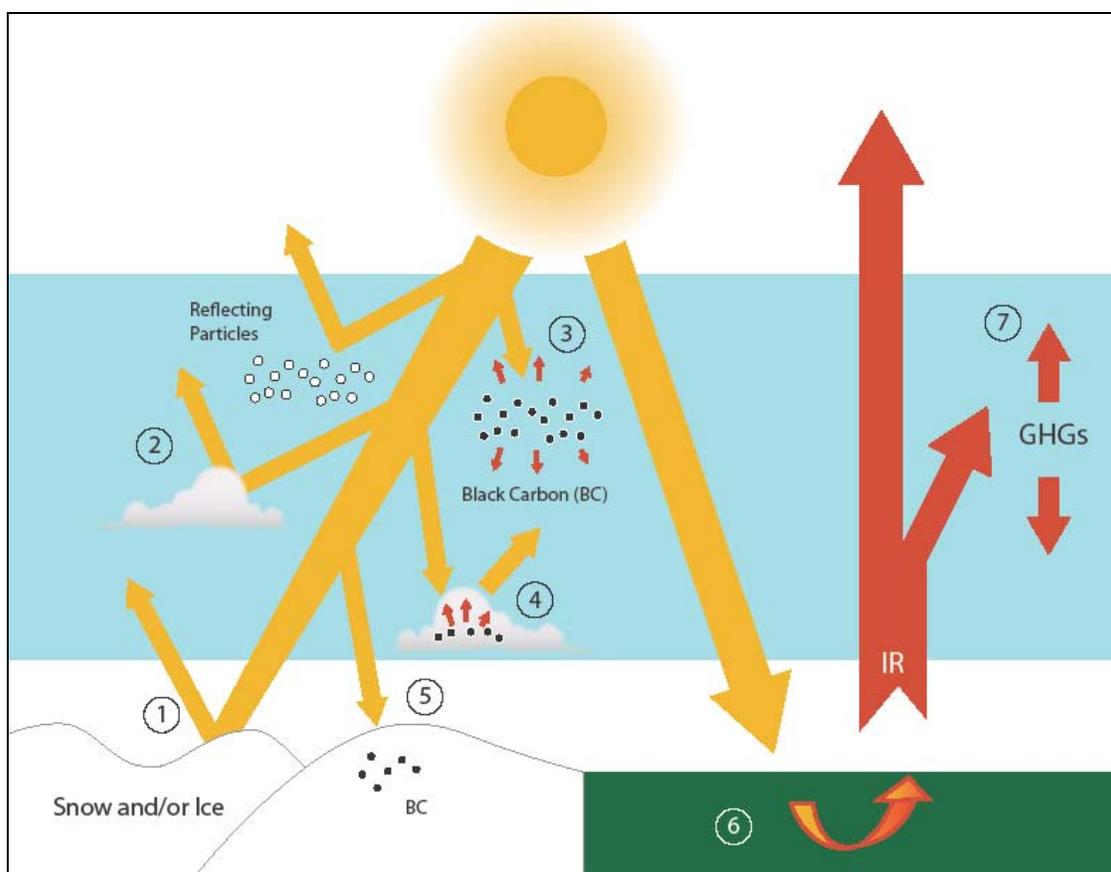
25 **2.2 Introduction**

26 There is a general consensus within the scientific community that black carbon (BC) is
27 contributing significantly to climate change at both the global and regional levels. Like carbon dioxide
28 (CO_2), BC is produced through the burning of carbon-based fuels, including fossil fuels, biofuels and
29 biomass. BC particles are part of the total mix of particulate matter (PM) released during incomplete
30 combustion of these fuels. BC influences climate by absorbing sunlight when suspended in the
31 atmosphere or when deposited on the Earth’s surface. The energy absorbed by BC is then released as
32 heat and contributes to atmospheric warming and the accelerated melting of ice and snow. In addition,
33 BC is capable of altering other atmospheric processes, such as cloud formation and precipitation
34 patterns.

35 The strong absorption, short-atmospheric lifetime, and other characteristics of BC make its
36 impacts on climate different from those of long-lived greenhouse gases (GHGs) like carbon dioxide (CO_2)
37 (see Figure 2-1). Because BC is involved in complex atmospheric physical and chemical processes, it is

1 difficult to disentangle all of BC’s impacts and to evaluate its net effect on climate. In addition, the
 2 combustion processes that produce BC also produce other pollutants, such as sulfur dioxide (SO₂),
 3 nitrogen oxides (NO_x), and organic carbon (OC). Since many of these compounds have a cooling effect,
 4 BC’s impacts are mixed with —and sometimes offset by—these co-emitted substances. This must be
 5 considered when evaluating the net effect of emissions sources.

6 This chapter focuses on how and to what extent BC influences the earth’s climate. Specifically,
 7 this chapter discusses approaches for defining BC and other light-absorbing particles, highlights the
 8 differences between BC and GHGs, describes the processes by which BC affects climate, and addresses
 9 the role of co-emitted pollutants. Further, this chapter summarizes recent scientific findings regarding
 10 the magnitude of BC’s impacts on global and regional climate, highlighting the effect of BC in sensitive
 11 regions such as the Arctic and other snow and ice-covered regions.



12

13 **Figure 2-1. Effects of Black Carbon on Climate, as compared to Greenhouse Gases.**

- 14 (1) Sunlight that penetrates to Earth’s surface reflects off bright surfaces, especially snow and ice.
 15 (2) Clean clouds and non-light absorbing (transparent) particles scatter or reflect sunlight, reducing the
 16 amount of solar energy that is absorbed by the surface.
 17 (3) Black carbon (BC) suspended in the atmosphere absorbs some incoming solar radiation, heating the
 18 atmosphere.
 19 (4) Clouds containing suspended BC can absorb some incoming solar radiation, reducing the quantity that is
 20 reflected. Clouds warmed by the absorbed energy have shorter atmospheric lifetimes and may be less

- 1 likely to precipitate compared to clean clouds.
- 2 (5) BC deposited on snow and/or ice absorbs some of the sunlight that would ordinarily be reflected by clean
3 snow/ice, and increases the rate of melting.
- 4 (6) Most solar radiation is absorbed by Earth's surface and warms it. Part of the absorbed energy is
5 converted into infrared radiation that is emitted into the atmosphere and back into space.
- 6 (7) Most of this infrared radiation passes through the atmosphere, but some is absorbed by greenhouse gas
7 (GHG) molecules like CO₂, methane, ozone and others. These gases re-emit the absorbed radiation with
8 half returning to Earth's surface. This 'greenhouse gas' effect warms the earth's surface and the lower
9 atmosphere.

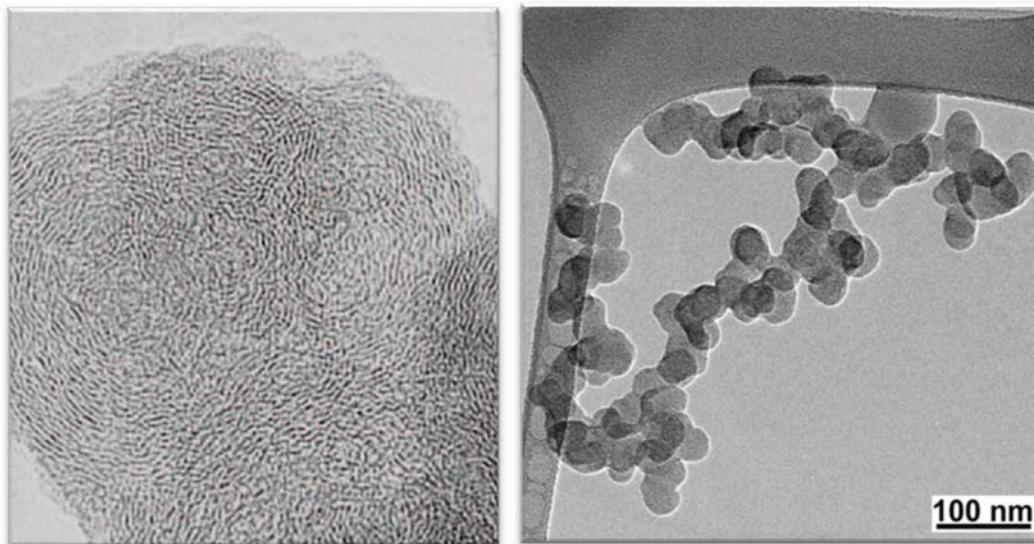
10 **2.3 Defining Black Carbon and Other Light-Absorbing PM**

11 All PM in the atmosphere can affect the Earth's climate by absorbing and scattering light.
12 Absorbed sunlight represents energy added to the Earth's system and leads to climate warming,
13 whereas scattering generally leads to increased reflection of light back to space, causing climate cooling
14 (Charlson et al., 1992; Forster et al., 2007; Seinfeld and Pandis, 2006). Some components of PM absorb
15 light more strongly than others. Carbon-based PM has typically been divided into two classes: black
16 carbon, which is also described as light-absorbing carbon, and organic carbon, which is often described
17 as non-light-absorbing. Neither BC nor OC has a precise chemical definition; rather, they constitute
18 complex mixtures of many different compounds. The classifications "black" carbon and "organic"
19 carbon are simplifications of that diversity.

20 BC is the most strongly light-absorbing component of PM: in this report, it is defined as the
21 carbonaceous component of PM that absorbs all wavelengths of solar radiation. Per unit of mass in the
22 atmosphere, BC can absorb a million times more energy than CO₂ (Bond and Sun 2005). This strong
23 absorptive capacity is the property most relevant to its potential to affect the Earth's climate. The
24 quantity of BC produced during combustion depends largely on the combustion conditions: BC is
25 emitted whenever insufficient oxygen and heat are available for complete combustion (see Text Box 2-
26 1). BC, which is mostly elemental carbon (EC) by mass, originates as very tiny spherules, ranging in size
27 from 0.001 to 0.005 micrometers (µm), and aggregates to particles of larger size (0.1 to 1 µm) (Figure 2-
28 2). Thus, BC is a constituent of fine particles (PM_{2.5}). Their size is important, since particles that are
29 similar in size to wavelengths of light are most likely to interact with light via scattering and absorption.
30 Because BC is directly emitted from sources, it is considered to be a *primary* particle. This makes it
31 distinct from *secondary* particles such as sulfates and nitrates which are formed in the atmosphere from
32 gaseous precursors like SO₂, NO_x and volatile organic compounds (VOC). Such secondarily formed
33 particles make up the bulk of PM_{2.5} mass in many areas in the U.S.

34 **BLACK CARBON (BC)** is the carbonaceous component of PM that absorbs all wavelengths of solar
35 radiation. BC is a product of incomplete combustion, and is commonly called "soot."

36



1
2 **Figure 2-2. Black Carbon Images.** LEFT: High resolution transmission electron microscopy (TEM) image
3 of black carbon spherule (notice wavy, graphene-like layers) (Pósfai and Buseck, 2010). RIGHT: TEM
4 image of aggregated BC spherules (Alexander et al., 2008).

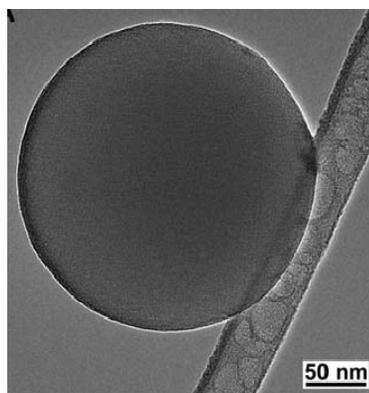
5 Many alternative definitions of BC exist in the literature. In fact, because BC is involved in
6 complex chemical processes, both during combustion and in the atmosphere, it has been studied and
7 evaluated by different disciplines for different purposes. This has resulted in a variety of definitions
8 related to chemical and/or physical particle properties, intended applications, and the different
9 measurement and estimation approaches, and has given rise to a confusing array of descriptive terms
10 such as “graphitic carbon”, “elemental carbon (EC)”, “equivalent black carbon (BCe)”, and “soot” which
11 are used in the literature as interchangeable with BC (see Chapter 5 and Appendix 1). Soot is actually a
12 mixture of PM produced during incomplete combustion; its very dark color indicates the presence of
13 high quantities of black (light-absorbing) carbon. Due to the confusion arising from the mixed use of
14 terminology, atmospheric chemistry and climate scientists have been proposing more specific language
15 to classify these materials (Bond and Bergstrom 2006; Andreae and Gelencsér, 2006).

16 For purposes of regional air quality management (e.g., human health studies related to air
17 quality, the evaluation of modeled estimates, and the attribution of emissions to sources), BC is
18 measured as a constituent of ambient $PM_{2.5}$ and is expressed in units of mass. For example, BC
19 emissions inventories, as discussed in Chapter 4, are generally expressed in mass units (e.g. tons/year).
20 Thus, light-absorption measurements are often converted into estimates of carbon mass. Due to
21 reliance on these mass-based indicators, BC is frequently labeled EC due to the long-standing use of
22 carbon measurement methods from which the air quality and emission estimates were derived.
23 Chapter 5 will provide a brief description of the various BC absorption-related and BC mass-related
24 measurement approaches. A more detailed discussion is then found in Appendix 1.

25 In addition to BC, other light-absorbing carbonaceous particles are emitted during combustion
26 processes. These compounds, however, are generally less strongly absorbing than BC (Novakov, 1995;
27 Alexander et al., 2008). In particular, the group of organic compounds referred to as brown carbon (BrC)

1 can be included in the category of light-absorbing carbon, which has the potential to warm the climate.
2 Like BC, BrC is a product of incomplete combustion (see Text Box 2-1). Many different forms of BrC
3 exist. For example, BrC can exist as “tar balls” which result from combustion of biomass and biofuels
4 and are carbon-based spherules with diameters between 0.03 and 0.5 μm (Pósfai et al., 2004).
5 Alexander et al. (2008), in their study of East Asian pollution plumes, identified very similar, light
6 absorbing carbon particles (see Figure 2-3). BrC is also added through atmospheric reactions of
7 particles and gases downwind of sources (Andreae, 2009). BrC compounds are diverse chemically, so
8 that their characteristic light absorption wavelengths – and colors – vary significantly. A mixture of
9 these compounds, because of the wide range of colors, appears brown to the human eye. BrC absorbs a
10 narrower range of wavelengths than BC, mainly within the visible and ultraviolet (UV) range (see Figure
11 2-4).

12 **BROWN CARBON (BrC)** refers to a number of organic carbon compounds that absorb light within the
13 visible and ultraviolet (UV) range of solar radiation. Like BC, it is a product of incomplete combustion.

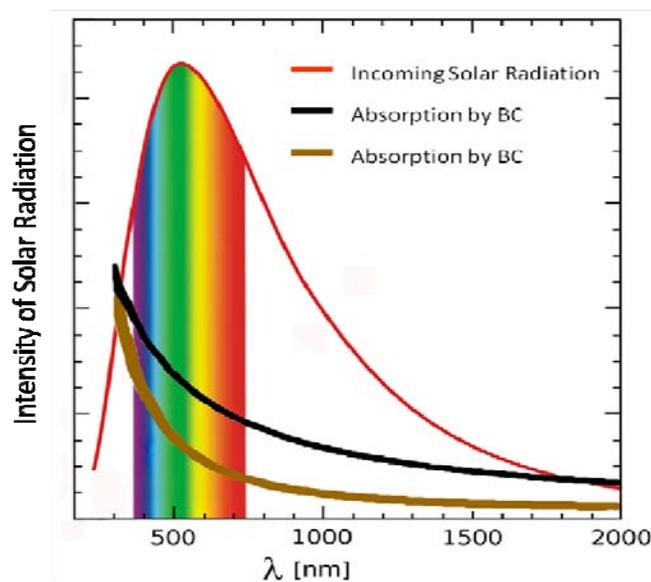


18 **Figure 2-3. Brown Carbon Image.** TEM image of a typical “tar ball” or “carbon sphere”
19 (Alexander et al. 2008).

20

21 The total quantity of solar energy absorbed by a BrC mixture depends upon the molecular
22 structure of the compounds present and the total mass of the material present in the combustion
23 plume. In general, the fuel type determines the quantity of BrC produced from a specific combustion
24 event. Some sources, such as open biomass burning, can produce substantially more BrC than BC. As a
25 result, the quantity of BrC may lead to a greater total solar energy absorbed by BrC than BC even though
26 BrC absorbs less energy than BC per unit mass.

27



1
2 **Figure 2-4. Light Absorption by Black Carbon and Brown Carbon.** The sun emits radiation with an
3 intensity that peaks in the visible wavelength range (denoted by the rainbow band). The black and brown
4 lines are idealized representations of how light absorption by BC and BrC differs as a function of
5 wavelength. The true quantity of light absorbed by either form of carbon depends upon the
6 concentration of these materials in the atmosphere. As indicated, BC absorbs all wavelengths of solar
7 radiation far more effectively than BrC and, will therefore have a greater warming effect, by mass.

8 Generally, BrC is not classified independently. It is typically considered part of the organic
9 carbon (OC) fraction of PM, which is often described as non-light-absorbing. The presence of light-
10 absorbing BrC within the OC fraction suggests that this partition of carbon-based particles into only two
11 classes is inadequate to characterize all the types of particles relevant to climate. Some authors have
12 suggested that a more realistic representation would be a continuum from light-absorbing to light-
13 scattering with BC at one end, most OC at the other, and the special class of BrC somewhere in the
14 middle.¹ It has also been suggested that scattering and absorption by BrC needs to be explicitly included
15 in climate models. For example, Alexander et al. (2008) argues that on a particle by particle basis, the
16 carbon spheres show light absorption commensurate to, or greater than, that of BC particles and that
17 their statistical measurements find that the carbon spheres are very abundant.

18 In addition, other gases such as SO₂ and NO_x emitted during the combustion process react in the
19 atmosphere to form non-carbonaceous PM. These particles are very effective at scattering solar
20 radiation. Since all of these combustion by-products are co-emitted with BC, they also offset some of
21 the climate impacts of BC particles, as discussed further later in this chapter in section 2.6.1.4.

¹ Particles containing iron and other calcium, aluminum, and potassium oxides also absorb light. Like BrC, metal oxides are very effective, more so than BC, at absorbing light at shorter wavelengths. Some metal oxides are derived from heavy fuel sources such as residual fuel oil (Huffman et al., 2000). High concentrations of such particles can result from windblown dust and may be significant during dust and sand storms that occur in Africa, China, and the Middle East. These fine particle constituents can travel long distances and may contribute to a positive radiative forcing to a limited degree (Prospero et al., 2010; Liu et al., 2008).

1

2 Text Box 2-1: Products of Incomplete Combustion

3 Most combustion on Earth (both anthropogenic and natural) involves carbon-based fuels, including fossil fuels (e.g.
4 coal, oil, natural gas), biomass (e.g. wood, crop residues) and biofuels (e.g. ethanol). Complete combustion of a
5 carbon-based fuel means all carbon has been converted to CO₂. Once ignited, the fuel must be well mixed with
6 oxygen at a sustained high temperature for this to occur. Incomplete combustion leads to a variety of particle
7 emissions, depending on the combustion conditions (oxygen availability, flame temperatures and fuel moisture)
8 and the type of fuel burned (gas, liquid, or solid).

9 BC is formed as part of a mixture called soot during the **flaming** phase of the combustion process. The amount of
10 BC emitted depends largely on combustion conditions. If there is sufficient oxygen and high temperatures, the soot
11 will be completely oxidized, and BC emissions will be minimal. Closed combustion systems (e.g., furnaces,
12 combustors, reactors, boilers, and engines) can be engineered to increase the mixing of air with the fuel and
13 insulated to ensure temperatures remain high. Open and uncontrolled burning produces large quantities of BC
14 because oxygen availability and temperatures within the fire can vary widely.

15 The form of the fuel also strongly influences the likelihood of complete combustion:

- 16 • *Gas phase fuels* (e.g. natural gas) can be readily mixed with oxygen, which minimizes the emission of
17 carbonaceous particles.
- 18 • *Liquid fuels* generally must vaporize in order to fuel flaming combustion (e.g. gasoline). If a liquid fuel
19 contains heavy oils, vaporization and thorough mixing with oxygen are difficult to achieve. The heavy
20 black smoke emitted by some marine vessels (which burn a sludge-like grade of oil known as “bunker
21 fuel”) is evidence of substantial BC emissions.
- 22 • *Solid fuels* require preheating and then ignition before flaming combustion can occur (e.g. wood). High
23 fuel moisture can suppress full flaming combustion, contributing to the formation of BrC particles as well
24 as BC (Graber and Rudich, 2006; Pósfai et al., 2004; Alexander et al., 2008).

25 Solid fuels are often preheated prior to ignition. This process involving the thermal breakdown of the fuel, known
26 as **pyrolysis**, produces a wide array of BrC materials. When sustained, pyrolysis converts solid fuels such as coal
27 and biomass into char. There is also a non-flaming process known as **smoldering** that is a slower, cooler form of
28 combustion which occurs as oxygen attacks the surface of heated solid fuel directly. The smoke that appears is
29 light-colored, consisting of a variety of small organic particles, including BrC. BC does not form, since temperatures
30 are too low. During open or uncontrolled burning of solid fuels, all stages of the burning process—pyrolysis,
31 smoldering, and flaming combustion—occur simultaneously, in different parts of the fuel pile, resulting in
32 emissions of BC and BrC. (Illustration to be added.)

33 **[End of Text Box 2-1]**

34

35 2.4 Key Attributes of BC and Comparisons to GHGs

36 The impact of BC on climate depends on a number of other factors in addition to its powerful
37 light-absorption capacity. These include atmospheric life span, the geographic location of emissions,
38 altitude and interactions with clouds (i.e., indirect effects), the presence of co-emitted pollutants, and
39 the influence of aging and mixing processes in the atmosphere. In many of these aspects, BC differs
40 substantially from long-lived GHG, as summarized in Table 2-1. These differences have implications for

1 how BC influences climate and how mitigation of BC differs from CO₂. Each of these dimensions is
 2 explored further below.

3 **Table 2-1. Comparison of Black Carbon to CO₂ on the Basis of Key Properties that Influence the Climate**

	BC	CO ₂
Atmospheric lifetime	Days to weeks	100+ years (some stays for millennia)
Distribution of atmospheric concentrations	Highly variable both geographically and temporally, correlating with emission sources	Generally uniform across globe
Direct radiative properties	Absorbs all wavelengths of solar radiation	Absorbs only thermal infrared radiation
Global mean radiative forcing (IPCC) (see section 2.6)	0.34±0.25 W m ⁻² direct forcing 0.1±0.1 W m ⁻² (snow/ice albedo forcing)	1.66±0.17 W m ⁻²
Cloud interactions	Multiple cloud interactions that can lead to warming or cooling, as well as effects on precipitation	No direct cloud interactions
Surface albedo effects	Contributes to accelerated melting of snow/ice and reduces reflectivity by darkening snow and ice, enhancing climate warming	No direct surface albedo effects
Contribution to current global warming	Likely 3 rd largest contributor (after CO ₂ and CH ₄), but large uncertainty	Largest contributor

4
 5 Particles in general have relatively **short atmospheric lifetimes** in comparison to GHGs. Particles
 6 of any type, including BC, are removed from the atmosphere within days to weeks by precipitation
 7 and/or dry deposition to surfaces. This short atmospheric residence time curtails their total
 8 contribution to the Earth’s energy balance, even for those particles like BC that have strong absorptive
 9 capacity. The efficiency with which particles are removed is influenced by their size and chemical
 10 composition. Atmospheric aging can increase the size of a particle or alter its chemical composition in a
 11 way that makes it an efficient nucleus for cloud droplet formation, facilitating its removal by
 12 precipitation. Despite short atmospheric lifetimes, fine combustion particles including BC can be
 13 transported up to thousands of miles from sources.

14 By contrast, most GHGs have longer lifetimes. This enables them to become well mixed in the
 15 atmosphere and to continue to absorb energy over many decades. Gases such as N₂O, CH₄, or the
 16 hydrofluorocarbons (HFCs) have lifetimes that range from as short as a year for some of the HFCs to as
 17 long as 50,000 years for CF₄ (a perfluorocarbon) (Forster et al., 2007). The carbon in CO₂ cycles between
 18 the atmosphere, oceans, ecosystems, soil, and sediments; therefore, CO₂ does not have a single defined
 19 lifetime. Computer models have indicated that about half of an emissions pulse of CO₂ will disappear

1 within 30 years, 30% within a few centuries, and the last 20% may remain in the atmosphere for
2 thousands of years (Denman et al., 2007).

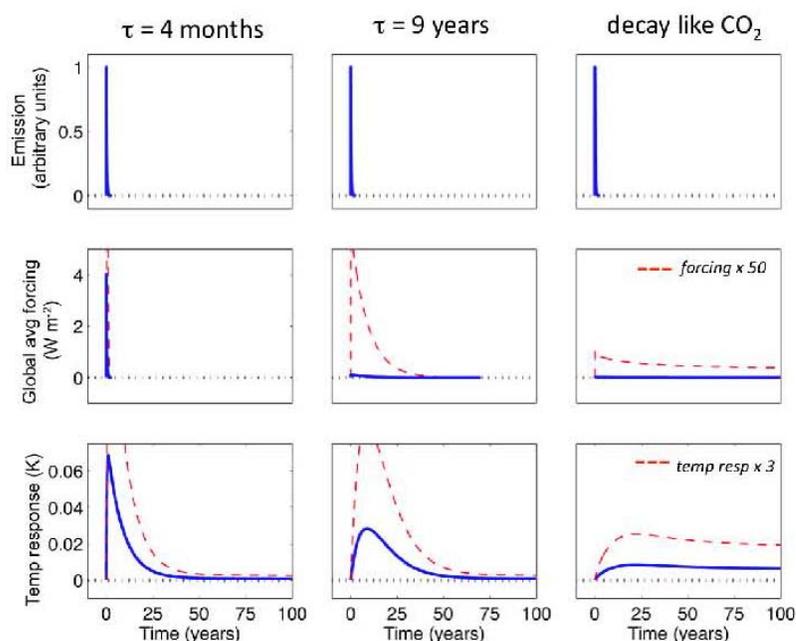
3 BC's short atmospheric lifetime means that atmospheric concentrations are highest near
4 significant emissions sources and during time periods and seasons of emissions releases. This **high**
5 **spatial and temporal variability** affects BC's impacts on climate. BC is a regional pollutant. CO₂ and
6 other GHGs are global pollutants with relatively uniform concentrations around the globe, regardless of
7 the location or exact timing of emissions. It is generally assumed that CO₂ emitted in one location has
8 approximately the same effect on climate as CO₂ emitted somewhere else. The same is not true with
9 BC.

10 **Geographic location and altitude** are also important determinants of the impact of BC on
11 climate. Particles have a greater effect on the net absorption of solar radiation by the atmosphere when
12 they are emitted or transported over light-colored, reflective (i.e. high "albedo") surfaces such as ice,
13 snow, and deserts. In the absence of PM, a high percent of sunlight would reflect off these surfaces and
14 return to space. Therefore, any absorption of either incoming or reflected light by PM above these
15 surfaces is more likely to lead to warming than for PM above darker surfaces. Even PM typically
16 classified as reflecting can darken these bright surfaces and contribute to warming (Scinocca et al.,
17 2006). This mechanism explains why studies have found the effects of BC to be magnified in the Arctic
18 and other alpine regions, as discussed in sections 2.6.4 and 2.6.5. In addition, the net radiative effects of
19 BC can be sensitive to altitude. As with particles suspended above a bright desert or glacier, particles
20 suspended above bright cumulus clouds can absorb both incoming and outgoing solar radiation,
21 increasing the net radiative effect of the light absorbing particle. When suspended between cloud
22 layers or beneath a cloud, the particle may be shielded from incoming light, therefore lessening its
23 potential radiative impact (Shultz et al., 2006).

24 Other key distinguishing features of BC include the wide **range of mechanisms** through which it
25 influences climate and its association with other **adverse public health and welfare effects**. In addition
26 to the direct radiative forcing found with both BC and GHGs, BC has significant interactions with clouds
27 that can result in both warming and cooling effects. It can also cause melting and warming via
28 deposition to snow and ice. All of these effects are discussed further later in this chapter. GHGs do not
29 have these indirect effects on clouds, snow and ice. GHGs influence climate through direct radiative
30 forcing only. BC and other particles are also directly associated with a host of other environmental
31 effects, such as changes in precipitation patterns and surface dimming. Finally, as a constituent of PM_{2.5},
32 BC is linked to a range of public health impacts (see Chapter 3). This, too, distinguishes it from long-lived
33 GHGs.

34 An important implication of BC's strong absorptive capacity, coupled with its short atmospheric
35 lifetime, is that when emissions of BC are reduced, atmospheric concentrations of BC will decrease
36 immediately and the climate, in turn, will respond relatively quickly. The **potential for near-term**
37 **climate benefits** (within a decade) is one of the strongest drivers of the current scientific interest in BC.
38 Mitigation efforts that permanently reduce BC emissions can halt the effects of BC on temperature,
39 snow and ice, and precipitation almost immediately. In contrast, when long-lived GHG emissions are

1 reduced, the climate takes longer to respond because atmospheric GHG concentrations—the result of
 2 cumulative historic and present-day emissions—remain relatively constant for longer periods (see, for
 3 example, **Figure 2-5**). It is important to recognize, however, that the short atmospheric lifetime of BC
 4 also means that reductions in current BC emissions will have little impact on long-term warming trends,
 5 which are driven by persistent species like CO₂. Only sustained reductions in long-lived GHGs can avert
 6 long-term climate change.



7

8 **Figure 2-5. Climate Response to Emissions of Pollutants with Different Lifetimes.** Figure illustrates
 9 forcing and temperature response to pulse emissions of three hypothetical substances with different
 10 atmospheric lifetimes (Column 1 illustrates short-lived species like BC, Column 2 illustrates species with
 11 medium atmospheric lifetime like methane, and Column 3 illustrates a long-lived species like CO₂).
 12 Dashed red trace shows forcing value times 50 to show more detail in the time dependence. (Source:
 13 Bond et al., 2011)

14 These differences between BC and GHGs have significant implications for BC mitigation
 15 decisions. Specifically, the effectiveness of a given mitigation effort depends on the timing and location
 16 of the emissions, the atmospheric processes, transport, and deposition rates of the emissions from the
 17 specific sources, the underlying surface (e.g. ice and snow), and the presence of co-pollutant emissions.
 18 This constitutes a significant difference from long-lived GHGs where the precise timing and location of
 19 emissions (or emissions reductions) does not matter significantly with respect to the climate impact.

20 2.5 The Role of Co-Emitted Pollutants and Atmospheric Processing

21 BC is never emitted into the atmosphere in isolation. Rather, it is part of a mixture of
 22 substances emitted during the combustion process. The composition of this mixture can vary
 23 significantly, depending on combustion conditions and fuel type. BC is generally accompanied by OC,

1 including BrC and other carbonaceous materials. In addition, an emissions plume may contain water,
2 inorganic potassium and sodium salts, ammonium nitrate and sulfate, gaseous constituents (e.g., SO₂,
3 NO_x and volatile organic compounds, or VOCs), various hazardous air pollutants (e.g. metals), and even
4 soil particles.

5 The absorptive properties of an emissions plume from a specific source will depend on all of the
6 **co-emitted pollutants**, and on how these constituents interact with one another in the atmosphere. BC
7 is co-emitted with OC and/or sulfate, nitrate and gaseous constituents (SO₂ and NO_x). Since OC and
8 sulfate and nitrate particles generally exert a net cooling influence, these pollutants play an important
9 role in determining the net absorptive capacity of the emissions plume. These other constituents,
10 however, may be emitted in greater volume than BC, counteracting the warming influence of BC. Thus,
11 estimating the climate impact of BC quantitatively requires accounting of the impact of these co-emitted
12 pollutants. Emissions from a single source can also vary over time. For example, the flaming phase of a
13 wildfire produces much more BC than its smoldering phase. Also, when diesel trucks are under load,
14 they produce more BC than during other parts of their driving cycle. Total **particle number** also impacts
15 scattering and absorption: the more particles present in a portion of the atmosphere, the greater the
16 probability that light rays will be scattered or absorbed by some of these particles.

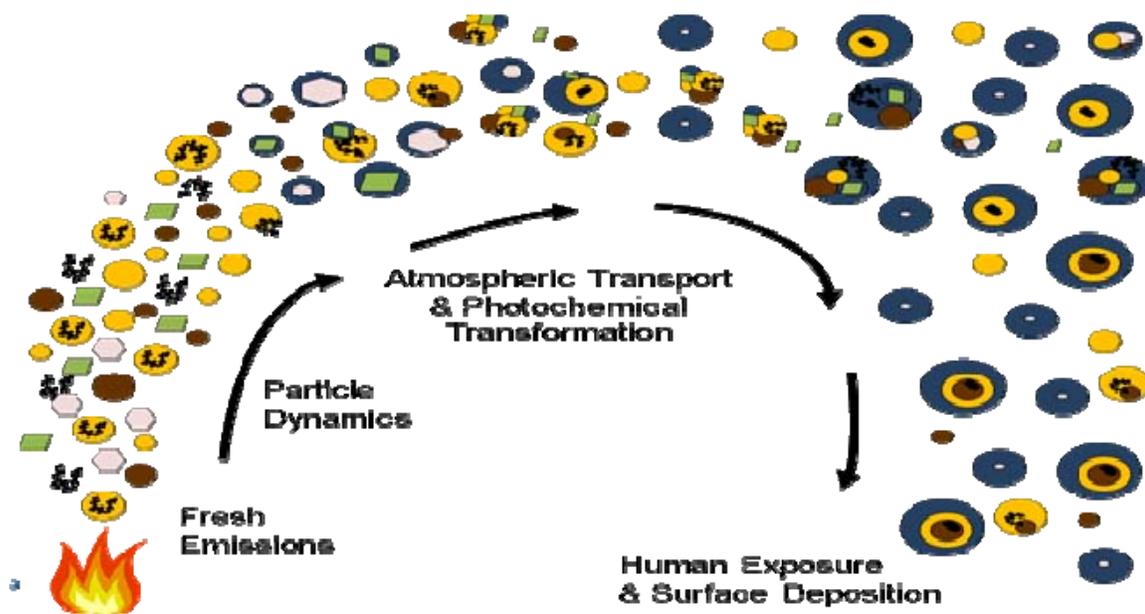
17 Despite these complications, emissions from particular sources are often characterized in terms
18 of their OC to BC ratio. Sources whose emissions mixtures are richer in BC relative to the amount of OC
19 emitted are more likely to be climate warming; therefore, mitigation measures focusing on these
20 sources (with lower OC:BC ratios) are more likely to produce climate benefits. These ratios are useful in
21 that they take the emissions mixture into account; however, they rely on crude accounting methods and
22 cannot provide precise measures of a particular source's impacts. A particular concern is the common
23 presumption that all OC is cooling, when in fact some components (especially BrC) are light-absorbing
24 and may contribute to warming associated with an emissions mixture. (The use of OC:BC ratios is
25 discussed further in Chapter 11.)

26 In theory, particles and gases of different types are emitted as distinct components in an
27 emissions plume. Atmospheric scientists refer to an emissions plume with this kind of high inter-particle
28 chemical variability as "**externally mixed**." The externally mixed plume, however, undergoes rapid
29 chemical and physical transformations. As the chemical composition of the particles approaches
30 uniformity, the particles are referred to as "**internally mixed**." Many mechanisms contribute to this
31 mixing. Particles colliding and sticking together (the process of coagulation) reduces the overall particle
32 number and reduces the differences in chemical composition among the individual particles in the
33 plume.

34 Condensation of gas phase pollutants and water vapor onto particles also plays an important
35 role. The degree of mixing influences the absorptive properties of the particle. Internal mixtures of
36 particles that include BC have been observed to absorb light more strongly than pure BC alone, by a
37 factor of approximately 1.5 to 2. In situ measurements indicate that emissions become internally mixed
38 within a few hours (Moffet and Prather, 2009). Whether BC is modeled as an externally or internally
39 mixed particle can have a large effect on resulting estimates of radiative forcing (see section 2.6.1).

1 Emissions plumes from different sources interact with each other as well as with
 2 surrounding atmosphere. As a combustion emissions plume rises into the atmosphere, it is diluted by
 3 ambient air (see Figure 2-6). The open atmosphere contains a number of reactive gases and particles
 4 originating from a wide variety of anthropogenic and natural sources. Physical and chemical changes
 5 resulting from coagulation, condensation, and other photochemical and atmospheric processes can alter
 6 the climate-forcing impact of a given emissions plume (Lauer and Hendricks, 2006) (see Table 2-2).
 7 These mixing and aging processes are complex. Excluding them from models of the climate impacts of
 8 BC, however, may yield incomplete or erroneous estimates. For example, coating of a BC particle by a
 9 clear (light-scattering) shell has been shown to enhance light absorption because the shell acts as a lens
 10 that directs more light toward the core (Lack and Cappa, 2010). Other authors have also found that
 11 light-absorption by BC is enhanced when BC particles are coated by sulfate or other light scattering
 12 materials (Moffett and Prather, 2009; Shiraiwa et al., 2009; Bond, Habib and Bergstrom, 2006; Sato
 13 2003). Other atmospheric processes, however, such as further chemical processing or particle growth
 14 through coagulation, may off-set this enhancement. Observations by Chan et al. (2010) at a rural site in
 15 Ontario, where BC particles were assumed to be coated, did not show enhanced light absorption.
 16 Furthermore, coated particles are more easily removed by cloud droplets and precipitation, decreasing
 17 their atmospheric lifetime (Stier et al., 2006).

18



19

20 **Figure 2-6. Particle Transformation in the Atmosphere, from Point of Emission to Deposition.**
 21 A variety of physical and chemical processes contribute to changing the light-absorption capacity
 22 of a fresh plume.

23

24

1

2 **Table 2-2. Examples of Particle Types and Mixtures Present in Combustion Plumes.** The size, shape,
 3 and chemical composition of a particle or particle mixture determine its radiative properties.

Particle	Type	Radiative properties*
	Black carbon ^a	Absorbing (all solar wavelengths)
	Brown (or yellow) carbon ^b	Absorbing (UV and some visible)
	Non-absorbing carbon ^b	Scattering
	Nitrate ^c	Scattering
	Sulfate ^c	Scattering
	Black carbon coated with brown or non-absorbing carbon ^d	Absorbing (enhanced by partial internal reflection of solar radiation); fractionally scattering
	Black carbon associated with sulfate or nitrate ^e	Absorbing plus some scattering
	Cloud and fog droplets ^f	Scattering
	Complex of several particles ^e	Absorbing and scattering
	Mixed particle (cloud processed) ^g	Absorbing (enhanced by partial internal reflection of solar radiation); fractionally scattering

4

5 **NOTES:**

- 6 a . Fresh BC is produced primarily during flaming combustion, and, to a lesser degree, from smoldering of solid fuels.
 7 b. Particles condense within a fresh combustion plume from pyrolytic BrC and yellow organic carbon. Oxidation of
 8 anthropogenic and biogenic VOCs produces non-light absorbing carbon particles, and may also produce BrC and yellow
 9 carbon PM.
 10 c. Emitted directly as a byproduct of combustion, or formed through the oxidation of SOx or NOx.
 11 d. In the exhaust gases of solid fuel fires, low volatility BrC and other organic compounds can condense on BC particles. In
 12 the ambient atmosphere, low volatility organics produced by oxidation of VOCs can also condense on BC.
 13 e. Forms when high particle concentrations lead to the coagulation of multiple particles
 14 f. Forms by condensation of water vapor onto acidic organic (carbon-based) and inorganic particles
 15 g. Forms when complex particles undergo the humidification and drying cycles characteristic of cloud formation and
 16 evaporation

17 **2.6 Global and Regional Climate Effects of Black Carbon**

18 BC affects climate through both direct and indirect mechanisms. The most extensively studied of
 19 these mechanisms is radiative forcing, which is directly linked to temperature change. Radiative forcing
 20 is a measure of how a pollutant affects the balance between incoming solar radiation and exiting

1 infrared radiation, generally calculated as a change relative to preindustrial conditions defined at 1750.
2 A pollutant that increases the amount of energy in the Earth's climate system is said to exert "positive
3 radiative forcing," which leads to warming. In contrast, a pollutant that exerts negative radiative forcing
4 reduces the amount of energy in the Earth's system and leads to cooling. The net radiative impact of a
5 pollutant since preindustrial times can be averaged over the Earth's surface and is expressed in Watts
6 per square meter ($W m^{-2}$). Global average radiative forcing is a useful index because it is related linearly
7 to the global mean temperature at the surface (Forster et al., 2007) and is additive across pollutants.
8 Radiative forcing also provides a consistent measure for comparing the effects of past and projected
9 future emissions. As a result, it has become a standard measure for organizations like the IPCC and the
10 U.S. Global Change Research Program (NRC, 2005).

11 **Radiative forcing:** The change in the energy balance between incoming solar radiation and exiting
12 infrared radiation, typically measured in watts per square meter ($W m^{-2}$), over a specific time period
13 (generally since preindustrial conditions in 1750). Positive radiative forcing tends to warm the surface of
14 the Earth, while negative forcing generally leads to cooling.

15 In addition to radiative forcing, BC is associated with other effects including surface dimming
16 and changes in precipitation patterns. While not directly linked to temperature change, these effects
17 also have important global and regional climate implications. Each of these effects is discussed in
18 greater detail later in this section.

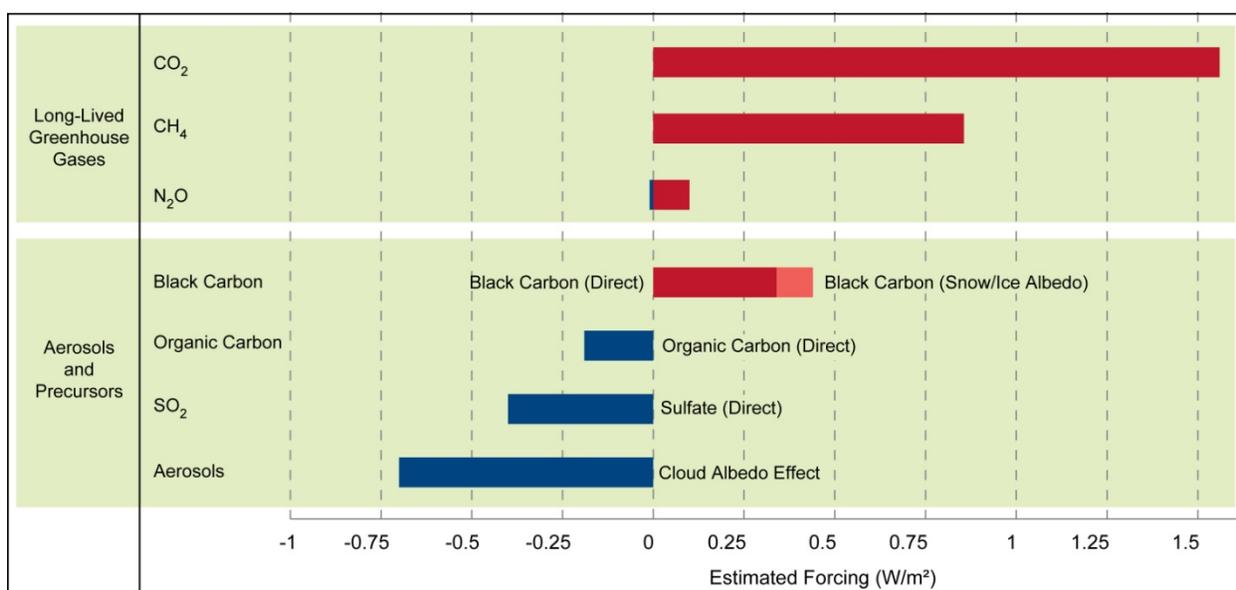
19 **2.6.1 Global and Regional Radiative Forcing Effects of BC: Overview**

20 This section provides an overview of the net impact of BC on radiative forcing based on the best
21 estimates in the current literature. Net radiative forcing for BC is actually the sum of several different
22 kinds of forcing, including **direct forcing** (direct absorption of solar or terrestrial radiation), **snow/ice**
23 **albedo forcing** (forcing that results from the darkening of snow and ice), and **indirect forcing** (a range of
24 forcing effects resulting from impacts on clouds, including changes in cloud lifetime, reflectivity, and
25 composition). These different kinds of forcing involve different mechanisms of action and can have
26 offsetting climate effects. For example, direct effects as associated with positive forcing, while most
27 (but not all) indirect effects are thought to result in negative radiative forcing. Since the effects can
28 counteract one another, it is important to consider all types of forcing when estimating the net radiative
29 impact of BC. However, due to lack of data, most studies of BC have focused on direct radiative forcing
30 only, or snow/ice albedo forcing. A more limited number of studies have estimated the indirect forcing
31 effects. These studies generally estimate indirect effects for all aerosol species together, rather than for
32 BC alone.

33 There is a range of quantitative estimates in the literature for global average radiative forcing
34 due to BC. Most studies indicate that due to the direct and snow/ice albedo effects, the net effect of BC
35 on climate is likely to be warming. However, because of the large remaining uncertainties regarding
36 interactions of BC with clouds, it is difficult to establish quantitative bounds for estimating global net
37 impacts of BC, or even to completely rule out the possibility of a net negative effect.

1 The direct and snow/ice albedo effects of BC are widely understood to lead to climate warming. The
 2 indirect effects of BC on climate via interaction with clouds are much more uncertain, but may partially
 3 offset the warming effects.

4 The most widely utilized estimates come from the IPCC’s Fourth Assessment Report, which was issued in
 5 2007. Based on a review of a large number of scientific studies available at the time, the IPCC estimated
 6 that BC has a direct radiative forcing of $+0.34 \text{ W m}^{-2}$, making BC third only to CO_2 and CH_4 . In addition,
 7 the IPCC estimated BC’s snow/ice albedo forcing to be $+0.1 \text{ W m}^{-2}$ (see Figure 2-7). Other aerosols were
 8 generally shown to have a cooling influence on climate: the IPCC estimates of negative direct radiative
 9 forcing due to OC and sulfates are also shown in Figure 2-7. Indirect effects are also estimated to result
 10 in net negative forcing due to increased reflectivity of clouds (“cloud albedo effect”).

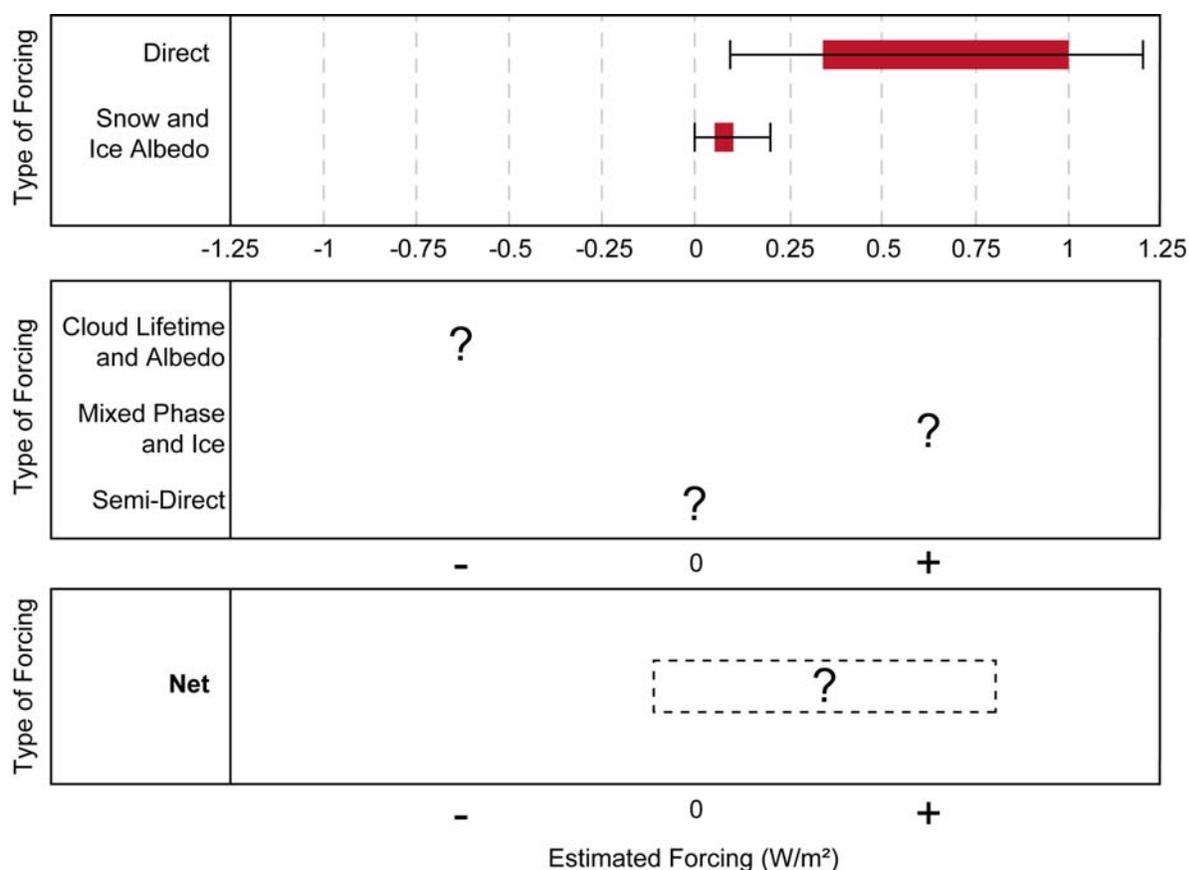


11
 12 **Figure 2-7. Components of Global Average Radiative Forcing for Emissions of Principal Gases, Aerosols,**
 13 **and Aerosol Precursors.** (Adapted from Figure 2.21 of Forster, et al., 2007.) Values represent global
 14 average radiative forcing in 2005 due to emissions and changes since 1750. Total radiative forcing for CH_4
 15 includes the effects of historical CH_4 emissions on levels of tropospheric O_3 and stratospheric H_2O , and the
 16 CO_2 oxidation product of CH_4 from fossil sources. Similarly, total radiative forcing for N_2O includes the
 17 effect of historical N_2O emissions on levels of stratospheric O_3 . The IPCC does not report an overall
 18 uncertainty for the net contribution to forcing of individual GHG emissions, but based on the uncertainties
 19 provided for the individual components of these contributions, the uncertainty in forcing from CO_2 and
 20 N_2O emissions is about 10% and from CH_4 emissions is about 20%. Uncertainty in direct forcing is $\pm 0.25 \text{ W}$
 21 m^{-2} for BC, and $\pm 0.20 \text{ W m}^{-2}$ for both OC and SO_2 . The range of forcing for the cloud albedo effect is -1.8
 22 to -0.3 W m^{-2} .

23 In addition to the estimates compiled by the IPCC (2007), many other studies have attempted to
 24 estimate the global average radiative forcing attributable to BC. An examination of the results of these
 25 studies as summarized in Figure 2-8 indicates that the direct effect and the snow/ice albedo effect of BC
 26 are positive, though the magnitude of these effects is uncertain. The figure shows the range of central
 27 estimates from the included studies (solid box) as well as the highest and lowest uncertainty estimates

1 from those studies (error bars) for both the direct effect and the snow and ice albedo effect. As
 2 discussed further below, a number of studies have estimated BC's direct radiative forcing to be higher
 3 than the IPCC estimate (Sato et al., 2003; Ramanathan and Carmichael, 2008).

4 There is a significant amount of uncertainty regarding the magnitude of the indirect effects of
 5 BC. (Indirect effects are discussed in detail in section 2.6.1.2.) The limited number of studies in the
 6 literature allow for statements on the direction (e.g. warming or cooling) of the forcing, but not its
 7 magnitude, as shown in Figure 2-8. The impact on cloud lifetime and albedo is likely cooling; however,
 8 Ramanathan (2010) asserts that the empirical evidence shows a positive forcing (warming) over land
 9 regions. The interactions with mixed-phase and ice clouds are likely to be warming. Semi-direct effects
 10 are so uncertain that it is not even possible to determine direction (though there are preliminary
 11 indications that semi-direct effects may be cooling on a global level).



12
 13 **Figure 2-8. Estimates of Radiative Forcing from Black Carbon Emissions Only.** The boxes indicate ranges
 14 of central estimates from the papers identified in this report. The error bars indicate the highest and
 15 lowest uncertainty estimates from those papers. Estimates are based on a synthesis of results from eleven
 16 studies that considered the direct forcing effects of BC emissions and six studies that considered
 17 reduction in snow and ice albedo from BC emissions. The range for the snow and ice albedo bar does not
 18 include the effects of the higher efficacy of the snow albedo effect on temperature change. The studies of
 19 indirect and semi-direct radiative forcing effects are not sufficiently comparable in scope and approach to
 20 combine the estimates. As a result, the likely direction of forcing is presented only. Similarly, this report

1 does not assign a range to the magnitude of the net effect beyond noting that it is very likely to be
2 positive (however, a negative effect cannot be excluded).

3 There are a number of factors that may contribute to the lack of consensus among modeled
4 estimates of net global average radiative forcing from BC. Koch et al. (2009) attributed the range of
5 estimates to differences in the aerosol microphysical calculations in the models (i.e., different estimates
6 of how much solar radiation each unit of BC absorbs). The authors also pointed out key differences in
7 models, such as the assumed values of various physical properties, and differences in the representation
8 of vertical transport and cloud effects. Differences in emissions inventories were not thought to be
9 significant.

10 Variability in the estimates may also arise due to differences in experimental design and how the
11 values are reported. Radiative forcing is commonly measured and reported as top-of-the-atmosphere
12 (TOA) radiative forcing which captures all variations in energy over the entire atmosphere. This
13 is appropriate for the well-mixed, long-lived GHGs, but perhaps not for BC, which exhibits high spatial
14 variability. For example, the vertical distribution of BC in the atmospheric column and interactions with
15 clouds leads to inputs of energy at different altitudes compared to the input of energy due to GHGs
16 (e.g., Ramanathan et al., 2001 and references therein). Climate effects are also sensitive to the location
17 of the BC emissions. For example, Arctic sea ice melting may be accelerated by BC emissions from
18 northern latitudes, as discussed later in this chapter. Finally, radiative forcing metrics that focus on
19 specific species do not generally capture co-pollutant interactions, which are very important for BC.

20 Studies focusing on global average radiative forcing may overlook key regional dynamics
21 associated with BC as a spatially heterogeneous pollutant. Many studies have found that BC's regional
22 climate impacts are more pronounced than the contributions of BC to global average temperature
23 change. In addition, certain regions of the world are more sensitive to or more likely to be affected by
24 BC forcing, either due to transport and deposition (i.e., the Arctic) or high levels of aerosol pollution in
25 the region (i.e., Asia). Global average radiative forcings for BC hide much of the regional variability in
26 the concentrations and impacts.

27 The following sections provide more detailed information regarding radiative forcing estimates
28 for BC at both the global and regional level. Direct forcing, snow/ice albedo forcing, and indirect forcing
29 effects are all discussed separately, with more detail regarding the findings of recent studies and fuller
30 explanation of the differences among estimates and key remaining uncertainties. In evaluating these
31 estimates, it is necessary to differentiate with respect to the baseline time period used to define the
32 radiative forcing estimates, the types of BC emissions included, and whether BC is estimated individually
33 or as part of an aerosol mixture. The radiative forcing estimates (and other climate effects) are often
34 expressed as a comparison to a given historical level rather than with respect to present day or in terms
35 of the anthropogenic influence compared to total forcing. However, these assumptions are not always
36 stated clearly. Also, many studies exclude open biomass burning. The inclusion or exclusion of BC from
37 wildfires and other sources of open biomass burning will affect the estimates of net BC effects. In
38 addition, some studies evaluate the climate effects of BC as it co-occurs with other aerosol chemical
39 species, such as OC, sulfates and nitrates, while others do not. When possible, it is indicated whether

1 the radiative forcing estimates include co-occurring OC and other species and how these other
2 pollutants influence estimates of BC's global and regional climate impacts.

3 **2.6.1.1 Direct Forcing**

4 The net *direct effect* of BC is to absorb solar radiation. As mentioned above and as shown in
5 Figure 2-9, the IPCC (2007) estimated the global average radiative forcing of BC from all sources at +0.34
6 (± 0.25) $W m^{-2}$. A subset of this forcing due to BC from fossil fuel combustion (mainly coal, petroleum
7 and gas fuels) was estimated to be +0.2 (± 0.15) $W m^{-2}$. Most studies published since the IPCC report
8 have reported higher direct forcing values. Additional work is underway to try to develop a new
9 central estimate for these direct impacts (IGAC/SPARC Bond, et al. study).

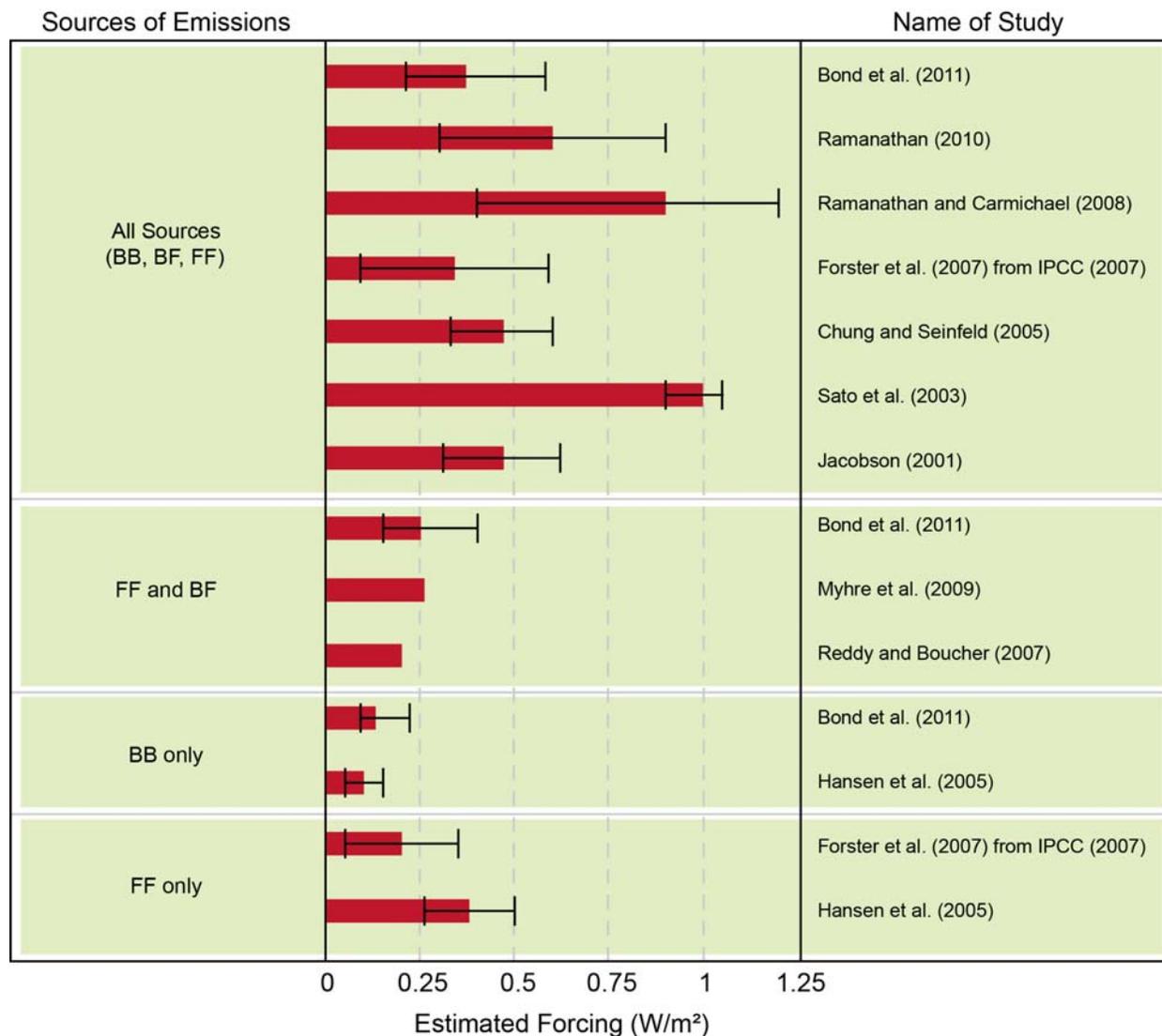
10 The assumptions about mixing state (e.g. internal/external) are critical to the results. As noted
11 in Section 2.5, studies that have incorporated internal mixing into the calculations of direct radiative
12 forcing for BC yield higher forcing than those that do not, and these models are considered to be more
13 realistic. Simulations by Jacobson (2001) found that accounting for internal mixing (core-shelled) of BC
14 in aerosols increases the absorption and warming by BC by a factor of two. Koch et al. (2009) accounted
15 for this underestimation of absorption by BC in older models by doubling the ensemble average from a
16 seventeen model intercomparison project (Schultz et al., 2006), resulting in a global average BC direct
17 radiative forcing of roughly +0.5 $W m^{-2}$. Bond et al. (2011) combined forcing results from 12 models to
18 use the best estimates for mixing and transport in those models. Based on this analysis, and using the
19 same emissions estimates used by the models assessed in the IPCC reports, Bond et al. found a total
20 forcing of +0.40 $W m^{-2}$, or 18% higher than the IPCC estimate, which they attributed to the fact that the
21 IPCC estimate includes some models that do not include enhanced absorption due to internal mixing.
22 Note that Bond et al. differentiate "anthropogenic emissions" (difference from 1750, including open
23 burning) and total BC emissions, calculating the total forcing from the latter to be +0.47 $W m^{-2}$.

24 In some cases, mainly in work based on observational constraints from the AERONET (Aerosol
25 Robotic Network) ground-based sunphotometer network, much higher values have been reported. Sato
26 et al. (2003) inferred a forcing of 1 $W m^{-2}$ based on these observational constraints. Chung et al. (2005)
27 and Ramanathan and Carmichael (2008) combined the AERONET results with satellite data and report
28 an estimated global average radiative forcing for BC of +0.9 $W m^{-2}$, with a range of +0.4 to +1.2 $W m^{-2}$.
29 While most recent studies find global forcing higher than the IPCC, a discrepancy remains between the
30 very high observationally constrained results and models (even those that include internal mixing and
31 therefore produce higher values). Bond et al. (2011) hypothesized that the higher forcing in the
32 observationally constrained results could result from higher emissions than in the model work. The
33 exact cause for these differences, however, has not been isolated.

34

35

36



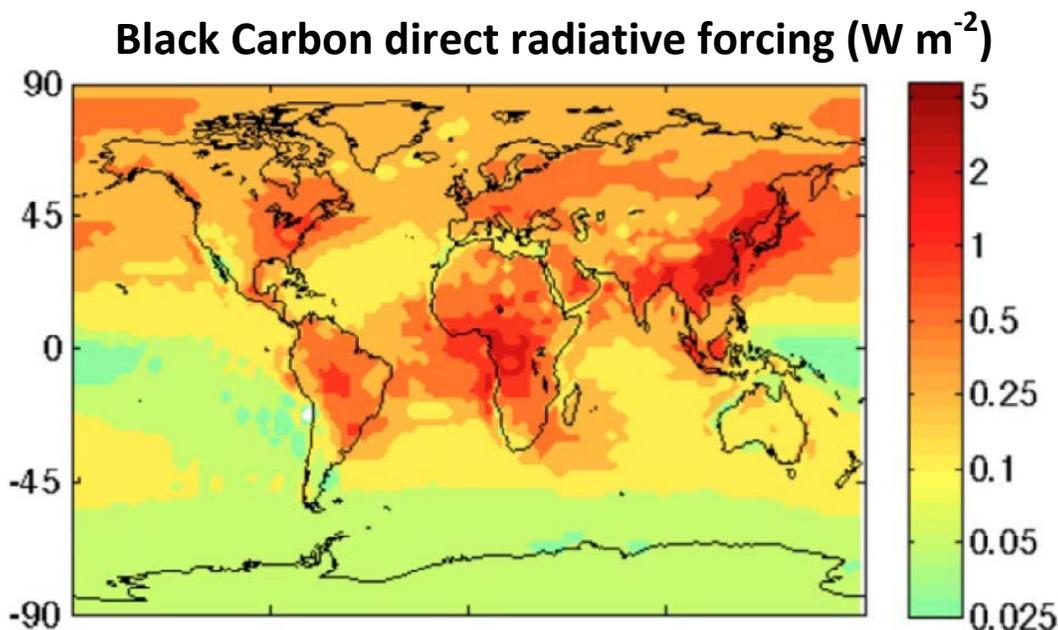
1

2 **Figure 2-9. Estimates of Direct Radiative Forcing from Black Carbon Emissions Only.** These values
 3 represent the range of estimates in the peer reviewed literature; however, they are not all directly
 4 comparable. Some are based on different estimates of BC emissions, include different sectors and present
 5 the forcing with respect to different baseline time periods (e.g. 1750, post-industrial, present day). **Note:**
 6 **BB = open biomass burning; BF = biofuels; and FF = fossil fuels.**

7

8 Compared to global radiative forcing, fewer studies have reported regional direct radiative
 9 forcing by BC. Studies such as Bond et al. (2011) show the geographic distribution of direct forcing from
 10 all sources of BC emissions (Figure 2-10). They found the largest forcing over South and East Asia and
 11 parts of Africa. Other work such as Chung and Seinfeld (2005) showed similar patterns with higher
 12 forcing in Central and South America (the Amazon basin) and sub-Saharan Africa due to the inclusion of
 13 biomass burning emissions. Chung and Seinfeld (2005) report a range of +0.52 to +0.93 W m⁻² for
 14 externally and internally mixed BC respectively, averaged over the Northern Hemisphere. Their earlier

1 work also suggests a strong seasonal cycle which peaks in May at $+1.4 \text{ W m}^{-2}$ (Chung and Seinfeld, 2002).
 2 For the Southern Hemisphere, Chung and Seinfeld (2005) estimate a range of $+0.15$ to $+0.23 \text{ W m}^{-2}$.
 3 Reddy and Boucher (2007) calculated the influence of regional BC emissions on the global average
 4 radiative forcing. The largest contribution to global TOA BC radiative forcing came from East Asia ($+0.08$
 5 W m^{-2}). The global average forcing due to North American BC emissions in this study was $+0.02 \text{ W m}^{-2}$.
 6 Myhre et al. (2009), who considered only fossil fuel and biofuel BC, also found largest forcing over South
 7 and East Asia.



8
 9 **Figure 2-10. Direct Radiative Forcing (W M^{-2}) of Black Carbon from All Sources**, simulated with the
 10 Community Atmosphere Model (Bond et al., 2011).

11

12 2.6.1.2 Indirect (Cloud) Forcing

13 The net *indirect effect* of particles on climate via impacts on clouds is highly uncertain (IPCC,
 14 2007). There are several different kinds of cloud effects that are important for radiative purposes, as
 15 summarized in Table 2-3. These cloud effects contribute to changes in the radiative balance of the
 16 atmosphere, and also influence climatic factors such as precipitation and dimming (section 2.6.3).

17 Since cloud droplets are formed when water vapor condenses onto a particle, many types of
 18 particles can affect the formation and microphysics of clouds. Emissions of aerosols into the
 19 atmosphere increases the number of particles on which cloud droplets can form, resulting in more and
 20 smaller cloud droplets. These additional cloud droplets produce brighter, more reflective clouds
 21 (Twomey, 1977). This generally results in surface cooling by preventing sunlight from reaching below the
 22 cloud to the Earth's surface (see also Section 2.6.3.1 on surface dimming). This increase in reflectivity of
 23 the clouds has been termed the "the first indirect effect" or the "**cloud albedo effect**". This effect leads

1 to cooling. In addition, the smaller cloud droplets are less likely to aggregate sufficiently to form rain
 2 drops, which changes precipitation patterns and increases cloud lifetime (Albrecht, 1989) (see also
 3 Section 2.6.3.2 on precipitation impacts). This has been called the “second indirect effect” or the “**cloud**
 4 **lifetime effect**,” and also leads to cooling. The magnitude and sign of the radiative effects depend on
 5 whether the clouds are composed of liquid droplets, ice particles, or a mix of ice and liquid droplets, and
 6 on the composition of the aerosol particles. In certain kinds of “mixed-phase clouds” (clouds with both
 7 ice and water), smaller droplets cause a delay in the freezing of the droplets, changing the
 8 characteristics of the cloud; however, the IPCC was not able to determine whether this “**thermodynamic**
 9 **effect**” would result in overall warming or cooling (Denman et al. 2007).

Effect	Cloud Types Affected	Process	Net effect	Potential Magnitude	Scientific Understanding
Cloud albedo effect	All clouds	Smaller cloud particles reflect more solar radiation	Cooling	Medium	Low
Cloud lifetime effect	All clouds	Smaller cloud particles decrease precipitation so increase lifetime	Cooling	Medium	Very low
Semi-direct effect	All clouds	Absorption of solar radiation by absorbing aerosols changes atmospheric stability and cloud formation	Cooling or warming	Small	Very low
Glaciation indirect effect	Mixed-phase clouds	An increase in ice nuclei due to some aerosols increases precipitation	Warming	Medium	Very low
Thermodynamic effect	Mixed-phase clouds	Smaller cloud droplets delay freezing causing super-cooled clouds to extend to colder temperatures	Cooling or warming	Medium	Very low

10 **Table 2-3: Overview of the Different Aerosol Indirect Effects.** This summary applies to all aerosols and is
 11 not BC-specific. Net effect refers to top-of-the-atmosphere radiative forcing. Scientific understanding is
 12 based on IPCC terminology. Adapted from IPCC (Denman et al. 2007), Table 7.10a.

13 The “**semi-direct effect**” and the “**glaciation indirect effect**” are specific to BC and other
 14 absorbing aerosols. The semi-direct effect refers to the heating of the troposphere by absorbing
 15 aerosols, affecting the relative humidity and stability of the troposphere, which in turn affects cloud
 16 formation and lifetime (IPCC, 2007; Ackerman et al., 2000). Older literature refers to the semi-direct
 17 effect as cloud burn-off (i.e., a decrease in cloud formation) from BC within the cloud layer. The
 18 definition was extended to include all effects on cloud formation and lifetime as other studies have
 19 found that humidity and stability effects from BC above and below clouds can cause both increases and
 20 decreases in clouds (Koch and Del Genio, 2010). The IPCC did not assign a sign to the net forcing of the
 21 semi-direct effect (Denman et al. 2007). More recently, Koch and Del Genio (2010) find in their review
 22 of the literature that most model studies generally indicate a global net negative effect (i.e., the effect of
 23 atmospheric heating by absorbing aerosols on cloud formation and lifetime causes net cooling). This
 24 was observed despite regional variation in the cloud response to absorbing aerosols (such as BC), and

1 resulting regional differences in warming and cooling from the semi-direct effect. By contrast, the
2 glaciation effect is a warming effect which occurs in some mixed-phase clouds only. This indirect effect
3 is caused by BC aerosols (and some other particles such as mineral dust) serving as ice nuclei in a super-
4 cooled liquid water cloud, thereby enabling precipitation rather than delaying it (Denman et al. 2007,
5 Lohmann and Hoose 2009).

6 Most estimates of the forcing from aerosol indirect effects are based on all aerosol species (e.g.
7 total PM) and are not estimated for individual species. The net indirect effect of all aerosols is estimated
8 as a negative value. The IPCC (Forster et al., 2007) estimated the total aerosol indirect effect to have a
9 radiative forcing of -0.7 W m^{-2} , with a 5 to 95% confidence range of -0.3 to -1.8 W m^{-2} and a low level of
10 scientific understanding (the IPCC definition of “level of understanding” is a qualitative measure based
11 on a combination of the quantity of evidence available and the degree of consensus in the literature).

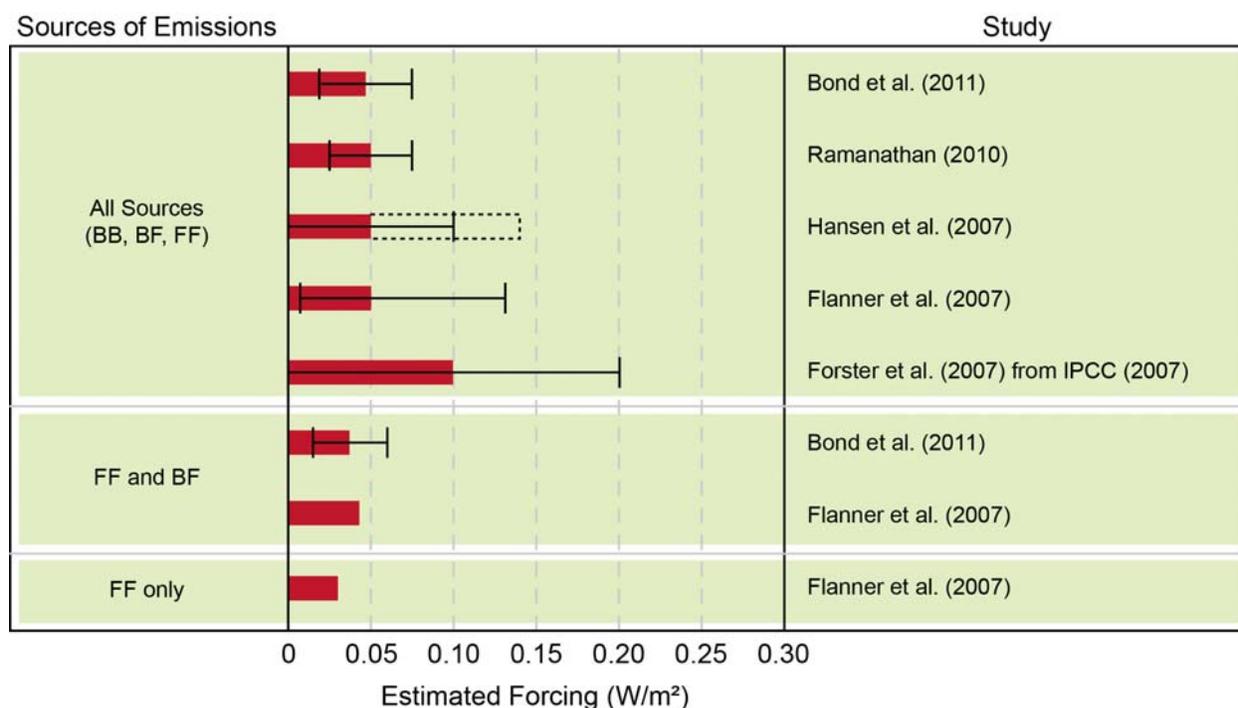
12 It is unclear to what extent BC contributes to the overall aerosol indirect effect. As a result, this
13 report does not assign any central estimate or even a range of possible values for the role of BC in the
14 overall indirect aerosol effect. BC’s role in the first and second indirect effects (cloud albedo and cloud
15 lifetime effects) is likely to be cooling, but possibly to a lesser extent than for other aerosols. Freshly
16 emitted, externally mixed BC particles are hydrophobic and would be less active cloud condensation
17 nuclei (CCN); however, aging may increase their ability to serve as CCN (Dusek et al. 2006). Recent work
18 (e.g., Bauer et al., 2010) using models with a more explicit representation of aerosol mixing than older
19 models suggests that the role of BC in the indirect effect may be greater than previously thought.
20 Similarly, BC may also participate in the thermodynamic indirect effect for mixed-phase clouds, but
21 whether this effect is net warming or cooling is still uncertain. BC has a primary role in the semi-direct
22 effect, and while this effect may produce warming or cooling depending on conditions, recent work
23 suggests that cooling prevails on a global scale. Finally, BC particles may contribute to warming from the
24 glaciation indirect effect in mixed-phase clouds, but the magnitude of this effect is uncertain. A
25 comprehensive, quantitative estimate of the net effect of BC would require an assessment of the likely
26 bounds of these indirect effects. No studies that estimate indirect or semi-direct radiative effects for BC
27 at a regional level were identified for this report.

28 **2.6.1.3 Snow and Ice Albedo Forcing**

29 BC deposited on snow and ice leads to positive radiative forcing. It darkens the surface which
30 decreases the surface albedo, and it absorbs sunlight, heating the snow and ice (Warren and Wiscombe
31 1980). The snow and ice albedo effect is strongest in the spring because snow cover is at its greatest
32 extent, and spring is a season with increased exposure to sunlight (Flanner et al., 2009). Brown carbon
33 has also been found to contribute to snow and ice albedo forcing (Doherty et al. 2010). Chapter 5 also
34 addresses observations of BC in snow in more depth.

35 There are a number of estimates of the magnitude of radiative forcing due to the snow albedo
36 effect (see Figure 2-11). In a modeling study, Hansen and Nazarenko (2004) estimated the global
37 average radiative forcing of BC on snow and ice to be $+0.16 \text{ W m}^{-2}$ for what they considered to be the
38 most realistic of the four cases that were simulated in their study. In later work, Hansen et al. (2007)

1 lowered this estimate to $+0.05 \text{ W m}^{-2}$, with a probable range of 0 to $+0.1 \text{ W m}^{-2}$ (Hansen et al. 2007).
 2 Relying on these studies, the IPCC (Forster et al., 2007) adopted a best estimate for the global average
 3 radiative forcing of deposited BC on snow and ice of $+0.10 (\pm 0.10) \text{ W m}^{-2}$, though acknowledged a low
 4 level of scientific understanding regarding this effect. In more recent work, Flanner et al. (2007)
 5 estimated the average forcing of BC on snow and ice (from fossil fuels (FF) and biofuels (BF)) at $+0.043$
 6 W m^{-2} , of which $+0.033 \text{ W m}^{-2}$ was attributed to BC from fossil fuels. When biomass burning was
 7 included in the calculation, the forcing of BC on snow and ice was estimated to be approximately $+0.05$
 8 W m^{-2} . Bond et al. (2011) estimated a global forcing of $+0.047 \text{ W m}^{-2}$, of which 20% was calculated to
 9 occur in the Arctic (defined as north of 60 degrees), and suggest that more mechanistic studies in
 10 general yield estimates lower than the central IPCC estimate of $+0.1 \text{ W m}^{-2}$.



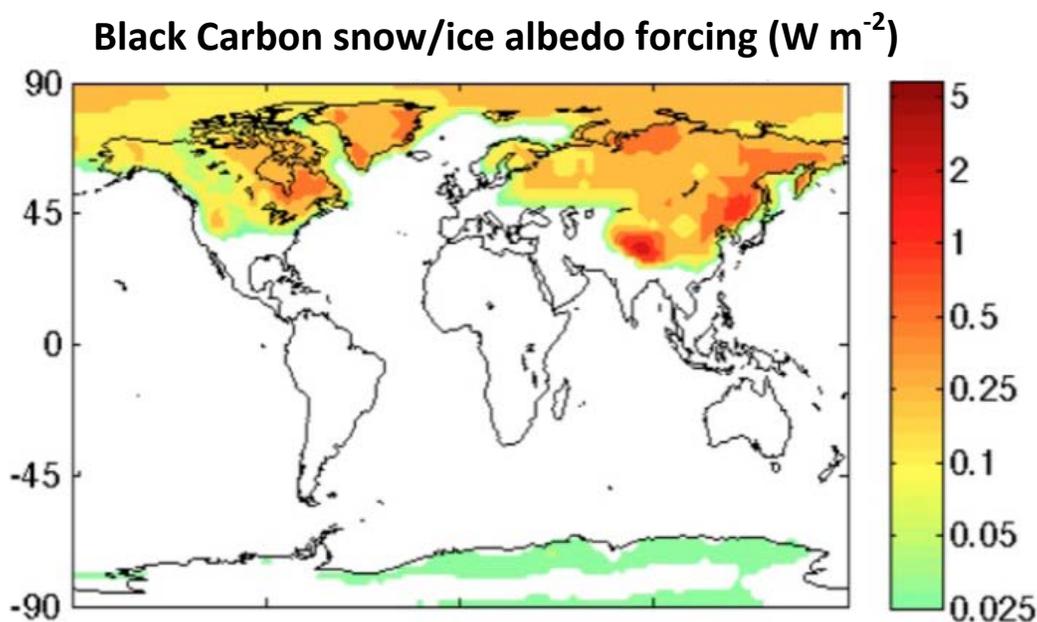
11
 12 **Figure 2-11. Estimates of Snow and Ice Albedo Radiative Forcing Effects from BC Emissions Only.** Note:
 13 BB = open biomass burning; BF = biofuels; and FF = fossil fuels. Dashed lines indicate the estimated range
 14 of snow and ice albedo radiative forcing when forcing efficacy is considered. Hansen et al. (2005) estimate
 15 an efficacy of 170% for the snow and ice albedo radiative forcing effect of BC emissions.

16 Hansen et al. (2007) also investigated the “effectiveness” (or “efficacy”) of the snow albedo
 17 forcing. This is a relative measure of positive feedback effects that occur with BC, compared to the
 18 feedbacks that occur with warming due to CO₂ forcing. They calculated that the radiative forcing from
 19 decreases to surface albedo is 2.7 times more effective at warming than radiative forcing from CO₂. This
 20 is a result of the energy absorption from the BC being directly applied to melting snow rather than
 21 spread throughout the height of the atmosphere. BC particles left behind in melting surface snow can
 22 concentrate and further reduce the surface albedo (see section 5.6). Melting snow can expose a dark
 23 surface, leading to a positive feedback. Flanner et al. (2007) found a larger efficacy of 3.2, with an
 24 uncertainty range of 2.1 to 4.5. Flanner et al. (2011) also found that observed Northern Hemisphere

1 snow retreat between 1979 and 2008 (from all causes) would be consistent with a total albedo feedback
2 on the order of $+0.45 \text{ W m}^{-2}$. This suggests that feedback is a larger process than represented in most
3 climate models.

4 For snow and ice, however, there is evidence that all atmospheric PM, including all mixtures of
5 BC and OC, increases the net solar heating of the atmosphere-snow column (Flanner et al., 2009). This
6 means that mixtures of BC and OC that are transported over snow-covered areas may have a net
7 warming influence regardless of the ratio of the two compounds (though this study did not include cloud
8 effects). This is in contrast to direct radiative forcing estimates which are strongly influenced by the
9 ratio of BC to other cooling PM components such as OC. Flanner et al. (2009) also found that fossil-fuel
10 and biofuel BC and OM emissions contributed almost as much to springtime snow loss in Eurasia as did
11 anthropogenic CO_2 . The size and composition of the deposited particles affects how long they remain
12 on or near the surface where they are able to reduce albedo.

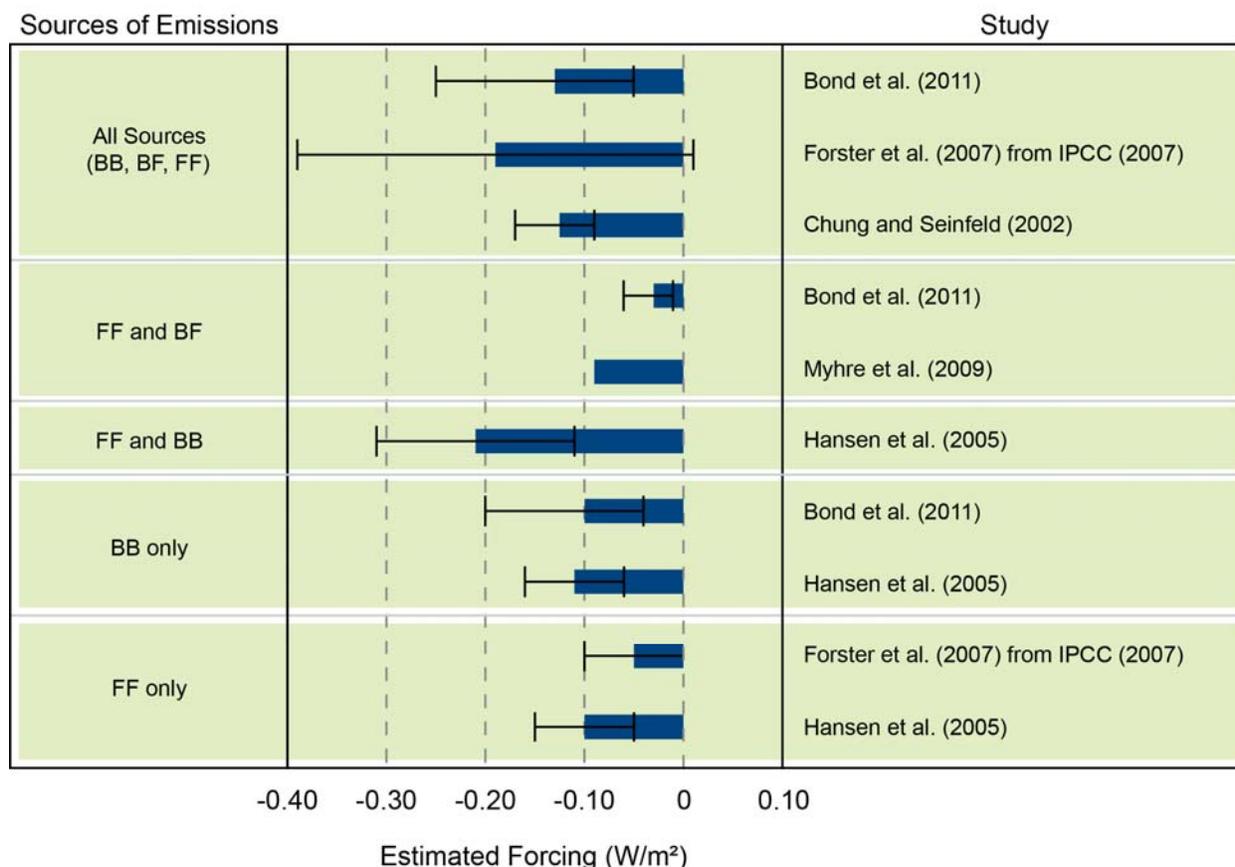
13 Snow and ice albedo forcing is confined to areas with snow and ice cover (approximately 7.5%-
14 15% of Earth's surface; see Chapter 5.6). Thus, global average forcing estimates do not convey the
15 significant spatial and temporal variability in the radiative forcing of BC on snow and ice. Radiative
16 forcing from changes in snow and ice albedo from BC are estimated to be much larger than the global
17 averages for much of Northern and Eastern Europe, Russia, and China. These effects are especially
18 pronounced in the Arctic and the Himalayas. Flanner et al. (2007) calculated an average forcing of BC on
19 snow and ice of $+1.5 \text{ W m}^{-2}$ in the Tibetan plateau, with instantaneous forcings of up to $+20 \text{ W m}^{-2}$ in the
20 spring. These high values are due to the large amount of mountain snow and ice cover as well as the
21 proximity to high emissions of BC from parts of China and the Indian subcontinent. Large radiative
22 forcing values have also been estimated over the Arctic. Hansen and Nazarenko (2004) calculated an
23 average forcing due to BC on snow and ice of $+1 \text{ W m}^{-2}$ in the Arctic and $+0.3 \text{ W m}^{-2}$ over the Northern
24 Hemisphere as a whole: however, these are based on global numbers that were reduced by a factor of
25 three in later papers (Hansen et al., 2007). The full spatial distribution of forcing by BC on snow and ice
26 as simulated by Bond et al. (2011) is shown in Figure 2-12. The effects of BC on the Arctic and the
27 Himalayas are described in more detail in sections 2.6.4 and 2.6.5, below.



1
 2 **Figure 2-12. Snow and Ice Albedo Forcing by Black Carbon**, simulated with the Community Atmosphere
 3 Model (Bond et al. 2011).

4
 5 **2.6.1.4 The Radiative Forcing Effects of Organic Carbon and other Co-Pollutants**

6 Although BC is mixed with other pollutants, both at the point of emission and in the
 7 atmosphere, most studies examine the impact of different types of aerosols in isolation. Only a limited
 8 number of studies consider the impacts of co-pollutants, and most of these studies have focused on OC
 9 rather than all aerosol species. Figure 2-13 shows estimates of direct radiative forcing for OC from a
 10 number of studies. As indicated in Figure 2-13, OC emissions from all sources are estimated to have net
 11 cooling impacts. For example, the IPCC (2007) estimated the negative direct radiative forcing of OC
 12 aerosols from all sources at $-0.19 (\pm 0.20) W m^{-2}$ and from fossil fuel alone at $-0.05 (\pm 0.05) W m^{-2}$.



1

2

Figure 2-13. Estimates of Direct Radiative Forcing from OC Emissions Only. Note: BB = open biomass burning; BF = biofuels; and FF = fossil fuels.

3

4

5

When OC and BC emissions are combined, the estimates of global average direct radiative forcing are generally positive. Figure 2-14 shows estimates for BC and OC combined from different sources. Here, the total direct radiative forcing from BC and OC emissions from all sources was estimated by IPCC (2007) at approximately $+0.15 \text{ W m}^{-2}$ (global average), and even biomass burning aerosols were estimated to have a positive net forcing of $+0.03 (\pm 0.12) \text{ W m}^{-2}$. Another study that calculated a net forcing from BC and OC from all sources reported a net global average forcing of about $+0.27 \text{ W m}^{-2}$ (Bond et al., 2011).

6

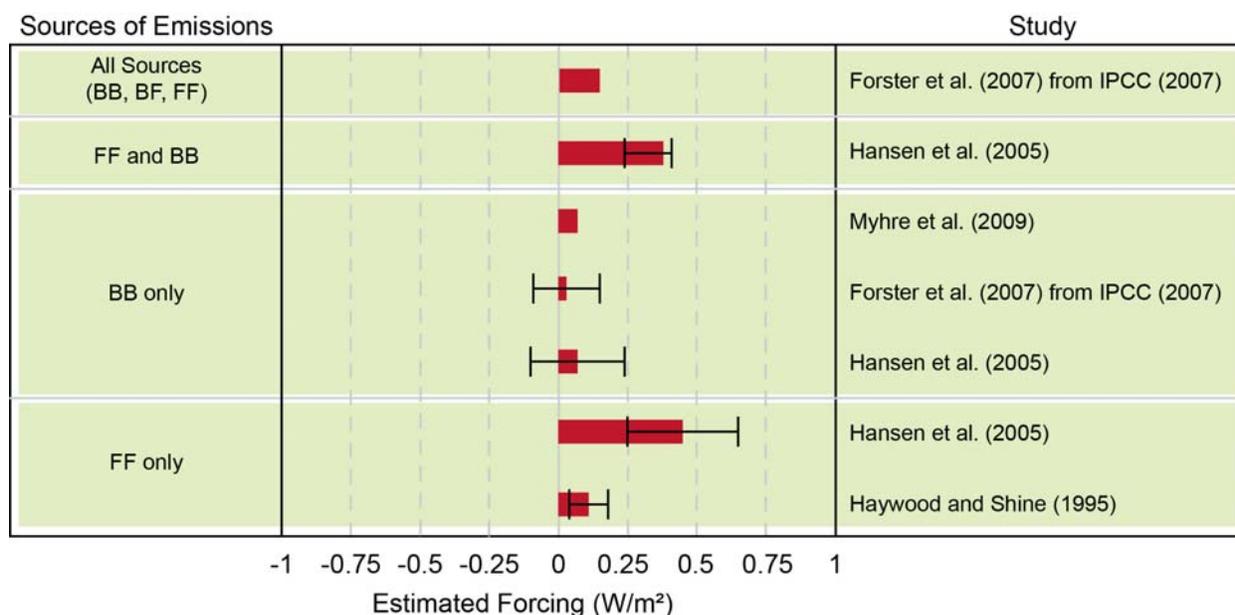
7

8

9

10

11

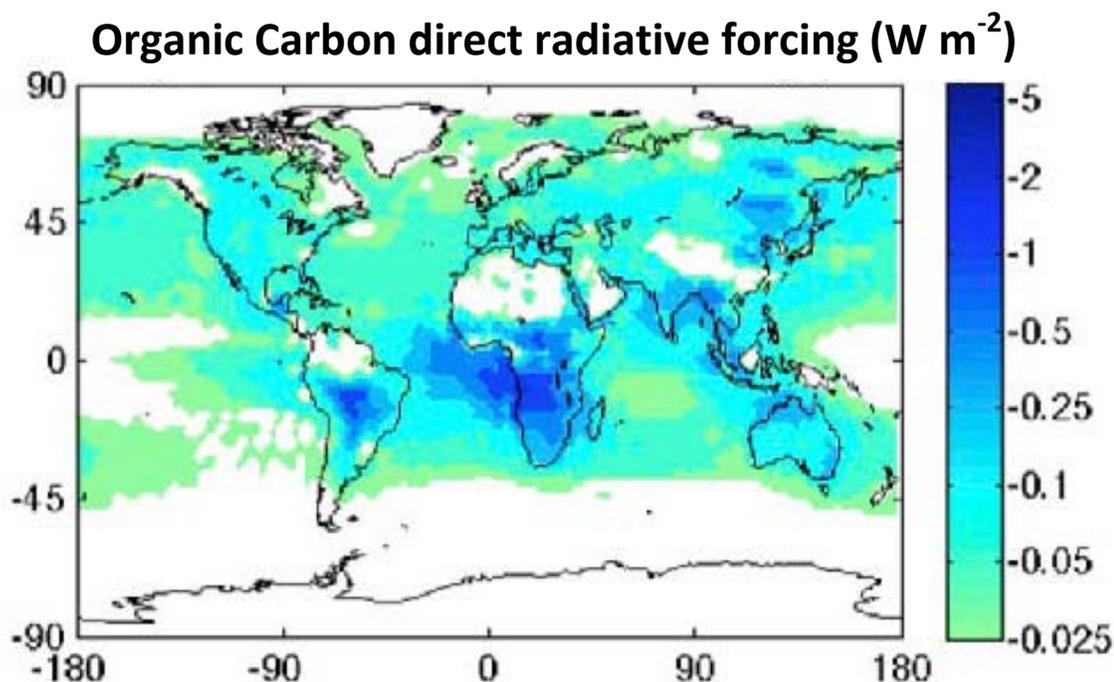


1
2 **Figure 2-14. Estimates of Direct Radiative Forcing from BC and OC Emissions.** Note: BB = open biomass
3 burning; BF = biofuels; and FF = fossil fuels. Forster et al. (2007) – from IPCC (2007) – estimate the
4 uncertainty surrounding estimates of direct radiative forcing from BC and OC independently. For this
5 reason, the uncertainty surrounding the combined estimated direct radiative forcing from BC and OC
6 emissions from all sources according to Forster et al. (2007) is omitted.

7 Several additional factors must be taken into consideration in interpreting these estimates,
8 however. First, it is important to note that like BC, OC exhibits high spatial variability in direct forcing
9 effects (see Figure 2-15). The regions of highest direct forcing by OC may not coincide with regions of
10 highest forcing by BC (see Figures 2-10 and 2-12 for comparison.) In addition, most studies evaluating
11 the net effects of BC and OC do not consider indirect effects, and inclusion of these effects will change
12 the net forcing estimates. One study, Chen et al. (2010), found that for one scenario reducing BC and
13 OC in a 2 to 3 ratio, the aerosol indirect effects were larger than (and opposite in sign to) the direct
14 effects. In addition, studies looking at forcing effects due to OC generally consider primary OC emissions
15 only. Secondary organic aerosols (SOA), however, can also make a substantial contribution to the
16 organic aerosols. SOA arises from the oxidation of gaseous volatile organic compounds (VOC). More
17 recently, Robinson et al. (2007) proposed a more dynamic evolution of aerosol OC in the atmosphere.
18 Based on measurements and models, they suggested that low volatility organic compounds, which are
19 emitted as PM, evaporate, oxidize and condense over time. The semi-volatile nature of the primary
20 emission of OC may have additional implications for our understanding of OC and BC/OC ratios on
21 climate, although this remains poorly understood (Jimenez et al, 2009).

22
23
24

1



2 **Figure 2-15. Direct Forcing by Organic Carbon from All Sources**, simulated with the Community
3 Atmosphere Model (Bond et al. 2011).

4 The inclusion of other species, mainly nitrate and sulfate aerosols, also tends to reduce the
5 estimate of net forcing. In particular, the presence or absence of sulfate and nitrate in calculations of
6 indirect effects, which together comprise a large fraction of aerosol mass, can dominate radiative
7 forcing calculations. Inclusion of both direct and indirect effects of aerosol species in the review by
8 Ramanathan and Carmichael (2008) led to an estimate of the total aerosol effect including direct and
9 indirect effects of -1.4 W m^{-2} , in contrast to a calculated BC direct forcing of $+0.9 \text{ W m}^{-2}$. However,
10 because much of the nitrate and sulfate precursor emissions come from sectors that are not rich in BC,
11 the net global effect of aerosols can be less important than the estimates of the net effects of aerosols
12 from a specific sector or measure (discussed further in 2.6.1.5). These aerosols also play a role in the
13 mixing state and therefore the direct radiative forcing effect of BC, as discussed in 2.5 and 2.6.1.1.
14 Therefore, ambient concentrations of these other aerosols can be important in determining the
15 influence of BC reductions. Using surface and aircraft measurements, Ramana et al. (2010) found that
16 the ratio of BC to sulfate was important in determining the net warming or cooling impact of pollution
17 plumes in China.

18 **2.6.1.5 Sector-Based Contributions to Radiative Forcing**

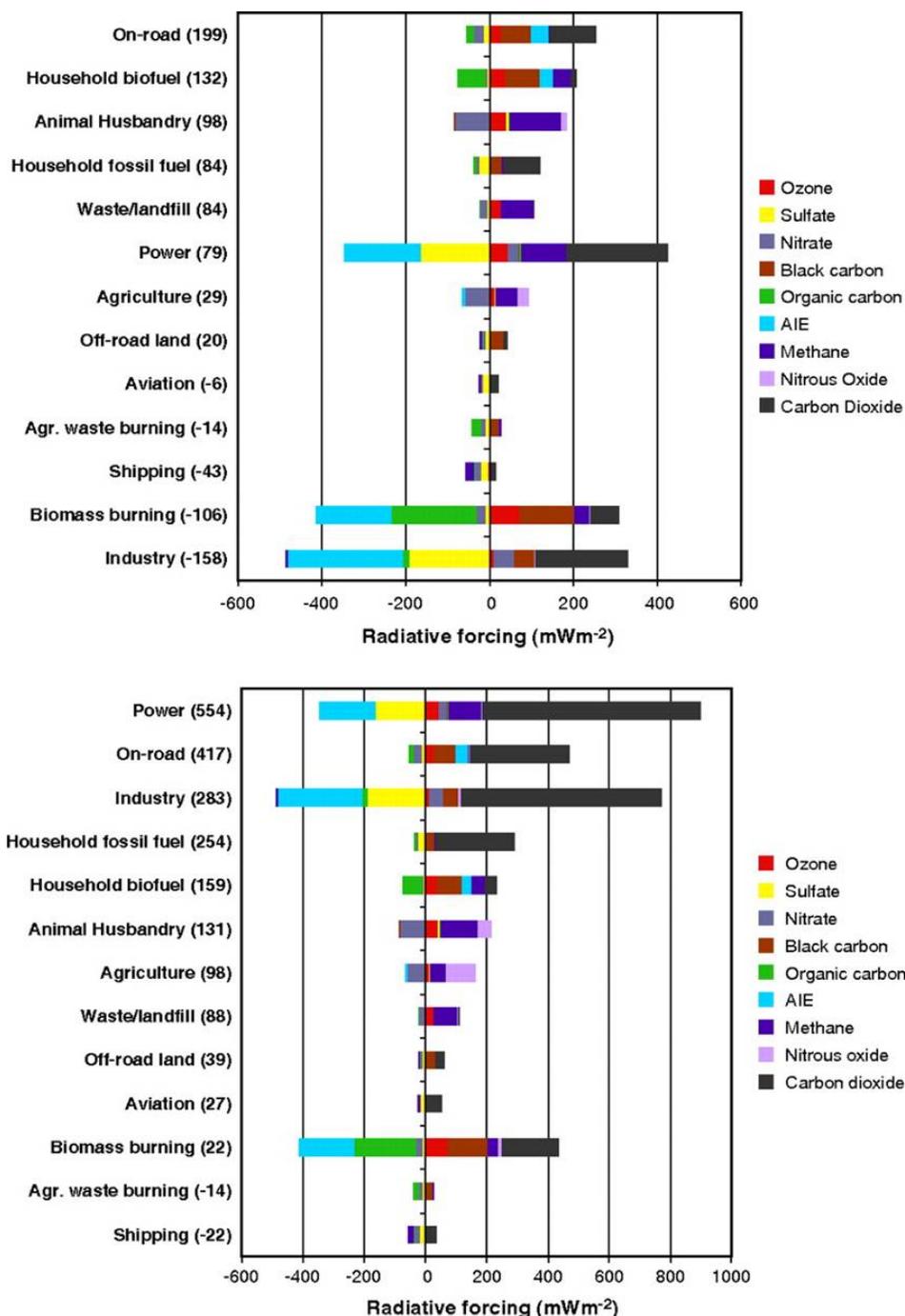
19 Some studies have attempted to quantify the radiative forcing effect of emissions mixtures
20 containing BC and other co-pollutants by estimating the radiative forcing of defined emissions sectors.
21 Comparisons between studies, however, is hindered by in part because each study uses different

1 estimates of the sector level contributions, the relative fraction of warming and cooling aerosols, and
2 their microphysical properties.

3 Unger et al. (2010) examined the impacts of sector-specific emissions on the short- and long-
4 term radiative forcing from a range of pollutants. Figure 2-16 shows that the mixture of emissions from
5 some of the largest BC emissions sources contributes considerably to total radiative forcing. On-road
6 transportation emissions are the largest contributor to radiative forcing in the short term (by 2020), due
7 to a combination of GHG and BC emissions. On-road transportation is also seen to be the second largest
8 contributor in the long term (by 2100), but this is largely the result of the significant GHG emissions from
9 this sector. Residential biofuel combustion is the second largest contributor in the short term due to the
10 contribution from BC and methane. Since these sources have lower net GHG emissions, they contribute
11 less to total global radiative forcing in the long term. However, these calculations have substantial
12 uncertainties owing to the details of aerosol physics and chemistry, the interactions of aerosols and
13 clouds, and the regional nature of the radiative forcing as discussed earlier in this chapter.

14 There is significant disagreement regarding the net impact of aerosol emissions from open
15 biomass burning on radiative forcing. As noted in the previous section, the IPCC estimated the net
16 direct radiative forcing impact from open biomass burning aerosols to be small, but positive at
17 $+0.03 \pm 0.12 \text{ W m}^{-2}$ (Forster et al., 2007). However, because of uncertainties regarding the extent and
18 composition of emissions from this source category, and the indirect radiative forcing effects of biomass
19 burning aerosols, it is not clear if this sector has an overall global warming or cooling effect. Kopp and
20 Mauzerall (2010) developed probability distributions from multiple studies to examine the likelihood of
21 warming from individual sectors. Based on existing evidence, they concluded that open burning in
22 forests and savannas is unlikely to contribute to warming, while the effect of open burning of crop
23 residues remains uncertain. The results of current analyses are sufficiently different that there is no
24 consensus on the likelihood of warming. Stohl et al. (2007) concluded that biomass burning has
25 “significant impact on air quality over vast regions and on radiative properties of the atmosphere” and in
26 particular “has been underestimated as a source of aerosol and air pollution for the Arctic, relative to
27 emissions from fossil fuel combustion.” As discussed further in Chapter 5.6, surface snow records
28 indicate that biomass burning is currently a major source of BC in Greenland and the North Pole (Hegg,
29 2010). Additional work is needed to improve scientific understanding of the radiative forcing impacts of
30 open biomass burning.

31



1
 2 **Figure 2-16. Global Radiative Forcing Due to Perpetual Constant Year 2000 Emissions, Grouped By**
 3 **Sector, at 2020 (Top) and 2100 (Bottom) showing the contribution from each species. The sum is shown**
 4 **on the title of each bar, with a positive radiative forcing means that removal of this emission source will**
 5 **result in cooling (Unger et al., 2010). AIE is the aerosol indirect effect.**

6 Among fossil fuels, diesel combustion for transportation is the largest contributor to global BC
 7 emissions. Several modeling experiments removing these emissions sources, such as Jacobson (2002,
 8 2005, 2010), Hansen et al. (2005), and Schultz et al. (2006), and observationally constrained studies such

1 as Ramanathan and Carmichael (2008), have found that carbonaceous aerosols from biofuel combustion
2 and fossil fuel combustion both contribute to warming. Kopp and Mauzerall (2010) concluded that
3 carbonaceous PM emissions from gasoline combustion are unlikely to contribute to warming,² while
4 diesel combustion and residential coal combustion are very likely to contribute to warming. These
5 authors also found mixed results with respect to the contribution of residential biofuel combustion for
6 the models included in their assessment.

7 A few studies highlight the substantial uncertainties regarding the contribution of biofuel
8 combustion and fossil fuel combustion to warming, given our limited understanding of how
9 carbonaceous aerosols affect cloud processes. In the modeling experiments by Chen et al. (2010),
10 reductions in fossil fuel carbonaceous aerosols (BC and OC) lead to decreases in cloud condensation
11 nuclei, leading to a decrease in cloud albedo, causing an increase in radiative forcing. The impact of
12 these cloud changes equal or exceed the direct radiative forcing impacts. This result contrasts with that
13 of Jacobson (2010) and Bauer et al. (2010) in which estimated warming from indirect effects did not
14 exceed the direct and other radiative forcing from fossil fuel emissions.

15 Fossil fuels burned for electricity generation contribute only a small fraction of carbonaceous
16 aerosol emissions, though this sector is a large source of long-lived, warming GHGs and short-lived
17 cooling sulfate aerosols (Shindell and Faluvegi, 2009). Thus, though their study found that the sector is
18 the largest single contributor to warming on the 100 year time scale, this is attributable to GHG
19 emissions rather than emissions of BC.

20 **2.6.2 Impact of BC Radiative Forcing on Temperature and Melting of Ice and Snow**

21 As mentioned in 2.6.1, global average radiative forcing is linearly related to the global mean
22 temperature at the surface (Forster et al., 2007). Radiative forcing from agents such as BC has similar
23 effects on global mean temperature as radiative forcing from CO₂ and other GHGs (Hegerl et al., 2007).
24 The relationship between radiative forcing and temperature has already been linked to a range of
25 climate impacts as identified in, for example, the 2009 USGCRP report, “Global Climate Change Impacts
26 in the United States.” This and other recent climate change assessments describe the risks and impacts
27 associated with climate change including degradation of air quality, temperature increases, changes in
28 extreme weather events, effects on food production and forestry, effects on water resources, sea level
29 rise, disruption to energy consumption and production, and potential harm to ecosystems and wildlife.
30 Though few studies explicitly link BC to all of these outcomes, to the extent that BC increases
31 temperature it may contribute to these impacts, including impairment of air quality and sea level rise
32 (via melting of ice, snow, and glaciers).

² For gasoline vehicles, it should be noted that the introduction of new engine technologies (e.g., some types of gasoline direct injection) in recent model years has increased BC/PM ratios in some new gasoline-powered motor vehicles (Smallwood et al. 2001), which may change the warming profile of emissions from these vehicles. See Chapter 7 for more discussion of this issue.

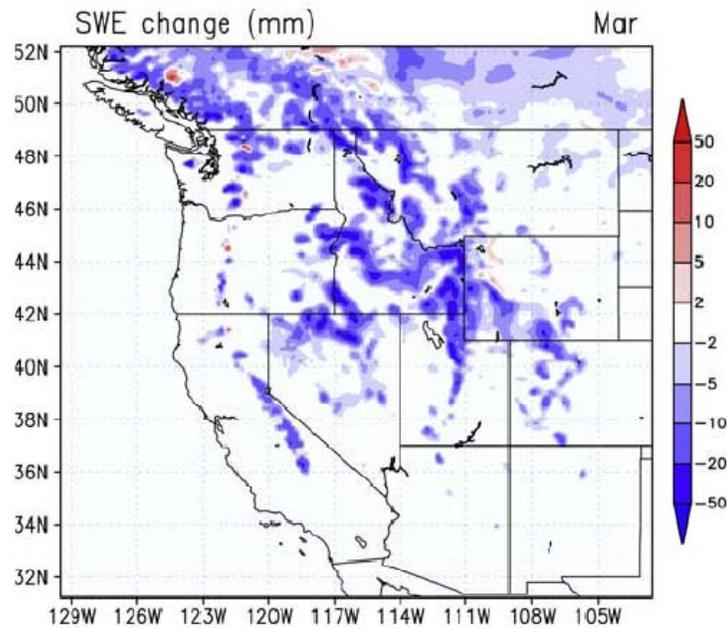
1 There have been some efforts to translate regional direct radiative forcing estimates into
2 regional changes in temperature. For example, Chung and Seinfeld (2005) estimate that externally
3 mixed BC leads to an average warming of 0.29°C in the Northern Hemisphere and 0.11°C in the Southern
4 Hemisphere. Internally mixed BC is estimated to result in an average warming of 0.54°C in the Northern
5 Hemisphere and 0.20°C in the Southern Hemisphere. Few studies have evaluated the North America-
6 specific temperature impacts associated with BC emissions. However, Qian et al. (2008) find that BC
7 emissions lead to warming of a tenth to a full degree Celsius over snow in the Western U.S. Simulations
8 show that BC absorption of solar radiation in the atmosphere leads to as much as 0.6°C of warming in
9 the lower and mid troposphere (Ramanathan and Carmichael, 2008). Additional estimates of regional
10 temperature effects associated with BC emissions in the Arctic and the Himalayas are discussed in
11 sections 2.6.4 and 2.6.5, respectively.

12 The snow/ice albedo effects of BC have been linked to accelerated melting of snow and ice.
13 While many glaciers around the world and Arctic sea ice have receded in recent decades, attribution of
14 melt to BC is challenging due to other global and local contributions to warming and precipitation
15 changes. Regardless of the deposited BC, the solar zenith angle, cloud cover, snow grain size, and depth
16 of the snow also influence the albedo (Wiscombe and Warren, 1980). The most common method has
17 been to utilize models, compare model runs with and without BC influences, and evaluate with
18 observations. Direct measurements are generated by melting and then filtering samples of snow and
19 ice. The filters provide an estimate of BC concentration by comparing their observed optical
20 transmissions to optical transmissions of known amounts of BC (Noone and Clarke 1988, Warren and
21 Clarke 1990). The mass is then used to estimate or compare to measured snow albedo, calculating the
22 influence of BC. Another approach has been to apply a known amount of soot to an area, and then
23 compare the measured albedo and melting rate to a nearby clean plot of snow.

24 Snow and ice cover in the Western U.S. has also been affected by BC. Specifically, deposition of
25 BC on mountain glaciers and snow packs produces a positive snow and ice albedo effect, contributing to
26 the melting of snowpack earlier in the spring and reducing the amount of snowmelt that normally would
27 occur later in the spring and summer (Hadley et al. 2010). This has implications for freshwater resources
28 in regions of the U.S. dependent on snow-fed or glacier-fed water systems. In the Sierra Nevada
29 mountain range, Hadley et al. (2010) found BC at different depths in the snowpack, deposited over the
30 winter months by snowfall. In the spring, the continuous uncovering of the BC contributed to the early
31 melt. A model capturing the effects of soot on snow in the western U.S. shows significant decreases in
32 snowpack between December and May (Figure 2-17, Qian et al. 2009). Snow water equivalent (the
33 amount of water that would be produced by melting all the snow) is reduced 2-50 millimeters (mm) in
34 mountainous areas, particularly over the Central Rockies, Sierra Nevadas, and western Canada.

35 Koch et al. (2007) found that biomass burning emissions in Alaska and the Rocky Mountain
36 region during the summer can enhance snowmelt. Dust deposition on snow, at high concentrations, can
37 have similar effects to BC. A study done by Painter et al. (2007) in the San Juan Mountains in Colorado
38 indicated a decrease in snow cover duration of 18-35 days as a result of dust transported from non-local
39 desert sources.

1



2

3

Figure 2-17. Spatial Distribution of Change in Mean Snow Water Equivalent (SWE, mm) for March (Qian et al. 2009)

4

5

6

These changes affect various types of surfaces and geographic locations throughout the world, including Arctic ice caps and sea ice, glaciers, and mountain snowpack (see section 2.6.4 for more detailed treatment of Arctic impacts). For example, Ming et al. (2009) suggest that reduced albedos in some glaciers in west China from BC deposition might accelerate the melt of these glaciers. Figure 2-18 shows a Chinese glacier and the concentration of BC that results from melting the upper layers of the snowpack until it is buried by fresh snowfall.

12

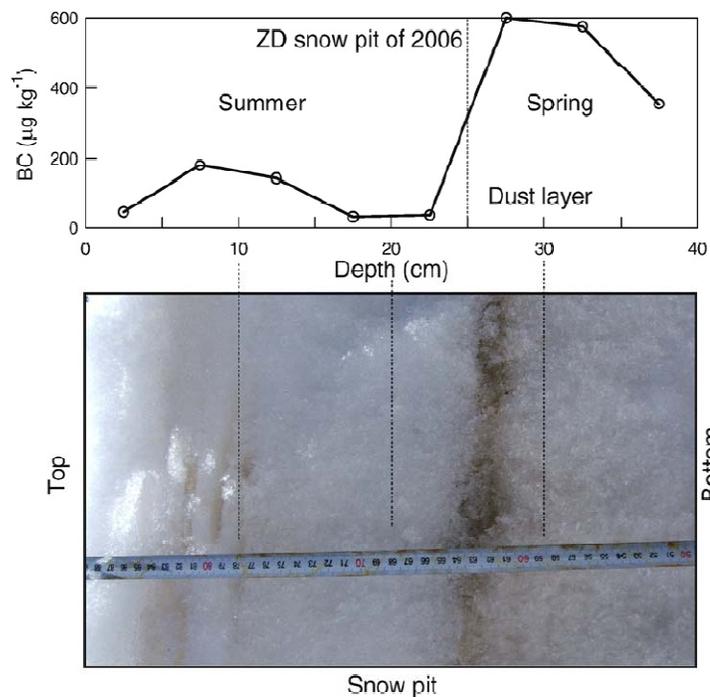


Figure 2-18. Black Carbon Concentrations (upper) Measured in the Snow Pit (bottom) of the ZD Glacier. Dust layer at 30 cm indicated the spring (melting season) of 2006. (From Ming, 2009, Figure 4.)

2.6.3 Non-Radiative Forcing Impacts of BC

In addition to its radiative effects, BC contributes to climate change through surface dimming and changes in precipitation patterns. These effects are associated other aerosols as well as BC. The dimming effect, which limits the amount of sunlight reaching the Earth’s surface, depends on the composition of the total atmospheric column above the earth’s surface. The effect of BC on precipitation depends on the location, the type of precipitation and background aerosol concentrations. Studies link aerosols to both increases and decreases in precipitation, as well as changes in timing and duration. GHGs are not associated with surface dimming, nor are they linked directly to changes in precipitation. Changes in precipitation from GHGs are mediated through changes in temperature.

2.6.3.1 Surface Dimming Effects

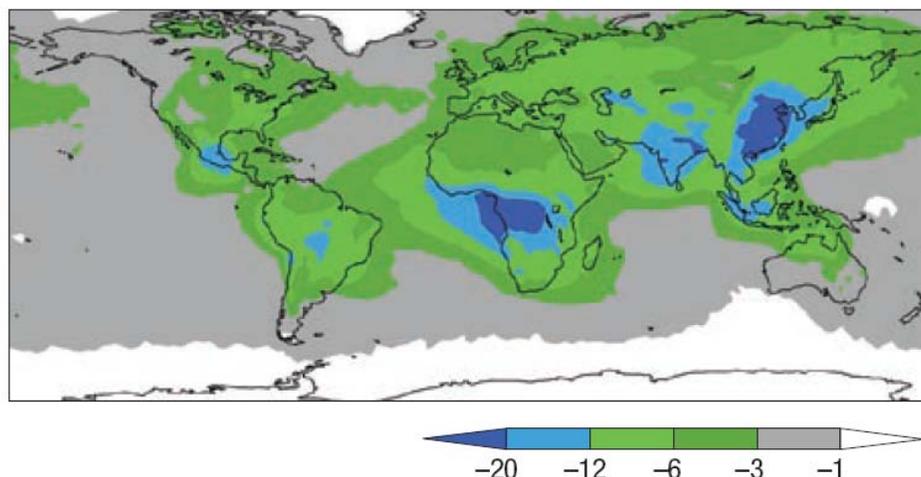
The absorption of incoming solar radiation by BC reduces the amount of solar radiation reaching the Earth’s surface, an effect referred to as *surface dimming* in many studies (e.g., Forster et al., 2007). This results in cooling at the surface (even though net forcing measured at the top of the atmosphere (TOA forcing) may be positive). A number of studies report evidence of global dimming between the 1960s and the 1980s, followed by an increase in the amount of sunlight reaching the surface during the 1990s to the present (sometimes referred to as *brightening*) (e.g., see review in Wild, 2009). Numerous studies suggest that the observed dimming and brightening trends are caused by changes in aerosol

1 emissions over time and the interaction of aerosol direct and indirect radiative forcing (Stanhill and
2 Cohen, 2001; Wild et al., 2005; Streets et al., 2006; Ruckstuhl et al., 2008). See Table 2-4 for a summary
3 of how aerosol interactions affect surface dimming.

4 Estimating the magnitude of this surface dimming effect is complicated, largely because the
5 distribution of BC in the vertical direction affects the impacts at the surface. Isolating the effect of BC is
6 also difficult since other non-BC aerosols (primarily sulfates) also scatter incoming solar radiation,
7 reducing the energy reaching the surface (Dwyer et al, 2010; Ramanathan and Carmichael, 2008). In
8 addition to the link between direct radiative forcing and surface dimming, the indirect effect of aerosols
9 on cloud albedo and cloud lifetime may also decrease solar radiation at the surface (Ramanathan and
10 Carmichael, 2008). Surface cooling combined with atmospheric heating from BC may increase the
11 stability of the boundary layer (e.g. the bottom layer of the troposphere that is in contact with the
12 surface of the earth) and reduce vertical mixing. This increase in atmospheric stability reduces natural
13 removal processes for air pollutants, resulting in worse air pollution episodes (Ramanathan and
14 Carmichael, 2008). As discussed in the next section, the increased stability in may also disrupt
15 established precipitation patterns in areas of high emissions and concentrations.

16 In some regions, BC, BrC, sulfates, organics, dust and other components combine to form
17 pollution clouds known as Atmospheric Brown Clouds (ABCs), which have been linked to global dimming
18 (Ramanathan et al., 2007; Ramanathan and Carmichael, 2008). Ramanathan and Carmichael (2008)
19 estimate the total global dimming effect from ABCs to be -4.4 W m^{-2} , with about -3.4 W m^{-2} from the
20 direct effect of aerosols (roughly half of which is attributed to BC) and the remaining -1 W m^{-2} from the
21 indirect effect. In this study, the -1.7 W m^{-2} of surface dimming from BC was found to be offset by $+2.6$
22 W m^{-2} of heating in the atmosphere. This resulted in a net TOA forcing estimate from this study of $+0.9$
23 W m^{-2} , as cited in section 2.6.1.1.

24 Dimming effects due to BC and the other aerosols are not spatially uniform (Figure 2-19). A
25 number of studies have found that dimming effects are particularly acute in certain regions associated
26 with high aerosol pollution levels and the presence of ABCs. These include major urban areas
27 (Ramanathan and Feng, 2009; Trenberth et al., 2007) and South Asia (Ramanathan and Feng, 2009;
28 Ramanathan et al., 2007; Ramanathan et al., 2005). The ABCs which cover large areas in the North
29 Indian Ocean and South Asia can reduce energy at the surface by 5-10% (Ramanathan et al., 2007;
30 Ramanathan and Carmichael, 2008). Some studies have estimated that the dimming associated with
31 ABCs can mask approximately half of the warming that would occur due to GHGs only at the surface in
32 the absence of ABCs, especially over Asia (Ramanathan et al., 2007; UNEP, 2008). Surface dimming
33 causes a reduction of approximately 6% in solar radiation at the surface over China and India when
34 compared to pre-industrial values (UNEP, 2008). The U.S. Global Change Research Program (CCSP, 2009)
35 estimated surface forcing values as low as -10 W m^{-2} over China, India, and sub-Saharan Africa.



1
2 **Figure 2-19. Surface Dimming by Anthropogenic Aerosols ($W m^{-2}$).** (Adapted from Chung et al.
3 (2005) and Ramanathan and Carmichael (2008).)

4 5 **2.6.3.2 Precipitation Effects**

6 Aerosols affect the processes of cloud and rain droplet formation. Some studies have linked
7 aerosols to reductions in rainfall, but these interactions are not well understood. A summary of the
8 aerosol interactions with clouds that cause changes in precipitation is provided in Table 2-4. According
9 to the IPCC (2007), the precipitation effects on a global scale attributed primarily to BC is from the semi-
10 direct effect (described in section 2.6.1.2) and the increased atmospheric stability resulting from that
11 effect (Ramanathan et al. 2005; Chung and Zhang, 2004; Menon et al., 2002). The increased stability
12 inhibits convection, affecting both rainfall and atmospheric circulation. As discussed in 2.6.1.2,
13 increased availability of cloud condensation nuclei (CCN) increases cloud lifetime thereby inhibiting
14 rainfall for a time period, which may be more important for shifting the location of rainfall than changes
15 in net global precipitation. There may also be increases in precipitation: BC in particular can stimulate
16 precipitation from ice clouds. However, because of the dependence of precipitation on complex and
17 localized conditions, scientific understanding of these effects is low and models often disagree on the
18 magnitude or sometimes even the sign of changes in precipitation due to factors such as warming or
19 aerosol emissions.

20 Surface dimming due to all types of aerosols may reduce precipitation by reducing the energy
21 available for evaporation from the Earth's surface (Liepert et al., 2004; Ramanathan et al. 2001).
22 Because rain is a major removal mechanism for BC from the atmosphere, large decreases in rainfall
23 could result in higher atmospheric concentrations of BC and other aerosols (Ramanathan and
24 Carmichael, 2008; Ramanathan et al. 2005).

25

Effects	Sign of Change in Surface Dimming	Potential Magnitude	Sign of Change in Precipitation	Potential Magnitude
Cloud albedo effect	Positive	Medium	NA	NA
Cloud lifetime effect	Positive	Medium	Negative	Small
Semi-direct effect	Positive	Large	Negative	Large
Glaciation indirect effect	Negative	Medium	Positive	Medium
Thermodynamic effect	Positive or negative	Medium	Positive or negative	Medium

Table 2-4: Overview of the Different Aerosol Indirect Effects and Their Implications for Global Dimming and Precipitation. This table applies to all aerosols, not just BC. Scientific uncertainty is very low for all effects except the cloud albedo effect (for which uncertainty is low). Table adapted from Table 7.10b in IPCC (Denman et al. 2007). For descriptions of the effects, see Section 2.6.1.2.

Ramanathan and Feng (2009) suggest that, on a global average basis, reduced precipitation caused by the surface dimming effects of aerosols is likely to be countered with increased precipitation from GHG-induced warming. The effect of aerosols on precipitation, however, varies by area, surface cover, and location. For example, in the tropics, the net effect of aerosols and GHG-induced warming may be reduced rainfall (Ramanathan and Feng, 2009). These shifts in rainfall patterns may have important implications for water availability.

In the U.S., Qian et al. (2009) found only small changes in the amount of precipitation in the western U.S. as a result of BC effects. While there is no evidence in North America that links any specific constituent of PM to changes in precipitation, there are studies that show correlations between total PM emissions and regional precipitation patterns. For example, Bell et al. (2008) find weekly patterns of emissions that correlate with weekly patterns in rainfall in the southeastern U.S. (Bell et al, 2008). Similar results have also been found for the East Coast of the U.S. (Cerverny and Balling, 1998).

There is stronger evidence linking aerosols to reduced precipitation in the tropics. Studies have indicated that surface dimming in this regions with high ABC's reduces evaporation (Hansen et al., 2007; Feingold et al., 2005; Yu et al., 2002). Other studies have found that the effect in the tropics may be unevenly distributed with increased precipitation just north of the equator (between 0 and 20N) and decreased precipitation just south of the equator (between 0 and 20S). This would shift the Intertropical Convergence Zone northward (Chung and Seinfeld, 2005; Roberts and Jones, 2004; Wang 2004). This northward shift may be caused by the enhanced temperature difference between the Northern and

1 Southern Hemispheres, which induces a change in circulation and convection in the tropics. Aerosols
2 have also been linked to impacts on regional precipitation in the Amazon basin (e.g., Martins et al.,
3 2009; Bevan et al., 2009). This is a region of high biomass burning emissions in the dry season. Further,
4 seasonal biomass emissions have been linked to larger changes in atmospheric circulation patterns by
5 affecting the global distribution of high-level clouds and convection precipitation (Jeong and Wang,
6 2010). Jeong and Wang (2010) also found that the climate response extends outside of the biomass
7 burning season. The effects of BC aerosols on precipitation may also extend beyond areas of high
8 concentrations. Wang (2007) found the largest change in precipitation occurs in the tropical Pacific
9 region which is far from the regions of largest BC forcing. The effect may be very similar to the pattern of
10 precipitation anomalies associated with the El Niño/Southern Oscillation.

11 There is also evidence that BC and ABCs slow down the monsoon circulation over South Asia.
12 Specifically, the surface dimming caused by BC aerosols (Meehl et al., 2008) and ABCs (Lau et al. 2006;
13 Ramanathan et al., 2005) alters both the north-south gradients in sea surface temperatures and the
14 land-ocean contrast in surface temperatures. These studies estimate an increase in pre-monsoon
15 rainfall during spring followed by a decrease in summer monsoon rainfall, in agreement with observed
16 trends.

17 Model studies of China have found increased rainfall in the south and reduced rainfall in the
18 north (Wu et al., 2008; Menon et al., 2002). Wu et al. (2008) simulated the regional climate effects in
19 Asia and found about a 0.6% increase in atmospheric water vapor over southern China, resulting in a
20 precipitation increase of 0.4–0.6 mm/day. In northern China, this study found about a 0.3% decrease in
21 water vapor and a resultant decrease in precipitation. Meehl et al. (2008) found small precipitation
22 increases over the Tibetan Plateau, but concluded that precipitation over China generally decreases due
23 to BC effects.

24 **2.6.4 BC Impacts in the Arctic**

25 BC emissions that are transported the Arctic are strongly linked to local warming (Reddy and
26 Boucher 2007) even if the globally averaged net climate impact of the total particulate emissions from
27 individual sources is uncertain. For example, Quinn et al. (2008) calculated that the contribution of
28 short-lived climate forcers (i.e., methane, tropospheric ozone, and tropospheric aerosols, including BC)
29 to Arctic warming is about 80% that of CO₂. BC can have significant snow albedo effects and the
30 magnitude of the indirect cooling effect over snow from co-emitted aerosols is reduced in the Arctic.
31 Studies using various climate model simulations suggest that as much as 50% of the observed retreat in
32 Arctic sea ice may be due to BC forcing (Hansen and Nazarenko, 2004; Flanner et al., 2007). Because
33 temperature in the Arctic has warmed at twice the global rate over the past 100 years (IPCC, 2007) and
34 because of the dramatic retreat of summer sea ice extent during the satellite observation period (see
35 Figure 2-20), there is interest in mitigation strategies that may slow the rate of climate change in this
36 region.

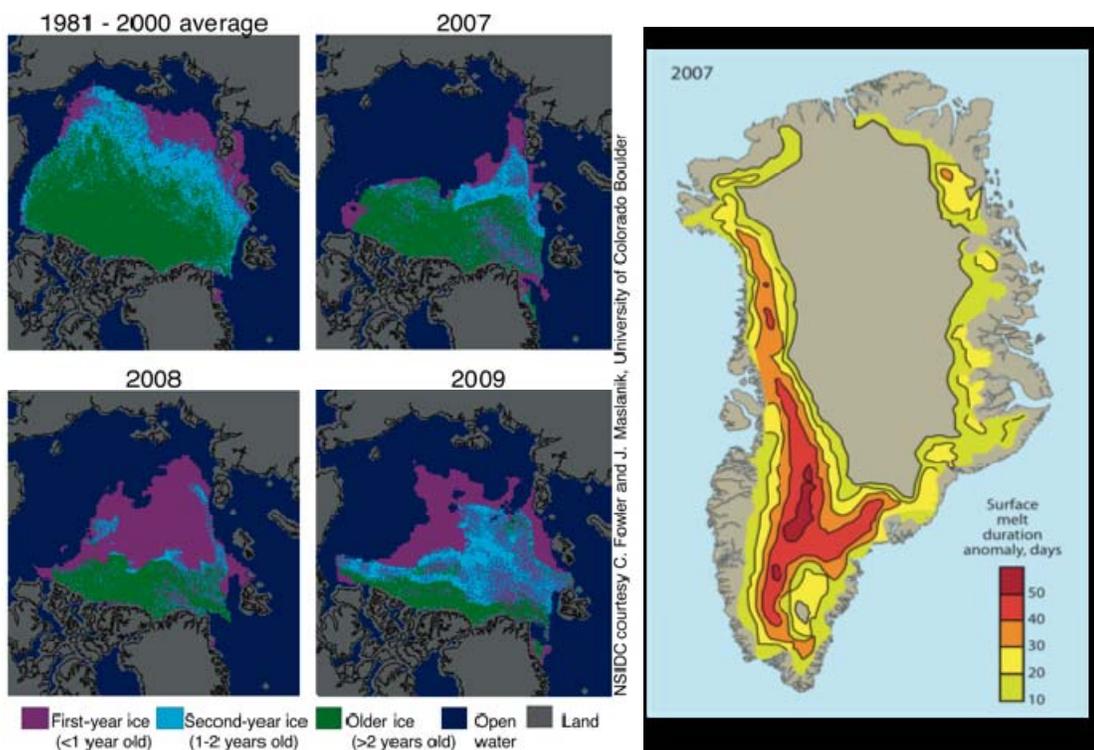


Figure 2-20. Evidence of Arctic Ice Melt. a) Extent of summer Arctic sea ice for 2007 -09 compared to 1981-2000 average. b) Duration of summer surface melt on Greenland in 2007 relative to 1973-2000 average (AMAP, 2009). Arctic summer sea ice has decreased by 40% since 1979, accompanied by in increasing discharge from the Greenland ice sheet. Natural variability may explain some of these changes, but the overall trend towards warming and melting has been attributed primarily to human induced climate change (Min et al., 2008; Holland et al., 2008). Summer sea ice melt creates a feedback loop that amplifies warming as reflective white ice/snow surfaces are replaced by darker ocean waters increased sunlight absorption. Recent work suggests a link between Arctic sea-ice melt and increased glacier runoff in Greenland. (Rennermalm et al., 2009)

Radiative forcing estimated from BC is larger over the Arctic than it is on average globally. Due to the lack of sunlight in winter months, the long days in summer, and the increased efficiency of transport of BC emissions from lower latitudes in spring there is also much larger seasonal variability in the estimates of radiative forcing from BC and other aerosols than there is from greenhouse gases (Quinn et al. 2008). Looking at forcing from fossil and biofuel emissions, Quinn et al. (2008) calculated a radiative forcing in the Arctic of $+1.2 \text{ W m}^{-2}$ in the spring, $+0.66 \text{ W m}^{-2}$ in the summer, $+0.16 \text{ W m}^{-2}$ in the fall, and only 0.09 W m^{-2} in the winter. Snow albedo forcing in the Arctic was calculated to add an additional $+0.53 \text{ W m}^{-2}$ in the spring, $+0.21 \text{ W m}^{-2}$ in the summer and negligible forcing in autumn and winter. This effect is amplified (e.g. increase in efficacy) by the hastening of the spring thaw that reveals darker ground and water (ocean/lake) surfaces.

The lifetime of aerosol particles in the Arctic is longer than other regions (see Garrett et al., 2004; Curry, 1995). This leads to a phenomenon known as Arctic haze which is the result of an accumulation of BC, OC, and sulfate particles in the atmosphere above the Arctic (Quinn et al., 2007). Strong surface-based temperature inversions and the dryness of the Arctic troposphere inhibit removal

1 of particles via deposition. Over a highly-reflective surface like the Arctic, BC particles absorb solar
 2 radiation and warm the atmosphere above and within the haze layer, while simultaneously reducing
 3 solar energy at the surface (i.e., surface dimming). Rather than a cooling effect from surface dimming,
 4 however, the atmospheric heating increases the downward longwave radiation and causes warming at
 5 the surface (Shaw and Stamnes, 1980; Quinn et al., 2008). Any warming particle above a highly-
 6 reflective surface can lead to heating of the entire surface-atmosphere aerosol column. The stable
 7 atmosphere above the Arctic prevents rapid heat exchange with the upper troposphere, increasing
 8 surface warming in the Arctic (Hansen and Nazarenko, 2004; Quinn, 2008).

9 Radiative forcing from both atmospheric concentration and deposition on the snow and ice has
 10 contributed to the surface temperature warming in the Arctic (Quinn et al., 2008). Simulations by
 11 Flanner et al. (2007) suggest that the deposition of BC from sources in North America and Europe on
 12 Arctic sea ice may have resulted in a surface warming trend of as much as 0.5 to 1°C. Similarly, Shindell
 13 and Faluvegi (2009) found 0.5 and 1.4°C of warming from BC in the Arctic since 1890. For the BC snow
 14 albedo effect, Quinn et al. (2008) estimated a warming of 0.24 to 0.76°C, varying by season. Warming
 15 due to BC heating in the atmosphere is estimated to be a further 0.24°C in spring, 0.15°C in summer, and
 16 nearly zero in autumn and winter. In Table 2-5, we show estimates of temperature increases in the
 17 Arctic from various BC emission sources (Shindell and Faluvegi, 2009; Flanner et al., 2007; Jacobson,
 18 2010). Part of these increases in temperature may also have been “unmasked” in recent years from
 19 reductions in sulfate aerosols and its gaseous precursor, sulfur dioxide (Shindell and Faluvegi, 2009).
 20 While sulfate aerosols have a negative radiative forcing, the reductions in sulfate aerosols have been
 21 strongly justified by improvements in air quality, acid rain, visibility, public health, and lessening of direct
 22 effects of sulfates on ecosystems. It has also been suggested the potential cooling effects of BC such as
 23 indirect radiative forcing and the ratio of BC to cooling components (e.g. OC) may not be as important in
 24 the Arctic since the snow and ice albedo darkening is so dominant (Mauritsen et al. 2010).

25 **Table 2-5. Arctic Temperature Impacts from Emissions of Black Carbon from Different Sectors.**

Scale of physical impact	Estimated Forcing or Temperature Change	Source of emissions	Aerosols	Model	Reference	Notes
Arctic	0.5 – 1.4 °C	100% of FF, BF, BB	BC, OC	GISS-ER	Shindell and Faluvegi (2009)	Aerosol indirect included “crudely.”
Arctic	0.5 – 1.6 °C	100% of FF, BF, BB	BC, OC	NCAR-CAM3 and SNICAR	Flanner et al. (2007)	Range results from using fire frequencies in high year (1998) and low year (2001)
Arctic	1.2 °C	100% of FF	BC, OC, minor inorganics	GATOR-GCMOM	Jacobson (2010)	
Arctic	1.7 °C	100% of FF, BF	Same	Same	Jacobson (2010)	

Arctic	Indirect > 0	100% of All sources	All aerosols	Idealized calculations, observations	Mauritsen, (2010)	Indirect only, no direct or snow albedo effect.
Key: FF = fossil fuel; BF = biofuel; BB = biomass burning.						

1

2 While there are strong qualitative indications of Arctic snow and ice melt from BC, quantitative
3 studies have only recently entered the peer-reviewed literature. Some studies have linked the local
4 warming measured on the Greenland Ice Sheet to observations of a gradual loss of ice, and modeled the
5 overall impact on the mass balance of the ice sheet. Box et al. (2004), for example, estimated the
6 modeled ice sheet mass balance at -76 km^3 per year, leading to a 2.2 mm sea level rise (contributing
7 15% of global sea level rise) during 1991- 2000. Hanna et al. (2005) considered a longer time period,
8 and estimated that the overall mass balance declined at a rate of -22 km^3 per year in 1961-1990 and -36
9 km^3 per year for 1998-2003, with melting during the past 6 years contributing 0.15 mm per year to
10 global sea level rise. Finally, Thomas et al. (2006) reported accelerating mass loss between an earlier
11 period (4-50 Gt per year, 1993-1999) and a more recent period (57-105 Gt per year, 1999-2004). In
12 a modeling study by Flanner et al. (2007), land snowmelt rates north of 50°N latitude (about 70 miles
13 north of the U.S./Canada border in Minnesota) increased by 28% in 1998 and 19% in 2001 in the month
14 preceding maximum melt when compared to control runs that did not include BC from large boreal fires
15 that occurred in 1998 and 2001. Strack et al. (2007) found soot deposition in the Alaskan Arctic tundra
16 created snow free conditions five days earlier than model runs without BC deposition. Ongoing studies
17 will help validate and constrain modeling effort. Importantly, American, Norwegian, Russian, and
18 Canadian research groups collaborated under the International Polar Year (2007-2008) program to
19 survey BC concentrations in snow and ice north of 65°N latitude in both the Eastern and Western Arctic
20 (Doherty, et al., 2010).

21 The location of the emission also matters for the magnitude of the effects in the Arctic, and as a
22 result, the benefits of mitigation. BC emissions from Northern countries have decreased since their peak
23 in the early 20th century. This is supported by a downward trend in the observed concentrations of
24 ambient and snow ice BC in the Arctic (see Chapter 5.6). However, BC deposited on areas covered
25 permanently with ice and snow, such as the Greenland Ice Sheet, tends to remain for long periods—
26 sometimes thousands of years—before being removed by surface run-off processes (Quinn et al., 2008).
27 As the Arctic warms, this BC deposited over decades is exposed, enhancing the current snow and ice
28 albedo effect. The effect of BC on the snow and ice albedo in the Arctic thus involves historical BC
29 deposition in the Arctic region. An important uncontrolled source of BC in Northern countries is open
30 biomass burning. Several recent studies have looked at the effect of these emissions on the Arctic. For
31 example, Stohl et al. (2006) found that North American boreal forest fires lead to elevated
32 concentrations of light absorbing aerosols including BC throughout the entire Arctic, with substantial
33 implications for Arctic warming and enhanced snow albedo effects. Other studies have linked open
34 biomass burning to reduced surface albedo and accelerated melting (Hegg et al., 2009; Generoso et al.,
35 2007; Kim et al., 2005). Following agricultural fires in Eastern Europe in spring 2006, Stohl et al. (2007)
36 measured record high air pollution levels and BC concentrations in parts of the Arctic above Europe.

1 Similarly, in a series of studies, Warneke et al. (2009 and 2010) found that spring fires in Russia (Siberia)
2 and Kazakhstan can more than double the Arctic haze that builds up during the winter months.

3 **2.6.5 BC Impacts in the Himalayas**

4 The world's third largest snowpack after Antarctica and the Arctic is found in the Hindu Kush–
5 Himalayan–Tibetan (HKHT) region. The mountain ranges that define this region fall primarily along the
6 borders of Pakistan, Afghanistan, India, Nepal, and China (UNEP, 2008). It is often referred to as the
7 Earth's "third pole." Atmospheric warming associated with BC is believed to be a significant factor in the
8 observed increases in melting rates of glaciers and snowpack in the HKHT (Thompson et al, 2003;
9 Barnett et al., 2005, Lau et al., 2010, UNEP 2008). Ramanathan and Carmichael (2008) and Ramanathan
10 et al. (2007) suggested that the advection over the Himalayas of air warmed by BC has played a role
11 comparable to that of GHGs in the observed retreat of Himalayan glaciers. A recent study by Carmichael
12 et al. (2009) also shows that BC throughout Asia has an atmospheric warming potential of about 55% of
13 that attributed to CO₂.

14 High radiative forcing estimates have been calculated for the Himalayas due to the large amount
15 of mountain snow and ice cover as well as the proximity to high emissions of BC from parts of China and
16 the Indian subcontinent. Flanner et al. (2007) calculated average forcing of +1.5 W m⁻² with short-term
17 forcing of up to 20 W m⁻² in the spring. Translating this to temperature, Flanner et al. (2009) attributed
18 an increase in the land-averaged March-May surface temperature of 0.93°C in Eurasia from BC and
19 organic matter in the atmosphere and deposited on the snow.

20 The state of the literature suggests that the effects of BC on snowpack and glaciers are
21 important. BC can alter snowpack and glacier extent and retreat through two mechanisms, the first
22 being increasing and decreasing precipitation as discussed in Section 2.6.1.3, and the second being local
23 warming, especially through deposition, increasing the rate of melt. Lau et al. (2010) found that heating
24 of the atmosphere by dust and BC leads to widespread enhanced warming over the Tibetan Plateau and
25 accelerated snowmelt in the western Tibetan Plateau and Himalayas. Menon et al. (2010) show
26 observed trends in snow cover in the Himalayas, with a spatially heterogeneous pattern of decreases
27 and increases of up to 17% from 1990 to 2001, where the area of decreases is much larger than the area
28 of increases. Menon et al. simulated similarly spatially heterogenous snow cover changes in a modeling
29 study due to aerosol emissions, showing that the influence of the aerosols was larger than the influence
30 of changing sea surface temperatures over that time period. Over Eurasia, Flanner et al. (2009)
31 conducted a modeling study that found the combination of strong snow albedo feedback and large fossil
32 fuel and biofuel emissions of BC and organic matter from Asia induce 95% as much springtime snow
33 cover loss as anthropogenic CO₂ alone. The effects on glaciers are not well quantified, but Xu et al.
34 (2009) found evidence that black soot aerosols deposited on Tibetan glaciers have been a significant
35 contributing factor to observed rapid glacier retreat. Changes in the timing and extent of melting may
36 adversely affect regional freshwater resources in region which relies heavily on this melt (Carmichael et
37 al., 2009).

1 **2.6.6 Summary of BC Impacts in Key Regions**

2 **Table 2-6. Climate Effects of Black Carbon in the United States, Asia, and the Arctic (Summary).**

	U.S.	Asia	Arctic
Radiative Forcing Effects	<ul style="list-style-type: none"> Estimates of direct radiative forcing of BC over the US range from 0.1- 0.7 W/m⁻² 	<ul style="list-style-type: none"> South and East Asia have some of the world’s highest estimates of radiative forcing, but large ABCs exert a counterbalancing dimming effect at the surface Average annual snow and ice albedo forcing in the Tibetan Plateau has been estimated to be 1.5 W m⁻², with local instantaneous forcing up to 20 W m⁻² 	<ul style="list-style-type: none"> Springtime Arctic forcing has been estimated to be 1.2 W m⁻² (direct) and 0.53 W m⁻² (snow albedo)
Temperature Effects	<ul style="list-style-type: none"> No studies were identified for U.S. temperature effects from BC Estimates of average warming from BC in the Northern Hemisphere range from 0.29°C to 0.54 °C 	<ul style="list-style-type: none"> Over the Himalayan region, atmospheric BC was estimated to result in up to 0.6°C of warming 	<ul style="list-style-type: none"> BC deposited on snow results in warming of roughly 0.4 to 0.5°C, varying by season Atmospheric BC was estimated to contribute roughly 0.2°C in spring, 0.1°C in summer, and nearly zero in autumn and winter
Precipitation Effects	<ul style="list-style-type: none"> One study found little change in the amount of precipitation in the western U.S. as a result of BC effects. Other studies have found that rainfall patterns in the eastern U.S. match particulate matter emissions. 	<ul style="list-style-type: none"> The cooling at the surface leads to reduced evaporation and precipitation as well as changes in sea-land temperature gradients. Precipitation and temperature gradient modifications can lead to shifts of regional circulation patterns such as a decrease in the Indian and Southeast Asian summer monsoon rainfall and a north-south shift in eastern China rainfall. 	<ul style="list-style-type: none"> No studies were identified for Arctic precipitation effects
Snow and Ice Effects	<ul style="list-style-type: none"> In the western U.S., BC deposition on mountain glaciers and snow produces a positive snow and ice albedo effect, contributing to the snow melt earlier in the spring. Early snowmelt reduces the amount of water resources that normally would be available later in the spring and summer, and may contribute to seasonal droughts. 	<ul style="list-style-type: none"> BC atmospheric warming is believed to be a significant factor in the melting of the HKHT glaciers and snow pack. The deposition of BC on glaciers and snow pack in Asia also has a strong snow and ice albedo positive feedback that accelerates melting of the glaciers and snow, with implications for freshwater availability and seasonal droughts. 	<ul style="list-style-type: none"> BC may increase snowmelt rates north of 50°N latitude by as much as 19-28%. As much as 50% of the observed retreat in Arctic sea ice may be due to BC forcing. Soot deposition in the Alaskan Arctic tundra created snow free conditions five days earlier than model runs without BC deposition.

1 2.7 Economic Value of BC Impacts on Climate

2 Methods for establishing the economic value of the climate damages associated with BC are still
3 being developed. Assessing the value of damages through a single metric (i.e., dollars) provides useful
4 information that can help inform policymakers regarding the scale and scope of the climate impacts of
5 BC and the benefits that can be gained from BC mitigation. However, no study to date has fully
6 monetized the climate impacts of BC. An analysis of this type would need to include the benefit of
7 avoiding risks and impacts associated with warming (especially near term warming and rate of
8 change), as well as the value of avoiding impacts such as accelerated ice and snow melt and changes in
9 precipitation induced by BC.

10 Currently, efforts to develop climate impacts valuation methods have focused on CO₂. In
11 computing the value to society of avoided climate damages, EPA assigns a benefits dollar value to CO₂
12 emission reductions using estimates of a “social cost of carbon” (SCC) developed by a U.S. federal
13 government interagency working group in 2010. The SCC is an estimate of the monetized damages
14 resulting from an incremental increase in CO₂ emissions in a given year; likewise, it can be thought of as
15 the monetized benefit to society of reducing one ton of CO₂. The SCC estimates are intended to include
16 an array of human-induced climate change impacts, such as changes in net agricultural productivity,
17 human health, property damages from increased flood risk, and the value of ecosystem services due to
18 climate change. Current SCC values, such as those utilized by EPA to analyze the benefits of the 2010
19 *Light-Duty Vehicle Greenhouse Gas Emission Standards and Corporate Average Fuel Economy Standards*
20 (U.S. EPA, 2010), are subject to a number of limitations, including the incomplete way in which the
21 underlying climate models capture catastrophic and non-catastrophic impacts, the incomplete
22 treatment of adaptation and technological change, uncertainty in the extrapolation of damages to high
23 temperatures, and assumptions regarding risk aversion. The SCC estimates developed for CO₂ have
24 been controversial due to the difficulty of estimating economic impacts across nearly every sector of the
25 economy as well as valuation issues regarding impacts on natural ecosystems. Furthermore, these
26 estimates were developed exclusively for CO₂ and are not directly transferrable to other GHGs or BC.³

27 It might be possible to use a similar approach to develop a social cost specific to BC using
28 integrated assessment models (IAMs) that combine economic growth, climate processes, and feedbacks

³ One approach that might appear tempting is to use existing estimates for the SCC for CO₂, and translate them into a social cost for BC using metrics such as the 100-year global warming potential, or GWP (see, for example, Copenhagen Consensus Center Reports). However, the damage functions used in the underlying models are sensitive to when and by how much the temperature changes – therefore, given the orders of magnitude shorter lifetime, a social cost calculated from first principles for BC could be very different than one that merely scales the social cost of CO₂ by the GWP. Again, regional dependence and impacts on precipitation patterns would not be captured by this method, nor would the regional dependence of snow and ice deposition and therefore special sensitivity of alpine and Arctic ecosystems to BC emissions. Therefore, the social cost of BC might not be well represented by using GWPs to scale an SCC. (See further discussion of the applicability of GWP metrics to BC in Chapter 11.) Given that warming profiles and impacts other than temperature change vary across climate forcers, the interagency SCC working group made a preliminary conclusion that transforming other climate forcers “into CO₂-equivalents using global warming potential, and then multiplying the carbon-equivalents by the SCC, would not result in accurate estimates of the social costs” of these non-CO₂ forcers (Interagency SCC Group, 2010), though it is unclear how large such an error would be.

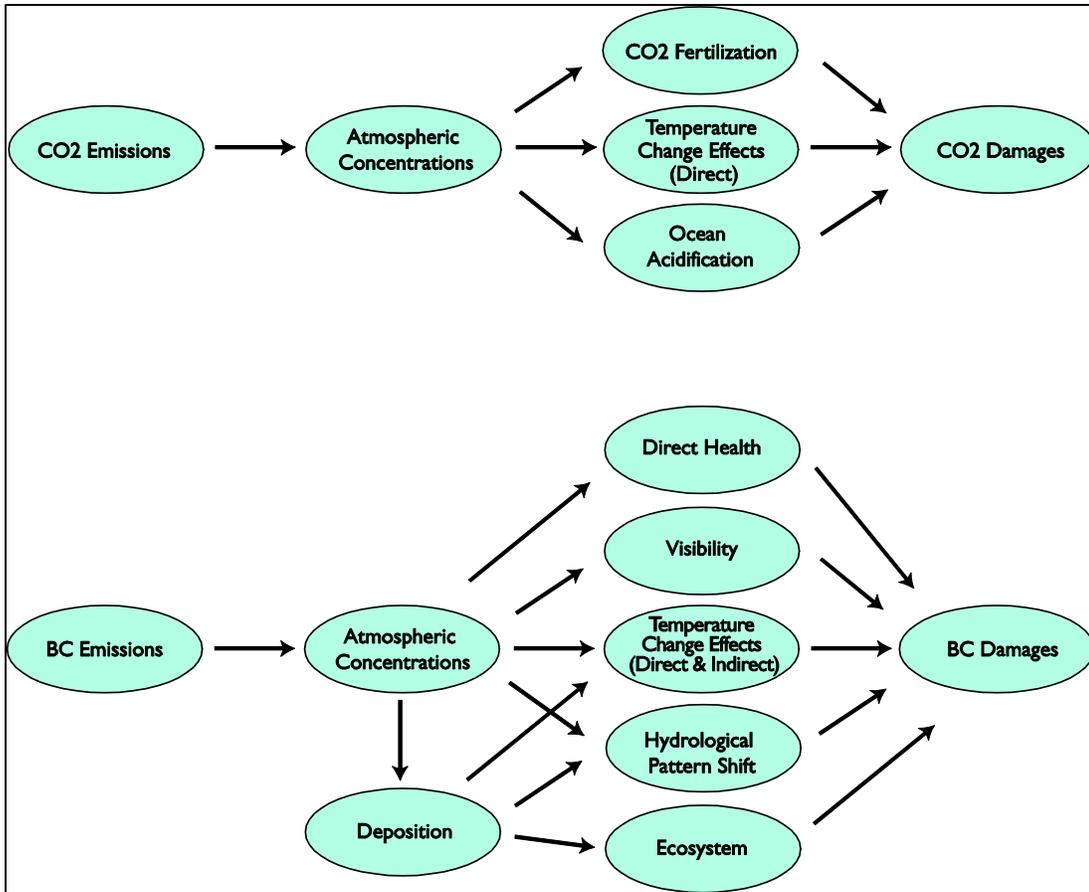
1 between the global economy and climate into a single framework to translate BC emissions into
2 economic damages. However, there are a number of factors which would complicate these calculations.
3 These difficulties stem partly from limitations in the capabilities of IAMs, and partly from the complexity
4 of the cause-effect chain needed to measure the physical links between emissions and climate change
5 impacts, and to calculate damages (see Figure 2-21).

6 Few IAMs are designed to demonstrate regional impacts, and currently these models do not
7 adequately consider the impact of BC and other short-lived climate forcers on the *rate* of climate
8 change. In addition, the feasibility of considering indirect climate effects such as the impact of BC on
9 snow and glacier melt and changes in precipitation patterns in the IAMs must be evaluated. In one
10 aspect, at least, calculating a social cost for BC might actually be easier than calculating a social cost for
11 CO₂: the short lifetime of BC and the relatively immediate nature of the climate impacts reduce the
12 extent to which social cost calculations would depend on the social discount rate selected. Due to the
13 complexities involved with valuing the climate benefits of BC reductions, a top-down approach using
14 IAMs may not be preferred. Rather, a bottom-up approach that considers location, emission profiles of
15 sources, and ambient concentrations and deposition of BC similar to the approaches used to quantify
16 health effects may be needed.

17 The cause-effect chain from emissions to impacts and damages is also complex for BC. The
18 regional nature of many BC impacts, the importance of location of emissions, and BC's impacts on
19 precipitation, snow/ice, and surface dimming add additional complexities to any such approach that are
20 not present for CO₂ SCC calculations. In addition, the peer reviewed literature lacks impact functions
21 and valuation methods necessary to assess many of these BC effects. Finally, because BC is emitted as
22 part of a mixture, incorporation of the climate impacts of reducing other co-emitted aerosols into a
23 social cost approach would reflect the net impact more accurately.

24

25



1
2
3
4

Figure 2-21. Cause-Effect Chains for CO₂ and Black Carbon from Emissions to Damages (Hartman 2010 modified)

1

2

3. Black Carbon Effects on Public Health and the Environment

3

4

3.1 Summary of Key Messages

5

- 6 • Short-term and long-term exposures to PM_{2.5}, of which BC is a constituent, are associated with a
7 broad range of adverse human health effects including respiratory and cardiovascular effects, as
8 well as premature death.
- 9 • The available scientific evidence suggests that the health effects associated with exposure to BC are
10 generally consistent with those observed for PM_{2.5}, with the most consistent evidence for
11 cardiovascular effects, and fewer studies supporting an association with respiratory effects or
12 mortality.
- 13 • At present, we have insufficient information to fully assess the health effects of BC relative to other
14 constituents of PM_{2.5}.
- 15 • PM_{2.5}, including BC, is linked to adverse impacts on ecosystems, to visibility impairment, to reduced
16 agricultural production in some parts of the world, and to materials soiling and damage.
- 17 • Techniques exist to quantify and place a monetary value on many health benefits associated with
18 reductions in PM_{2.5}. However comparable methods are not as well developed for non-climate
19 welfare benefits with the exception of visibility. A “damage-function” approach is the typical
20 method used to calculate the benefits of the modeled changes in environmental quality.

21

3.2 Introduction

22

23 This chapter assesses the current scientific knowledge relating to the public health and non-
24 climate welfare effects (air quality effects) associated with short-term and long-term exposure to BC.
25 The methodologies used to assess air quality benefits to society from reducing BC emissions are also
26 addressed. As discussed below, these assessment methods rely on well established concentration-
27 response functions developed for fine particles and a wide-range of health endpoints as a basis for
28 evaluating the benefits of reducing emissions from sources of BC. The total reductions in PM_{2.5}
29 achieved by BC-oriented mitigation strategies include reducing emissions and associated risks of directly
30 emitted organic particles and gaseous precursors of PM, as well as BC. The impacts of air quality
31 programs targeted at reducing BC in terms of human health and non-climate welfare benefits are
32 discussed more fully in Chapter 6 of this report.

1 **3.3 Health Effects Associated with Exposure to PM_{2.5}, including BC**

2 Since 1997, EPA has recognized the need to regulate fine and coarse-fraction particles
3 separately. Current national ambient air quality standards (NAAQS) use PM_{2.5} as the indicator for fine
4 particles, and PM₁₀ as the indicator for thoracic coarse particles. At present, EPA is undertaking another
5 periodic review of these standards. As part of this review, EPA has completed an Integrated Science
6 Assessment for Particulate Matter (“ISA”) (U.S. EPA, 2009) providing a concise evaluation and
7 integration of the policy-relevant science pertaining to the health and environmental effects of ambient
8 particles. Discussion below is based on evidence assessed and conclusions drawn in the ISA.

9 A large body of scientific evidence links exposures to fine particles (i.e., ambient PM_{2.5} mass
10 concentrations) to an array of adverse effects, including premature mortality, increased hospital
11 admissions and emergency room visits for cardiovascular and respiratory diseases, and development of
12 chronic respiratory disease (U.S. EPA, 2009). Recent evidence provides a greater understanding of the
13 underlying mechanisms for PM_{2.5} induced cardiovascular and respiratory effects for both short- and
14 long-term exposures, providing biological plausibility for the effects observed in epidemiological studies.
15 This evidence links exposure to PM_{2.5} with cardiovascular outcomes that include the continuum of
16 effects ranging from more subtle subclinical measures (e.g. changes in blood pressure, heart rate
17 variability) to premature mortality. These health effects may occur over the full range of PM_{2.5}
18 concentrations observed in the long- and short-term epidemiological studies and the EPA has concluded
19 that no discernable threshold for any effects can be identified based on the currently available evidence.

20 In reviewing the studies regarding health effects of PM_{2.5}, EPA has recognized that it is highly
21 plausible that the chemical composition of PM would be a better predictor of health effects than
22 particle size alone (U.S. EPA, 2009, 6-202). Differences in ambient concentrations of PM_{2.5} constituents
23 observed in different geographical regions as well as regional differences in PM_{2.5}-related health effects
24 reported in a number of epidemiological studies are consistent with this hypothesis (U.S. EPA, 2009,
25 section 6.6). Over the past decade, the scientific community has focused increasingly on trying to
26 identify the health impacts of particular PM_{2.5} constituents or groups of constituents associated with
27 specific source categories of fine particles. The growing body of evidence for the health impacts of
28 specific PM_{2.5} constituents includes evidence of effects associated with exposure to BC and associated
29 OC. However, the ISA concludes that the currently available scientific information continues to provide
30 evidence that many different constituents of the fine particle mixture, as well as groups of constituents
31 associated with specific source categories of fine particles, are linked to adverse health effects. While
32 there is “some evidence for trends and patterns that link specific PM_{2.5} constituents or sources with
33 specific health outcomes... there is insufficient evidence to determine if these patterns are consistent or
34 robust” (U.S. EPA, 2009, p. 6-210). Consequently, research and data collection activities focused on
35 particle composition could improve our understanding of the relative toxicity of different fine particle
36 constituents or groups of constituents associated with specific sources of fine particles to inform future
37 regulatory activities and benefits assessments.

38 Some community epidemiology studies have included BC or EC measurements as one of several
39 indicators of fine particulate air pollution. The effects observed with BC in these studies are similar to

1 those observed for PM_{2.5} and some other PM constituents (e.g., nickel, vanadium), suggesting that these
2 effects are not attributable solely to BC. Indeed, it would be difficult to separate the contribution of BC
3 to these associations from those of co-emitted OC and other correlated and co-emitted primary
4 pollutants in such studies. Still, these studies provide generally consistent evidence for an association
5 between cardiovascular morbidity and BC concentrations. For example, a number of studies have
6 reported associations between short-term exposure to BC and cardiovascular effects. A series of
7 analyses found that changes in blood pressure (Delfino et al. 2010c; Mordukhovich et al. 2009; Wilker et
8 al. 2010) and heart rate variability (HRV) (Adar et al., 2007; Chuang et al., 2008; Gold et al., 2005; Huang
9 et al., 2003; Park et al., 2005; Schwartz et al., 2005) were associated with increases in ambient mean BC
10 concentration. The ST-segment of an electrocardiograph represents the period of slow repolarization of
11 the ventricles and ST-segment depression can be associated with adverse cardiac outcomes, including
12 ischemia. Delfino et al. (2010a) found positive associations between ST-segment depression and BC
13 concentrations. Homocysteine, a sulfur-containing amino acid formed during metabolism of
14 methionine, is a risk factor for atherosclerosis, myocardial infarction (MI), stroke, and thrombosis.
15 Similarly, lower blood DNA methylation content is found in processes related to cardiovascular
16 outcomes, such as oxidative stress and atherosclerosis. Several studies observed an association
17 between BC concentration and elevated plasma total homocysteine (Park et al. 2008; Ren et al. 2010).
18 An additional study (Baccarelli et al. 2009) observed an association between lower blood DNA
19 methylation content and BC concentrations. Cardiac arrhythmia (a broad group of conditions where
20 there is irregular electrical activity in the heart) was associated with increased concentrations of BC in
21 studies conducted in Boston (Dockery et al., 2005; Dockery et al., 2005; Rich et al., 2005; Zanobetti et al.
22 2009), but not in Vancouver, Canada (Rich et al., 2004). Another series of analyses has reported
23 inconsistent associations between BC and blood markers of coagulation and inflammation, with some
24 studies finding an effect (Delfino et al., 2008; Delfino et al., 2009; O'Neill et al., 2007), and others finding
25 no effect for a blood marker with large intra-individual variability (i.e., BNP)(Wellenius et al., 2007) or no
26 effects for acute lag periods (i.e., 48 hours or 1 week) (Zeka et al., 2006). Concentrations of BC (Peters
27 et al., 2001; Zanobetti and Schwartz, 2006) and EC (Bell et al., 2009; Peng et al., 2009, Sarnat et al. 2008;
28 Tolbert et al. 2007) were also found to be associated with hospital admissions and emergency
29 department visits for cardiovascular outcomes.

30 The most noteworthy new cardiovascular-related revelation in the past six years with regards to
31 PM exposure is that the systemic vasculature may be a target organ (U.S. EPA, 2009). Endothelial
32 dysfunction is a factor in many diseases and may contribute to the origin and/or exacerbation of MI or
33 ischemic heart disease, as well as hypertension. Endothelial dysfunction is also a characteristic feature of
34 early and advanced atherosclerosis. O'Neill et al. (2005) reported that increases in mean BC
35 concentration were associated with decreased vascular reactivity among diabetics, but not among
36 subjects at risk for diabetes. Madrigano et al. (2010) observed that BC was associated with a marker of
37 endothelial function and inflammation, and that genes related to oxidative defense might modify this
38 association. Consistent with these findings, animal toxicological studies have shown that BC can affect
39 heart rate variability (Tankersley et al. 2004; 2007), cardiac contractility (Tankersley et al. 2008) and
40 oxidative stress response (Tankersley et al. 2008), providing biological plausibility for a long-term effect
41 on cardiovascular health.

1 Fewer studies have examined the effects of BC (often measured as EC in this context) with
2 respiratory effects. Delfino et al. (2006) found associations between airway inflammation and EC among
3 asthmatic children, while Jansen et al. (2005) reported an association with a marker of pulmonary
4 inflammation and BC among older adults. These results are supported by toxicological studies reporting
5 evidence of airway inflammation (Godleski et al., 2002; Saldiva et al., 2002). Kim et al. (2004) and
6 McConnell et al. (2003) reported significant associations of bronchitic symptoms among children with
7 asthma and EC. Suglia et al. (2008) reported that BC was associated with decreased lung function
8 among urban women. A recent analysis evaluated the effect of PM_{2.5} components on respiratory
9 hospital admissions and found statistically significant associations between the county-average
10 concentrations of EC and effect estimates for respiratory hospital admissions (Bell et al., 2009).

11 A few recent studies have examined the association between mortality and short-term exposure
12 to components of PM_{2.5}, including BC or EC. Franklin et al. (2008) did not observe an effect of EC on
13 mortality. Lippmann et al. (2006) reported that nickel, vanadium, and EC were the best predictors,
14 respectively, of PM₁₀ risk estimates for mortality. Lipfert et al. (2006; 2009) found positive associations
15 between EC and all cause mortality, while Ostro et al. (2007; 2008) found positive associations between
16 EC and cardiovascular mortality. These associations (Ostro et al. 2007; 2008) were higher in individuals
17 with lower educational attainment and of Hispanic ethnicity. A systematic review and meta-analysis of
18 short-term exposure time-series studies of black smoke (a surrogate for BC) and daily mortality detected
19 significant, positive associations with all-cause, cardiovascular, and respiratory mortality (Smith et al.
20 2009).

21 Some studies have attempted to trace PM health effects back to specific sources using source
22 apportionment techniques. A number of these studies have linked BC-rich sources, including motor
23 vehicles and traffic, with adverse cardiovascular and respiratory health outcomes (USEPA 2009, Section
24 6.6.2). Sarnat et al. (2008) also found consistent positive associations with sources related to biomass
25 combustion and metal processing. However, in general the uncertainties associated with source
26 apportionment methods have not been well characterized.

27 BC is a component of indoor air pollution, which has been implicated in the array of health
28 effects affecting the approximately 3 billion people worldwide who rely on solid fuels for everyday
29 cooking and heating, mostly in the form of biomass (e.g., wood, animal dung, or crop wastes) but also
30 coal (mainly in China) (Rehfuess et al. 2006). Exposure to indoor air pollution from solid fuel use has
31 been linked to approximately 2 million deaths per year (WHO, 2009). The use of solid fuels in poorly
32 ventilated conditions results in high levels of indoor air pollution, most seriously affecting women and
33 their youngest children (Bruce et al. 2000). Recent observational studies have suggested that indoor air
34 pollution from biomass fuel is associated with respiratory morbidity, including acute lower respiratory
35 tract infections in children (Smith et al., 2000) and COPD in women (Ezzati, 2005; Orozco-Levi et al.,
36 2006; Smith et al., 2004, Rinne et al. 2006; Ramirez-Venegas et al. 2006; Liu et al. 2007; Sumer et al.
37 2004; Kiraz et al. 2003; Regalado et al. 2006.). Biomass smoke in Guatemalan women has been shown to
38 increase diastolic blood pressure (McCracken et al., 2007). Evidence also exists that implicates exposure
39 to biomass fuel smoke in adverse effects on different birth outcomes, including low birth weight and
40 stillbirth (Boy et al., 2002; Sram et al., 2005; Pope et al. 2010).

1 Finally, it is important to note that a variety of hazardous air pollutants (HAPs) including PAHs,
2 dioxins and furans, are co-emitted with BC (Allen et al., 2006; Shun-I Shih et al., 2008; Hedman et al.,
3 2006; Vinay Kumar Yadav et al., 2010; Omar Amador-Munoz et al., 2010; Walgraeve et al., 2010). These
4 HAPs are associated with adverse health effects including cancer and respiratory effects, among others.
5 Reductions in HAP emissions occurring in conjunction with BC mitigation programs will help reduce
6 these health risks. Furthermore, these toxic pollutants are generally persistent once they are emitted
7 into the environment, so these co-benefits can be expected to have long lasting beneficial impacts (Kai
8 Hsien Chi et al, 2010; Quiroz, 2010).

9 **3.4 Non-Climate Welfare Effects of PM_{2.5}, including BC**

10 Non-climate welfare effects resulting from BC emissions are discussed in terms of PM_{2.5}
11 exposure and deposition. PM_{2.5} has been linked to adverse impacts on ecosystems, primarily through
12 deposition of PM constituents. Crop yields may also be adversely affected by exposure to and deposition
13 from PM_{2.5}. Visibility impairment, which is caused by light scattering and absorption by suspended
14 particles and gases, is also a non-climate welfare effect of BC. In addition, deposition of PM is
15 associated with damages to materials and buildings.

16 Ecological effects of PM include direct effects to metabolic processes of plant foliage (Naidoo
17 and Chirkoot, 2004; Kuki et al. 2008); contribution to total metal loading resulting in alteration of soil
18 biogeochemistry (Burt et al. 2003; Ramos et al. 1994; Watmough et al. 2004), plant growth (Audet and
19 Charest, 2007; Kucera et al. 2008; Strydom et al. 2006) and animal growth and reproduction (Gomot-de
20 Vaufleury and Kerhoas, 2000; Regoli et al. 2006); and contribution to total organics loading resulting in
21 bioaccumulation and biomagnification across trophic levels (Notten et al. 2005).

22 Crop yields can be sensitive to the amount of sunlight received. As discussed in detail in Chapter
23 2.6.3, BC and other airborne particles contribute to surface dimming, and crop losses have been
24 attributed to increased airborne particle concentrations in some areas of the world (Chameides et al.
25 1999). Auffhammer et al. (2006) found that fossil fuel and biomass burning contributes to reduced rice
26 harvests in India. Decreases in rice and winter wheat yields have also been attributed to regional scale
27 air pollution in China (Chameides, et al. 1999).

28 Building materials (metals, stones, cements, and paints) undergo natural weathering processes
29 from exposure to environmental elements (wind, moisture, temperature fluctuations, sunlight, etc.).
30 Deposition of PM is associated with both physical damage (materials damage effects) and impaired
31 aesthetic qualities (soiling effects) for building materials. Wet and dry deposition of PM can physically
32 affect materials, adding to the effects of natural weathering processes, by potentially promoting or
33 accelerating corrosion of metals, by degrading paints and by deteriorating building materials (Haynie,
34 1986; Nazaroff and Cass, 1991). Fine particles may coat building materials, damaging the appearance of
35 homes, public buildings, and historic landmarks (Hamilton and Mansfield, 1991). Studies have been
36 conducted by a number of authors identifying the anthropogenic sources of soiling and materials
37 damages to monuments and historical buildings (Sabboni 1991 and 1995, Ghedini 2003, Bonazza 2006).
38 For example, Bonazza evaluated deposition to the London Tower and found that “deposition of

1 elemental carbon darkens surfaces and has importantly aesthetic implications for buildings.” Reduction
2 of PM deposition is beneficial in terms of reduced cleaning, maintenance, and restoration expenditures
3 for buildings and structures.

4 **3.4.1 Visibility**

5 Particles are the dominant air pollutant responsible for visibility impairment, e.g. “haze”, in both
6 urban and remote areas. In the same way that particles influence the Earth’s radiative balance, by
7 scattering and/or absorbing solar radiation, they influence the quantity of light received by the human
8 eye and, therefore, one’s ability to see long distances. Aerosol-based light extinction can be estimated
9 using the IMPROVE algorithm that multiplies the ambient concentration of PM components by typical
10 component-specific light extinction efficiencies (see <http://vista.cira.colostate.edu/improve/>).¹ BC and
11 crustal minerals are the only included components that contribute to light absorption. Under low
12 humidity conditions, BC and OC have the greatest effect on visibility among the major PM species. Per
13 unit mass, the algorithm specifies that BC is 2.5 times more effective at absorbing light than organic
14 carbon is at scattering.

15 Carbonaceous PM is responsible for a large fraction of regional haze, particularly in the
16 Northwest, where annual average concentrations for 2000-2004 accounts for 40-60% of the aerosol-
17 based light extinction. Most of this average carbonaceous visibility impairment throughout the US is
18 associated with OC (in both rural and urban areas) because of relatively high OC concentrations
19 compared to BC. Regional haze in the eastern US generally contains even higher concentrations of
20 carbonaceous PM and light-absorbing BC plays a relatively larger but still minor role compared to OC
21 (Debell, 2006).

22 As described in Chapter 5, urban areas have more carbonaceous PM than nearby remote (rural)
23 areas in the same region (U.S. EPA 2003, U.S. EPA 2004). Western urban areas have more than twice the
24 average concentrations of carbonaceous PM than remote areas sites in the same region (Debell, 2006).
25 As shown in Figure 5-4, average urban PM_{2.5} is composed of roughly equal proportions of carbonaceous
26 and sulfate components in some eastern areas. At the high relative humidity common in the eastern
27 US, hydrated sulfate dominates as the constituent responsible for most urban haze on the haziest
28 summer-time days (U.S. EPA 2009).

29 The 1977 Clean Air Act Amendments called for the development of regulations to address
30 regional haze (visibility impairment) in 156 National Parks and wilderness areas in the United States. The
31 EPA promulgated a Regional Haze Rule (RHR) in 1999 in response to this CAA mandate. Implementation
32 of the RHR entails planned emissions reductions to ensure that by 2064, the worst haze days in these
33 protected areas will improve to natural conditions without degrading visibility conditions for the best
34 haze days. In addition to the RHR aimed at achieving visibility improvements in protected National Park

¹ For two major PM_{2.5} components, sulfate and nitrate, water growth factors are included to account for enhanced light extinction due to relative humidity.

1 areas, the NAAQS program has been successful at achieving visibility improvements in rural areas, as
2 well as in urban areas where people live and work.

3 **3.5 Valuation Techniques for Assessing Air Pollutant Impacts of BC**

4 EPA routinely calculates the health and non-climate welfare benefits associated with reductions
5 in PM_{2.5} and other pollutants. In conducting such analyses, EPA utilizes a “damage-function” approach
6 to calculate the total benefits of the modeled changes in environmental quality. This approach
7 estimates changes in individual health and welfare endpoints (specific effects that can be associated
8 with changes in air quality) and assigns values to those changes assuming independence of the
9 individual values. Total benefits are calculated simply as the sum of the values for all non-overlapping
10 health and welfare endpoints. The “damage-function” approach is the standard method for assessing
11 costs and benefits of environmental quality programs and has been used in several recent published
12 analyses (Levy et al., 2009; Hubbell et al., 2009; Tagaris et al., 2009).

13 EPA uses the Value of a Statistical Life (VSL) and other economic indicators to place a monetary
14 value on the economic benefits of avoided mortalities due to of US air quality regulations. The VSL is
15 determined by studies of individuals’ willingness to pay (WTP) for avoided mortality health
16 improvements associated with reductions in air pollution. Similar analyses using global air quality
17 models and country specific economic data could provide useful data on which the worldwide
18 community could base decisions. However, uncertainties and ethical concerns surround attempts to
19 apply valuation statistics to global health impact assessments. For example, inequities in salaries
20 between developed and developing countries may result in vastly different reported WTP and VSLs
21 around the world, causing premature deaths in developing countries to be valued lower than the same
22 number of premature deaths in developed countries. Issues such as these must be addressed when
23 placing a monetary value on benefits of BC reductions internationally (Casper 2008).

24 Visibility benefits from PM reductions are a non-climate welfare effect that EPA has had some
25 success in quantifying and valuing. Visibility directly affects people’s enjoyment in a variety of daily
26 activities and their overall sense of wellbeing. Individuals value visibility both in the places they live and
27 work, in the places they travel to for recreational purposes, and at sites of unique public value, such as
28 the Great Smokey Mountains National Park. Economic benefits are believed to consist of use values and
29 nonuse values. Use values include the aesthetic benefits of better visibility, improved road and air
30 safety, and enhanced recreation in activities like hunting and bird watching. Nonuse values are based on
31 people’s beliefs that the environment ought to exist free of human-induced haze. Nonuse values may be
32 more important for recreational areas, particularly national parks and monuments.

(This page intentionally left blank)

4. Emissions of Black Carbon

4.1 Summary of Key Messages

- Emissions of BC from U.S. sources total about 0.65 million tons (580 Gg) in 2005, which represents about 8% of the global total. Mobile sources account for a little more than half (52%) of the domestic BC emissions. Nearly 90% of the mobile source total is from diesel sources. Open biomass burning is the next largest sources in the United States, accounting for about 35% of the total. In general, BC is concentrated in urban areas, where populations are largest, making health an important issue in addition to climate in BC mitigation strategies.
- OC is a significant co-emitted pollutant among the major BC emitting sources. The United States is estimated to emit about 1.7 million tons (1905 Gg) of OC. The ratio and mass of BC and OC varies by source. Diesel combustion emissions produce the largest fraction of BC while emissions from open biomass burning are dominated by OC. More research is needed on how OC/BC ratios can be used to characterize the net climate impacts of different sources.
- Diesel sources have a low OC/BC ratio, making them strong candidates for mitigation. By 2030, domestic diesel emissions will be reduced by the phase-in of recent national mobile source emission standards, and other categories, such as open biomass burning, will emerge as top emitters of BC in the United States.
- More than two-thirds of the almost 8 million tons (7300 Gg) in global BC emissions come from open biomass burning and residential sources. The regions of the world responsible for the majority (nearly 75%) of BC emissions world-wide are Africa, Asia, and Latin America. In developing countries, biomass burning and residential sources are the dominant sources of BC, while in developed countries, emissions of BC are lower and are often dominated by transportation and industry.
- Long-term historic trends of BC in the United States reveal a dramatic increase in emissions from contained combustion sources from the mid 1880s to approximately the 1920s followed by a decline over the next 8 decades. The decline can be attributed to changes in fuel use, more efficient combustion of coal, and implementation of PM controls. In contrast, developing countries (e.g., China and India) have shown a very sharp rise in BC emissions over the past 50 years.
- Characterization of domestic and global BC emissions and the subsequent development of BC emission inventories are based on a limited number of existing source measurements. Better information is needed on chemical composition of PM for some critical emission sources to improve estimates of BC in the inventories.

1 **4.2 Introduction**

2 Emissions inventories provide valuable information about major sources of BC both domestically
3 and internationally, and the trends in BC over time. This chapter covers domestic and global emissions
4 of BC and OC. In the case of domestic emissions, the discussion begins with source measurements that
5 generate speciated emission profiles and ends with a description of the current U.S. emissions inventory
6 for BC and OC by source category, with particular attention to mobile sources, open biomass burning,
7 and fossil fuel combustion¹. This chapter also provides an overview of key emissions estimates from
8 available global inventories as well as inventories for key world regions such as China and India, and
9 evaluates historical trends in global emissions. This chapter includes a comparison of the US portion of
10 the global BC inventory to the EPA developed estimates. In addition, this chapter discusses the
11 implications of long-range transport of aerosols, which contributes to total BC in the column of air above
12 an area. Based on the discussion in this chapter, key emission research needs for BC and other light
13 absorbing aerosols are incorporated into the recommendations discussed in Chapter 12.

14 **4.3 U.S. Black Carbon Emissions**

15 16 **4.3.1 Summary of Emissions Methodology**

17 Currently, the U.S. EPA does not require the states to report emissions of BC and other PM
18 constituents (OC, nitrates, sulfates, crustal material) as part of the National Emissions Inventory (NEI).
19 Rather, the U.S. emissions inventory uses total PM_{2.5} emissions to derive estimates for direct emissions
20 of carbonaceous particles, including BC and OC, for all sources except on-road mobile sources.
21 Therefore, all of the available emissions inventory information on carbon emissions in the United States
22 is restricted to those source categories with sufficient PM_{2.5} emissions estimates to support this
23 derivation. The methods used to generate U.S. emissions inventories are described in detail in Appendix
24 2.

25 In general, EPA estimates emissions of BC and OC by appropriately matching PM_{2.5} emission
26 estimates from EPA's NEI with source profiles contained in EPA's SPECIATE database (see Appendix 4 for
27 details). SPECIATE is the EPA's repository of particulate matter (PM) speciation profiles of air pollution
28 sources. The speciation profiles contain weight fractions of chemical species (e.g., BC and OC) of PM for
29 specific sources. Applying these profiles to PM emission inventories provides estimates of how much BC
30 and OC is emitted by specific source categories. There are about 300 profiles in the SPECIATE database
31 that are of sufficient quality for this purpose. The mapping of how these ~300 profiles to the over 3,400
32 source categories available in EPA's NEI for PM_{2.5} is described in Appendix 2 and more details are
33 available in the literature (Reff 2009, Simon 2010). For all non-mobile source and non-open biomass

¹ Most estimates of source emissions in the United States utilize thermal optical methods which estimate BC as elemental carbon (EC). However, for purposes of this chapter, all emissions estimates will be referred to as BC. This issue is addressed for ambient measurements in Chapter 5 and covered in more detail in Appendix 1.

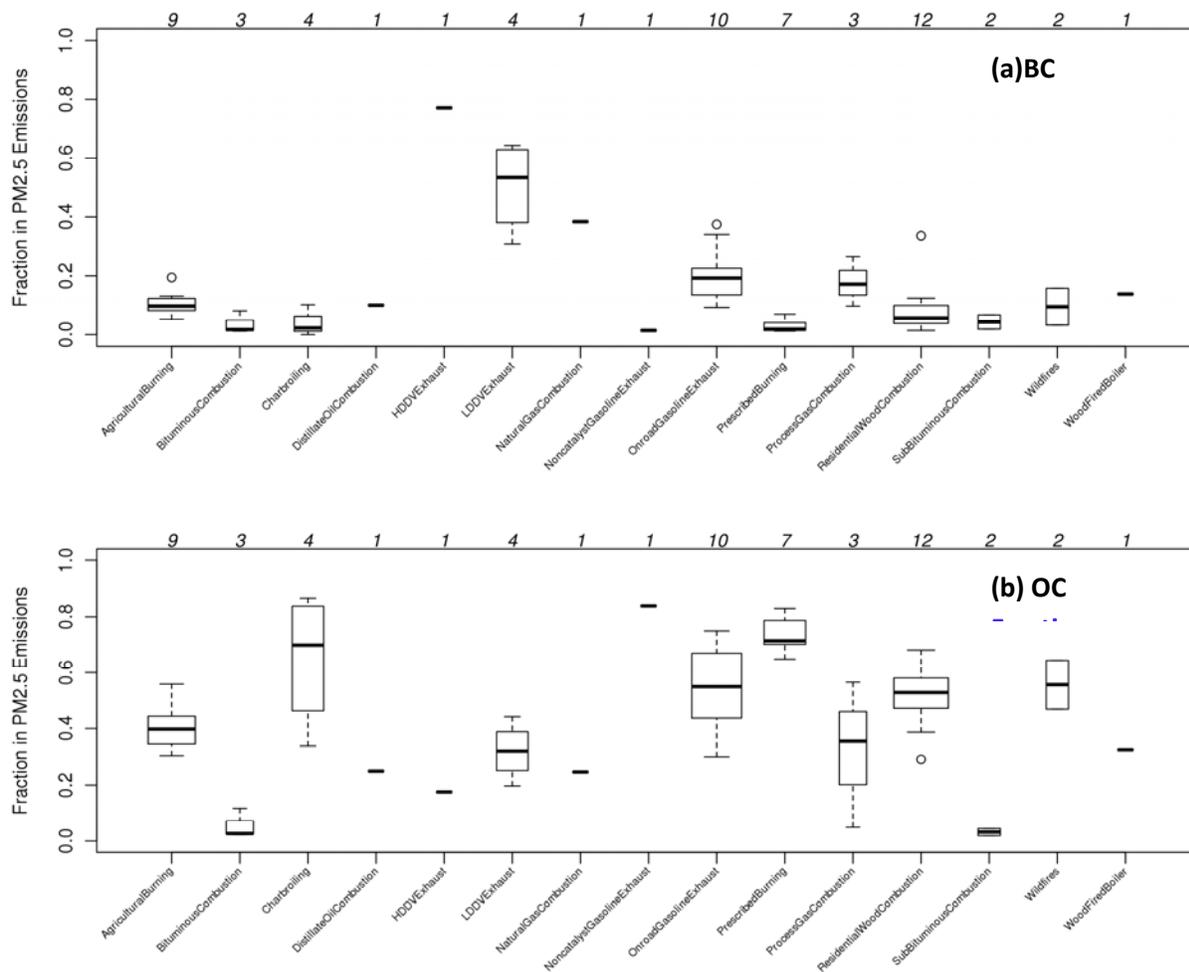
1 emission estimates, all BC and OC estimates are based on EPA's 2005 modeling inventories (termed
2 "2005CK" inventories), which rely on the 2005 NEI for PM_{2.5}.

3 As noted above, for on-road mobile source categories (e.g., cars and trucks), BC is predicted
4 directly without using SPECIATE. For on-road gasoline and diesel vehicles, emissions estimates are
5 generated directly through models. Appendix 2 provides details on how these emissions were
6 calculated using EPA emissions models for on-road and nonroad vehicles/engines, and also discusses
7 other important issues like high emitters, deterioration of PM emissions (i.e., increase in PM mass) with
8 age, and increased PM emissions at lower temperatures. All three of these issues are important and
9 available data on them are incorporated into EPA's emissions models. There are more data on these
10 issues for gasoline PM than for diesel PM.

11 PM_{2.5} emissions from open biomass burning (wildfires, agricultural burning, and prescribed
12 burning) come from an emissions inventory compiled by the Regional Planning Organizations (RPOs) in
13 2002 (RPO, 2002). There are five RPOs in the United States which are set-up to address regional haze
14 and related issues across the country. Due to the need to accurately represent local/regional fire
15 emissions, each RPO has invested time in including greater regional/local specificity resulting in
16 development of more accurate fire inventories, thereby making them more accurate than national
17 estimates developed by EPA. In addition, these RPO estimates have received more wide spread review
18 and acceptance by the states, RPOs and other federal agencies. Though these emission estimates are
19 for the year 2002, the difference between the year of estimates matters less than the accuracy and
20 review of the estimates because there is very little year to year variation in all categories except for
21 wildfires. In the case of wildfires, these 2002 estimates are consistent with an average of wildfire
22 activity over a ten year period from 2001 to 2010. BC and OC emissions were then estimated based on
23 these PM_{2.5} estimates using the same methodology explained above.

24 It is also important to note that the BC and OC inventories do not account for secondary
25 formation of particles in the atmosphere. While not significant for BC, a significant amount of OC can be
26 formed in the atmosphere from biogenic and anthropogenic emissions of volatile organic chemicals.
27 Most air quality and climate models rely on estimates of OM (which is OC plus the mass that accrues to
28 primary OC through photochemistry in the atmosphere), rather than OC, to calculate atmospheric
29 reactions and impacts.

30
31
32
33



1

2

3

4

5

6

7

Figure 4-1. (a) BC and (b) OC Fractions of PM_{2.5} Emissions for the Top 15 Black Carbon Emitting Source Categories in the United States. The box represents the 25th to 75th percentile range and the whiskers represent the 10th and 90th percentile points of the emissions source test data as it exists in EPA's SPECIATE database. The horizontal lines within the box represent the average values (median) for that source category.

8

9

10

11

12

13

14

15

Figure 4-1 displays the number of resulting profiles (the numbers on the top of the graphs) and their distribution of BC and OC fractions of PM_{2.5} by source category. The number of individual profiles by source category can be quite limited---sometimes only a single value is known. Figure 4-1(a) reveals that heavy duty diesel vehicles have the largest fraction of PM_{2.5} that is BC (about 77%). This fact is supported by the EPA's diesel health assessment document from 2002 in which the chemical composition of diesel engine exhaust is identified as that shown in Figure 4-2, with black carbon contributing 75% of the total PM_{2.5} composition. However, figure 4-1(a) shows that light-duty gasoline vehicles have a much smaller fraction (about 20-25%) of PM that is black carbon.

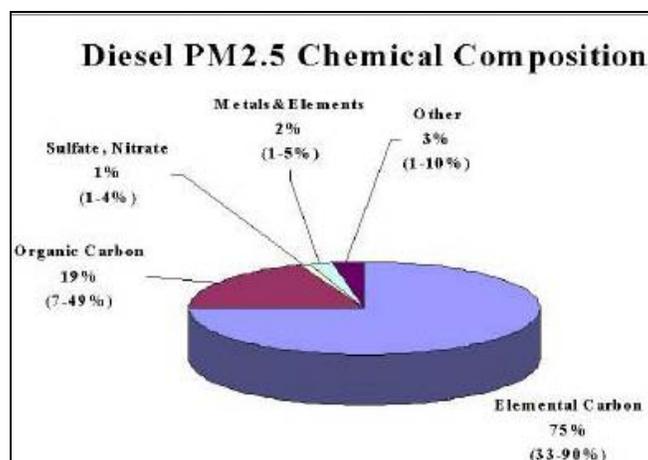


Figure 4-2. Heavy-Duty Diesel PM_{2.5} Emissions Profile. (Diesel Health Assessment Document, 2002)

4.3.2 U.S. Black Carbon Emissions: Overview and by Source Category

In 2005², the United States is estimated to have emitted about 5.5 million tons³ (or about 5,000 Gg) of primary PM_{2.5} of which about 0.65 million tons (12%) was BC and about 1.7 million tons (30%) was primary OC⁴. Thus at a national level, more than twice as much OC is emitted from domestic sources as BC. The domestic emissions of 0.65 million tons represents about 8% of the world’s total BC emissions (i.e., 8.4 million tons) making the United States the 7th largest global BC emitter (Lamarque et al., 2010). The majority of U.S. BC emissions come from mobile sources (predominantly diesel) and open biomass burning. In 2005, about 65% of total U.S. BC was emitted in urban counties and, in the case of mobile sources, more than 70% of the total U.S. BC emissions occur in urban counties.

Figure 4-3 displays the percentage of total U.S. emissions of primary PM_{2.5}, BC, and OC for six “mega” source categories:

- “Open biomass burning” (agricultural burning, wildfires, and prescribed burning)
- “Residential” (any combustion for residential activities regardless of fuel burned)
- “Energy/power” (EGUs and other power generation sources)
- “Industrial”
- “Mobile sources” (includes all on-road, nonroad, tire wear, and brake wear) and
- “Other”

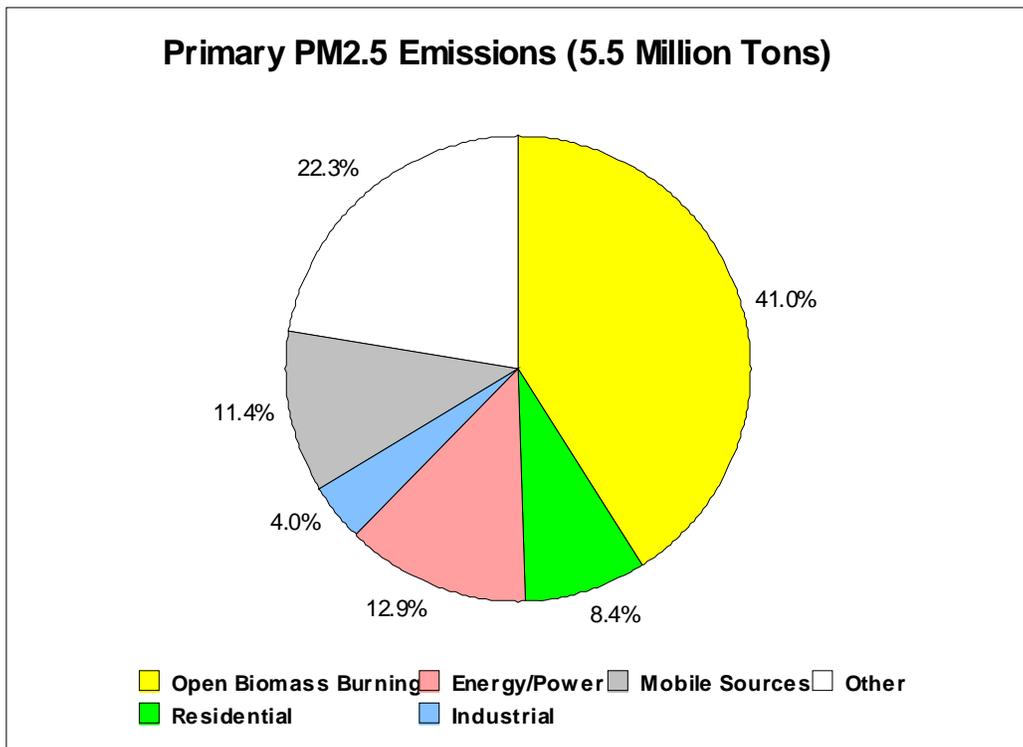
² Most U.S. emissions presented in this chapter are for the year 2005. However, all emissions estimates for “open biomass burning” categories (wildfires, agricultural burning, and prescribed burning) are based on a 2002 inventory developed by the five Regional Planning Organizations (RPOs) across the United States, who are partially funded by EPA. More details are provided later as to why these estimates are used in lieu of EPA estimates.

³ Unless otherwise specified, the term “tons” refers to short tons throughout this report. 1102 short tons = 1 Gigagram

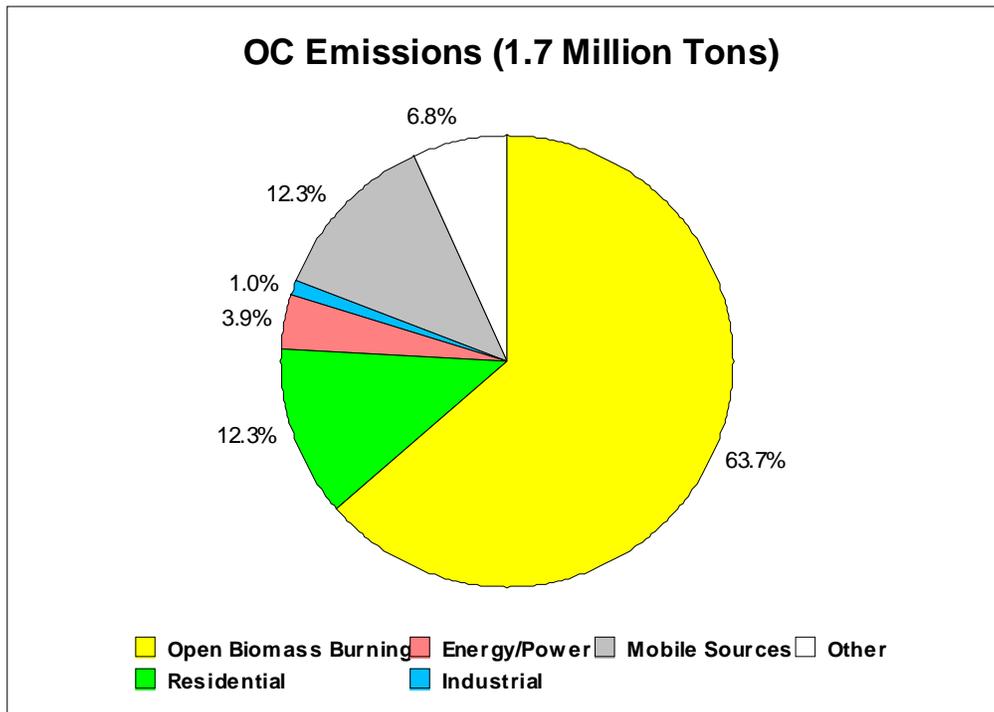
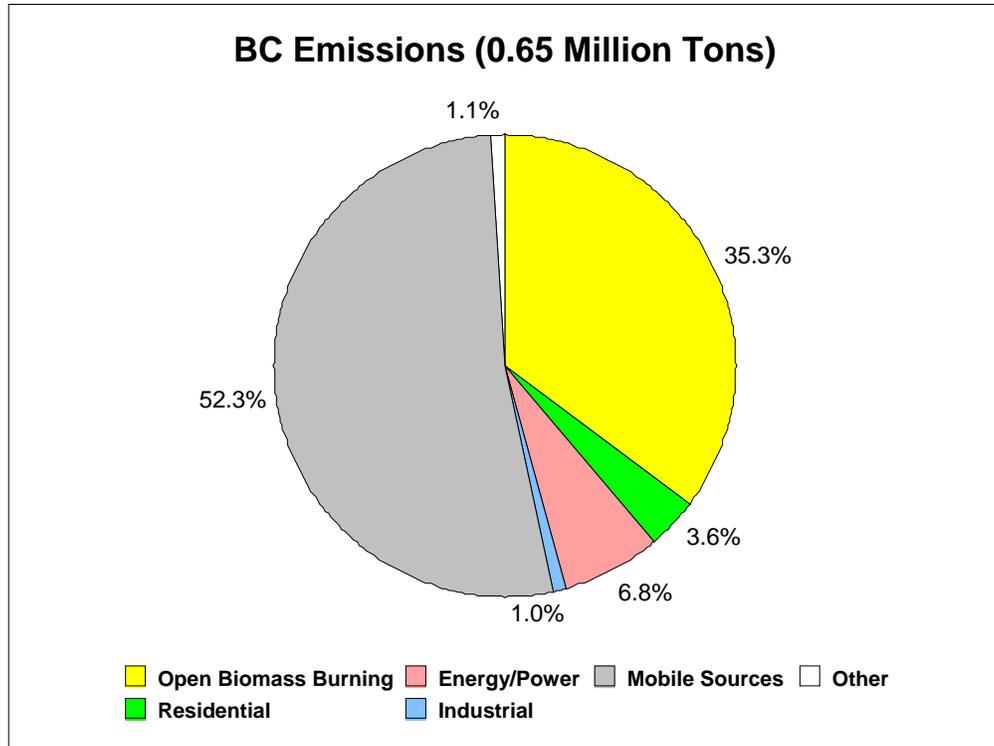
⁴ This does not account for other components of organic PM emissions, such as oxygen and hydrogen

1 Table 4-1 shows the actual tons per year of BC, OC, and direct PM_{2.5} for these source categories, as well
2 as some key emissions ratios. In the last row, emissions in Gigagrams (Gg) are shown in parenthesis,
3 since Gg are standard units for reporting global emissions.

4 Figure 4-3 clearly shows mobile sources are the dominant contributor to total BC emissions in
5 the United States in 2005. Mobile sources contribute 52% of the total BC emissions, followed by open
6 biomass burning (35%), and energy/power (7%). All other categories are about 4% or less. Additional
7 detail on the specific sources that comprise these mega source categories is provided later in this
8 section.



9



3 **Figure 4-3. Primary PM_{2.5}, EC, and OC Emission Contribution by Mega Source Categories.**

4

1

Table 4--1. 2005 Emissions and Ratios of Emissions by Mega Source Category.

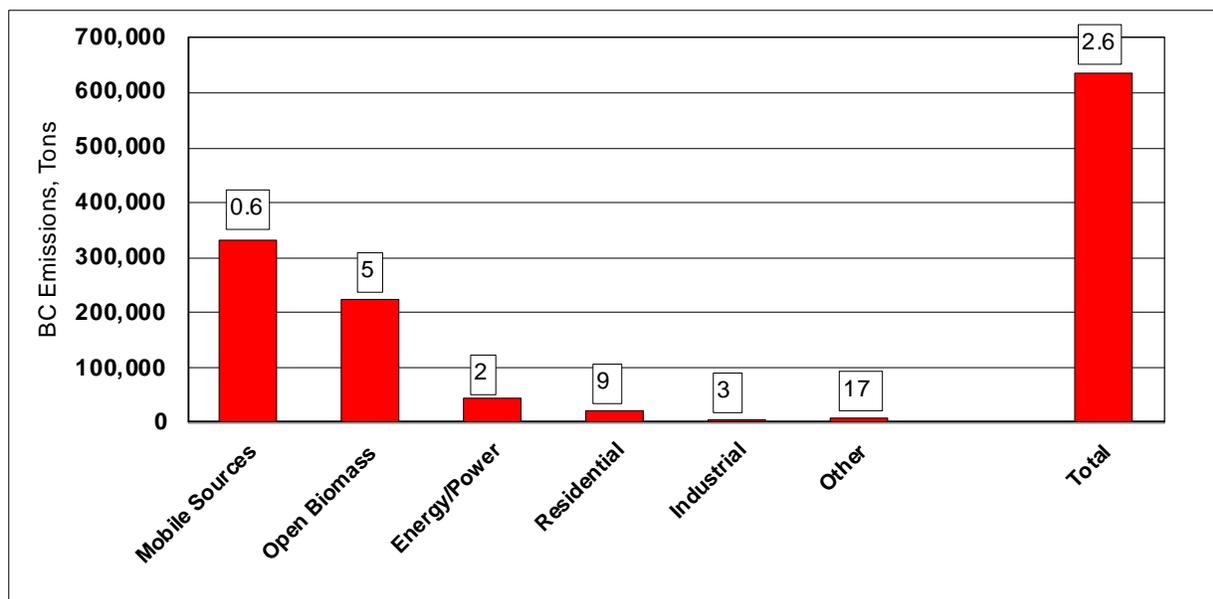
Mega ^a Source Category	PM2.5	BC	OC		OC/BC	BC/PM2.5
Open Biomass Burning	2,266,513	224,608	1,058,494		4.7	0.10
Residential	464,063	22,807	204,160		9.0	0.05
Energy/Power	712,438	43,524	65,138		1.5	0.06
Industrial	219,460	6,085	16,234		2.7	0.03
Mobile Sources	626,859	333,405	205,171		0.6	0.53
Other	1,232,123	6,743	112,967		16.8	0.01
Totals (Short Tons)	5,521,456	637,172	1,662,164		2.61	0.12
Gigagrams (Gg)	5,009	578	1,508			

2

3

4 As shown by the ratios in Table 4-1 (OC/BC and BC/PM_{2.5}), the composition of primary PM_{2.5}
 5 emissions varies significantly among source categories. As discussed in Chapter 2, such differences have
 6 important implications for climate. For example, diesel-powered mobile sources emit significantly more
 7 BC than OC, while the opposite is true for open biomass burning and residential sources. Figure 4-4
 8 displays the OC/BC ratios for the various source categories along with the total BC emissions for the
 9 different source categories. The data in Table 4-1 also show that for some source categories, BC and OC
 10 together make up less than 50% of total PM_{2.5} emissions, indicating that there are significant amounts of
 11 other/unidentified primary co-pollutants (such as direct emissions of nitrates and sulfates) in the
 12 emissions mixture.

13



14

Figure 4-4. BC Emissions (tons) and OC/BC Ratios for Major Source Categories (2005, NEI).

15

16

1

2 The mega source categories can be subdivided into more specific categories. Table 4-2 shows
3 the national-level emissions of primary PM_{2.5}, BC, and OC emissions for about 90 specific sub-categories
4 of sources in the United States. Table 4-2 also shows OC/BC and BC/PM_{2.5} ratios for each of the specific
5 source categories. Some of this data is drawn from the National Emissions Inventory (NEI), EPA's
6 "bottom-up" compilation of estimates of air pollutants discharged on an annual basis and their sources
7 (EPA, 2005). As discussed previously, the "open biomass burning" categories shown in yellow come
8 from an emissions inventory compiled by the Regional Planning Organizations (RPOs) in 2002 (RPO
9 references).

1 **Table 4-2. U.S. Emissions of PM2.5, BC, and OC (short tons).**

Mega Source Category	Specific Category	Total Primary PM2.5	BC	Primary OC	OC/BC	BC/PM2.5	
Open Biomass	Wildfires	1,600,358	151,855	738,997	5	0.09	
	Prescribed Burning	535,627	58,525	268,826	5	0.11	
	Agricultural Burning	130,528	14,228	50,671	4	0.11	
Residential	Residential Wood Combustion	379,878	21,194	200,645	9	0.06	
	Residual Oil Combustion	78,672	787	787	1	0.01	
	Residential Coal Combustion	2,648	634	1,187	2	0.24	
	Residential Natural Gas Combustion	2,865	192	1,541	8	0.07	
Energy/Power	Natural Gas Combustion	64,239	24,668	15,867	1	0.38	
	Bituminous Combustion	394,853	6,697	10,387	2	0.02	
	Sub-Bituminous Combustion	143,383	6,028	4,514	1	0.04	
	Distillate Oil Combustion	23,718	2,372	5,930	3	0.1	
	Wood Fired Boiler	56,289	2,088	19,764	9	0.04	
	Process Gas Combustion	9,457	1,378	2,850	2	0.15	
	PMSO2 Controlled Lignite Combustion	20,499	293	5,826	20	0.01	
Industrial	Stationary Diesel	4,476	3,452	786	0.2	0.77	
	Cement Production	17,523	514	2,221	4.3	0.03	
	Ind Manuf - Avg.	46,501	416	3,422	8.2	0.01	
	Mineral Products - Avg	23,632	347	1,242	3.6	0.01	
	Kraft Recovery Furnace	21,222	325	1,111	3.4	0.02	
	Chem Manuf - Avg	17,526	320	1,608	5	0.02	
	Lime Kiln	7,002	162	466	2.9	0.02	
	Heat Treating	14,439	144	1,011	7	0.01	
	Aluminum Production	5,730	132	223	1.7	0.02	
	Ferromanganese Furnace	1,240	125	64	0.5	0.1	
	Surface Coating	9,165	64	1,903	29.7	0.01	
	Cast Iron Cupola	3,479	32	222	6.9	0.01	
	Electric Arc Furnace	4,317	16	140	8.8	0	
	Secondary Aluminum	6,057	12	91	7.6	0	
	Sintering Furnace	5,739	10	157	15.7	0	
	Pulp & Paper -Avg.	6,569	7			0	
	Catalytic Cracking	8,864	6	1	0.2	0	
	Secondary Copper	1,137	1	11	11	0	
	Ammonium Nitrate Production	1,025				0	
	Secondary Lead	410				0	
	Petroleum Ind - Avg	6,224		218		0	
	Copper Production	432				0	
	Ammonium Sulfate Production	65				0	
	Open Hearth Furnace	6,686		1,337		0	
	Mobile Sources	On-road diesel	208,473	153,477	44,423	0.3	0.74
		Nonroad diesel	145,289	112,058	30,618	0.3	0.77
		Locomotive	30,910	22,495	5,130	0.2	0.73
Commercial Marine (C1 & C2)		28,119	21,652	4,937	0.2	0.77	
On-road gasoline		75,924	14,510	59,657	4.1	0.19	
Nonroad gasoline		55,834	5,444	46,734	8.6	0.1	
Commercial Marine (C3)		56,028	1,681	6,303	3.7	0.03	
Tire		5,325	1,198	3,060	2.6	0.22	
Brakewear		17,801	475	2,321	4.9	0.03	
Aircraft		3,156	410	1,988	4.8	0.13	
Other		Charbroiling	64,124	2,601	42,975	16.5	0.04
	Wood Products - Drying	8,113	649	4,057	6.3	0.08	
	Paved Road Dust	54,481	569	5,308	9.3	0.01	
	Dairy Soil	9,862	509	3,139	6.2	0.05	
	Wood Products-Sawing	12,355	469	5,498	11.7	0.04	
	Overall Average Manufacturing	10,577	466	927	2	0.04	
	Unpaved Road Dust	419,648	409	22,897	56	0	
	Charcoal Manufacturing	5,578	290	100	0.3	0.05	
	Solid Waste Combustion	14,965	228	1,258	5.5	0.02	
	Wood Products - Sanding	2,257	135	790	5.9	0.06	
	Asphalt Manufacturing	2,160	124	93	0.8	0.06	
	Fiberglass Manufacturing	4,641	93	1,299	14	0.02	
	Agricultural Soil	334,515	67	10,310	153.9	0	
	Fly Ash	1,733	30	21	0.7	0.02	
	Phosphate Manufacturing	992	27	78	2.9	0.03	
	Industrial Soil	2,011	23	20	0.9	0.01	
	Food & Ag - Handling	10,331	18	418	23.2	0	
	Urea Fertilizer	589	12	183	15.3	0.02	
	Potato Deep-Frying	192	8	121	15.1	0.04	
	Glass Furnace	7,803	5	55	11	0	
	Calcium Carbide Furnace	314	4	23	5.8	0.01	
	Sludge Combustion	163	2	14	7	0.02	
	Crustal Material	1,160	2	62	31	0	
	Brick Grinding and Screening	1,272	1	31	31	0	
	Auto Body Shredding	129	1	10	10	0.01	
	Inorganic Fertilizer	78	1	2	2	0.01	
	Asphalt Roofing	1,872	0	1,129		0	
	Limestone Dust	1,912				0	
	Sand & Gravel	134,885				0	
	Construction Dust	96,669		4,463		0	
	Meat Frying	12,216		7,012		0	
	Lead Production	33				0	
	Synthetic Residential Wood Combustion	345				0	
	Sandblasting	1,673		8		0	
	Steel Desulfurization	259				0	
	Inorganic Chemical Manufacturing	4,161				0	
	Gypsum Manufacturing	1,395				0	
Food&Ag-Drying	5,551		666		0		
Boric Acid Manufacturing	11				0		
Coke Calcining	811				0		
Sea Salt	287				0		

2
3 NOTES:

- 1) All emissions are for 2005 except those for “open biomass burning,” which are based on 2002 RPO estimates
- 2) This table represents all emissions in column D as BC; however, they were derived from thermal-optical monitoring techniques and reported as Elemental Carbon (EC).
- 3) Aircraft inventories only include emissions from landings and take-offs and does not include in-flight emissions.
- 4) In this table, the mobile source inventories are for all 50 states. Wildfire emissions are for the 48 contiguous states plus Alaska. All other estimates are only for the 48 contiguous states (AK and HI are expected to be minor BC and OC contributors for all these sources).
- 5) BC emissions from “Agricultural Burning” are very dependent on the types of burning activity included (e.g., range land, crop residue, and other types of burning activity). Recent work using satellite-imagery shows the total “agricultural emissions” in the United States (averaged over 5 years) are somewhat lower than the BC emission estimates shown here (McCarty, 2011). McCarty’s estimates for BC emissions from agriculture burning are based on the inclusion of crop residue burning only, a limited definition of “agricultural burning” that others also feel is appropriate. Working with USDA, EPA is in the process of evaluating this work as well as more its own recent work on a 2008 fires inventory that relies on updated remote sensing methods to estimate emissions from agricultural burning.

4.3.2.1 Emissions from Mobile Sources

Mobile sources account for about 52% of total U.S. BC emissions in 2005. Within this category, emissions from diesels (both nonroad and on-road) dominate, accounting for about 92% of BC. Gasoline vehicles/engines are responsible for the remaining 8% of BC emissions from the mobile source category. Figure 4-5 shows this more detailed breakout of mobile source BC emissions. In general, diesel $PM_{2.5}$ consists of about 70-80% BC and about 20% OC.⁵ Gasoline $PM_{2.5}$, in contrast, consists of about 20% BC with the remainder being mostly OC. Diesel PM is thus unique in having a very high ratio of BC to OC. The total light absorbing capacity of the specific compounds and the resultant mixture emitted in diesel or gasoline exhaust is not known. However, a limited number of mobile source measurements suggests that particle emissions from both gasoline and diesel vehicles are strongly light absorbing. (Japar et al., 1984; Strawa et al., 2010; Adler et al., 2010). It should be noted that while mobile sources represent about 52% of the national total of BC emissions, they represent about 69% of all non-wildfire BC emissions in the United States.

⁵ The estimate shown applies to the total diesel PM inventory. However, under low loads (e.g., idle), BC constitutes a smaller fraction of PM emissions (i.e., 20-40%). Emissions in these conditions contribute a relatively small fraction of total PM.

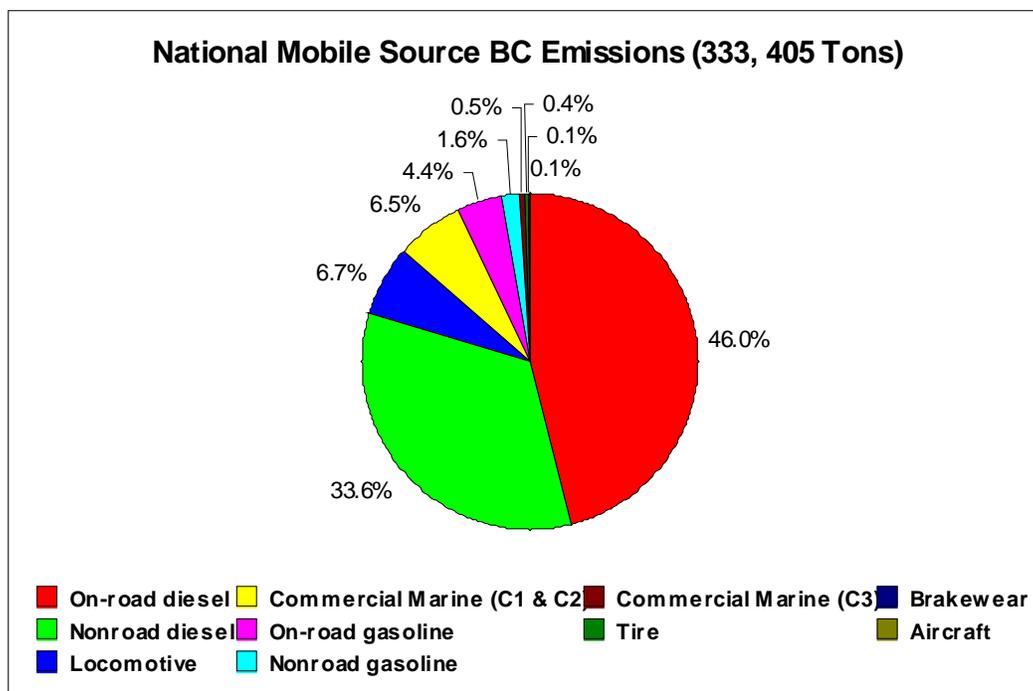
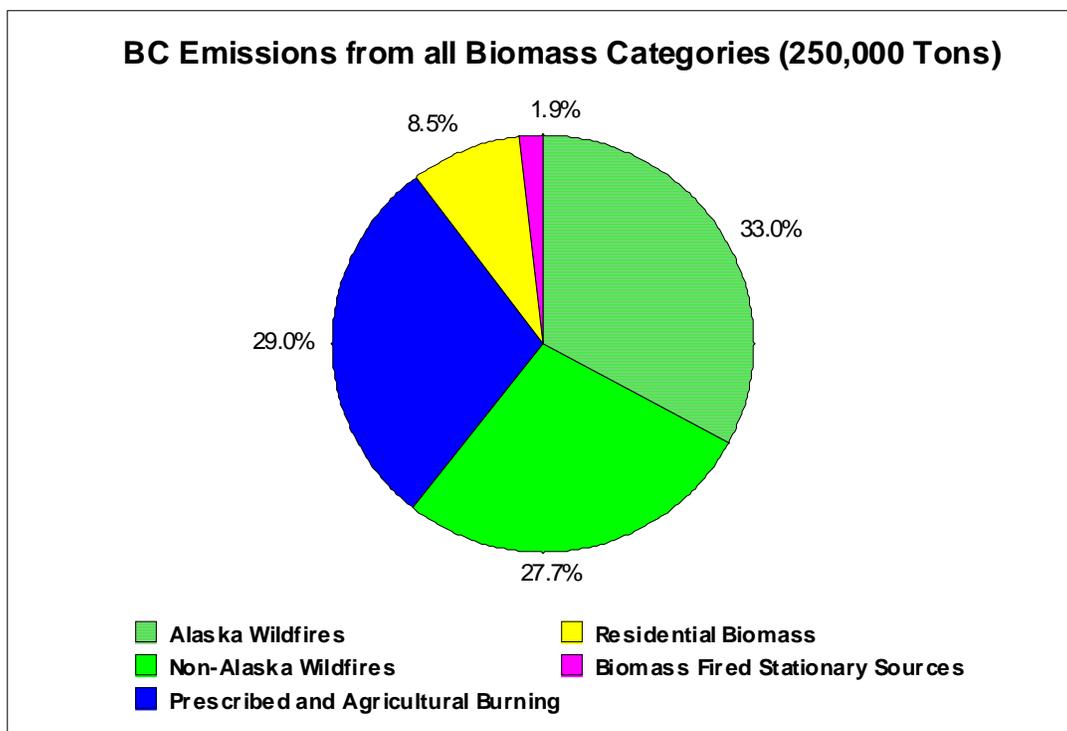


Figure 4-5. National Mobile Source BC Emissions by Detailed Sectors (2005 NEI).

While mobile sources dominate the U.S. inventory currently, significant reductions in emissions of both BC and OC have been achieved since 1990, and existing vehicle regulations are expected to produce further reductions in coming years as they are implemented. Most of these BC reductions are a direct result of EPA’s regulations on diesel PM, but reductions in total carbon emissions, mostly OC, are also due to regulations on emissions from gasoline vehicles. Due to these regulations, the mobile source contribution to BC compared to other sources has declined on both an absolute basis and a fractional basis since 1990. As reductions continue through 2030 and beyond, the pie chart shown in Figure 4-3 will continue to change, showing an increasingly smaller contribution of mobile sources to overall U.S. BC emissions. Chapter 7 summarizes mobile source EC inventories for various years from 1990 through 2030, and the control programs that are expected to result in these emissions reductions by 2030.

4.3.2.2 Emissions from Biomass Combustion

Several source categories in Table 4-2 include emissions from “wood based” (biomass) combustion. Based on an approach suggested by Bond et al. (Bond 2004a, 2007) to facilitate consideration of mitigation options, elements of these source categories: “open biomass burning”, “residential heating/cooking”, and “biomass fired stationary sources” have been combined into a “biomass combustion” category for this discussion. Table 4-3 summarizes the sources categories included this “biomass combustion” category and their associated emissions.



1
2 **Figure 4-6. BC Emissions from all “Biomass Combustion” Source Categories.**

3 Often, global inventories define a broad “Contained Burning” source category that includes all of the
4 sources listed under “Residential” in Table 4-2, the “wood fired boiler” source in the “Energy/Power” mega-
5 category, and “charbroiling” in the “Other” mega category (Bond 2004a, Bond 2007). For the United States,
6 sources within the “Contained Burning” category combined to emit about 27,000 tons of BC representing about
7 11% of the BC emissions and about 20% of the OC emissions from all biomass combustion (open and contained)
8 that occurs in the United States.

9 **4.3.2.3 Emissions from Energy/Power Sector**

10 The energy/power source category contributes approximately 7% of U.S. BC emissions and
11 includes a range of emission categories, as shown in red in Table 4-2. In general, emissions from these
12 sources are split fairly evenly between BC and OC. The largest fossil fuel combustion source of BC
13 emissions according to the 2005 NEI is natural gas combustion; however, estimates of the amount of BC
14 compared to OC in direct PM_{2.5} emissions from this source category are highly uncertain.⁶ The
15 bituminous and sub-bituminous coal categories, both of which primarily represent electricity generating
16 units (EGUs) but may also reflect small contributions from commercial and institutional sources,

⁶ Specifically, EPA applies just one speciation factor to convert direct PM_{2.5} emissions from natural gas combustion sources to estimated EC emissions. This single factor is a BC/PM_{2.5} ratio of 0.38 which leads to a relatively large BC emissions estimate (about 25,000 tons). Though not currently available in the literature, some unofficial source testing has suggested the BC/PM_{2.5} ratio is in the range of 6 to 10% (corresponding to speciation factors of 0.06 to 0.10) indicating that both the combustion process used as well as presence of controls on the unit will affect the amount of BC in PM_{2.5} emissions from this source type. Future work will include further investigation into speciation for this source type.

1 represent relatively small contributions to BC emissions in the United States (a little more than 1%
2 each). This small BC contribution is quite different from these sources' contribution to emissions of
3 long-lived GHGs, where they dominate the inventory (e.g., EGUs account for 40% of CO₂ emissions).

4 5 **4.3.2.4 Emissions from "Other" Source Categories**

6 Table 4-2 shows that the remaining mega categories, "Industrial Sources," and "Other Sources"
7 (in blue and white, respectively), combine to comprise about 2% of total BC emissions domestically. As
8 is explained in more detail in Chapter 8, direct PM_{2.5} emissions from industrial sources in the United
9 States are small compared to emissions of other co-emitted pollutants such as NO_x, HAPs and CO₂. This
10 is the result of effective control technologies for PM emissions on a variety of stationary/industrial
11 sources. One industrial source of potential interest for additional PM controls is stationary source diesel
12 engines (generators, emergency equipment, etc.), which as shown in Table 4-2, has a low OC/BC ratio
13 and contributes more than half of the EC emissions in the "Industrial Sources" category. Existing EPA
14 regulations for new engines in this category will result future BC reductions through the use of diesel
15 particulate filters (DPF) , although these controls will not apply to existing engines . Included in the
16 "other" category are many manufacturing type activities as well as fugitive dust emission sources and
17 charbroiling.

18 **4.4 Global Black Carbon Emissions**

19 Global inventories are important for providing information on the distribution of BC emissions
20 world-wide and for identifying key differences between regions, both in terms of total quantity of
21 emissions and major sources. There are a few global BC inventories available currently, and those from
22 Bond et al. (Bond et al., 2004; Streets 2004a) are the most widely used and referenced. Compiling a
23 global BC inventory is difficult for several reasons: varying emissions among similar sources, varying
24 measurement techniques, different PM size cut points used in the measurements, and the definition of
25 BC itself (as discussed in other parts of this Report) used in the inventories. The most up-to-date of
26 these inventories is for the year 2000 and has been developed to support climate modeling needs in the
27 Intergovernmental Panel on Climate Change's (IPCC) Fifth Assessment report (termed "AR5"). These
28 estimates have been published in the literature (Lamarque, 2010) and form the basis for all the
29 discussion in this section. These estimates effectively serve as "current" year global BC inventories.

30 In general, these global BC inventories are compiled using fuel-consumption data to estimate
31 emissions from particular source categories. A few global inventories are based on a "top-down"
32 concept (Parrish, 2004; Penner 1999) in which emissions are inferred from concentration and ancillary
33 measurements in the ambient air, usually downwind from the source or calculated from generalized
34 emission factors and national or regional activity indicators. Most global inventories, including those of
35 Lamarque, Bond et al. used as the basis for this section are based on "bottom-up" type processes. EPA's

1 inventories are also based on “bottom-up” approaches.⁷ In this method, emissions are measured or
 2 computed directly by concentration, mass flow, and or stream velocity observations at the source or
 3 emissions are calculated (using specific emission factors and activity levels) on a source-by-source or
 4 localized basis. Details on methods used to generate both global and domestic BC/EC emissions can be
 5 found in Appendix 2, including more details on “bottom-up” approaches.

6 Global BC inventories have clear advantages when comparing emissions across world-regions,
 7 countries, and sectors because the methodology used is consistent across the spatial domain. Global
 8 inventories, however, can sometimes overlook important but subtle differences between countries
 9 through reliance on default-type information to estimate emissions where actual data are not available.
 10 Regional or country-specific inventories, on the other hand, generally contain more accurate emissions
 11 information for the domain in question because of the availability of more relevant and more specific
 12 fuel composition data, technology differences in sectors, regulations, emission factors, and activity
 13 levels. In this way, the relative importance of certain sources, especially smaller ones, can be
 14 incorporated with more accuracy into the final emissions estimates. Unfortunately, each regional
 15 inventory tends to employ different methods, making comparisons across regional inventories more
 16 difficult. Ideally, regional inventory information could be combined with global inventories to fill in the
 17 gaps where global inventories are weakest. While that harmonization has not yet fully occurred, the BC
 18 inventories described by Lamarque et al., below, make an attempt to combine some of the information
 19 across global and regional inventories.

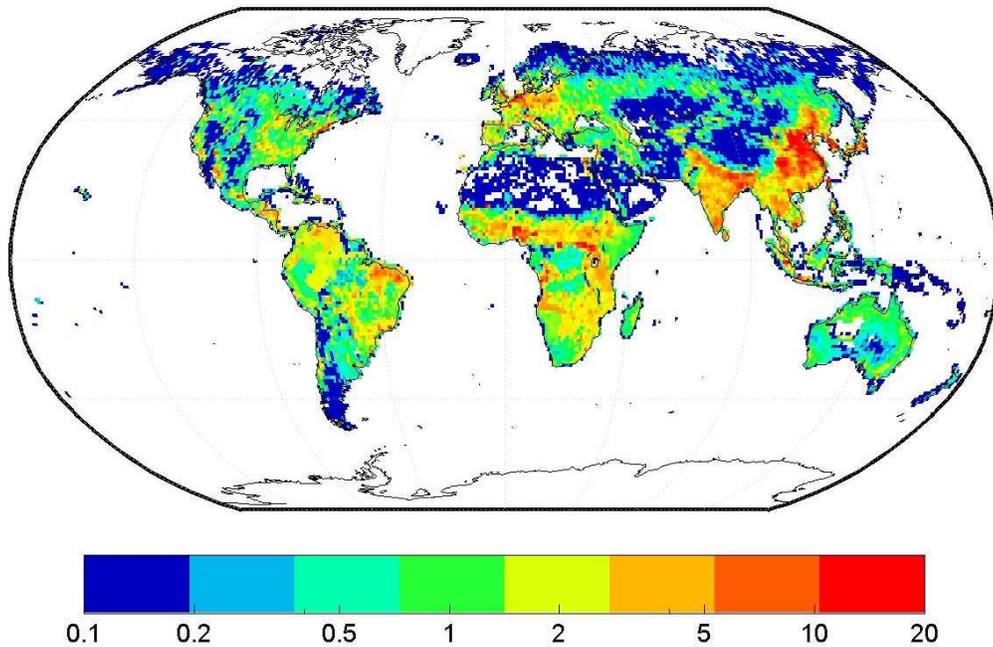
20 This next section provides details on global BC inventories, including the AR5 inventory. It also
 21 explores available regional inventories and compares them to global inventory estimates for the same
 22 regions. The focus of the regional comparisons will be on Asia, where numerous regional efforts are on-
 23 going.

24 **4.4.1 Summary of Global Black Carbon Emissions by Region and Source Category**

25 Total global BC emissions for 2000 are estimated to be about 7,600 gigagrams (about 8.4 million
 26 tons) for 2000. The spatial distribution of these emissions represented in a density map below (Figure 4-
 27 7) shows Asia, parts of Africa, and parts of Latin America (Central and South America) to be among the
 28 regions emitting the largest amounts of BC. Figure 4-8 shows global estimates disaggregated into the
 29 these three major world regions responsible for 75% of worldwide BC emissions: (1) Asia (China, India,
 30 Southeastern Asia, South Asia, Thailand, Asia-“Stan”, Taiwan, Japan, and N. Korea world regions); (2)
 31 Africa (Western Africa, Southern Africa rest of, Eastern Africa, Northern Africa, South Africa world
 32 regions); and (3) Latin America (South America, Mexico, Central America, Argentina, Venezuela, and
 33 Brazil world regions). Asia accounts for about 40% of the global BC emissions, Africa for about 23%, and
 34 Latin America for about 12%, as shown in Figure 4-8 below. Based on these AR5 estimates, domestic

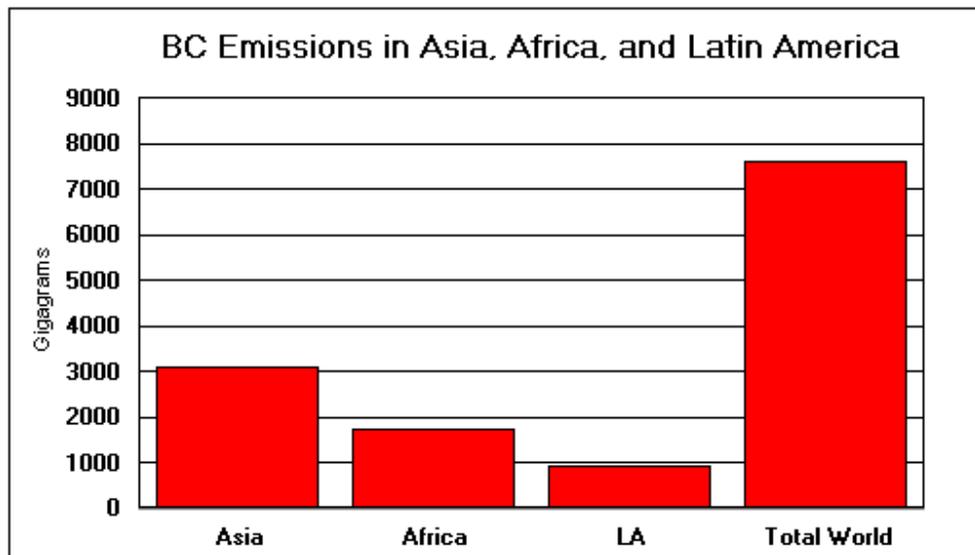
⁷ As an example of how these methods arrive at similar conclusions, EPA’s motor vehicle emissions model (MOVES) accurately predicts national consumption of gasoline and diesel fuels based on vehicle population and activity data. Differences between EPA and global inventories may therefore be related to differences in underlying emission rates per unit activity or fuel consumption.

1 emissions account for approximately 6% of the global total (i.e., the United States is the 7th most
2 significant region in the world in terms of BC contribution).⁸



3
4
5

Figure 4-7. Global BC Emissions based on Year 2000 Estimates (Bond, 2007), scale in Gg



6
7

Figure 4-8. BC Emissions by Selected World Region, 2000 (Gg).

⁸ EPA estimates of U.S. BC emissions are about 49% higher than those from AR5 resulting in an 8% contribution to global BC. These differences are more fully discussed later in this chapter.

1 Table 4-4 displays total global BC emissions by 37 world regions and by 8 major source
2 categories. Similarly, Table 4-5 shows the distribution of the roughly 35,700 Gg (about 39 million tons)
3 in global OC emissions by these same world regions and source categories. The OC emissions from the
4 United States make up about 3% of the global total.⁹

5 The last column in Table 4-4 shows the ratio of BC emissions from each country or region to
6 those estimated for the United States. For example, China (which comprises nearly all the “East Asia”
7 Region) emits 3.5 times as much BC as the United States.¹⁰ Developed world regions like Europe, Japan,
8 and the Middle East have very low BC emissions. In these regions, like in the United States,
9 transportation is the dominant emissions sector. Japan also has significant contributions to BC from
10 industrial sources. In identifying mitigation options, care must be exercised when relying on
11 classifications of world regions/countries as either “developed” or “developing” as surrogates for BC
12 emission intensity or source to determine how “climate-beneficial” controls might be. China, for
13 example, has the fastest growing economy in the world, yet has a developing country’s level of per
14 capita income. While China shares the high BC emissions levels of less developed countries, its sources
15 of BC are not the same as those of less developed areas (biomass burning). A crucial difference
16 between China and other developing areas is its use of coal in residential combustion as well as poorly
17 controlled industry, and apparently a much lower reliance on burning in agriculture than is typical. This
18 makes the contribution to potential warming due to BC emissions greater for China but the mitigation
19 approach different than that which would be required in other developing countries.

⁹ As was the case for domestic BC emissions, EPA estimates OC emissions at about 4% of the global total.

¹⁰ If EPA-based estimates are substituted for the AR5 estimates of U.S. BC emissions the ratio of China to U.S. BC emissions is closer to 2.3.

Table 4-4. Global BC Emissions in 2000 (in Gg)* (Lamarque, 2010).

*Transport includes aircraft and shipping

Region	Energy	Industry	Transport	Residential/Domestic	Ag Waste Burning	Waste	Grassland Fires	Forest Fires	Total	Country/US Ratio
China+ (Rest of)	12	669	72	539	44	7	5	9	1358	3.48
Western Africa	0	20	15	127	8	3	505	105	784	2.01
India	4	108	74	324	4	2	5	15	538	1.38
Brazil	1	53	91	30	3	2	70	215	465	1.19
Southern Africa (Rest of)	0	8	5	68	2	0	373	4	460	1.18
Indonesia+	1	28	34	73	12	1	7	252	407	1.04
USA	3	85	216	55	6	3	9	13	390	1.00
Russia+	5	33	32	102	7	1	35	145	360	0.92
Eastern Africa	0	5	7	119	4	1	210	7	353	0.90
Southeastern Asia (Rest of)	1	30	45	101	3	1	6	166	353	0.90
South America (Rest of)	0	20	34	30	5	1	42	45	178	0.46
Australia	0	11	12	7	4	0	120	20	174	0.45
Western Europe (Rest of)	1	36	88	17	1	1	6	1	150	0.39
Central Europe (Rest of)	3	26	40	54	2	1	2	3	131	0.34
Japan	2	49	61	7	1	1	0	1	123	0.32
South Asia (Rest of)	0	13	30	68	0	1	1	2	116	0.30
Middle East	3	37	62	2	6	1	0	0	111	0.29
South Korea (Republic of Korea)	3	55	36	9	3	1	0	0	106	0.27
Mexico	3	13	36	6	5	1	8	28	99	0.26
Northern Africa	0	11	36	37	1	1	0	0	87	0.22
Central America	0	15	16	12	1	1	2	35	84	0.21
Thailand	0	20	33	12	2	0	3	12	83	0.21
Canada	0	17	19	4	2	0	5	31	78	0.20
France	1	10	48	11	0	0	0	0	71	0.18
Ukraine+	0	14	9	40	5	0	1	1	71	0.18
Argentina	0	12	26	6	7	0	14	4	70	0.18
Germany	1	13	48	5	0	1	0	0	68	0.17
Asia-"Stan"	0	10	2	27	2	0	25	0	67	0.17
South Africa	1	10	14	16	1	0	16	0	58	0.15
United Kingdom	1	10	31	4	0	0	0	0	46	0.12
Italy	2	9	31	2	0	0	1	0	46	0.12
Taiwan	1	18	12	2	0	0	0	0	32	0.08
Venezuela	0	5	7	0	1	0	8	9	30	0.08
Turkey	1	12	10	2	4	0	0	0	30	0.08
North Korea (Democratic Peoples Republic)	0	11	1	16	0	0	0	1	29	0.07
Baltic States (Estonia, Latvia)	0	1	3	11	0	0	0	1	15	0.04
New Zealand	0	1	3	1	0	0	0	0	6	0.01
World Total	54	1497	1340	1947	146	35	1481	1128	7628	

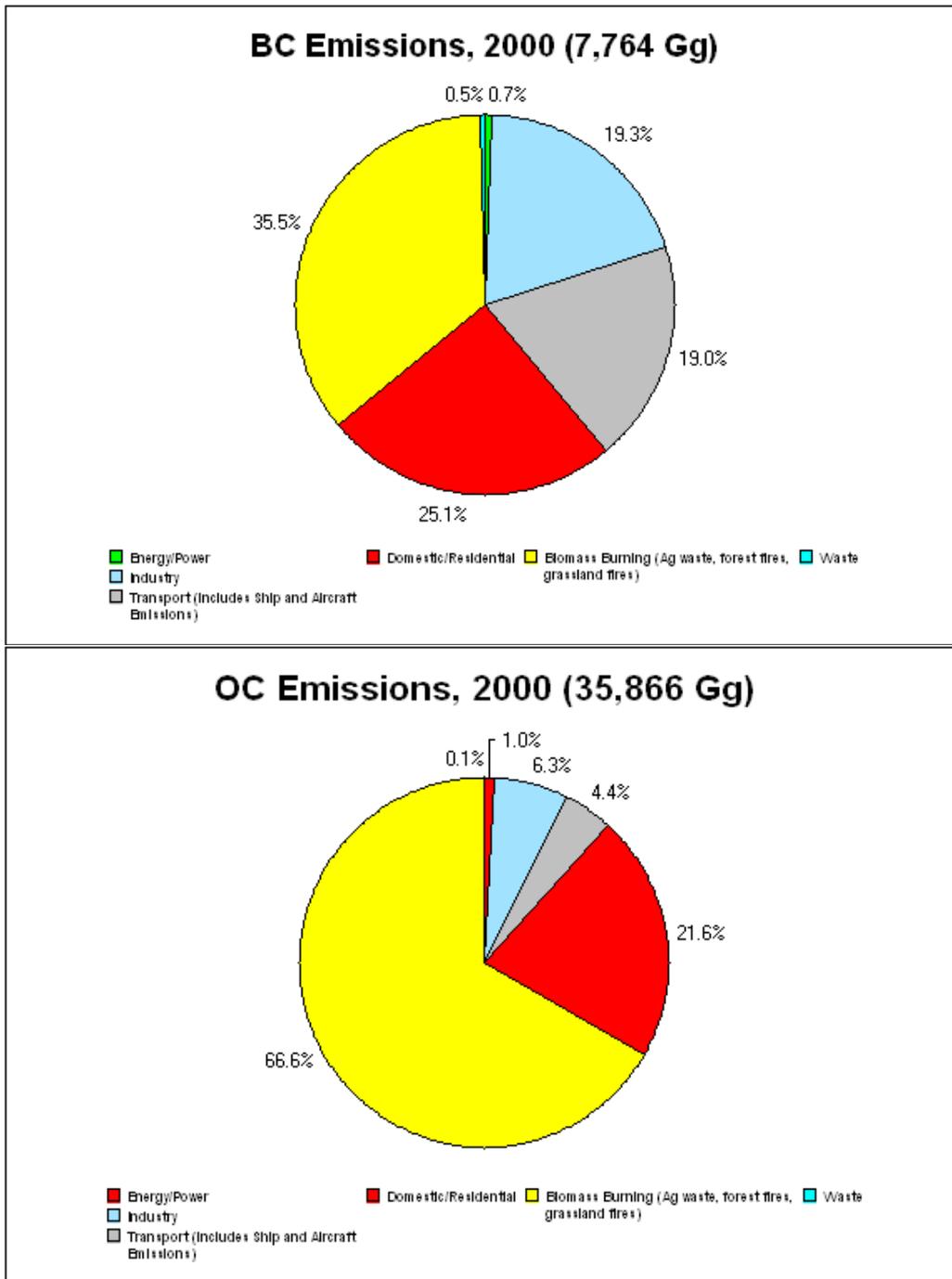
Table 4-5. Global OC Emissions in 2000 (in Gg)* (Lamarque, 2010).

* Transport includes only aircraft. Global OC emission estimates not available for Shipping.

Region	Energy	Industry	Transport	Residential/Domestic	Ag Waste Burning	Waste	Grassland Fires	Forest Fires	Total	Country/US Ratio
Western Africa	1	104	43	538	41	3	3679	882	5291	6.37
Indonesia+	5	34	63	327	57	1	51	3060	3595	4.33
Russia+	25	23	33	550	34	3	338	2582	3588	4.32
China+ (Rest of)	39	877	72	1812	208	7	37	122	3174	3.82
Southern Africa (Rest of)	0	24	9	275	7	0	2732	34	3083	3.71
Brazil	8	203	103	85	14	2	487	1788	2690	3.24
Eastern Africa	0	22	10	525	20	1	1461	56	2095	2.52
Southeastern Asia (Rest of)	8	70	80	428	15	1	41	1405	2048	2.46
India	15	260	63	1301	20	2	38	146	1846	2.22
Australia	3	8	7	27	19	0	836	165	1066	1.28
South America (Rest of)	4	60	54	116	26	1	312	392	966	1.16
USA	72	60	143	198	28	5	97	227	831	1.00
Canada	7	13	14	19	8	0	56	551	669	0.81
Mexico	7	20	107	39	22	1	52	265	513	0.62
Central America	3	49	29	62	7	1	19	294	463	0.56
South Asia (Rest of)	1	45	21	315	1	1	9	27	420	0.50
Central Europe (Rest of)	9	19	25	250	10	3	15	49	380	0.46
Asia-*Stan"	2	6	5	157	11	2	179	3	364	0.44
Ukraine+	2	13	7	224	22	1	6	21	297	0.36
Thailand	4	51	38	38	8	0	19	102	261	0.31
Argentina	2	28	23	8	31	0	103	53	249	0.30
Middle East	27	14	171	5	28	1	2	0	248	0.30
Western Europe (Rest of)	23	29	36	85	5	2	40	10	230	0.28
South Africa	4	17	46	38	6	0	110	0	222	0.27
Venezuela	5	6	40	2	3	0	68	76	199	0.24
Northern Africa	6	10	51	104	5	1	1	0	179	0.22
South Korea (Republic of Korea)	15	71	24	11	13	1	0	4	137	0.17
Japan	19	36	30	14	6	1	0	23	130	0.16
France	8	6	18	53	0	1	2	4	92	0.11
Baltic States (Estonia, Latvia,	1	1	2	64	0	0	1	11	79	0.10
Turkey	8	9	19	15	17	0	2	0	70	0.08
North Korea (Democratic Peoples Republic)	0	13	1	28	0	0	0	16	58	0.07
Germany	11	8	21	13	0	1	1	0	56	0.07
Taiwan	5	29	12	1	0	0	0	1	46	0.06
Italy	9	5	13	8	0	1	5	3	45	0.05
United Kingdom	8	6	14	7	0	1	1	0	36	0.04
New Zealand	0	1	1	4	1	0	0	0	8	0.01
World Total	368	2249	1447	7746	696	47	10800	12372	35725	

1 Figure 4-9, summarizing global emissions into six broad source categories, indicates that global
2 BC totals are dominated by open biomass burning, and residential cooking and heating sources. Roughly
3 35% of the total global emissions of BC are from open biomass burning, while the domestic (or
4 residential) sector contributes 25% of the global total. In developing countries, most of the residential
5 (domestic) emissions come from cook stoves that burn biomass, dung or coal resulting in significant
6 emissions of BC. China, India, and Africa to contribute nearly two-thirds of the total BC emissions from
7 this source category, an issue discussed in more detail in Chapter 9.

8



1

2

3

Figure 4-9. Global Distribution of BC and OC by Major Source Categories.

4

Table 4-6 displays the global BC, OC, and OC/BC ratios for 6 major source categories.

5

Transportation sources show the lowest OC/BC ratios, while burning categories are seen to be

6

dominated by OC emissions and industrial sources are somewhere in the middle. All these sources also

7

emit CO₂ and other greenhouse gases as well as sulfur emissions that transform into SO₂, NO_x emissions

8

that transform into nitrates and contribute to ozone, and other particles.

1

Table 4-6. OC/BC Ratios by Broad Source Categories.

Source Category	BC (Gg)	OC (Gg)	OC/BC
Energy/Power	54	368	7
Industry	1,497	2,250	2
Transport	1,471	1,587	1
Domestic/Residential	1,947	7,746	4
Biomass Burning	2,755	23,869	9
Waste	35	47	1
Totals	7,759	35,867	4.6

2

3

4 Figure 4-10 ranks BC emission estimates for the 37 world regions shown in Table 4-4,
5 highlighting the relative contribution of open biomass (grassland and forest fires) and anthropogenic
6 sources. With this AR5 BC inventory, regions like Africa, Brazil, and Australia are dominated by open
7 biomass burning sources whereas countries like the United States, China, and India are dominated by
8 anthropogenic sources. Chapter 7 provides discussion of an alternative classification of fires as “natural”
9 or “controllable” recognizing that not all fires classified as “forest” or “wild” are uncontrollable.

10 Figure 4-11 details the relative contribution of emissions for the 8 sectors in each of the 37
11 regions ranked in Figure 4-9. Forest fires, grassland fires, industry, and transportation are all major
12 sources of BC depending on world region. Areas like Asia have significant emissions from industry,
13 domestic, and transportation sectors. Africa and South America are generally dominated by open
14 biomass burning sources. Developed regions like the Middle East, Japan, Europe and the United States
15 are dominated by transportation sources. In the international inventory, “non road” emissions are
16 included in the industry category, whereas in the domestic inventory these emissions are counted in the
17 mobile source category. It is not possible to determine what percentage of “industry” emissions are
18 actually “non road” emissions in the AR5 inventory.

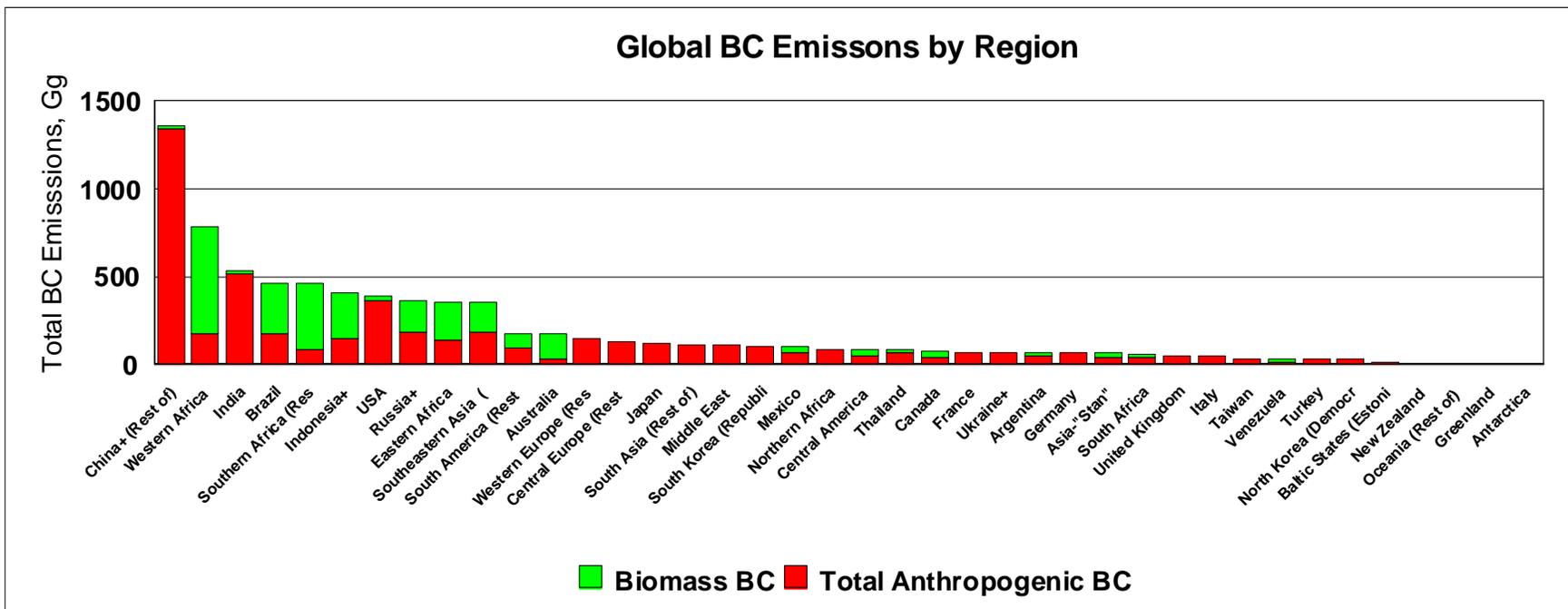


Figure 4-10. BC Emissions by World Region, 2000 (Gg).

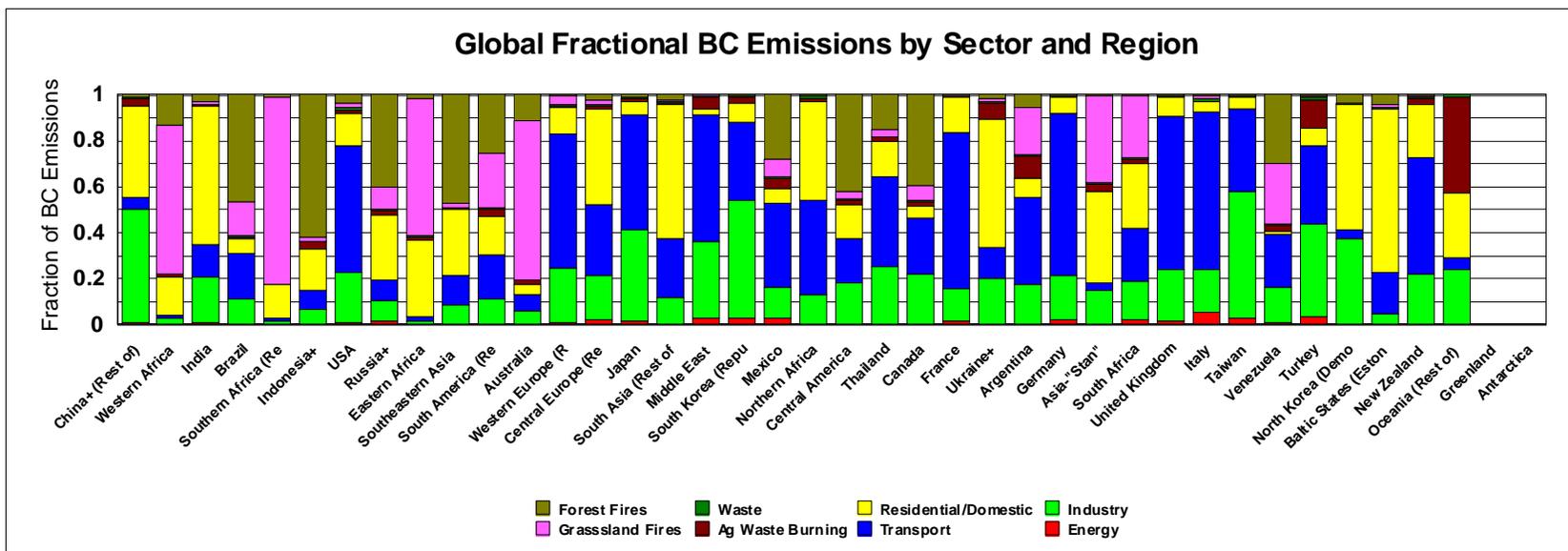


Figure 4-11. Global BC Emissions by Source Categories and Region.

1 Emission estimates for BC and OC are generally far more uncertain compared to estimates for
2 CO₂, SO₂ or other pollutants primarily because BC is emitted by a large number of small, dispersed
3 sources with irregular operating conditions, such as cook stoves, biomass burning, traffic, and
4 construction equipment. Low technology-combustion (e.g., “open burning”) contributes greatly to both
5 the emissions and uncertainties. There has not been a lot of work done on estimating uncertainties with
6 BC emission estimates. However, Bond et al. (2004b) do present a bottom-up estimate of uncertainties
7 in source strength by combining uncertainties in PM emission factors, emissions characterization, and
8 fuel use patterns. They judge the precision of total BC emissions to be within a factor of two. Advances
9 in emissions characterizations for small residential, industrial, and mobile sources and top-down analysis
10 combining field measurements and modeling with iterative inventory development will likely be
11 required to reduce these uncertainties further. The general “factor of 2” in overall uncertainty
12 estimated by Bond et al. (2004b) is comparable to the range of estimates of climate forcing by BC given
13 in the 4th IPCC assessment (IPCC, 2001).

14 **4.4.2 Black Carbon Emissions North of the 40th Parallel**

15 Emissions north of the 40th parallel are thought to be particularly important for BC’s climate-
16 related effects in the Arctic (Shindell, 2007; Ramanathan and Carmichael, 2008). The 40th parallel north
17 is a circle of latitude that is 40 degrees north of the Earth’s equatorial plane. Globally, it crosses Europe,
18 the Mediterranean Sea, Asia, the Pacific Ocean, North America, and the Atlantic Ocean. In the United
19 States, the 40th parallel approximately bisects New York City in the East and San Francisco in the West,
20 passing near Trenton, NJ, Philadelphia, PA, Columbus, OH, Indianapolis, IN, Springfield, IL, Kansas City,
21 MO, and Denver, CO.

22 Global inventories indicate that most BC emissions, particularly from fossil fuels, occur in the
23 Northern Hemisphere. Therefore, emissions north of 40°N latitude may be of particular concern in
24 understanding the impacts of BC on climate. Figure 4-12 presents the magnitude of global BC emissions
25 and source contributions by latitude. Transportation is the largest source of global BC emissions north
26 of the 40th parallel, though open burning, residential burning, and industrial sources all contribute
27 emissions north of 40°N in the Bond et al. inventory. These patterns have implications for assessing the
28 contribution of source regions to snow melt in the Arctic as well as total BC-related forcing in the
29 Northern Hemisphere.

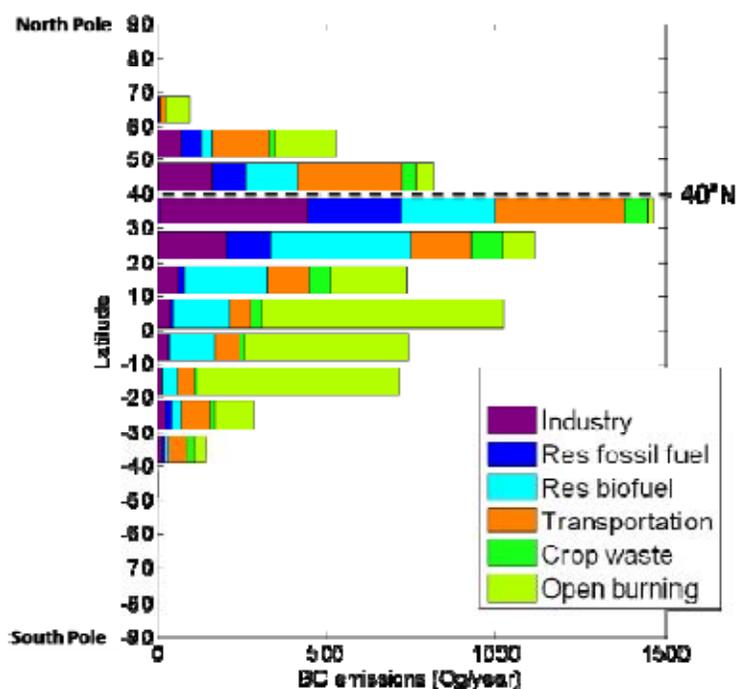


Figure 4-12. Geographical distribution of global BC emissions by latitude (Bond, 2008).

BC emissions from U.S. sources north of the 40th parallel are displayed below in Table 4-7. About 260,000 out of the 637,000 tons (41%) is estimated to be emitted in areas north of the 40th parallel. With the exception of the Fossil Fuel Combustion and Biomass Combustion categories, most of the other categories show BC emissions contributions north of the 40th parallel are equivalent to the number of U.S. counties in that region (about 38%). “North of 40” emissions from biomass burning are seen to be 51% of the domestic total mainly attributable to the wildfire emissions from Alaska (as discussed earlier in this chapter). These emissions from Alaska are likely to influence the Arctic given the close proximity. The contribution from mobile sources to domestic BC emissions north of 40th parallel is proportionally greater than it is to total BC emissions nationally, because BC emissions from fossil fuel combustion north of 40th parallel represent such a small percentage (only 6%) of all emissions across the United States for this source category. Biomass combustion (with a heavy influence from Alaskan wildfire emissions) and mobile sources make up nearly 97% of the BC emissions estimated to occur north of the 40th parallel in the United States.

Table 4-7. A Comparison of BC Emissions Nationally to Those from Sources “North of 40th Parallel” in 2005 (short tons).

	Total US BC Emissions	BC Emissions Estimated North of 40th Parallel	Fraction of BC Emissions north of the 40th Parallel	Fraction of TOTAL BC north of 40th parallel
Biomass Combustion	250,499	128,501	0.51	49%
Fossil Fuel Combustion	43,049	2,794	0.06	1%
Fugitive Dust Sources	1,609	483	0.30	0%
Industrial Sources	6,085	1,574	0.26	1%
Mobile Sources	333,405	125,784	0.38	48%
Other Minor Sources	2,525	755	0.30	0%
Totals/avgs	637,172	259,891	0.41	

1 **4.4.3 Alternative Estimates of Global and Regional Emissions**

2 In addition to the widely used Lamarque/Bond inventory discussed above, there are other global
3 BC and OC emissions inventories compiled by other researchers. Seven other global BC and OC
4 inventories are available in the published literature (Penner 2000; Cooke and Wilson, 1996; Lioussé et al.
5 1999; Cooke et al 1990; Cofala et al 2007). The total BC emissions estimated in these inventories fall in
6 the “factor of 2” error range estimated in Bond’s BC inventory, which signals that these estimates are
7 generally consistent with the estimates presented above. Most of these alternative emissions are
8 developed using “bottom-up” approaches, similar to that used by Bond et al. and Streets et al. These are
9 summarized and discussed further in Appendix 2. The alternative emissions inventories do not provide
10 as much detail or as comprehensive an explanation of uncertainty in the estimates as the Bond
11 inventories employed in this chapter.

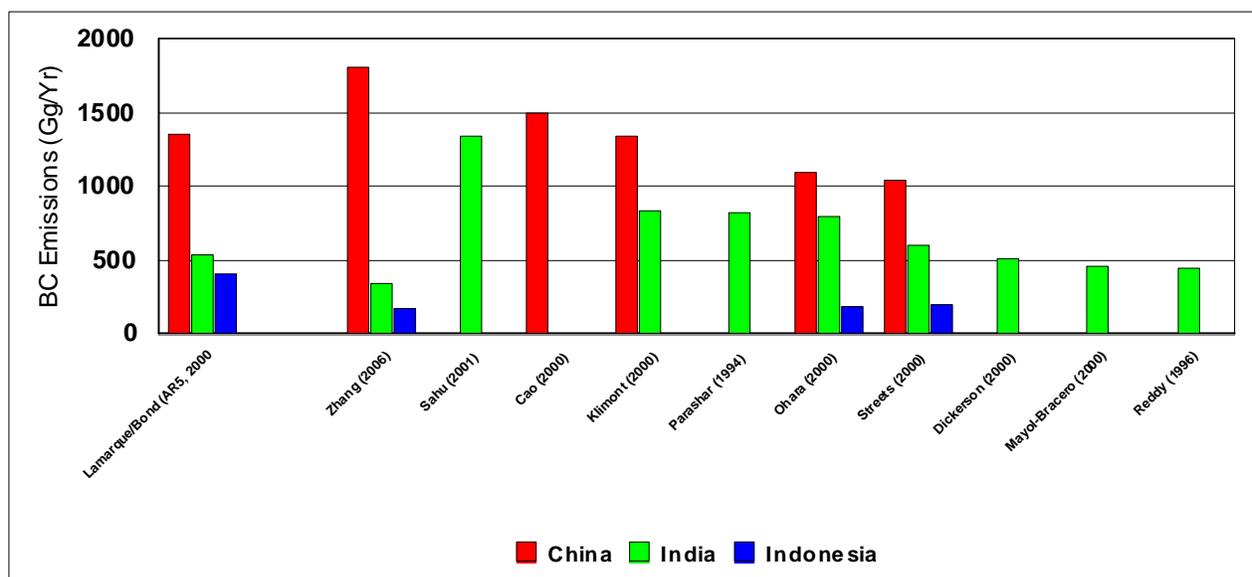
12 An advantage of global inventories is that the emissions estimates are compiled using consistent
13 definitions and methods across all regions. The global inventories, however, do not necessarily employ
14 region or country specific emission factors, activity levels, and other surrogates. Regional emissions
15 inventories, constructed for specific regions, nations, or local areas, often make use of more accurate
16 data from local and government sources. This may allow for improved BC emissions estimates relative
17 to data drawn from models or global energy databases. Regional inventories are more likely to account
18 for differences in the composition of the fuel burned, the diversity of technologies (especially in
19 developing countries), and the importance of smaller sources that can often be overlooked in global
20 inventories. Some of these regional inventories are based on “top-down” type approaches while others
21 are based on the traditional “bottom-up” approaches described earlier. Reconciling the global
22 inventories with regional inventories is complicated by differences in methods used for each inventory.
23 Good regional inventories, however, may still be used to evaluate the global estimates, and can be used
24 to inform future versions of those global inventories.

25 Most of the regional BC inventory efforts to date have focused on the Asian sub-region (Zhang,
26 2006; Sahu 2001; Cao 2000; Klimont 2000; Parashar 1994; O’hara 2000; Streets 2000; Dickerson 2000;
27 Mayer-Bracero 2000; Reddy 1996) likely due to high emissions of BC and OC from diverse sources there.
28 There are fewer regional BC inventories available for European countries. In general, global emissions
29 inventories have to be used to estimate European BC emissions. Recent work by the Arctic council to
30 estimate BC and OC emissions for Arctic nations may provide useful information on regional inventories
31 in those nations. A full list of available regional inventories, along with additional details about the
32 methods used, is available in Appendix 2.

33 Figure 4-13 compares some of the different regional BC emissions estimates for China, India and
34 Indonesia to the estimates from AR5 inventories. In general, even though the base inventory year
35 (indicated on x-axis label in Figure 4-12) is different in most cases, these inventories are seen to be fairly
36 consistent with one another, and also with the Bond global inventory. The range of emissions for a
37 country from these various inventories also gives an indication of the amount of uncertainty in BC

1 emission estimates for a given region. All of the regional estimates are within the error bounds
 2 estimated by Bond et al. for BC emissions.

3 In addition, for future work to improve the global estimates, these regional estimates can be
 4 used to “bound” estimates for a given world region or country. Finally, it is important for countries to
 5 begin developing regional inventories of BC and OC, to better identify sources and their BC emissions,
 6 and to supplement global inventories that sometimes rely on “default” type information to develop
 7 regional estimates. Having more accurate “localized” inventories will enable better and more effectively
 8 designed mitigation strategies for specific sources in specific world regions.



9
 10 **Figure 4-13. Comparison of regional inventories for China, India, and Indonesia with AR5 estimates.**

11 **4.4.4 Inventory Comparisons for the U.S. Black Carbon Emissions**

12 Table 4-8, compares the U.S. portion of the 2000 global AR5-based BC and OC emission
 13 estimates of Lamarque/Bond et al. (in green) to the EPA’s BC estimates for 2002/2005 (in red). U.S.
 14 emissions from the global inventory are aggregated to the highest level of source category detail
 15 possible to facilitate comparisons with EPA-based BC estimates¹¹. The degree of difference between the
 16 EPA inventory and the AR5 inventory for U.S. emissions is depicted as a percentage in the light blue
 17 column.

¹¹ In general, aggregating of emissions from different inventories to this level of broad source categorization introduces uncertainties since an accurate matching of individual source categories to these larger source categories is not always possible. The specific source types included in the more broad categories in the AR5 inventories (and used in Table 4-8) are unclear and details were not available for this Report. More work is needed in comparing region-specific inventories from global estimates to regionally developed inventories, and especially to better understand the sources that make up the larger sectors that are generally depicted in Reports and publications.

1 **Table 4-8: Comparison of BC and OC Emissions (in Gg) for the United States between AR5 Global**
 2 **Inventories and EPA Inventories.**

BC Emissions (Gg) in AP5 and EPA Inventories				
AP5 Source Description	BC	EPA Source Description	BC	EPA estimates High By:
Energy	3	Energy/Power	39	1200%
Industry	85	Industrial	6	-93%
Transport	216	Mobile Sources	302	40%
Residential	55	Residential	21	-62%
Agricultural Waste Burning	6	Agricultural Burns	13	117%
Waste	3			
Grassland Fires	9	Prescribed Burns	53	489%
Forest Fires	13	Wildfires	138	962%
Totals:	390		572	47%
OC Emissions (Gg) in AP5 and EPA Inventories				
AP5 Source Description	OC	EPA Source Description	OC	EPA estimates High By:
Energy	72	Energy/Power	59	-18%
Industry	60	Industrial	15	-75%
Transport	143	Mobile Sources	186	30%
Residential	198	Residential	185	-6%
Agricultural Waste Burning	28	Agricultural Burns	46	64%
Waste	5			
Grassland Fires	97	Prescribed Burns	244	151%
Forest Fires	227	Wildfires	670	195%
Totals:	830		1,405	69%

3
 4 Total BC emissions for the United States are estimated to be about 390 Gg in the AR5 inventory,
 5 and about 572 Gg in the EPA inventories¹². Most of this approximately 50% difference is driven by EPA
 6 estimates for open burning and (to a lesser extent) for mobile sources in the United States that are
 7 higher than those from the global inventories. As discussed previously, wildfire emissions can vary
 8 greatly from year to year, and this may explain some of the difference between the estimates for open
 9 burning since the AR5 estimates are based on the year 2000 and the EPA estimates on the year 2002.
 10 Also, EPA estimates include all nonroad and on-road emissions in the transportation source category,
 11 while global inventories group emissions from some of the smaller nonroad sources into the “Industry”
 12 category. This could account for global inventory estimates of U.S. emissions being lower for
 13 “transport” and higher for “industry” compared to the EPA estimates. In the case of OC emissions, Table
 14 4-8 shows that the AR5 total is about 830 Gg while the EPA estimates are seen to be about 1,405 Gg, a
 15 difference of about 69%. As with BC, most of this discrepancy stems from fire emissions that EPA
 16 estimated to result in more OC than do the AR5 estimates for the United States. It is likely that fire
 17 emissions (both OC and BC) from many countries are under-estimated due to the methods used to

¹² EPA’s estimate of the domestic BC emissions in Table 4-8 (572 Gg) is a bit smaller than the total BC emissions estimates shown earlier in this chapter of 578 Gg. This difference stems from (1) most of the sources in the “Minor Sources” mega source category were not included in this comparison; (2) while an emissions estimate (albeit small) for a “Waste” category is provided in the global inventories, no such estimate was included in the U.S. derived inventory.

1 estimate fire emissions in global inventories including an insufficient accounting for emissions from
2 smaller fires.

3 The comparison of BC emissions from the most often used global estimates by Lamarque/Bond
4 et al. to BC inventories developed by EPA reveal important differences that necessitate further
5 investigation. A key focus of any future examination is how these differences may influence the
6 estimates of regional effects from global climate models. However, as noted in Chapter 2, emission
7 uncertainties are not thought to be as important as other factors in determining climate impacts from
8 model output (Koch, 2010). In addition to better understanding the role of uncertainty in emission
9 estimates on impacts simulated by models, more work is needed to better understand the source make-
10 up of sectors with large differences between the two inventories (e.g., biomass combustion sources,
11 mobile sources, and some parts of the residential sectors). In addition, it is necessary to clarify the
12 characterization of the uncertainties associated with global BC and OC emissions (and “the factor of 2”
13 often discussed) estimated by Bond et al.

14 **4.5 Long-Range Transport of Emissions**

15 Aerosols emitted in a particular region can be transported long distances through the
16 atmosphere to other regions of the globe. Therefore, BC emitted in one place can affect radiative
17 forcing in other locations downwind. Furthermore, the climate impacts of black carbon, such as effects
18 on temperature and precipitation, do not necessarily occur where the radiative forcing occurs and may
19 occur downwind of the source region (Shindell et al 2008, TF HTAP, 2010). The relationships between
20 where pollutants are emitted and where their impacts are experienced are often characterized as
21 “source-receptor” relationships. Emissions in a source region are transported, or lead to formation of
22 additional aerosols that then are transported, and eventually deposit or affect the receptor regions
23 downwind. Long-range or intercontinental transport of aerosols may occur in the planetary boundary
24 layer (PBL), which is the layer of the atmosphere that is in contact with the earth’s surface, or in the free
25 troposphere, which is the layer of the atmosphere just above the PBL but below the stratosphere.
26 Aerosols that have been lofted above the boundary layer into the free troposphere can be transported
27 long distances due to the relatively small amount of precipitation and high wind speeds. In the mid-
28 latitudes of the Northern Hemisphere, long-range transport is largely from west to east, due to the
29 prevailing winds. However, different transport patterns are dominant in other parts of the world.

30 The Task Force on Hemispheric Transport of Air Pollution (TF HTAP) organized under the
31 Convention on Long-range Transboundary Air Pollution conducted a multi-model assessment of long-
32 range transport of aerosols and other pollutants from four main source regions in the Northern
33 Hemisphere approximating the populated regions of North America, Europe, South Asia, and East Asia
34 (TF HTAP, 2010). The models included in the study produced widely varying estimates of the absolute
35 amount of intercontinental transport of aerosols. Most of the diversity in model estimates appears to
36 be due to differences in the representation of physical and chemical transformations that aerosols
37 undergo in the atmosphere, which leads to differences in the estimated atmospheric lifetime of
38 aerosols. Uncertainties in emissions estimates and atmospheric transport algorithms also contribute to

1 the diversity of estimates. A systematic comparison between the TF HTAP ensemble estimates and
2 observations in the mid-latitudes has not been conducted.

3 Although the absolute estimates in the TF HTAP ensemble are quite different, the relative
4 contributions of the four continental source regions to concentrations or deposition downwind are
5 more consistent. In the North American region, it was estimated from the ensemble of simulations that
6 about 80% ($\pm 25\%$) of the BC deposited in North America is from anthropogenic sources in North
7 America. Open biomass burning, largely forest fires, across North and Central America contribute about
8 12% ($\pm 17\%$). Other emission sources from outside North America contribute about 8% ($\pm 17\%$) of the BC
9 deposited within the North American study region.

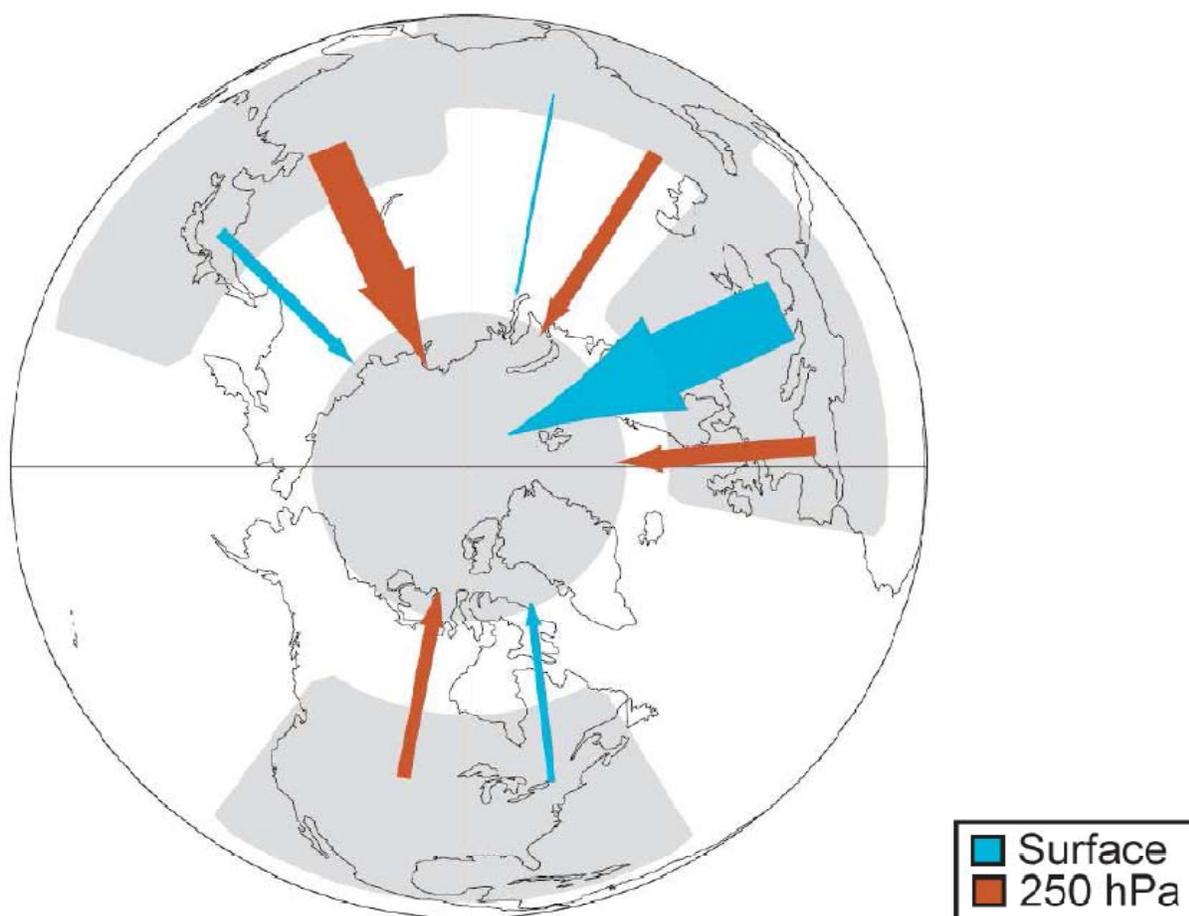
10 The TF HTAP multi-model study also examined the impact of intercontinental transport on total
11 atmospheric column concentrations, aerosol optical depth, and aerosol radiative forcing. The TF HTAP
12 concluded that intercontinental transport associated with anthropogenic sources of BC (not including
13 open biomass burning) accounted for roughly 30% of the aerosol optical depth and direct aerosol
14 radiative forcing over North America. Similarly, anthropogenic BC emissions from North America are
15 likely to contribute 10-30% of the black carbon radiative forcing over other regions of the Northern
16 Hemisphere. This ensemble study would suggest that long-range transport of BC is a minor contributor
17 to surface concentrations over North America, but a major contributor to the radiative forcing and
18 regional climate impacts of BC. It is worth noting that the results were calculated using rather coarse
19 global-scale models and variations within the North American region were not investigated.

20 The results of the TF HTAP multi-model experiments are consistent with previous modeling
21 results that showed that sources outside North America make a relatively small contribution to surface
22 aerosol levels in North America (Chin et al. 2007, Koch et al. 2007) and that intercontinental transport of
23 BC emissions, particularly from South and East Asia, is more important for surface concentrations or
24 deposition at high altitudes (Hadley et al 2007) and for total column loadings and climate impacts (Koch
25 et al., 2007; Reddy and Boucher, 2007).

26 In recent work, Kopacz et al. (2010) estimated the contribution of BC emission sources to BC
27 concentrations and deposition in the Himalayas and Tibetan Plateau and the associated direct and
28 snow-albedo radiative forcing. They conclude that emissions from northern India and central China and
29 from western and central China contribute most of the BC in the Himalayas and Tibetan Plateau,
30 respectively, although the contributions of different locations varies with season. However, they also
31 show that the Himalayas and Tibetan Plateau region can receive significant contributions from very
32 distant sources including biomass burning in Africa and fossil fuel combustion in the Middle East. They
33 estimate that the snow-albedo effect of BC deposition on snow in the region results in a warming
34 influence that is an order of magnitude larger than the direct radiative forcing influence.

35 Given the paucity of anthropogenic sources of BC in the Arctic, a large fraction of the climatic
36 impact of BC in the Arctic can be attributed to long-range transport. Shindell et al. (2008) examined the
37 results of the TF HTAP multi-model experiments for insights about transport to the Arctic. Comparing to
38 observations of BC at Barrow, Alaska, and Alert, Canada, all of the models appeared to underestimate

1 the transport of BC to the Arctic. Consistent with the findings for the source-receptor relationships at
 2 mid-latitudes, they found that the models varied widely in terms of the absolute estimates of the
 3 contribution of different source regions, but were similar in their estimates of the relative contributions.
 4 The ensemble results suggest that European emissions are the largest contributors to surface BC in the
 5 Arctic (due to the high latitude, and therefore Arctic proximity, of many European sources), while East
 6 Asia is the largest contributor to BC in the upper troposphere (Figure 4-14) (Shindell, et al., 2008).
 7 Additional source apportionment analysis under the TF HTAP (2010) concluded that anthropogenic
 8 emissions from Europe and open biomass burning emissions from Eurasia both contributed about 35%
 9 of the surface BC in the Arctic. Anthropogenic emissions from the North American study region, not
 10 including open biomass burning, accounted for an average of 5% of surface BC in the Arctic region, with
 11 model estimates spanning the range from 2% to 10% (TF HTAP, 2010). However, unlike the rest of the
 12 Arctic, deposition of BC in Greenland, location of the second-largest ice sheet in the world, is most
 13 sensitive to North American emissions (Shindell et al., 2008).

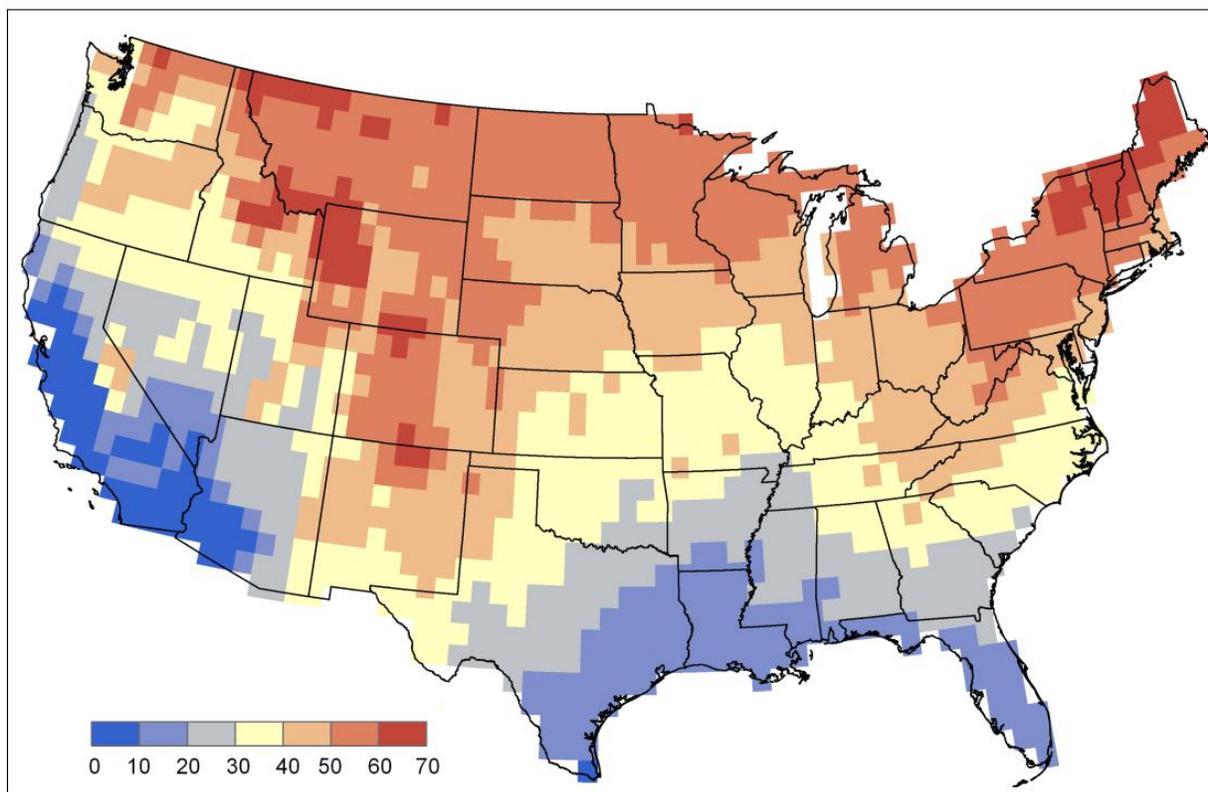


14
 15 **Figure 4-14. Relative Importance of Different Regions to Annual Mean Arctic BC Concentrations at the**
 16 **Surface and in the Upper Troposphere (250 hPa).** Values are calculated from simulations of the response
 17 to 20% reduction in anthropogenic emissions of precursors from each region (using NO_x for ozone).
 18 Arrow width is proportional to the multi-model mean percentage contribution from each region (shaded)
 19 to the total from these four source regions. (Shindell et al., 2008)

1 In addition to the TF HTAP approach of largely using grid-based models, trajectory-based models
2 have also been employed to quantify transport to the Arctic. These models show a much stronger
3 influence of sources in Northern Eurasian locations to Arctic surface concentrations and deposition, and
4 much less influence from more distant sources. The exception is for high altitude sites in Greenland,
5 which may be influenced by very different sources than the rest of the Arctic (Hirdman et al., 2010).

6 The contribution of both open biomass burning and fossil fuel combustion to BC deposition in
7 the Arctic has been confirmed by detailed chemical analysis of surface snow and ice cores. However, the
8 observational evidence would suggest that open biomass burning, including crop burning, is the
9 dominant source of BC deposition in the Arctic (McConnell, et al., 2007, Hegg, et al., 2009; Hegg, et al.,
10 2010). The relative contribution of different source types and locations, however, varies significantly
11 across receptor locations and seasons.

12 Within the United States, the potential for transport of domestic BC emissions to the Arctic is
13 known to vary by location and season. Given its proximity to the Arctic, BC emission sources in Alaska
14 are likely to have an impact on the Arctic, depending on the synoptic weather conditions. For emission
15 sources in the contiguous United States, recent backward trajectory modeling work by the Joint Fire
16 Science Program (Larkin, 2011) has shown that the probability of emissions impacting the Arctic is
17 critically dependent on the specific injection height into the atmosphere and the specific synoptic
18 weather patterns prevalent at the time. Figure 4-15 displays the percentage of spring days (March-May)
19 when 8-day back-trajectories starting at the Arctic Circle pass through the boundary layer (below
20 2000m) over locations in the contiguous United States, based on synoptic patterns observed over a ten
21 year period (1999-2008). This analysis suggests that the potential for springtime transport of BC ground-
22 level emissions from the contiguous United States to the Arctic can be significant. Over the southern
23 portion of the United States, the potential for transport to the Arctic is relatively infrequent (< 25%). For
24 locations in the northern part of the United States and other higher-altitude locations, the analysis
25 indicated that the potential for transport trajectories to the Arctic is common (> 50%). However, even in
26 areas which show a large seasonal and climatological potential for transport, it is possible to identify
27 multi-day periods where transport to the Arctic is limited. The dependency on source location and
28 synoptic weather conditions may have implications for understanding source apportionment and for
29 implementing mitigation strategies.



1
2 **Figure 4-15. Percentage of Spring Days (March through May) in which Eight-day Trajectories Reaching**
3 **the Arctic Circle Passed through Locations in the Contiguous United States within the Boundary Layer**
4 **(below 2000m).** These percentages are based on 10 years of trajectories using meteorological data from
5 1999-2008.

6 **4.6 Historical Trends in Black Carbon Emissions**

7
8 **4.6.1 U.S. Black Carbon Emissions Trends**

9
10 Historic trends and future projections of BC emissions provide an indication of the relative
11 importance of different sectors over time and can help focus future mitigation efforts. Establishing
12 emissions trends requires the use of a consistent estimation method. Most domestic inventories
13 discussed earlier in this chapter are derived from methods that have changed as measurement and
14 models have improved. As a result, care must be taken in interpreting trends of over time. However, it
15 is possible to observe large scale changes. Specifically, the data show that U.S. emissions of BC
16 increased steadily from the mid 1800's through 1920, and then declined over the next 8 decades. This is
17 likely attributable to changes in fuel use from coal to cleaner fuels, more efficient combustion of coal,
18 and implementation of PM controls. In more recent years, EPA's introduction of the NAAQS for fine
19 particles in 1997 and strengthening of that NAAQS in 2006 necessitated PM_{2.5} reductions that likely
20 contributed to BC emission reductions as well. In addition, since 1990, due to mobile source emission
fine particle regulations, there have been substantial reductions in BC emissions from those sources.

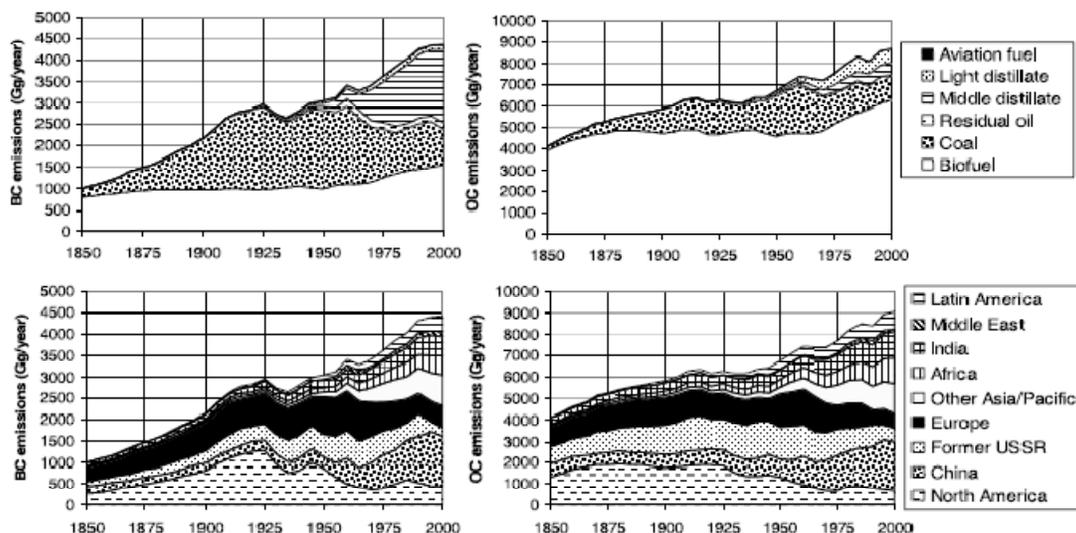
1 Since mobile source emissions are modeled, a time series of BC emissions can be generated
2 more easily for this source category than for other U.S. source categories. Mobile sources have
3 experienced a 30% reduction in BC, a 51% reduction in OC, and a 36% reduction in PM_{2.5} emissions from
4 1990 - 2005. From 1990 to 2005, BC emissions decreased by 79%, 30%, and 25% for on-road gasoline,
5 on-road diesel, and nonroad diesel sources, respectively. Continued reductions are expected for mobile
6 sources in the next two decades, as discussed further in Chapter 6. BC emissions for nonroad gasoline
7 sources, though extremely small, did not change from 1990 to 2005.

8 BC emissions trends for the other major source categories (open biomass burning, industry, and
9 energy/power) are difficult to estimate due to lack of data and inconsistent measurements and methods
10 over time. The methods used to estimate emissions from 1990 to 2008 have changed significantly, as
11 has the way PM_{2.5} estimates are used to derive BC emissions estimates. There are no BC estimates
12 available for any non-mobile source categories for the year 1990. From 1990 to about 1998 there was
13 about a 30% reduction in direct PM_{2.5} emissions from EGUs and other power-generation sources due to
14 controls on direct PM_{2.5}. It is expected that some of these reductions in direct PM_{2.5} led to decreases in
15 emissions of BC, but this is difficult to verify without consistent speciation data for the entire time
16 period. In 1999, there was a major change in the methods used estimate PM_{2.5} emissions. Based on
17 these new methods, from 1999 to 2008 an additional 21% reduction in direct PM_{2.5} is seen from this
18 source category. In contrast, direct PM_{2.5} emissions from industrial sources are estimated to have
19 declined only 6% during the entire 1990-2008 period.

20 Trends in emissions from biomass burning categories (wildfires, prescribed burns, and
21 agriculture burns) are not available due to significant year-to-year changes in the methods used to
22 estimate emissions. For that reason, in the modeling inventories “average fires” are used to represent
23 emissions from this source category.

24 **4.6.2 Global Black Carbon Emissions Trends**

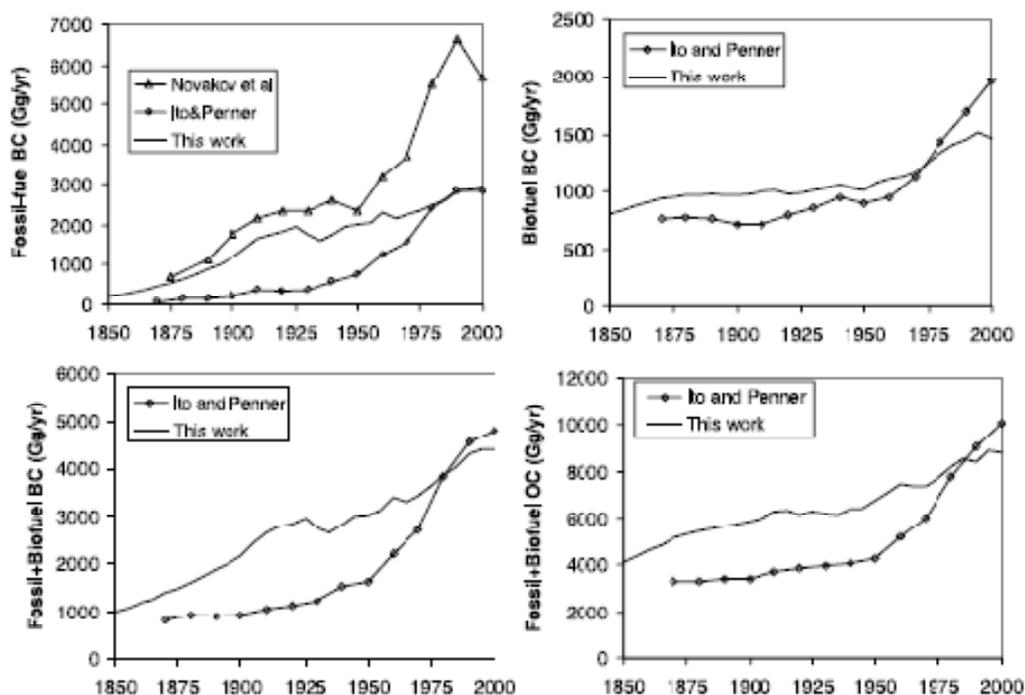
25 There are a number of studies available which have looked explicitly at global BC emissions
26 trends over time (e.g., Bond et al. 2007; Ito 2005, and Novakov et al. 2003). Figure 4-16 (Bond et al.
27 2007) shows the growth in global BC emissions during the period between 1850 and 2000. The figure
28 shows that emissions of BC have increased almost linearly, totaling about 1000 Gg (approximately
29 1.1 million tons) in 1850, 2200 Gg (approximately 2.4 million tons) in 1900, 3000 Gg (approximately 3.3
30 million tons) in 1950, and 4400 Gg (approximately 4.8 million tons) in 2000. The slower growth between
31 1900 and 1950 may be due to economic circumstances and also the introduction of cleaner technology,
32 especially in developed countries. OC shows a similar pattern of linear growth that is slightly slower in
33 the mid-1900s.



1
2 **Figure 4-16. Emissions of BC (left) and OC (right).** Emissions segregated by fuel (top) and world region (bottom)

3 Figure 4-17 relates BC emission trends from Bond et al. (2007) to earlier work done by Ito and
 4 Penner (2005), and by Novakov et al. (2003).¹³ Ito and Penner show a very similar trend and magnitude
 5 in BC emissions from biofuel, but the magnitude of fossil-fuel BC emissions is much lower. In the late
 6 1900s, Bond et al.’s biofuel emissions increase less (about 30% between 1960 and 2000 vs. 100% for Ito
 7 and Penner). By contrast, Novakov et al. estimated higher fossil-fuel BC emissions than Bond et al., in
 8 the early 1900s. Novakov et al.’s work was based on total BC aerosol, while Bond et al. and Ito and
 9 Penner’s work focused on fractions of PM less than 1 micron in diameter. Novakov et al. shows flat BC
 10 emissions between about 1910 and 1950, similar to Bond et al.

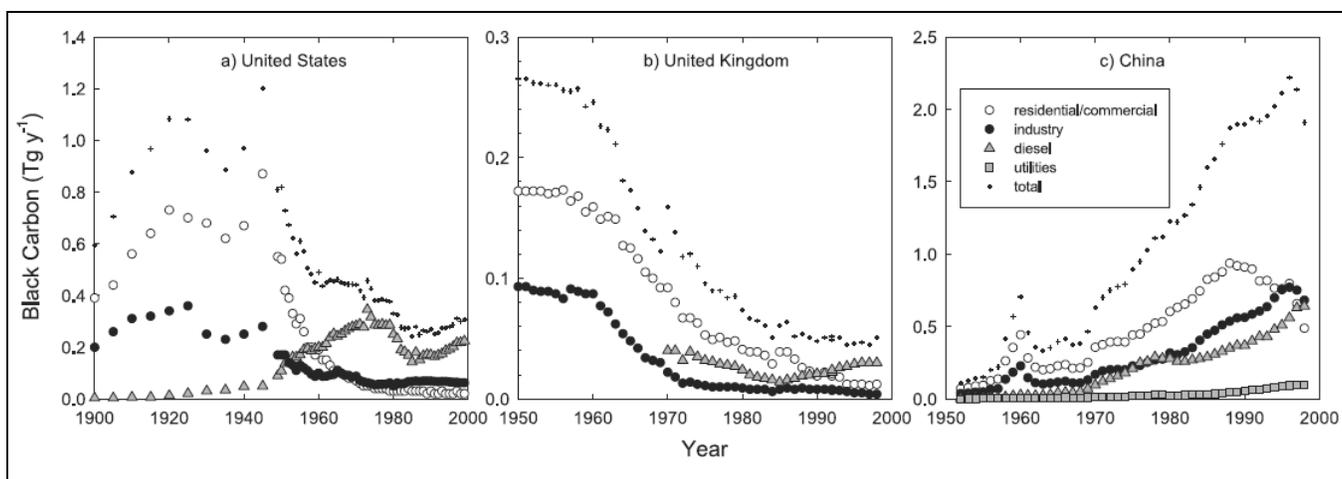
¹³ Novakov et al. looked at BC from fossil fuel combustion only.



1
2 **Figure 4-17. Comparison of Bond et al. 2007 (“this work”) Historical Reconstruction of Global Emission**
3 **Trends with Previous Studies.**

4
5 The greatest difference between the more recent Bond et al. (2007) work and the earlier Ito and
6 Penner and Novakov et al. work is the more gradual transition in the latter half of the 20th century.
7 Both of the earlier studies considered the introduction of cleaner diesels and some changes in sectoral
8 divisions. Bond et al. modeled shifts to cleaner burning through increases in consumption in cleaner
9 sectors. Bond et al. indicate the shift to cleaner burning coal explicitly for the first time, reducing BC
10 emissions from this sector. It is likely that the difference between the three studies is largely
11 attributable to the choice of emission factors, which entails some implicit assumptions about technology
12 choices.

13 Figure 4-18 shows the estimated BC emissions trends for the U.K., United States, and China
14 (Novakov, 2003). According to these data, emissions from the United States peaked in 1920, while
15 Europe peaked in 1950 and has declined about 90%. Total global emissions of black carbon, however,
16 have been steadily increasing since 1875 (Novakov, 2003). Presently, global BC emissions total
17 approximately 9 million tons. Almost all of the increase in recent decades is from developing countries
18 in Asia, Latin America, and Africa. China and India contribute nearly 25% of global BC emissions.



1
 2 **Figure 4-18. BC emissions (Tg /y) in the United States, United Kingdom, and China (Novakov, 2003).** BC
 3 emissions are estimated from annual consumption data for the principal BC producing fossil fuels and BC
 4 emission factors disaggregated by utilization sector. BC from biofuels and open biomass burning are not
 5 included.

6 Together, these emissions trends studies and other works suggest that developed countries
 7 dominated global BC emissions until the adoption of pollution control technologies and fuel-use shifts
 8 began to slow growth and eventually to result in significant reductions after mid-century (Bachmann
 9 2007, 2009; Ramanathan, 2007). Available data suggest that BC emissions from developed countries
 10 have declined substantially over the past several decades, while emissions from developing countries
 11 have been growing. Today, the majority of EC emissions are from developing countries (Bond, 2006a)
 12 and this trend is expected to increase (Jacobson, 2009, IIASA).

13

(This page intentionally left blank)

5. Observational Data for Black Carbon

5.1 Summary of Key Messages

- Estimates of black carbon (BC) are made with a variety of instrumentation and measurement techniques. Most ground level estimates of BC are reported as mass concentrations based on thermal-optical and filter-based optical techniques. Published studies show that BC estimates derived by commercial instrumentation are generally within 30%, but may disagree by as much as a factor of two. Further research is needed to standardize ambient and emissions measurement methods and to develop factors that harmonize existing measurements produced from different sampling and analytical techniques.
- Ground-level BC measurements across the globe indicate estimated concentrations ranging from < 0.1 $\mu\text{g}/\text{m}^3$ in remote locations to $\sim 15 \mu\text{g}/\text{m}^3$ in urban centers. Although monitor locations are sparse globally, available observations suggest ambient levels in China are almost 10 times higher in urban and rural areas than those in the North America or Europe. A comparison of urban concentrations to corresponding regional background levels reveals an urban increment of up to 2 $\mu\text{g}/\text{m}^3$ in the North America and Europe compared to an urban increment of $\sim 6\text{-}11 \mu\text{g}/\text{m}^3$ in China.
- In the United States, BC comprises $\sim 5\text{-}10 \%$ of urban PM_{2.5} mass.
- Long-term records of historical black carbon concentrations, derived from sediments or ice cores, valuably supplement available ambient data. Long term trends in estimated ambient concentrations derived from BC in sediments of New York Adirondacks and Lake Michigan show recent maximum concentrations occurred in the early- to mid-1900s and it appears concentrations have since decreased, which are attributed to decreased U.S. fossil fuel BC emissions. Ice core measurements in Greenland reveal a similar maximum BC level in the early 1900s, related to industrial emissions, but also show that biomass burning emissions contribute significantly to deposited BC in the Arctic. Globally, Northern Hemispheric ice core BC trends vary with location; some ice cores have BC values increasing to present-day, while other areas show maximum levels reached earlier in the 1900s.
- Over the past 2 decades when U.S. ground level ambient BC measurements are available, BC concentrations have declined, due to corresponding reductions in mobile source emissions during the 1990s into the early part of the last decade, and since 2007 due to recession-related decreases in vehicular travel and industrial output.
- Estimates of the total atmospheric column using remote sensing qualitatively show the same variability in absorbing aerosol levels across the globe. Like ground level measurements, the relative Aerosol Absorption Optical Depth (AAOD) values in China are also ten times higher than those observed in the United States. Remote sensing measurements that utilize multiple wavelengths also

1 show that the absorbing particle mixture varies globally among areas dominated by urban-industrial
 2 sources, biomass burning, and wind-blown dust.

3 **5.2 Black Carbon and Other Light-Absorbing Carbon: Measurement Methods**

4 In the atmosphere, black carbon is a light-absorbing carbonaceous component of particulate
 5 matter. Current measurement techniques generally attempt to estimate BC either by light absorption
 6 characteristics or by thermally isolating a specific carbon fraction. The techniques used currently to
 7 estimate black carbon mass concentrations are summarized in Table 5-1. These two general categories
 8 of BC measurement techniques can be viewed as different indicators of the chemical and physical
 9 properties of black carbon¹. This is discussed further in Appendix 1. The two most common BC
 10 measurement techniques are thermal-optical and filter-based light absorption as denoted in the table
 11 below.

12 **Table 5-1. Description of BC Measurement Techniques.**

Method Type	Method Description * Indicates prevalence of use
Light absorption/optical	<p><i>Filter-based:</i> Light absorption by particles is measured through a filter loaded with particles; BC is quantified using factors that related light absorption to a mass concentration. ***</p> <p><i>Photoacoustic:</i> Light absorption by particles is measured by heated particles transferring energy to the surrounding air and generating sound waves; BC is quantified using factors that relate light absorption to a mass concentration.*</p> <p><i>Incandescence:</i> Incandescent (glowing) particle mass is measured; BC is quantified by calibrating the incandescent signal to laboratory-generated soot.*</p>
Isolation of specific carbon fraction	<p><i>Thermal-Optical:</i> BC is measured as the carbon fraction that resists removal through heating to high temperatures and has a laser correction for carbon that chars during the analysis procedure; BC is quantified as the amount of carbon mass evolved during heating. ***</p> <p><i>Thermal:</i> BC is measured as the carbon fraction that resists removal through heating to high temperatures; BC is quantified as the amount of carbon mass evolved during heating.*</p>

13

¹ In current practice, measurements produced from light absorption/optical methods are expressed as BC while those produced from thermal-optical or thermal methods are referred to as elemental carbon or EC. To simplify the discussion, this differentiation in characterization of BC by measurement method is not repeated. Instead, since both measurement types are essentially estimating the same parameter (i.e., BC) albeit via different method orientation, and to make clear that light absorption measurements do not necessarily provide a ‘better’ indicator of BC than thermal methods, the term BC is used to describe all measurements. In Appendix 1, where this topic is more thoroughly explored, the BC measurements produced by light absorption/optical methods are referred to as apparent BC or “BCa”, and those produced by thermal or thermal-optical methods are referred to as apparent EC or “ECa”.

1 Thermal-optical measurements involve exposing a particle-laden filter to a series of heating
2 steps (Huntzicker et al., 1982; NIOSH 1999; Birch and Cary, 1996; Chow et al., 1993; Chow et al., 2007;
3 Peterson and Richards 2002). The carbon fraction that evolves first is termed organic carbon (OC) and
4 the particulate carbon that evolves in the second heating phase is termed EC. Despite the seeming
5 simplicity of thermally separating particulate carbon into two fractions, there is considerable uncertainty
6 in assigning carbon mass to either OC or EC fractions. For example, charring of particles during the
7 thermal analysis has produced erroneous OC and EC assignments (Cadle et al., 1980; Huntzicker et al.,
8 1982; Yu et al., 2002). In addition, there are several different commonly used temperature protocols
9 that cause variation in the OC and EC assignments. Long-standing reliance on the thermal optical
10 methods has resulted in an extensive observational record based on OC /EC splits, and the frequent use
11 of EC as an estimate for BC.

12 While EC is directly quantified as the mass of carbon atoms that evolve during a thermal or
13 thermal-optical analysis, optical techniques observe the light-absorbing properties of the particles to
14 estimate BC. Filter-based, optical instruments are relatively low cost, readily available, simple to
15 operate, and thus frequently field deployed to measure BC. Filter-based instruments measure the
16 quantity of light transmitted through a filter loaded with particles (Hansen et al., 1982; Lin et al., 1973;
17 Rosen et al., 1983). For filter-based optical instruments, the detected light absorption by particles is
18 converted to an estimated BC mass concentration. There are two main uncertainties associated with
19 the quantification of filter based BC using optical methods: 1) a filter loading artifact and 2) the
20 selection of an appropriate conversion factor. Several studies have shown that filter-based BC
21 measurement can be affected by the amount and composition of particles loaded onto the filter. This
22 artifact can be accounted for by applying a correction algorithm, but the detection and accurate
23 correction of this artifact is an area of development (Virkkula et al., 2007). In addition, the selection of
24 the conversion factor to relate light absorption to mass is a significant issue of debate. There are a
25 variety of conversion factors that have been published in scientific literature and are commonly applied
26 to estimate BC (Novakov 1982; Gundel et al., 1984; Liousse et al., 1993; Petzold et al., 1997; Bond and
27 Bergstrom 2006). There is also a common suggestion that an ideal solution would be to quantify BC in
28 light absorption terms, which is the strength of the optical techniques.

29 While the terms “black carbon or BC” and “elemental carbon or EC” are frequently associated
30 with measurements from the two general categories of specific commercial instruments in the scientific
31 literature, both of these measurement techniques provide estimates of black carbon concentrations
32 (Wolff et al., 1982; Andreae and Gelencsér, 2006). Ambient monitoring studies that simultaneously
33 utilized light adsorption and thermal-optical methods show that the estimates of BC by the two
34 techniques are generally within 30%. Ambient inter-comparison studies have found that estimates of
35 BC from thermal measurement methods are usually reliable predictors of ambient BC estimated via light
36 absorption techniques and vice versa. While the estimates from the two techniques are highly
37 correlated and display similar concentration values, they can vary by up to a factor of two among the

1 limited number of studies available². Further discussion of these comparisons can be found in Appendix
2 1.

3 **5.3 Ambient Concentrations of Black Carbon**

4 Observational data representing ambient BC concentrations is very limited. Currently, few
5 countries have robust networks for ambient measurement of PM_{2.5}. Most available global ambient BC
6 data are produced in the United States, Canada, Europe, and China, and the vast majority of these data
7 are based on the more widely available thermal measurement techniques (see Section 5.2). In the
8 United States and Europe, limited light absorption measurements are available to supplement these
9 thermal measurements. And there is also a modest network of BC monitoring sites across the globe in
10 remote areas to provide information about background levels.

11 **5.3.1 Major Ambient Monitoring Networks**

12 Figure 5-1 provides a map showing the extent of known BC monitoring networks around the
13 globe. The existing networks in the United States, Canada, Europe (EUSSAR, EMEP), and Asia
14 (CAWNET), as well as those with global coverage (GAW, ESRL/GMD) and ad hoc collections of special
15 study data are shown. The map separately shows locations using light adsorption, thermal or both
16 measurement techniques. Most locations shown are in North American and these monitors mostly
17 utilize thermal measurement techniques.

² Comparable studies of the relationship between measured estimates of BC from light absorption and thermal techniques have not been conducted for direct measurements of source emissions.

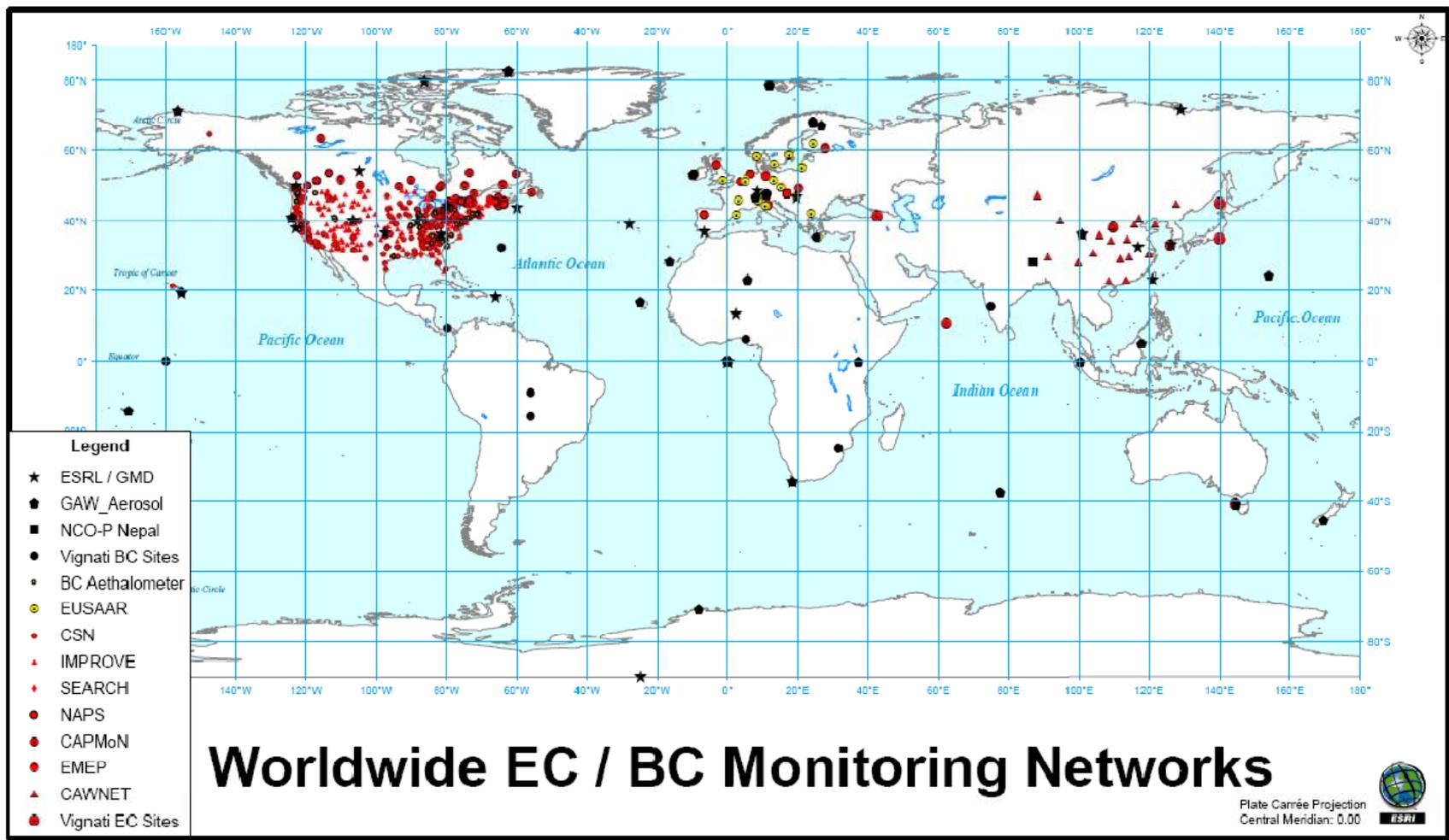


Figure 5-1. Ambient BC Measurement Locations Worldwide. Light absorption measurement locations are colored black. Thermal measurement locations are colored in red. A small subset of locations with both measurements is colored yellow.

1 Ambient BC data in the United States are mostly available from PM_{2.5} urban and rural speciation
2 monitoring networks which use thermal measurements. The Interagency Monitoring of Protected
3 Visual Environments (IMPROVE) network started collecting data in the late 1980s, and the urban
4 Chemical Speciation Network (CSN) started in the early 2000s. Urban BC is measured through the CSN
5 network of approximately 200 monitors located in major urban areas.³ In rural environments such as
6 national parks and wilderness areas, the United States relies on the IMPROVE network to characterize
7 air quality. This network consists of approximately 160 monitors. Like the CSN, the IMPROVE network
8 utilizes thermal measurement technologies.⁴ Other U.S. data includes supplementary measurements
9 from approximately 45 light absorption monitors (operational in 2007); 5 semi-continuous carbon
10 measurements; and smaller networks of thermal optical and light absorption monitors (SEARCH, Super-
11 sites). See Appendix 1 for more details.

12 **5.3.2 Global Ambient Concentrations**

13 Table 5-2 summarizes data from a number of studies and monitoring networks that help
14 illustrate the range of BC concentrations across the globe. The table also indicates the BC measurement
15 methods: [thermal (T) and light absorption (LA)] for each study/monitoring network. While BC
16 measurements for urban and rural areas are similar in North America and Europe, the reported
17 concentrations for China are much higher. Both urban and rural BC concentrations in China are
18 approximately 10 times higher than urban and rural concentrations in the United States, respectively.⁵

19

³ Measurements are based on integrated 24-hr samples, mostly collected every three days, and were mostly analyzed for EC between 2001-2007 using an EPA NIOSH-type TOT protocol. EPA started to transition CSN measurements to the IMPROVE_A TOR protocol for EC in May 2007.

⁴ Measured every three days. The IMPROVE program slightly modified the protocol in 2005, which resulted in higher quality data and slightly higher EC as a fraction of total measured carbon. The IMPROVE network data for 2005-2007 are produced using the newer IMPROVE_A TOR protocol.

⁵ As discussed in the Chapter 4, the ratio between China and U.S. measured BC concentration is 2 to 3 times higher than their national BC emissions.

1 **Table 5-2. Summary of Selected Global BC Ambient Concentrations for Urban and Rural/Remote Areas.**

Range of Annual Average Concentrations ($\mu\text{g}/\text{m}^3$)					
Region	Networks	Year	Method Type	Urban (# sites)	Rural/Remote (# sites)
U.S.	CSN ^a / IMPROVE ^b	2005-2007	T	0.3 to 2.5 (~200 sites)	0.1 – 0.6 (~150 sites)
	SLAMS ^k	2007	LA	0.3 to 3.0 (~ 45 sites)	
Canada	NAPS ^c	2003-2009	T	0.9 – 1.8 (12 sites)	0.4 – 0.8 (4 sites)
Europe	EMEP ^d	2002-2003	T	1.4 – 1.8 (2 sites)	0.2 – 1.8 (12 sites)
Europe	EUSAAR ^e	2006	T	1.5 (2 sites)	0.1 – 0.7 (4 sites)
			LA	2.7 (1 site)	0.2 – 0.5 (4 sites)
UK	BC Network ^f	2009	LA	1.0 – 2.9 (19 sites)	
China	CAWNET ^g	2006	T	9.3 – 14.2 (5 sites)	0.3 – 5.3 (13 sites)
Nepal	NCO-P ^h	2006-2008	LA		0.16 (1 site)
Global Background	NOAA GMD Sites ^j Mauna Loa Point Barrow South Pole	1990-2006	LA		0.01 – 0.02
		1988-2007	LA		0.02 – 0.07
		1987-1990	LA		0.002 – 0.004
Other Arctic Sites	Alert (Canada) Zeppelinfjell ^e (Svalbard, Norway)	1989-2008	LA		0.04 -- 0.1
		2002-2009	LA		0.02 - 0.06
UK	Black Smoke(BS) ⁱ	2006	LA	5.0 – 16.0 (12 sites)	

2

3 Notes:

4 a. CSN – Primarily urban network sites.

5 b. IMPROVE – Rural network sites.

6 c. Personal communication with Tom Dann (Environment Canada).

7 d. Monitoring was for the period 07/02 – 06/03 from Yttri et al., 2007. Elemental and organic carbon in PM10: a one year
8 measurement campaign within the European Monitoring and Evaluation Programme EMEP, *Atmospheric Chemistry and Physics* 7
9 (22):5711-5725. <http://www.atmos-chem-phys.org/7/5711/2007/acp-7-5711-2007.pdf>10 e. Data taken from <http://ebas.nilu.no/> For EUSAAR, the sites assigned to be urban are Ispra, IT (BC) and Melpitz, DE. Although not
11 part of EUSAAR, the urban sites also include Ring A10, NL (EC). The northern EUSAAR remote location of Zeppelinfjell, NO site is
12 included with other Arctic sites listed separately.13 f. Urban network sites from 2009 Annual Report for the UK Black Carbon Network (May 2010 Draft); Curbside site at London
14 Marylebone Road reported $10\mu\text{g}/\text{m}^3$.

- 1 g. Data and urban/regional/remote classification was for the period 2006 from Zhang et al, 2008. Carbonaceous aerosol composition
2 over various regions of China during 2006, *Journal of Geophysical Research*, 113, D14111.
- 3 h. Monitoring was for the period 03/06 – 02/08 from Marinoni et al, 2010. Aerosol mass and black carbon concentrations, two year-
4 round observations at NCO-P (5079m, Southern Himalayas), *Atmospheric Chemistry and Physics Discussions*, 10, 8379-8413.
- 5 i. Data taken from http://www.airquality.co.uk/reports/cat05/1009031405_2009_BC_Annual_Report_Final.pdf; curbside site at
6 London Marylebone Road reported an average of $\sim 40\mu\text{g}/\text{m}^3$ for each year.
- 7 j. NOAA Global Monitoring Division Sites – For this table, we modified reported numbers in absorption units using a nominal mass
8 extinction coefficient of $10\text{m}^2\text{g}^{-1}$. One year from each site was eliminated as non-representative.
- 9 k. BC data at State and Local Air Monitoring Stations from AQS, mostly with Magee Aethalometers.
- 10

11 The United Kingdom shows higher BC concentrations at the upper range than the United States
12 likely due to the influence of local sources on the individual monitoring sites. In general, roadside or
13 near-source monitors yield higher values, as demonstrated by the curbside monitors in London which
14 report considerably more BC than the urban-wide locations (Butterfield et. al, 2010). The “Black Smoke”
15 data for the UK that provide the basis for the five-decade trend discussed in Section 2.4.2 are 3 to 4
16 times higher than co-located measurements of BC (Quincey, 2007).

17 The global background sites that are part of the National Oceanic and Atmospheric Administration
18 (NOAA) network reveal BC concentrations that are 1 to 2 orders of magnitude lower than those typically
19 observed in either urban or rural continental locations. The presence of BC in these remote locations
20 without any nearby sources is indicative of long range transport and is used to evaluate intercontinental
21 transport processes in global model.

22 5.3.3 Comparison of Urban and Rural Concentrations Globally

23 Available data suggest that BC concentrations vary substantially between urban and rural areas.
24 Specifically, urban areas tend to have higher concentrations. The global BC data (for 2005-07 average
25 or calendar year 2006) displayed in Figure 5-2 contrast the annual average rural and urban
26 concentrations for North America, China, and Europe.^{6,7} The ambient rural concentrations provide an
27 indicator of regional background concentrations resulting from regional emissions and transported
28 aerosols. Levels in urban areas reflect the higher average concentrations resulting from the
29 combination of local emissions and regional emissions. The portion of urban concentrations due to local
30 emissions can also be described as the “urban increment” or “urban excess”.⁸

31 As demonstrated in Figure 5-2, urban BC measurements in North America are generally much
32 higher than the nearby regional background levels. This suggests that there can be a substantial

⁶The data are aggregated and displayed on the 1.9 x 1.9 degree resolution which is widely used by global climate models. This coarse grid does not allow us to see sharp gradients which tend to exist within urban areas. Also, note that these grid-based displays use a logarithmic scale to show the order of magnitude range of concentrations for BC across the globe.

⁷The map shows the 40th parallel, the importance of which is discussed further in Chapter 4.

⁸Because of strong regional homogeneity among background measurements, urban grid squares without measurements were estimated from nearby cells to permit an estimate of urban excess. These estimated values may be higher than surrounding regional measurements. Spatial interpolation here is based on inverse distance weighting of the nearest neighbors (Ref: Abt Associates, 2005).

1 increment of local emissions in urban areas. For the period 2005 to 2007, the urban increment ranged
2 from zero to 2.2 $\mu\text{g}/\text{m}^3$ (i.e., up to 92% of the total urban BC concentrations). In general, average urban
3 concentrations are relatively consistent across North America, though the larger populated regions of
4 the eastern United States, eastern Canada, and California contain most of the highest concentrations.
5 However, western United States and western Canada have lower regional background concentrations
6 and therefore relatively larger urban increments, while higher rural concentrations in the eastern North
7 America result in smaller urban increments (more similar regional and urban average values). The
8 higher regional background levels across the eastern North America suggest higher and more consistent
9 levels of BC emissions from sources across the region, and/or greater transport from clustered cities to
10 surrounding rural areas.

11 Figure 5-2 also shows that Europe's measurement data are quite similar to those for North
12 America. However, both China's regional and urban BC concentrations are much higher than those seen
13 in North America and Europe and its urban increments are approximately four times larger. This can be
14 attributed in part to larger urban and regional emissions sources in China compared to North America
15 and Europe.

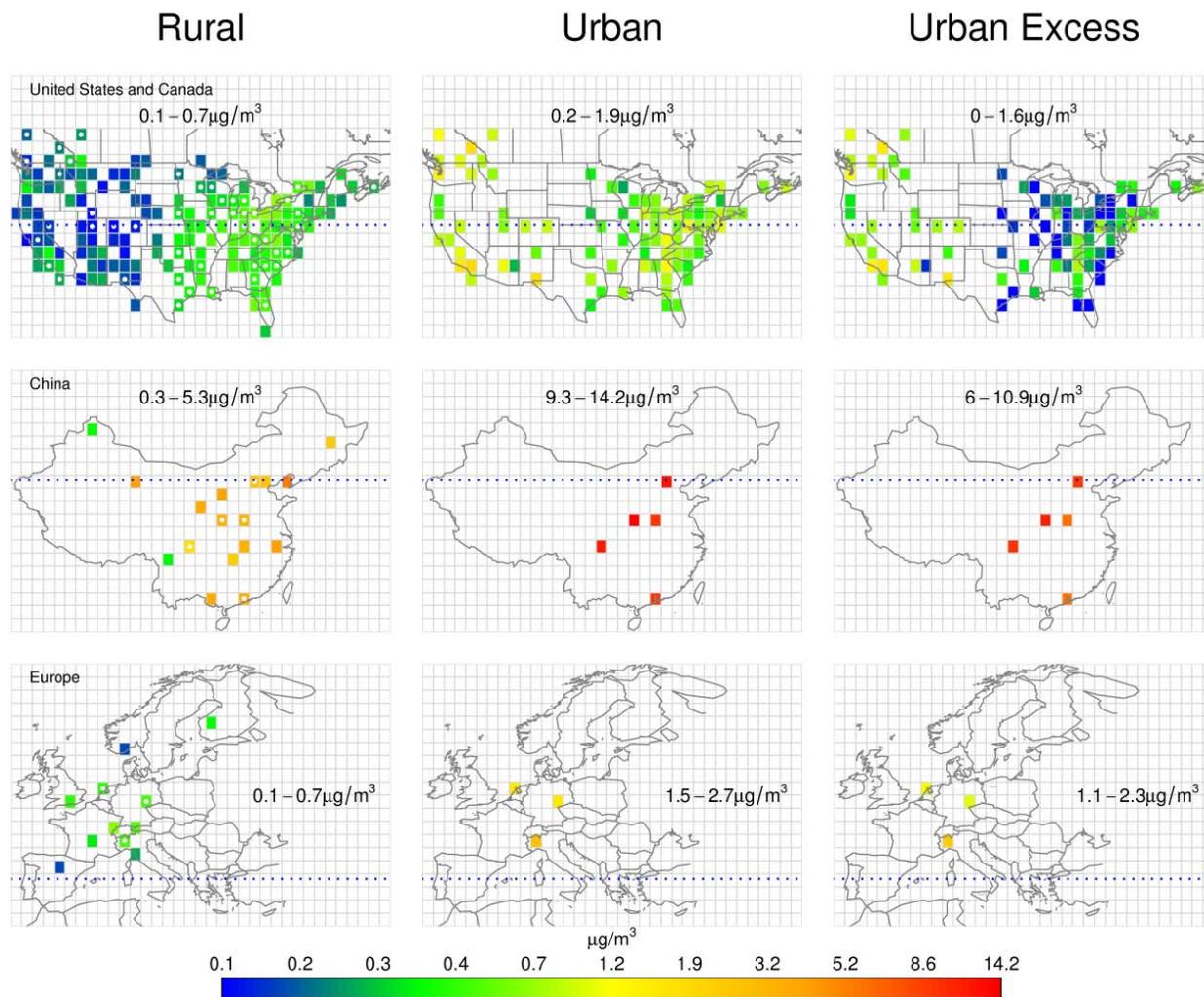
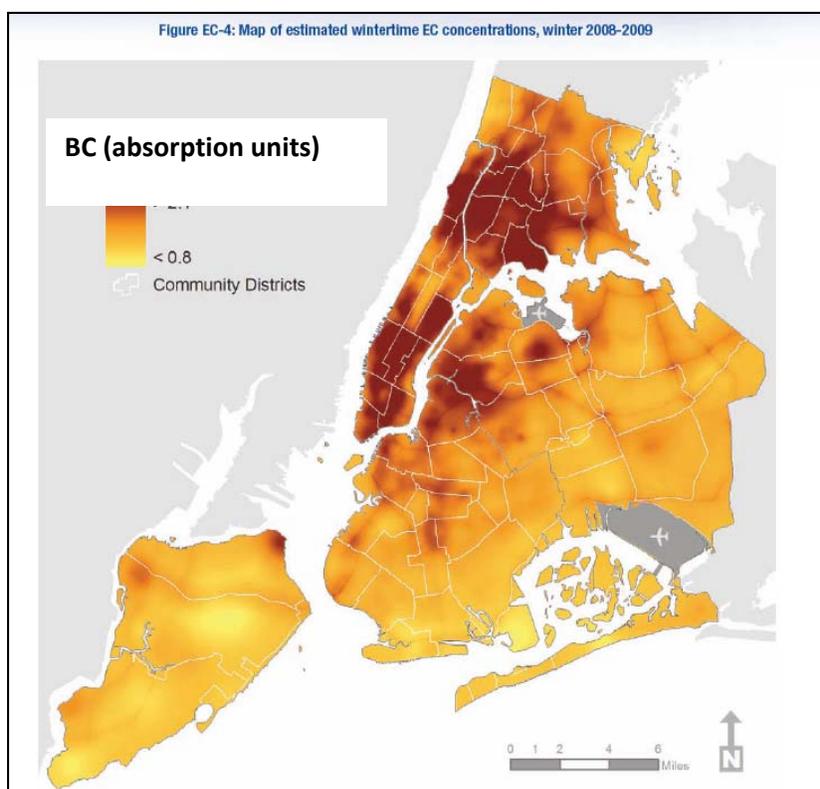


Figure 5-2. Spatial distribution of global BC data. Rural, urban, and urban excess concentrations for 2005-2007. Grid squares with a white dot represent estimated rural concentrations from spatial interpolation of the nearest neighbors with measurement data. The 40th parallel is shown as a dotted line.

1 In addition to the differences in urban and rural BC concentrations, there can be substantial
 2 spatial variation in BC concentrations within a given city. Because global representations of BC
 3 concentrations are typically based on limited monitoring locations and are generally presented as
 4 average concentrations (often across monitors hundreds of miles apart), it is important to realize that
 5 ambient concentrations of BC in any urban area can vary widely from location to location within the city.
 6 BC concentrations can vary spatially within an urban area because the magnitude of monitored BC
 7 concentrations is dependent on the proximity of the monitor to roadways and other nearby sources.
 8 Therefore, concentrations measured may not be representative of other locations. Figure 5-3 illustrates
 9 the estimated spatial variability of BC in New York City⁹. This special study used 150 monitoring sites to
 10 reveal large gradients in BC concentrations. While most of the high concentration zones can be
 11 attributed to mobile source emission density, this study also revealed significant BC emission sources
 12 associated with residential oil combustion.



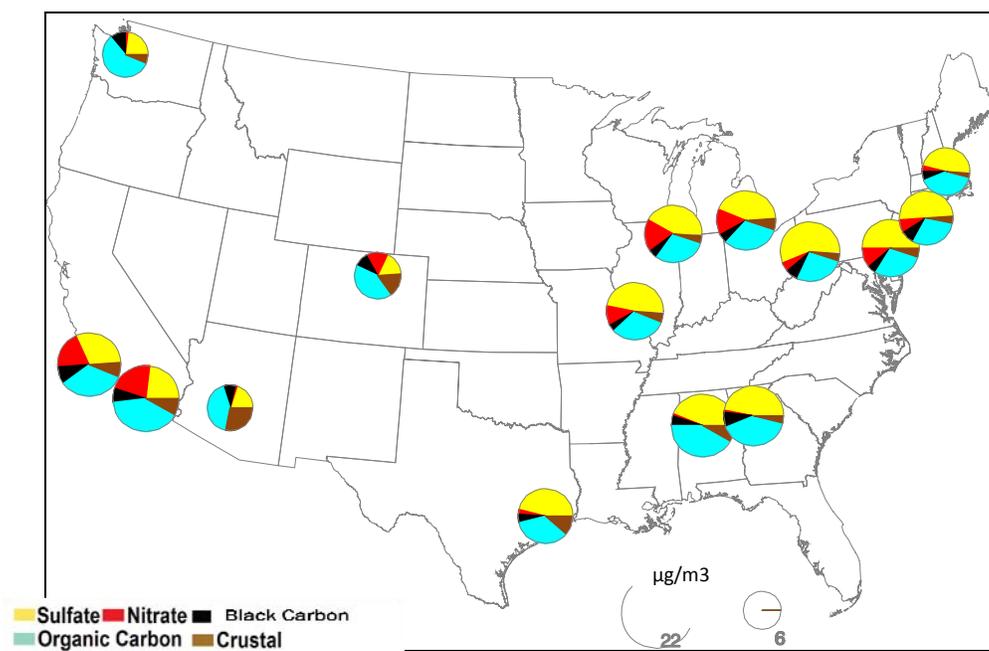
13

14 **Figure 5-3. Urban Gradients in BC for NYC.** (The New York City Community Air Survey, Results from
 15 Winter Monitoring 2008-2009,
 16 http://www.nyc.gov/html/doh/downloads/pdf/eode/nyccas_master_report_12_15_09.pdf)

⁹ Based on 150 filter-based portable samplers and optical absorption measurements with the smoke stain reflectometer.

1 **5.3.4 Black Carbon as a Percentage of Measured Ambient PM_{2.5} Concentrations in the**
 2 **United States**

3 Because total PM_{2.5} mass is the basis for regulation of fine particles in the United States and also
 4 serves as the basis for BC emission estimates, it is informative to estimate the contribution to total
 5 PM_{2.5} mass from BC. However, given the limited BC data available on a global scale, this evaluation is
 6 based solely on data for urban areas in the United States that are regionally representative of large U.S.
 7 cities. Compared to U.S. rural locations, urban locations contain a higher percentage of BC and OC.
 8 While urban nitrate concentrations are also higher than surrounding rural areas, carbonaceous aerosols
 9 are responsible for most of the urban PM_{2.5} increment. Other components, such as dust, are similar in
 10 both urban and rural environments (U.S. EPA, 2004). Figure 5-4 shows the BC fraction of PM_{2.5} mass for
 11 15 selected U.S. urban areas. The values represent average concentrations among monitoring locations
 12 in the area. The average BC concentrations range from 0.6 µg/m³ in St. Louis to 1.2 µg/m³ in Atlanta.
 13 The percentage of PM_{2.5} that is BC ranges from 4% in St. Louis to 11% in Seattle.¹⁰



14
 15 **Figure 5-4. Composition of PM_{2.5} for 15 Selected Urban Areas in the United States.** Annual
 16 average PM_{2.5} concentrations (µg/m³) are presented where the circle size represents the
 17 magnitude of PM_{2.5} mass. The BC and Organic Carbon Mass (OM) fractions are illustrated. OM

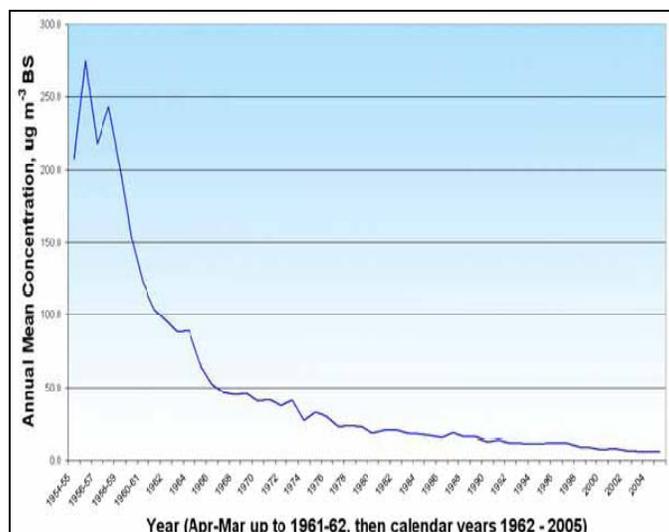
¹⁰ Approximately 20-80% of the estimated ambient organic matter (OM) is directly emitted. [Carlton, 2009] The other portion, termed secondary organic aerosol (SOA), is formed through chemical reactions of precursor emissions after being released from the sources [Chu (2004), Cabada (2004), Saylor (2006), Carlton, 2009]. OM is typically 1.4 to 1.8 times higher than measured OC levels in urban areas, with an even larger multiplier of OC levels measured in rural areas [Turpin (2001), Bae (2006)]. The OM-to-OC ratio tends to be higher with an aged aerosol (resulting from transported, atmospheric-processed, and aged particles), SOA, or directly emitted OM from biomass combustion. Although we are not able to quantify the amount of OM that may be LAC, it is worth noting that average OM for the 15 selected cites represents 26% to 55% of PM_{2.5} and the OM-to-BC ratio ranges from 4 to 9.

1 represents OC together with its associated non-carbonaceous mass (e.g., hydrogen, oxygen and
 2 nitrogen), estimated by a material balance approach (Frank, 2006).

3 5.4 Trends in Ambient BC Concentrations

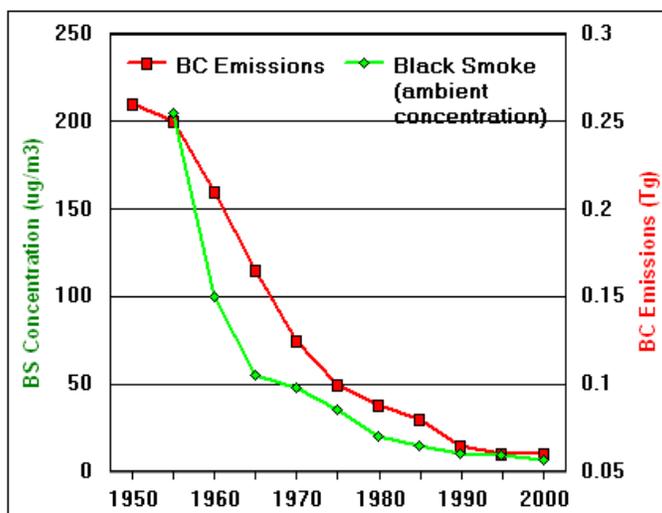
4 5.4.1 Trends in Ambient BC Concentrations in the United States and the United Kingdom

5 Measurement data necessary for assessing long-term ambient trends in BC are limited even for
 6 the areas with currently robust monitoring programs such as the United States.¹¹ However, although
 7 limited, some information on changes in ambient concentrations over longer time periods is available
 8 and these data are useful in evaluating and corroborating emissions trends. Since most BC is directly
 9 emitted rather than the formed chemically from precursors in the atmosphere, ambient BC
 10 concentrations respond directly to emissions changes. Figure 5-5a shows the dramatic reduction in
 11 measured “Black Smoke” (BS) in the UK since the 1950s. This dramatic decline is attributable to a
 12 number of factors, including fuel switching, the introduction of cleaner fuels and technologies, and
 13 successful smoke control legislation (*Air Pollution in the UK, 2008*). Figure 5-5b overlays these BS
 14 measurements and estimated BC emissions for the UK for the same time period revealing a
 15 corroborating similarity in temporal patterns.



16
 17 **Figure 5-5a. Trends in Black Smoke Measurements in the United Kingdom, 1954-2005.** The BS
 18 measurements are highly correlated with optical BC, although BS is 3 to 4 times higher than BC
 19 under current U.K. aerosol conditions.

¹¹ Assessment of longer term trends in BC is possible by analyzing ice core and lake sediment data. These data reflect historical archives from which BC concentrations can be estimated and used to supplement more recently available direct ambient air quality measurements. A discussion of these data and the corresponding results is the focus of section 5.6 of this chapter.



1

2

3

Figure 5-5b. Comparison of Ambient Black Smoke Measurements in the United Kingdom with Estimated BC emissions.

4

5

6

7

8

9

10

11

12

13

14

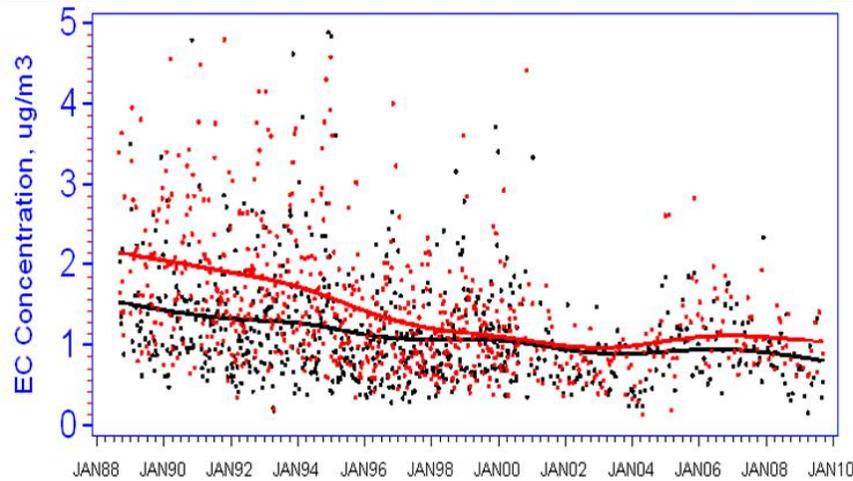
15

16

17

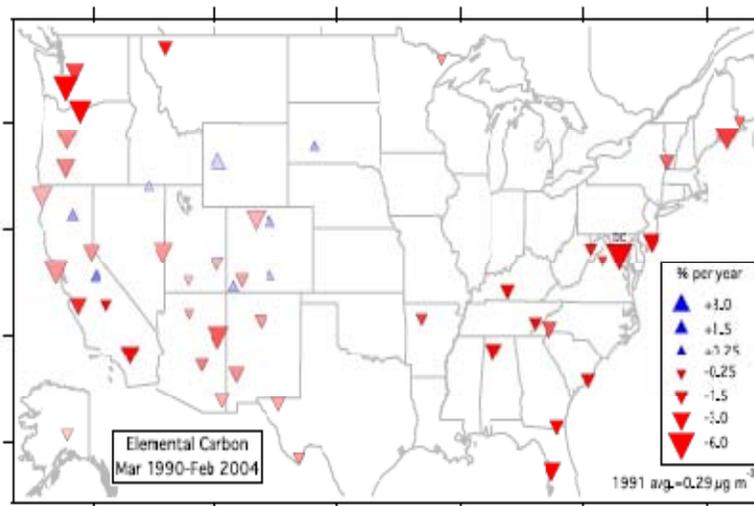
18

For more recent time periods, there is a great deal more data available to assess ambient trends in the United States than there is for longer-term historical trends. A variety of measurements from the IMPROVE and CSN networks, as well as other monitoring locations, provide important data for assessing recent changes in ambient BC concentrations domestically. Figure 5-6 shows the 1988-2009 (22-year trend) for BC in Washington D.C. as measured by the IMPROVE program. IMPROVE’s urban Washington, D.C. monitoring site has one of the longest BC monitoring records in the United States. These data show a substantial decline during the 1990s, followed by a more level trend over the past 5-10 years. The higher BC concentrations and more substantial BC reductions during the week as compared to the weekend may correspond to the influence of the reduction in diesel emissions. Nationwide reductions in average BC concentrations have also been observed in rural areas during this same time period (Figure 5-7). Concentrations in the United States decreased by over 25% between 1990 and 2004. Although not shown in this figure, percentage decreases were much larger in winter suggesting that emissions controls have been effective in reducing concentrations across the entire United States [Murphy, et al, 2011]. Some of the largest annual average decreases in rural areas occurred in California where 50% reductions from 1989 to 2008 are reported. [Bahadur, 2011]



1
2
3
4
5
6
7

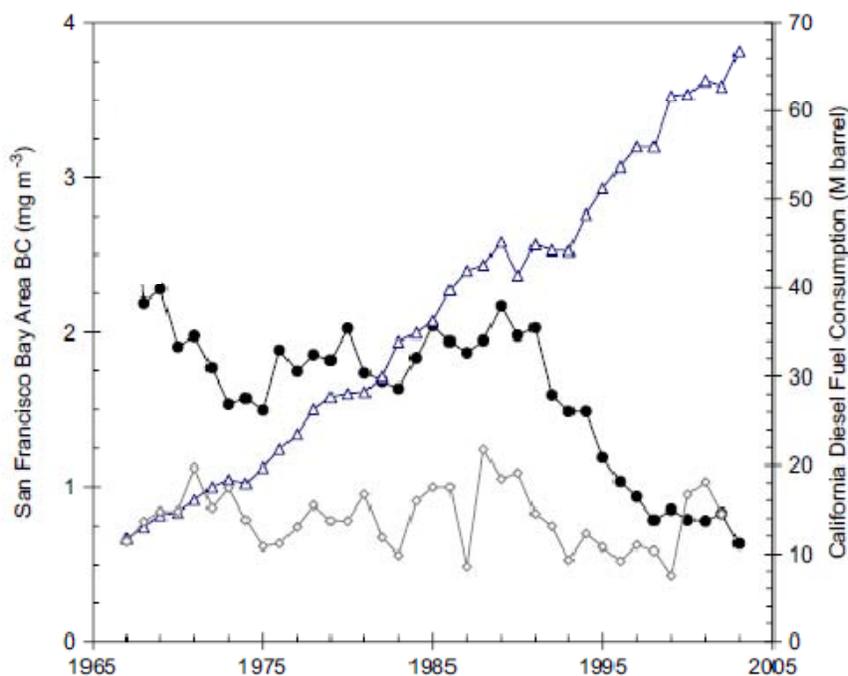
Figure 5-6. Ambient BC Trends in Washington D.C. The red data points and line represent measurements from Wednesdays (as a proxy for weekdays) and the black data points and line represent measurements from Saturdays (as a proxy for weekends). This monitoring site changed its sampling protocol from twice per week (Wednesday and Saturday) to once every 3 days in September 2000. The apparent small increase in BC concentration after January 2005 is due to a change in analytical protocol.



8
9
10
11
12
13
14
15
16

Figure 5-7. Trends in BC at All IMPROVE Network Stations with Sufficient Data between 1 March 1990, and 29 February 2004. Marker size indicates the magnitude of the trend. Triangle direction and blue or red color corresponds to the sign of the trend. Color saturation is proportional to the average concentration in 1991 with full saturation at twice the national median. The only urban site in Washington DC is marked. Averages in the bottom right corner exclude Washington DC. [Adapted from Murphy et al., 2011]

1 Figure 5-8 juxtaposes estimated annual average BC concentrations in the San Francisco Bay with
 2 annual consumption of diesel fuel in California (Kirchstetter et al., 2008).¹² Kirchstetter notes that the
 3 contrast in the trends in BC concentration and diesel fuel use is striking, especially beginning in the early
 4 1990s when BC concentrations began markedly decreasing despite sharply rising diesel fuel
 5 consumption. This contrast suggests that control technologies to reduce BC emissions have been
 6 successful (see Chapter 7). Similarly, Figure 5-9 shows a data set from Boston, MA which displays a
 7 decline in BC concentrations during the period 2000-2007. The decline of BC concentration at this site
 8 has been attributed to diesel retrofits in Boston, but is no doubt also reflective of fleet wide changes in
 9 emissions especially due to diesel emissions standards (U.S. EPA, 2004).



10

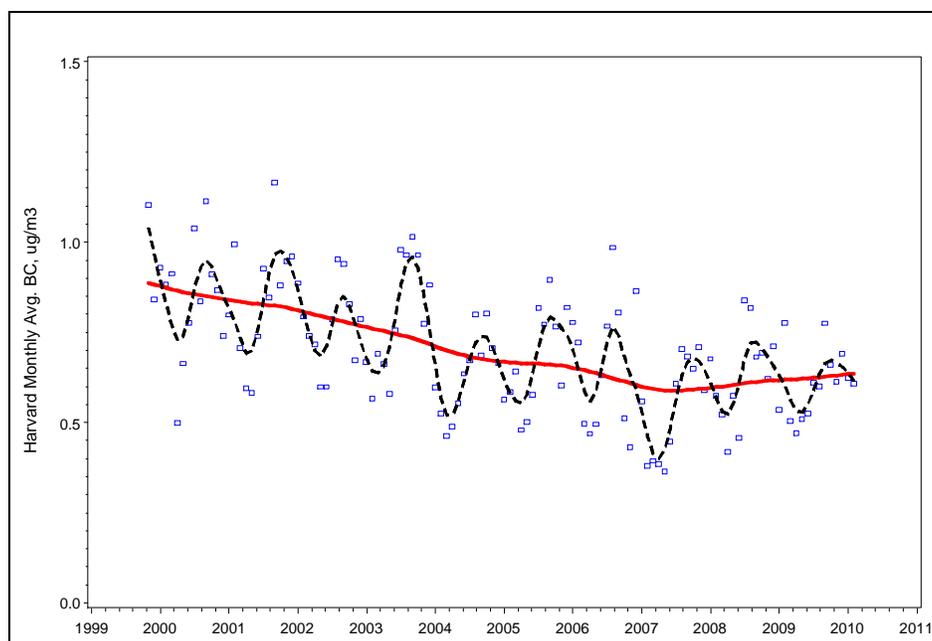
11

12

13

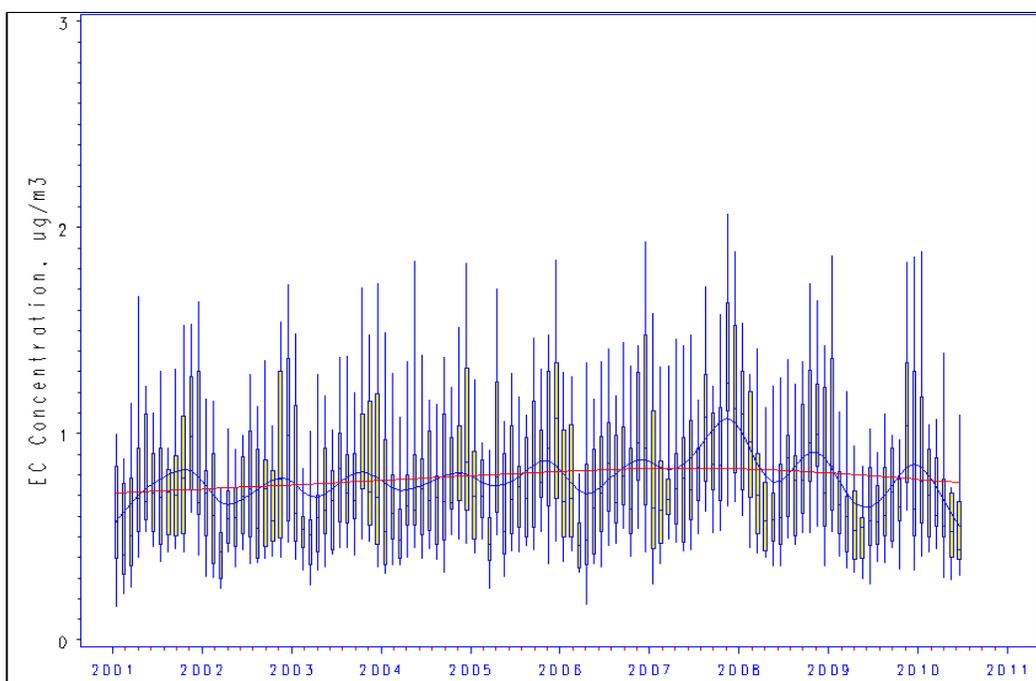
Figure 5-8. Estimated annual average ambient BC concentrations in the San Francisco Bay.
 Area vs. diesel fuel sales (Kirchstetter et al., 2008). BC is shown as black dots. California on-road
 and nonroad diesel fuel sales are shown as triangles and diamonds.

¹² BC was estimated using Coefficient of Haze measurements which are shown to be highly correlated with optical BC. See Appendix 1 for further details regarding COHs.



1
 2 **Figure 5-9. Ambient BC trends in Boston (Harvard School of Public Health location).** BC derived
 3 with aethalometer.

4
 5 Figure 5-10 shows that BC concentrations have not substantially changed, on average, for an 18-
 6 site subset of EPA’s national urban CSN network sites with the longest historical record. The figure
 7 shows the range of monthly average concentrations and the typical seasonal pattern among the group
 8 of 18 continental U.S. locations. In general, the CSN sites are representative of neighborhood, urban-
 9 wide, and regional-scale emissions and may not necessarily reflect local scale emission changes. The
 10 apparent decrease between 2007 and 2010 may be due in part to the CSN transition to a different
 11 carbon monitoring protocol and different samplers. However, based on a somewhat limited comparison
 12 of parallel carbon sampling at 11 large urban areas using the old and new EPA monitoring protocols,
 13 average EC concentrations were quite similar (see Appendix 1). Thus the BC decline over the past 3 years
 14 in urban areas may be real and due to the effects of the recent economic recession which has resulted in
 15 reduced vehicle miles traveled and industrial activity.



1

2

Figure 5-10. Ambient BC Trends (2001-2010), based on monthly distribution of average concentration among 18 CSN monitoring locations in the United States.

3

4

Based on the evidence provided above from a variety of recent BC indicator measurements (and derived from different data sources), it appears that ambient concentrations of neighborhood/urban and regional scale concentrations of BC in the United States declined from the mid 1980s until the early part of the 2000s. Further BC reductions in United States appear to have occurred since 2007 due to recession related decreases in vehicular travel and industrial output. However, this finding cannot be confirmed with the limited emissions estimates available for 1990 and 2005.

8

9

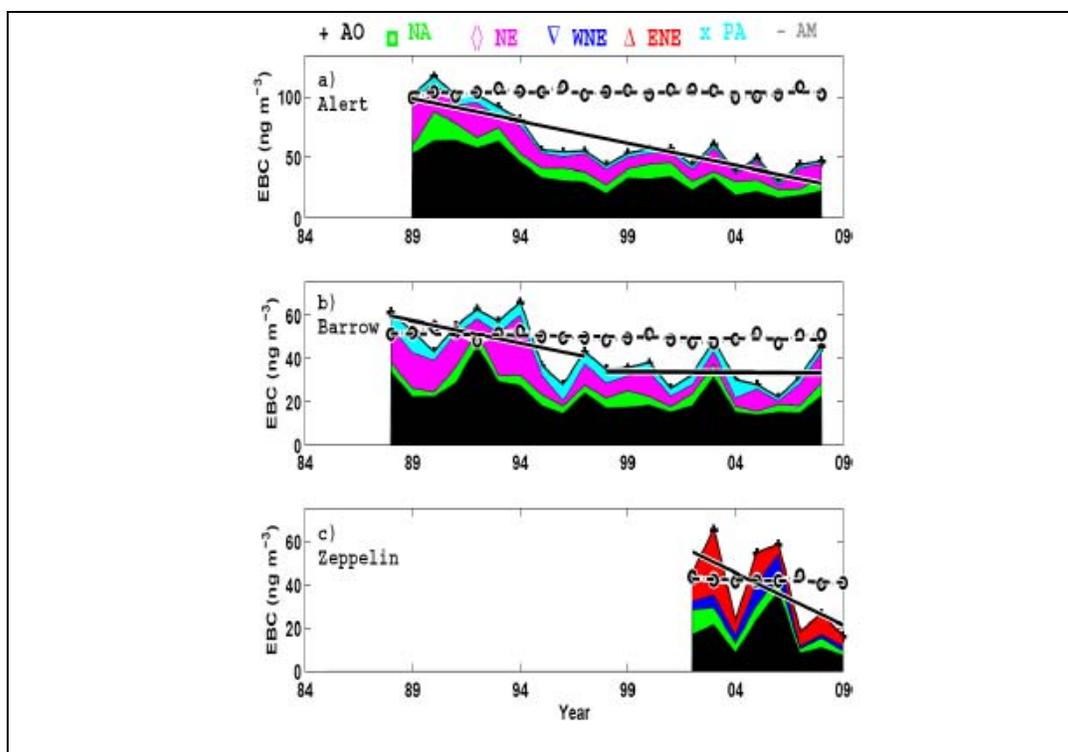
10 **5.4.2. Trends in Ambient BC Concentrations in the Arctic**

11

Trends in BC concentrations at three Arctic locations (Alert Canada, Barrow Alaska, and Zeppelin Norway) are presented in Figure 5-11 (Hirdman, 2010). As stated by the authors, there is a general downward trend in the measured BC concentrations at all stations, with an annual decrease of -2.1 ± 0.4 ng/m^3 per year (for the years 1989-2008) at Alert and -1.4 ± 0.8 ng/m^3 per year (2002-2009) at Zeppelin. The decrease at Barrow is not statistically significant. Based on transport analysis the authors conclude that northern Eurasia (the NE - Northeast, WNE - West Northeast, and ENE - East Northeast clusters) is the dominant emission source region for BC and decreasing emissions in this region drive the downward trends. However, there are indications that the BC emissions from ENE in wintertime have increased over the last decade, probably reflecting emissions increases in China and other East Asian countries. Emissions associated with the other clusters (Arctic Ocean - AO, North America - NA, Pacific-Asia - PA, and west northeast Eurasia - WNE) have been stable or decreasing over the time periods in this study.

20

21



1
2 **Figure 5-11. The Annual Mean BC Concentrations Measured at Alert (a), Barrow (b), and Zeppelin (c)**
3 **and Split into Contributions from the Four Transport Clusters.** The annual mean concentrations
4 measured at Alert(a), Barrow (b), and Zeppelin (c) are split into contributions from four transport clusters.
5 The solid line shows the linear trend through the measured concentrations. The circles show the annual
6 mean BC concentrations when the cluster-mean concentrations are held constant over time (means over
7 the first three years). This line is influenced only by changes in the frequencies of the four clusters. The
8 dashed line shows the linear trend of these data. [Hirdman, 2010]

9 5.5 Remote Sensing Observations

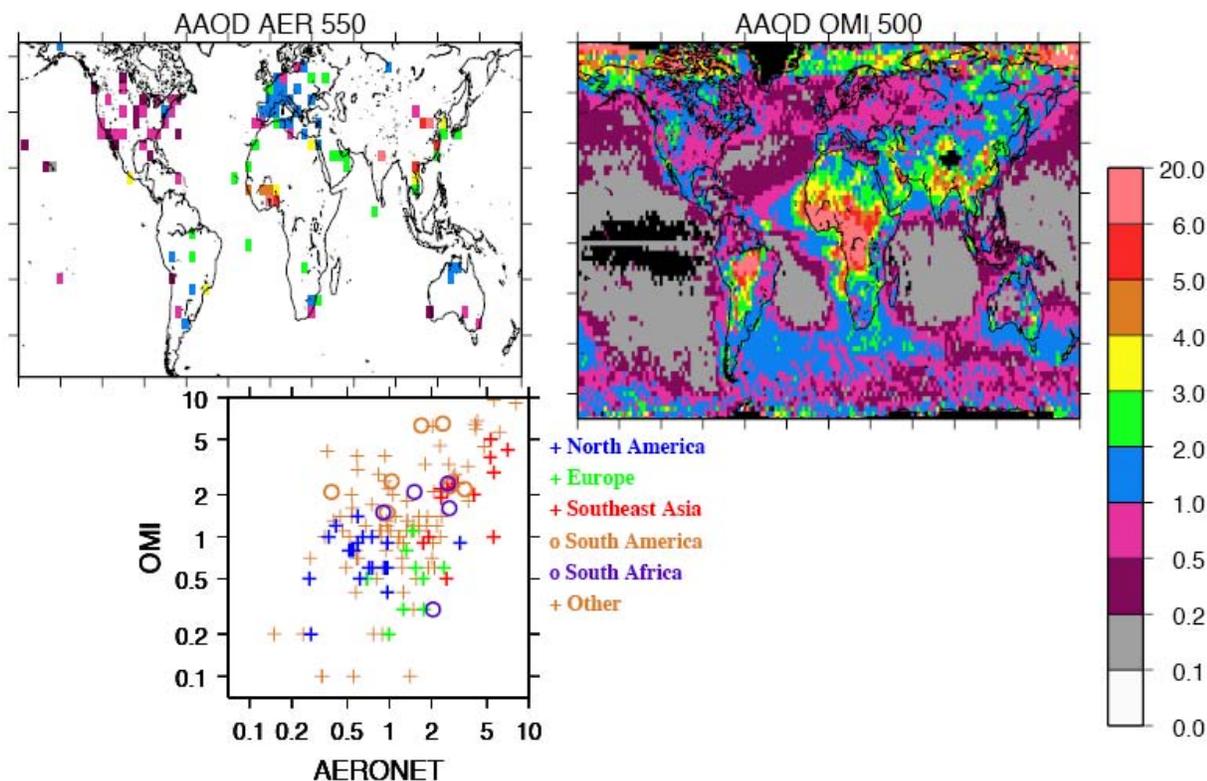
10 Measurements from satellite and ground-based remote sensing are useful in describing global
11 aerosol and, in particular, black carbon absorption. Satellites systems designed with aerosol remote
12 sensing capability include MODIS and MISR on Terra and Aqua, as well as GLAS and CALIPSO lidars which
13 describe aerosol layer heights and other satellite instruments such as the Total Ozone Mapping
14 Spectrometer (TOMS). The ground-based remote-sensing Aerosol Robotic Network (AERONET) has
15 provided information on aerosol distribution, seasonal variation and absorption properties since 1963
16 (Holben, 1998; Kahn, 2007; Kahn, 2009, Kahn, 2010; Kazadzis, 2009; Winkler, 2007).

17 Unlike spatially-discrete ambient BC monitors, remote sensing observations are global and thereby
18 offer greater spatial surface coverage of BC levels and provide important estimates of BC where surface
19 ambient measurements are not available. In addition, remote sensing is not limited to characterizing
20 surface concentrations but provides important information on differences in concentrations in BC and
21 aerosols throughout the total atmospheric column. Combining these new data sources with traditional
22 ground based (ambient) measurements has been used to derive the complete aerosol effect on the
23 environment and climate (Falke, 2001; Husar, 2011). Integrated data sets of aerosol based extinction

1 have relied heavily on AERONET sun photometer measurements in remote locations with low
2 concentration and relatively homogeneous aerosol [Kaufman et al 2001], while downwind of pollution
3 or dust sources they have relied on MODIS characterization of the aerosol spatial distribution over the
4 ocean and dark surfaces [Remer et al., 2002b] and on TOMS over bright surfaces [Torres et al 2002b].

5 AERONET derived estimates of total column aerosol optical depth (AOD) at 4 wavelengths (440,
6 670, 870 and 1020) can further characterize other aerosol optical properties, including an estimate of
7 Aerosol Absorption Optical Depth (AAOD) throughout the absorption spectrum (Holben et al., 1998;
8 Dubovik and Kings, 2000). Similarly, aerosol measurements from the Ozone Monitoring Instrument
9 (OMI) of TOMS also provide a measure of AAOD.

10 Koch compares estimated AAOD for 1996–2006 based on AERONET with OMI satellite retrievals
11 for 2005–2007 (Torres et al., 2007). The two data sets broadly agree with one another. However, the
12 OMI estimate is larger than the AERONET value for South America (with UV sensitive biomass
13 combustion) and smaller for Europe and Southeast Asia which are dominated by BC. The AERONET
14 AAOD and OMI observations qualitatively agree with the ground level concentrations of BC for the
15 United States, Europe and Asia presented in Figure 5-12, and clearly increases the spatial
16 characterization of aerosol absorption. As discussed below, aerosol absorption may not necessarily be
17 associated with anthropogenic source emissions.

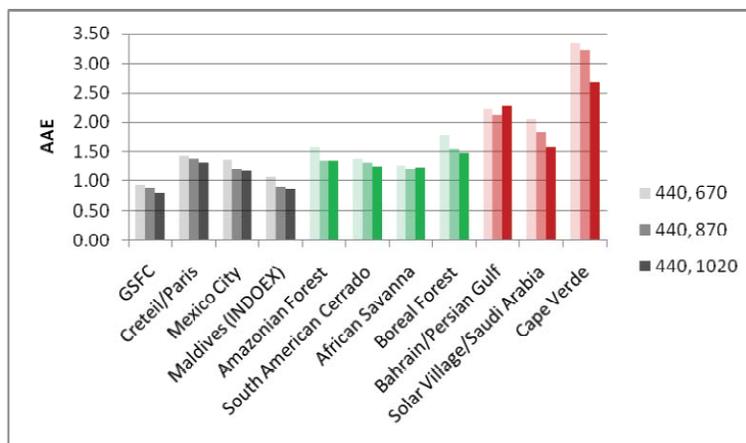


1
 2 **Figure 5-12. Aerosol Absorption Optical Depth (AAOD) from AERONET (1996-2006) and OMI (2005-**
 3 **2007).** Top: Aerosol absorption optical depth, AAOD, (x100) from AERONET (at 550 nm; upper left), OMI
 4 (at 500 nm; upper right). Bottom: scatter plot comparing OMI and AERONET at AERONET sites. (Koch,,
 5 2010)

6 Multi-wavelength instruments, such as AERONET can also characterize the wavelength
 7 dependence of absorption (often expressed as Absorption Angstrom Exponent, or AAE) to provide an
 8 indicator of the absorbing aerosol mixture. Using pairs of wavelength specific absorption
 9 measurements, Russell et al. find AAE values near 1 (the theoretical value for black carbon) for
 10 AERONET-measured aerosol columns dominated by urban-industrial aerosol, larger AAE values for
 11 biomass burning aerosols, and the largest AAE values for Sahara dust aerosols. Using these observations
 12 from multi-wavelength sensors can help distinguish the types of absorbing aerosols (Figure 5-13). It also
 13 demonstrates that the global AAOD observations presented in Figure 5-12 do not exclusively represent
 14 BC from anthropogenic sources¹³.

15

¹³ The illustrative remote sensing observations presented in Section 5.5 will be considerably strengthened when geostationary GLORY satellite with broad spectrum solar sensors to determine the global distribution of aerosol and cloud properties is deployed. Glory will provide 9-wavelength single-scattering albedo (SSA) and aerosol optical depth (and therefore aerosol absorption optical depth (AAOD) and AAE), as well as shape and other aerosol properties. [Mishchenko, 2007; Russell, 2010].



1

2

3

4

5

Figure 5-13. Absorption Angstrom Exponent (AAE) Values for AAOD Spectra Derived from AERONET Data. Black: Urban/Industrial or Mixed; Green: Biomass Burning; Red-Brown: Desert Dust. Shading for each location indicates wavelength pair (in nm) for AAE calculation. GSFC=Goddard Space Flight Center, Greenbelt, MD. (Russell 2010)

6

7

8

9

10

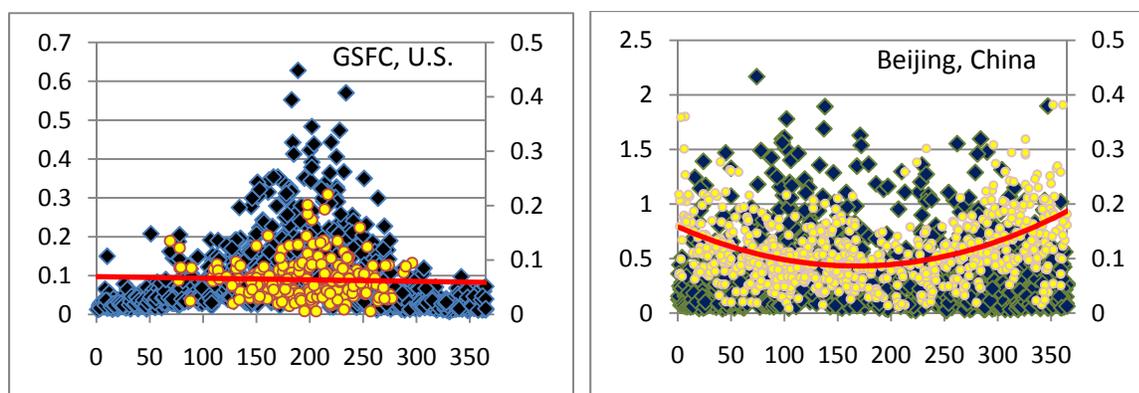
11

12

13

14

While a common limitation of remote sensing (which depends on solar light) is its general representativeness of day-time and cloudless sky conditions, AAOD is additionally only representative of higher extinction periods required to make the needed absorption calculations. Consequently, AAOD for the United States and Europe is not based on measurements during the winter when atmospheric extinction is lower than the minimum computational threshold. Similarly, AAOD are not as well represented during the monsoon periods in Asia when AOD measurements are not available. These issues may be partially addressed by using seasonally- or monthly-weighted averages. Figure 5-14 illustrates the issue of incomplete data records and the contrast of AAOD levels across the globe.



1
2 **Figure 5-14. AERONET AOD and AAOD as a Percentage of AOD.** Left: GSFC in eastern United States.
3 Right: Beijing, China. May 2005 to April 2009. Blue diamonds represent AOD (at 870nm) shown on left
4 vertical axis and yellow circles represent AAOD percent of AOD on the right vertical axis. Horizontal axis
5 shows the day of the year from 1 to 366. The red line is an estimate of the seasonal pattern of the %
6 AAOD as derived from the available observations. Note that Winter-time AAOD observations are not
7 available for GSFC and similar locations with low AOD values. (AERONET are based on Version 2, Level 2.0
8 inversion products with permission of Brent Holben, NASA and Hong-Bin Chen Chinese Academy of
9 Sciences.)

10 Figure 5-14 presents AAOD as a percentage of AOD by date for two example locations (GSFC in
11 the eastern United States, and Beijing, China) from AERONET. It is apparent that winter time values of
12 AAOD are not available for GSFC due to low AOD. This is typical of the eastern United States and any
13 location where high AOD results from secondarily formed scattering aerosols in the summer. Average
14 percent AAOD for GSFC is ~6% AOD, but this only represents the warmer months of the year. On the
15 other hand, AOD is sufficiently high year-round in Beijing, and AAOD ranges from ~10% in the summer
16 to 20% in the winter. Without estimates of winter-time AAOD for GSFC, the estimate of annual average
17 AAOD and the ratio to Beijing is very uncertain. Understanding these issues and any bias they may
18 result in for reported AAOD is critical if these data are used to directly corroborate model estimates
19 unless models similarly limit their calculations to the same portions of the observational record.
20 However, it is worth noting these data do suggest that Beijing's estimated summertime AAOD is ~ 10 xs
21 higher than that for GSFC. This exactly matches the ratio seen for ground level BC measurements in
22 Figure 5-2 further reinforcing the value of remote sensing estimates to expand the spatial extent of
23 ground level measurements for model evaluation and corroboration of emission inventories.
24 Interestingly, these data also suggest that China's BC emissions (as indicated in Chapter 4 as only four
25 times the emissions in the United States) may be underestimated and not reflect emissions growth by a
26 factor of 2.5 as suggested elsewhere (Koch, 2010). However, as stated in Chapter 2, the exact cause for
27 these differences has not been isolated (Bond, 2011).

28 **5.6 Black Carbon Observations from Surface Snow, Ice Cores, and Sediments**

29 Snow and ice cover approximately 7.5-15% of Earth's surface, depending upon the time of year
30 (Kukla and Kukla, 1974). The sunlight that reaches the snow surface typically penetrates about 10-20 cm
31 into the snow, with the topmost 5 cm receiving the most sunlight and where light-absorbing impurities
32 can significantly alter the amount of solar energy reflected by the snowpack (e.g., Galbavy et al., 2007).

1 Black carbon measurements in snow, and related surface reflectivity measurements, are critical to
2 accurately estimate climate forcing due to snow-bound black carbon. In addition, ice core
3 measurements of BC provide an important record of natural and anthropogenic black carbon emissions
4 transported to snow-covered regions. Lake and marine sediments also pose an opportunity to derive
5 historical trends in BC emissions prior to the point of time when air monitoring data are available.

6 **5.6.1. Measurement Approach**

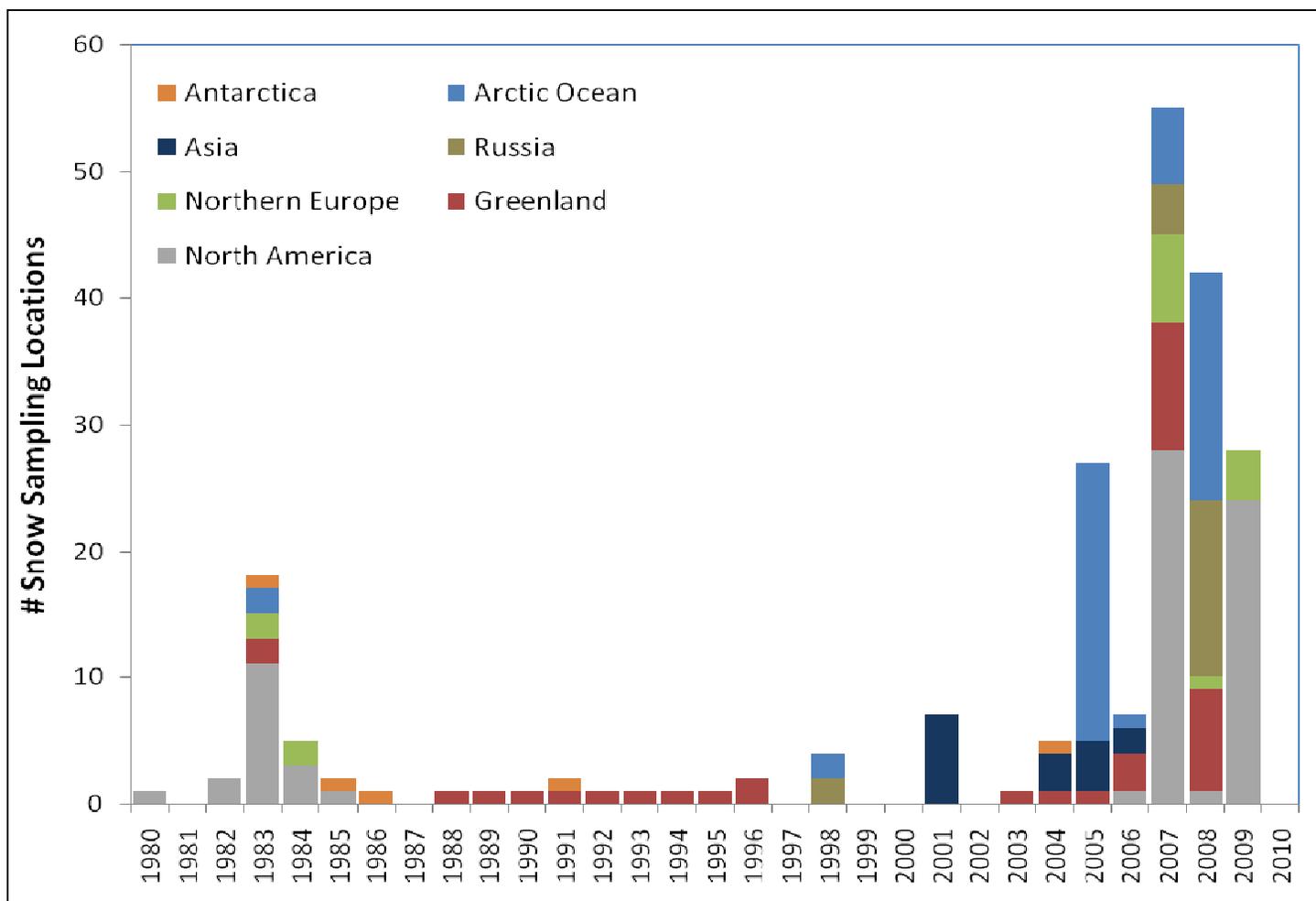
7 Measurement of BC in snow or ice is a laborious process that initiates with careful manual
8 collection of snow or drilling an ice core. A sample of snow or ice is then melted and BC is quantified
9 through several analytical approaches. The majority of researchers filter the melted snow or ice,
10 collecting BC to the filter matrix and estimating BC by observing how light at certain wavelengths are
11 absorbed by the particles (Grenfell et al., 1981; Clarke and Noone, 1985) or through a thermal or
12 thermal-oxidative process (Ogren et al., 1983; Chylek et al., 1987). In addition, one newer approach
13 avoids filtering the snow and quantifies BC by laser-induced incandescence (McConnell et al., 2007).
14 The mass of the sample meltwater is measured and the final concentration units are usually in mass of
15 BC per mass of snow or ice (e.g., ng BC/g snow).

16 Quantification of BC in sediments is an emerging field of study. The measurement technique is
17 more complex than for snow or ice samples, as BC particles are embedded in sediment material that
18 contains significant amounts of organic material. The sampling process usually involves extracting a
19 sediment core and then slicing the core into layers. The BC particles are subsequently isolated for a
20 given sample by applying a series of chemical and/or thermal treatments designed to remove non-BC
21 material (Smith et al., 1973; Lim and Cachier, 1996; Khan et al., 2009). Once the non-BC material is
22 removed to the degree possible, BC concentrations are quantified via similar techniques utilized in ice
23 core or ambient samples – measured by light absorption or through thermal processes. Microscopic
24 analysis of carbon particles has also been employed to qualitatively determine the source type from the
25 particle shape and surface texture (Smith et al., 1973; Kralovec et al., 2002).

26 **5.6.2. Surface Snow Data**

27 Measurements of black carbon in the shallow surface layer of snow have been conducted since
28 the 1980s by research teams at locations throughout the Northern Hemisphere and in Antarctica,
29 although the measurements were sporadic (Figure 5-15). Two large field studies, Clarke and Noone,
30 (1985) and Doherty et al., (2010), significantly boosted the number of sampling locations during two
31 windows of time (1983-1984, 2006-2009). However, to put this into perspective, even the largest
32 number of locations ever measured in one year across the globe (55 in 2009) is on par with the number
33 of air monitoring locations in a single mid-sized state in the United States. Recent model estimates by
34 Flanner et al., (2007), seek to fill in the missing measurement gap with predictions of surface snow BC
35 concentrations in the northern hemisphere, estimating values ranging five orders of magnitude (<1 to
36 >1000 ng BC/g snow).

37



1

2

3

4

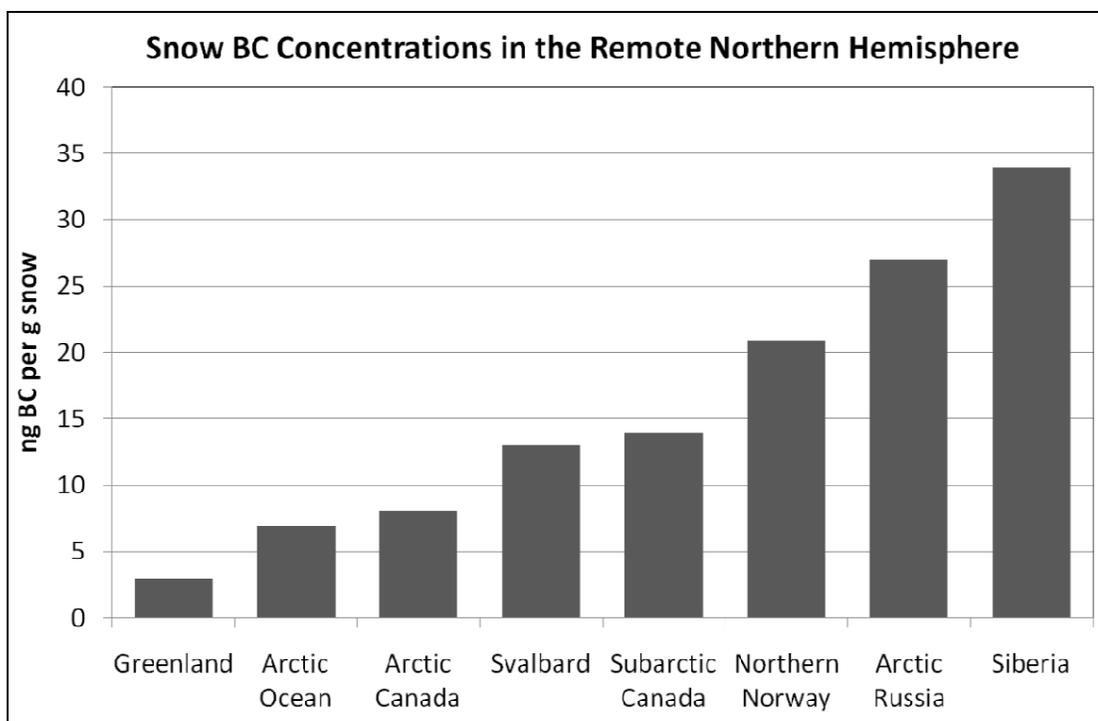
5

Figure 5-15. Locations of BC Measurements in Surface Snow and Shallow Snow Pits (snow pits are indicated for each year covered in the pit depth). These data are reported in Cachier and Pertuisot (1994), Cachier (1997), Chylek et al., (1987, 1999), Clarke and Noone, (1985), Doherty et al., (2010), Grenfell et al. (1981, 1994, 2002), Hagler et al., (2007a,b), Hegg et al., (2009, 2010), Masclat (2000), Ming et al., (2009), Perovich et al., (2009), Slater et al. (2002), Warren and Clarke (1990), Warren et al., (2006), and Xu et al., (2006).

1 Recent surface snow results from Doherty et al., (2010) show that BC concentrations range over
2 an order of magnitude in remote areas of the Northern Hemisphere (Figure 5-16). Even higher BC
3 values in snow were reported for the Tibetan Plateau and throughout western China, up nearly another
4 order of magnitude (Ming et al., 2009, Xu et al., 2006). BC removal from the atmosphere is primarily
5 driven by precipitation (Ogren et al., 1984), thus BC concentrations in snow or ice are a function of the
6 atmospheric concentration of BC above the surface and the frequency and amount of snowfall in a
7 particular area. For example, Xu et al., (2009a) noted that BC concentrations on the Tibetan plateau
8 were high during nonmonsoon periods with low precipitation, which they related to regional particulate
9 pollution (“Asian Brown Cloud”) elevating during the dry nonmonsoon period and then highly
10 concentrating the infrequent precipitation with impurities. An additional important factor, discussed by
11 several studies (Doherty et al., (2010), Flanner et al., (2007), Grenfell et al., (2002), Xu et al., (2006)), is
12 the potential increase in surface snow BC levels when melting snow leaves behind BC particles, further
13 darkening the topmost layer of snow.

14 It is important to note that certain non-BC particulate species have been shown to absorb light
15 when deposited to snow or ice. While dust is not as strong of a light absorber per unit mass as black
16 carbon, dust can play a significant role in reducing snowpack reflectivity at high concentrations (Warren
17 and Wiscombe, 1980). In addition, brown carbon in snow has been suggested to significantly absorb
18 light (Doherty et al., 2010). Given that studies suggest that organic material in snow may undergo
19 chemical transformation and loss from the snowpack due to sunlight-driven reactions (Grannas et al.,
20 2007, Hagler et al., 2007a), brown carbon may absorb light to an even greater degree in fresh
21 precipitation than what has been measured in aged snow samples.

22



1
2
3
4

Figure 5-16. BC Concentrations in Surface Snow in Arctic and Subarctic Areas of the Northern Hemisphere. These results were derived from recent measurements reported in Doherty et al., (2010).

5 **5.6.3. Ice Core Data**

6 Measurements of BC in ice cores are critical to understand the longer-term trends of human
 7 influence on snow reflectivity. Ice cores, produced by drilling into permanent ice and carefully
 8 extracting a column of ice, have been collected and analyzed for black carbon at a number of locations
 9 in the Northern Hemisphere (Figure 5-17). In addition, an Antarctic BC ice core record spanning the past
 10 two and a half thousand years has just been completed as part the National Science Foundation WAIS
 11 Divide deep ice core project (Ross Edwards, personal communication). The ice cores with continuous BC
 12 data available primarily cover the past few hundred years, with the exception of the Dye 3 ice core in
 13 Greenland and the WAIS Divide core in Antarctica which extend back several thousand years. The layers
 14 of the ice core are dated using several strategies, including measuring certain chemical species with
 15 known seasonal variation, looking for certain known historical events that had unique chemical
 16 signatures (e.g., volcanic eruptions, nuclear explosions), and observing the visible layering of ice
 17 throughout the core (e.g., Hammer et al., 1978).

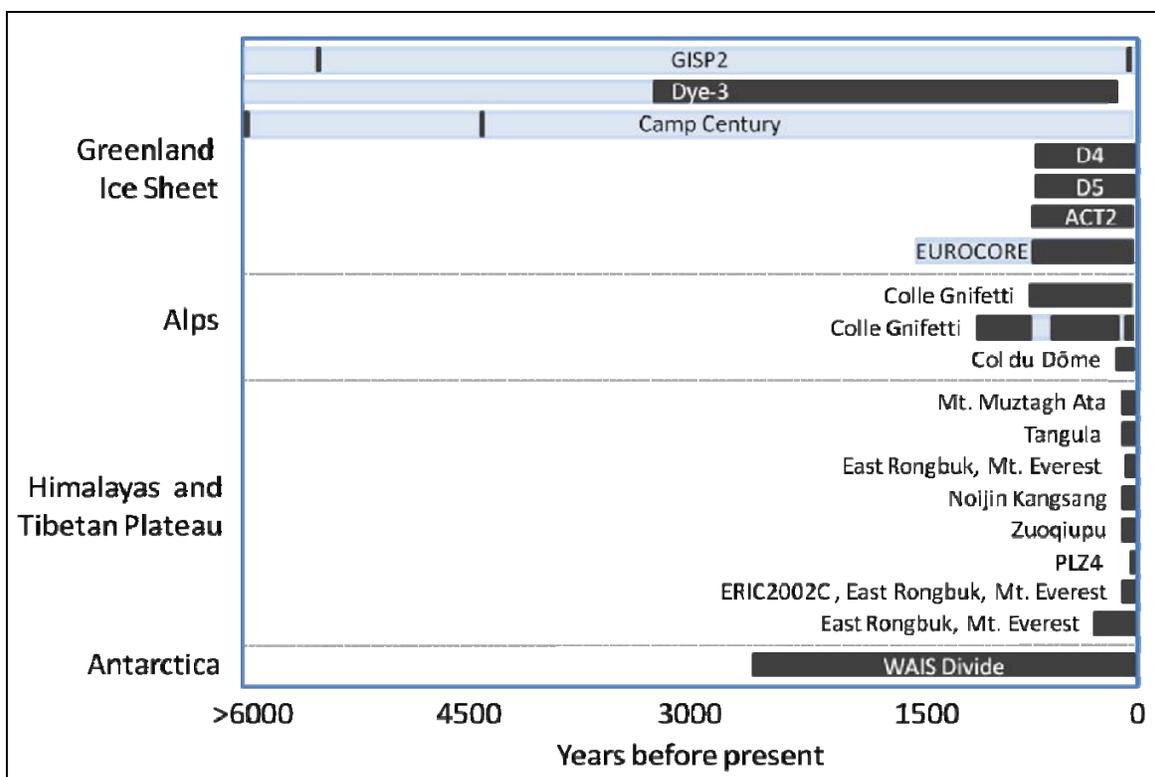


Figure 5-17. BC Ice Core Records Worldwide. Labeled by their identifying name. The extent of the bars shows the depth of the ice core, with the black regions showing sections of the ice core that had BC concentrations reported. These data are reported in Cachier and Pertuisot (1994), Chylek et al., (1987, 1992, 1995), Ross Edwards (personal communication regarding WAIS Divide ice core), Kaspari et al., (2011), Lavanchy et al., (1999), Legrand et al., (2007), Liu et al., (2008), McConnell et al., (2007), McConnell and Edwards (2008), Ming et al., (2008), Thevonon et al., (2009), and Xu et al., (2009a,b).

The concentrations of BC in a certain ice core reflect the past atmospheric concentrations above the region, which in turn relate to short- and long-distance transport of BC emissions. Thus, the ice core results vary from location to location. For example, on the remote Greenland Ice Sheet, McConnell (2010) showed a peak in BC concentrations in the 1910-1920 time range, decreasing in concentration from that point to present day. Meanwhile, ice cores in the European Alps show BC concentrations increasing significantly past the 1910-1920 period, with highest concentrations recorded in the 1950-1960 time frame (Lavanchy et al., 1999; Legrand et al., 2007). Finally, Xu et al., (2009b) and Ming et al., (2008) reveal variable results for multiple shallow ice cores collected in the Himalayas and Tibetan Plateau that date from the 1950s to 2004 – several ice cores have highest BC levels in the 1960s and lower levels from that point forwards, while another ice core had continuously increasing levels until present day. The ice core data collected to date have made associations between elevated BC and human activities, however, the trends vary significantly by location.

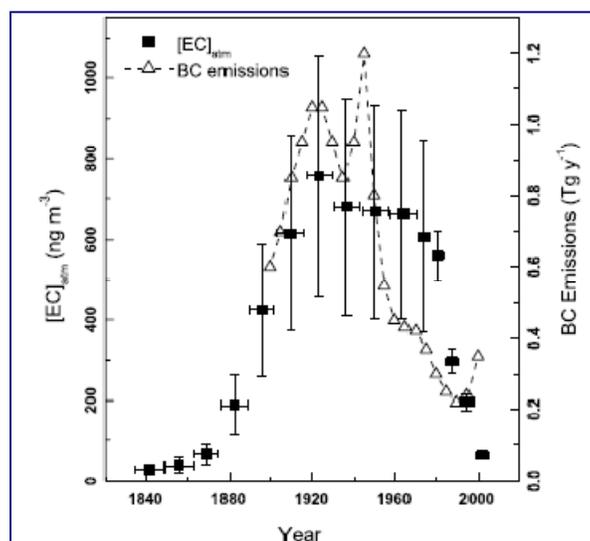
5.6.4. Sediment Data

With ice core records only available in remote, high-altitude locations in the world, undisturbed lake sediments provide additional spatial coverage of BC historical trends and may demonstrate higher

1 associations with local emissions. In addition, deep ocean marine sediments reveal ancient BC trends
 2 related to natural emissions. Similar to ice cores, BC records in sediments initiate from the deposition of
 3 BC from the atmosphere, which relates to the atmospheric transport of BC emissions to a particular
 4 location. After depositing to the surface of a water body, the BC particles eventually transport
 5 downwards and, if the sediment is undisturbed, may form a permanent archive in the layers of
 6 sediment.

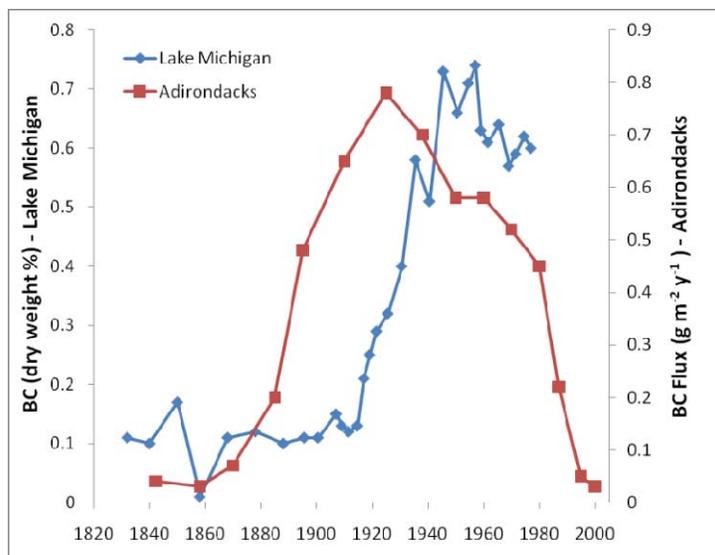
7 Lake sediment BC records have been quantified for several interior lakes in North America –Lake
 8 Michigan, dating 1827-1978 (Griffin and Goldberg, 1983) and in four lakes located in the Adirondacks of
 9 New York, dating 1835-2005 (Husain et al., 2008). Total carbon particles, associated with specific
 10 sources by particle shape, as also been measured in Lake Erie sediments, dating 1850-1998 (Kralovec et
 11 al., 2002). Historical BC records have also been obtained for a number of lakes in the Alps of northern
 12 Slovenia (Muri et al., 2002, 2003) and in ancient marine sediments, aged approximately 100 million to
 13 5000 years before present, spanning southern to far northern latitudes of the Pacific Ocean and at
 14 several locations in the Atlantic Ocean (Smith et al., 1978).

15 The findings by Smith et al., (1978) reveal an approximate 10-fold increase in ancient BC
 16 deposited levels moving from the equator northward to 60 degrees (bisecting Canada), which they
 17 related to the increase in natural wildfire emissions moving from the equator northward. These trends
 18 lay the base pattern of deposited BC, to which anthropogenic emissions BC would be added. Figure 5-18
 19 presents estimates of atmospheric BC derived from sediment core measurements in the Adirondack
 20 region of New York State for deposition from approximately 1835 to 2005 (Husain et al., 2008) and
 21 overlays these estimates with long-term U. S. BC emissions data developed by Novakov (2003). The
 22 derived BC ambient estimates are well correlated with the historical BC emissions estimates for fossil
 23 fuel combustion in the United States, and Husain et al., (2008) attributed the decrease from 1920-2000
 24 to reduction in BC emissions from United States fossil fuel combustion.



25
 26 **Figure 5-18. Atmospheric BC determined by Husain et al. (2008), for the Adirondack Region from 1835**
 27 **to 2005.** The measurements are compared with U.S. BC emissions (Novakov et al., 2003).

1 Figure 5-19 combines the ambient BC determined from Adirondack lake sediments by Husain et
 2 al., (2008), shown above, with those from Lake Michigan sediments from Griffin et al., (1983). The
 3 archives in both North American lake sediments reveal over an order of magnitude increase in
 4 deposited BC levels after the late 1800s (Figure 5-19). The Lake Michigan time series shows an apparent
 5 peak in the record around 1940-1960, while the Adirondacks show a peak near 1910-1930. The
 6 differences in the trends may be attributed to local source impacts on the deposited BC, measurement
 7 approaches, and sediment deposition processes. Regardless, both show a similar degree of increase
 8 during the course of the industrialization period.



9

10 **Figure 5-19. BC Trends in Lake Sediments Located in the Adirondacks (Husain et al., 2008) and in Lake**
 11 **Michigan (Griffin et al., 1983).** The Griffin et al. data were reported in units of % charcoal per mass of
 12 sediment and the Husain et al. data were reported as EC flux and estimated from Figure 5 in their paper.
 13 Both descriptive terms are converted for simplicity to BC here.

14

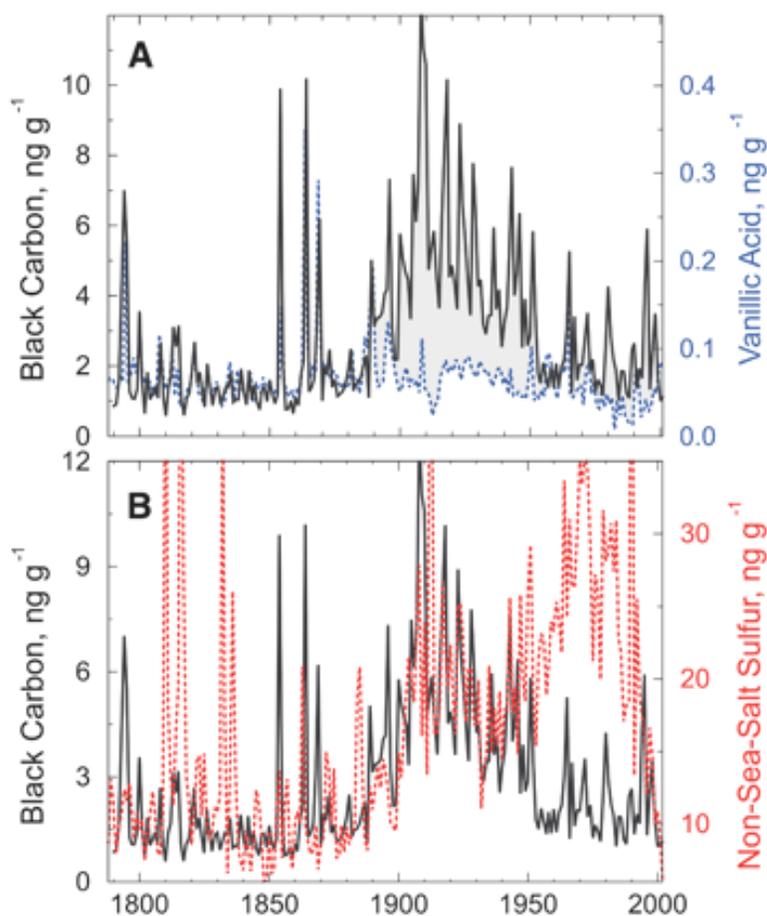
15 5.6.5. Arctic BC Snow and Ice Data – Source Identification

16 Impacts of BC emissions on the Arctic are of particular interest given the climate-sensitive
 17 nature of the region. BC emissions from particular source types or regions and transport to the Arctic
 18 have been explored through modeling studies and field measurements. This section discusses the
 19 findings in observational BC data from Arctic snow and ice. Connections between snow and ice BC data
 20 and source types are generally made by measuring additional species in the snow (i.e., ions, metals,
 21 organics, and isotopes) and comparing trends between the multiple data sets.

22 Historical trends in Arctic ice cores collected on the Greenland Ice Sheet improve our
 23 understanding of the historical impact of anthropogenic and natural emissions of BC on the Arctic.
 24 McConnell and Edwards (2008) and McConnell et al. (2007) provide monthly-resolution BC data in ice
 25 cores on the Greenland Ice Sheet. Similar to the lake sediment findings for the Adirondack Mountain
 26 region, the maximum BC concentrations in Arctic ice in the past hundred years occurred in the early

1 1900s corresponding to increases in a number of species associated with industrial emissions (e.g.,
 2 cadmium, cesium, thallium, lead). McConnell et al., (2007) compare vanillic acid (VA), non-sea-salt
 3 sulfur (nss-S), and BC trends to apportion the BC due to industrial versus forest fire emissions (see Figure
 4 5-22). VA is considered an indicator of forest fire emissions, while nss-S relates to industrial emissions
 5 and volcanic eruptions. In the postindustrial era, BC anthropogenic emissions contributed roughly 50-
 6 80% of the total BC loading in the ice during early 1900s and over past few decades the industrial input
 7 was on the order of 20-50% (estimated from Figure 5-20, originally published in McConnell et al., 2007).
 8 While nss-S correlated highly with the increasing BC during the late 1800s to mid-1900s, the trends did
 9 not match later, which may be related to changes in industrial emission factors. This study associates
 10 the high BC concentrations in the early 1900s with North American fossil fuel emissions and suggests
 11 that Asian emissions may play an important role past the mid-1900s.

12

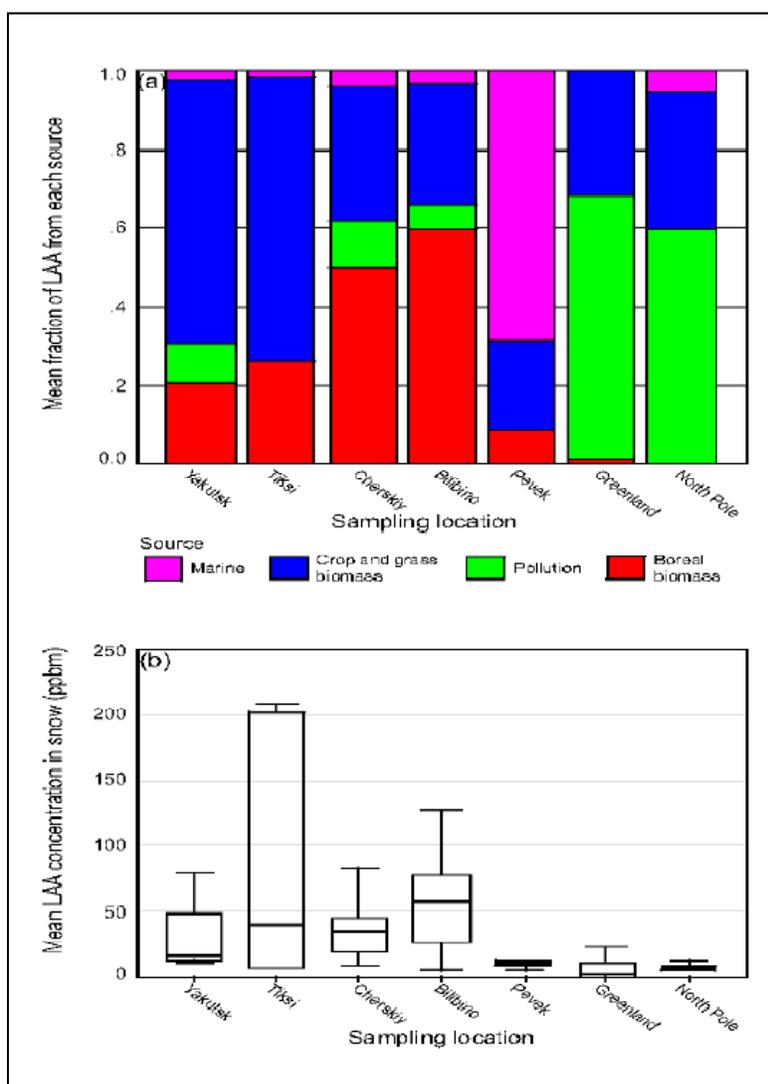


13

14 **Figure 5-20 Annual Average Concentrations of BC and VA (A) and of BC and Non-Sea-Salt**
 15 **Sulfur (nss-S) (B).** Adapted from McConnell (2007). The gray shaded region (between the black
 16 and blue dotted line) in the top figure represents the portion of BC attributed to industrial
 17 emissions, not boreal forest fires.

18 A recent study by Hegg et al. (2010), estimated fractional source contributions to light absorbing
 19 particles (i.e., black carbon plus additional light absorbing particles) by collecting a large number of

1 surface snow samples throughout the Arctic, which they measured for detailed chemical composition.
 2 Statistical analyses revealed that the measured species grouped into four unique factors with source-
 3 defining chemical characteristics (for example, sodium and chloride indicating a marine environment),
 4 which the authors labeled as marine (or having transported over ocean regions), boreal biomass, crop
 5 and grass biomass, and pollution. Depending on the location of the sample within the Arctic and time
 6 of year, the estimated contribution from these four sources varied considerably (Figure 5-21). In Siberia,
 7 emissions from biomass burning were significant drivers of BC and other absorbing species. However,
 8 on the Greenland Ice Sheet and at the North Pole, pollution and crop/grass biomass were found to be
 9 the primary sources.



10

11 **Figure 5-21. Fractional Source Contributions to Light Absorbing Aerosol (LAA) Snow**
 12 **Concentrations in Siberia (Pevek, Billirio, Cherskiy, Tiksi, Yakutsk), the Greenland Ice Sheet,**
 13 **and the North Pole(a). Concentrations of LAA at each location, with error bars indicating the 95%**
 14 **confidence interval (b). This figure was originally published in Hegg et al., (2010).**

15

1 **5.7 Limitations and Gaps in Current Ambient Data and Monitoring Networks**

2 The primary limitation in existing ambient monitoring data is the sparse geographic coverage of
3 existing BC monitoring locations. There are parts of the world where there currently are no
4 measurements; and where they do exist, the measurements are not archived into a consolidated
5 database. The differences in average BC concentrations between countries (global scale), among
6 regions (regional scale) and also within cities (local scale) are all much larger than the differences across
7 monitoring methods. These geographic variations are also larger than the inter-annual changes that
8 may occur within a 5-to-10-year period. To help develop and corroborate emission inventories and to
9 evaluate global models (see Chapters 3 and 4), additional ambient measurements are needed at more
10 locations. Existing geographically dense, filter-based PM_{2.5} measurements in the United States (and
11 elsewhere if available) can be used to cost-effectively supplement the measurements from more specific
12 and expensive BC monitors. Also currently there are insufficient measurements characterizing the BrC
13 component of OC. The addition of more multiple wavelength optical analyzers or use of optical
14 measurements from existing PM_{2.5} filter samples can be useful (Hecobian, 2010; Chow 2010).

(This page intentionally left blank)

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
27
28
29
30
31
32
33
34

6. Mitigation Overview: Climate and Health Benefits of Reducing Black Carbon Emissions

6.1 Summary of Key Messages

- Available studies indicate that growth or decline in future BC emissions will likely vary significantly by region and sector. While overall BC emissions are likely to decrease globally, this trend will be dominated by emissions reductions in developed countries and may be overshadowed by emissions growth in key sectors (transportation, residential) in developing countries, depending on growth patterns.
 - Developed nations have already made significant progress in reducing BC emissions, and further reductions are expected to occur through 2030 with full implementation of existing regulations.
 - Emissions projections for developing countries are more variable, with studies indicating that emissions are likely to increase in some sectors and regions and decrease in others.
- Available control technologies can provide cost-effective reductions in BC emissions from many key source categories, resulting in some near-term climate benefits, especially at the regional level.
 - BC emissions reductions are generally achieved by applying technologies and strategies to improve combustion and/or control direct PM_{2.5} emissions from sources. Benefits in sensitive regions like the Arctic, or in regions of high emissions such as India and Asia, may include reductions in warming and melting (ice, snow, glaciers), and reversal of precipitation changes.
 - BC reductions could help reduce the rate of warming soon after they are implemented. However, available studies also suggest that BC mitigation alone would be insufficient to change the long-term trajectory of global warming (which is driven by GHGs). A recent draft of a UNEP assessment that is currently underway evaluated BC reductions as part of a larger strategy for short-lived climate forcers and in conjunction with slower acting CO₂ programs. The draft assessment (still under review by UNEP) indicates that a combined approach focusing on emissions reductions of both short-lived forcers and CO₂ might significantly postpone or possibly avoid temperature increases greater than 2°C.
- These cost-effective mitigation strategies will also provide substantial public health co-benefits.
 - Reductions in directly emitted PM_{2.5} can substantially reduce human exposure, providing large public health benefits that often exceed the costs of control. In the United States, the average public health benefits associated with reducing directly emitted PM_{2.5} are estimated

- 1 to range from \$270,000 to \$1.1 million per ton PM2.5 in 2030. The cost of the controls
 2 necessary to achieve these reductions is generally far lower. For example, the costs of PM
 3 controls for new diesel engines are estimated to be less than \$13,000 per ton PM2.5.
 4 Globally, the health benefits of mitigation strategies aimed at BC would be even larger,
 5 potentially involving hundreds of thousands of avoided premature deaths each year.
- 6 • Considering the location and timing of emissions and accounting for co-emissions will improve the
 7 likelihood that mitigation strategies will be beneficial for both climate and public health.
 - 8 ○ PM mitigation strategies that focus on sources known to emit large amounts of BC—
 9 especially those with a high ratio of BC to OC, like diesel emissions—will maximize climate
 10 co-benefits. The timing and location of the reductions are also very important. The largest
 11 climate benefits of BC-focused control strategies may come from reducing emissions
 12 affecting the Arctic, Himalayas and other ice and snow-covered regions.
 - 13 ○ The effect of BC emission reductions on human health are a function of changing exposure
 14 and the size of the affected population. The largest health benefits from BC-focused control
 15 strategies will occur locally near the emissions source and where exposure affects a large
 16 population.
 - 17 • The United States will achieve substantial BC emissions reductions by 2030, largely due to
 18 forthcoming controls on mobile diesel engines. Diesel retrofit programs for in-use mobile sources
 19 are also helping to reduce emissions. Other source categories, including stationary sources,
 20 residential wood combustion, and open biomass burning, have more limited mitigation potential
 21 due to smaller remaining emissions in these categories, or limits on the availability of effective BC
 22 control strategies.
 - 23 • Other developed countries have emissions patterns and control programs that are similar to the
 24 United States, though the timing of planned emissions reductions may vary. Developing countries
 25 have a higher concentration of emissions in the residential and industrial sectors, but the growth of
 26 the mobile source sector in these countries may lead to an increase in their overall BC emissions and
 27 a shift in the relative importance of specific BC emitting sources over the next several decades.

28 **6.2 Introduction**

29 This chapter provides an overview of key factors affecting which mitigation strategies and
 30 approaches for reducing BC emissions have potential to provide climate and public health benefits over
 31 the next several decades. The chapter examines what is known about the overall impact of existing or
 32 planned control programs on emissions of BC; how current BC emissions are projected to change over
 33 the next several decades in response to these control programs and/or economic growth and
 34 development; and the potential climate, public health, and environmental benefits of reducing BC
 35 emissions (both in general and in particular sectors). The closing section of the chapter provides an
 36 overview of the mitigation options available in each of four major emissions sectors—mobile sources,
 37 stationary sources (including both power generation and industry), residential heating and cooking, and
 38 open biomass burning. Additional detail on each of these sectors is provided in subsequent chapters.

1 Chapters 7-10 describe projected changes in emissions in the United States and globally, available
2 control technologies and strategies and their associated costs, and implementation challenges for each
3 sector individually. Before considering the details of each sector, however, it is valuable to consider
4 some of the key factors that will determine the effectiveness of mitigation efforts more generally, and
5 the full magnitude of the benefits that might be achieved.

6 **6.3 Effect of Existing Control Programs**

7 Many existing control programs have been highly effective in reducing BC, particularly controls
8 affecting emissions from mobile and stationary sources. However, it is important to note that BC is not
9 the *direct* target of any currently existing emissions control program. Rather, it has been reduced in
10 conjunction with control programs focused on reducing ambient PM_{2.5} concentrations or direct particle
11 emissions in general. As discussed throughout this report, BC is always co-emitted with other particles
12 and gases. Therefore, determining the effect of various mitigation strategies on BC emissions requires
13 an understanding of the entire emissions mixture coming from a given source and the extent to which
14 the BC fraction is reduced by specific control technologies or strategies. Currently, there is only limited
15 information about effective control strategies for reducing BC in a targeted fashion and the associated
16 costs of those strategies.

17 Controls on direct PM_{2.5} emissions do affect emissions of BC and other constituents such as OC.
18 This is clear from the limited emissions testing data and observational record which link declining BC
19 concentrations to the PM_{2.5} control program (see Chapters 4 and 5). However, the extent to which BC
20 has been controlled as a component of an overall PM_{2.5} mixture has depended somewhat arbitrarily on
21 the proportion of BC in the emissions mix from a particular source category and the specific control
22 strategy applied. Some programs in some sectors (such as mobile source emissions standards) do
23 effectively reduce BC emissions as much or more than other constituents, while in other instances, BC
24 reductions may be proportionally smaller. The relative effectiveness of a particular control technology
25 for reducing specific constituents is often unknown, which means that for most sectors, it is not clear
26 whether PM_{2.5} controls will reduce BC preferentially or even proportionally to other constituents.
27 Ongoing research will help to clarify this issue.

28 In general, available estimates of BC emissions reductions are calculated from analyses of PM_{2.5}
29 controls. As discussed in Chapter 4, EPA's trends report (2009) shows U.S. emissions of direct PM_{2.5}
30 have declined by 58% since 1990, a reduction of over 1.3 million tons. Over half of this reduction (57%)
31 has come from controls on stationary fossil fuel combustion, with substantial reductions also occurring
32 in emissions from industrial processes (25%) and mobile sources (17%). Using speciation factors, it is
33 possible to calculate BC reductions in these sectors, but these estimates are generally rough. Precise,
34 measured data about the effectiveness of specific controls for reducing BC emissions is often not
35 available. As described in Chapter 5, however, recent ambient BC measurements do appear to indicate
36 a decline in neighborhood/urban and regional scale concentrations of BC in the United States between
37 the mid 1980s and the present (see section 5.4.1).

1 It is also important to point out that the overarching PM_{2.5} criteria pollutant control program for
2 stationary sources in the United States and Europe has focused mainly on secondarily formed particles
3 such as sulfates and nitrates, rather than direct PM_{2.5} emissions. This is because PM controls motivated
4 by public health and environmental goals are focused on reducing total PM mass at least cost. PM
5 controls oriented toward climate would have to consider the light absorbing and scattering properties of
6 the various PM constituents.

7 While control strategy information and cost data for BC mitigation approaches are limited, this
8 also varies by sector and location, with some of the best available information available for mobile
9 source controls. Analyses conducted for recent regulatory actions in the United States provide a solid
10 foundation for understanding applicable technologies and costs, and related implementation issues. For
11 other sectors where less information is available, for example open biomass burning, better information
12 on BC-specific control strategies, effectiveness and costs is needed. EPA has historically evaluated PM
13 control strategies for specific sectors as part of the regulatory impact analyses for specific rulemakings.
14 These analyses generally include best-available information on control options, effectiveness, and costs,
15 and some of them do include information on controls for specific PM constituents, but this rarely
16 includes BC.

17 Despite what is known from analysis conducted in the United States, many of the strategies that
18 have been applied domestically differ in important ways from control strategies that have been adopted
19 internationally. Some of the strategies utilized by developed countries have also been undertaken in
20 developing countries or could be adopted on a broader scale internationally. In other cases, developing
21 countries have a different mix of sources that will require different types of control strategies. These
22 issues are discussed further in the sections that follow, and in the conclusion to this chapter.

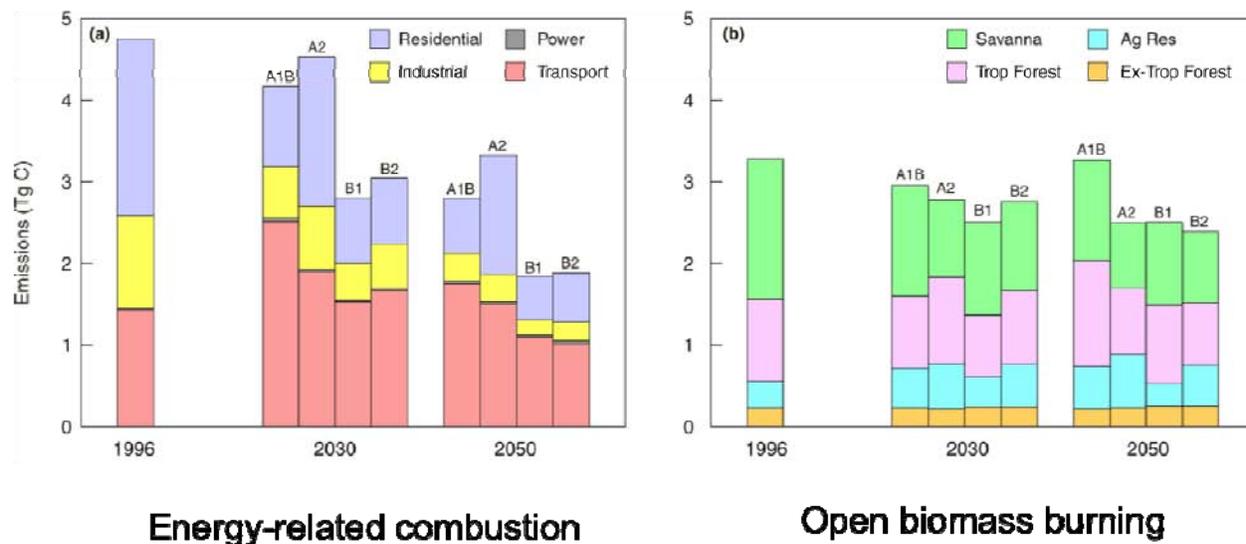
23 **6.4 Future Black Carbon Emissions**

24 The influence of BC on climate and public health in the future, and the need to pinpoint more
25 precisely the effectiveness of various mitigation strategies for reducing BC, depend in large part on the
26 magnitude of future emissions. This section describes what is known regarding these future emissions.

27 Developed nations have already made significant progress in reducing direct PM emissions, and
28 further reductions are expected to occur through 2030 with full implementation of existing regulations.
29 In particular, substantial BC reductions have been achieved through controls in the mobile source
30 sector, and that additional reductions will continue to be realized over the next two decades. In the
31 case of stationary sources, the most substantial BC emissions reductions in the United States and other
32 developed countries were achieved decades ago (often through fuel switching away from coal).

33 Several recent studies (Streets et al., 2004; Cofala et al., 2007; Jacobson and Streets, 2009;
34 Rypdal et al., 2009) provide a snapshot of potential future BC emissions trends. These studies have
35 produced a range of estimates for future BC emissions depending on assumptions about economic
36 growth, population levels, and development pathways. In an analysis of future BC emissions trends
37 based on the IPCC SRES scenarios, Streets et al. (2004) projected BC emissions to decrease globally, by 9

1 to 34% by 2030 relative to 1996 levels. However, there was considerable variation among the different
 2 sectoral projections depending on the SRES scenario examined (Figure 6-1). Thus, while aggregate
 3 emissions were generally projected to decline under alternative growth scenarios, emissions growth was
 4 projected for certain sectors or regions. The sectors where Streets indicates a potential for future
 5 emissions growth include residential emissions in Africa, open biomass burning emissions in South
 6 America, and transportation emissions in the developing world (for example, where fuel sulfur levels are
 7 still too high for implementation of DPFs—see Chapter 7 and Appendix 3). In general, industrial
 8 emissions were projected to decline, as were transport emissions in developed countries.



9

10 **Figure 6-1. Global BC Emissions Forecasts for Various Sectors under Alternative IPCC SRES Scenarios** (in
 11 teragrams (Tg) of carbon). Scenarios generally show a modest decrease in BC emissions from all sectors
 12 as compared to 1996 baseline emissions. (Streets et al., 2004)

1 An analysis by Jacobson and Streets (2009) found that under the assumptions embedded in the
 2 A1B scenario for IPCC, total global BC emissions may increase substantially. Again however, this analysis
 3 indicates that projected emissions growth or decline varies significantly among regions and sectors, as
 4 Figure 6-2 illustrates. In general, BC emissions in developed countries are projected to decline, while
 5 emissions in developing countries may grow. Transportation (mobile source) emissions in particular are

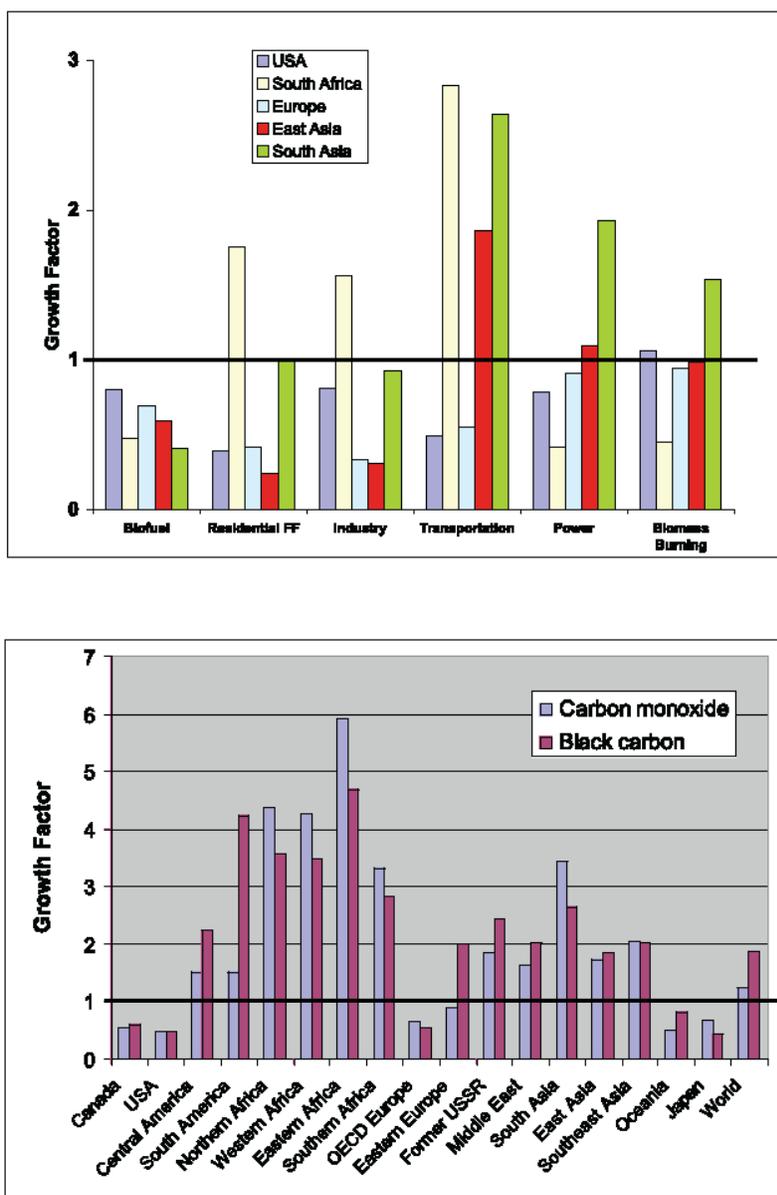
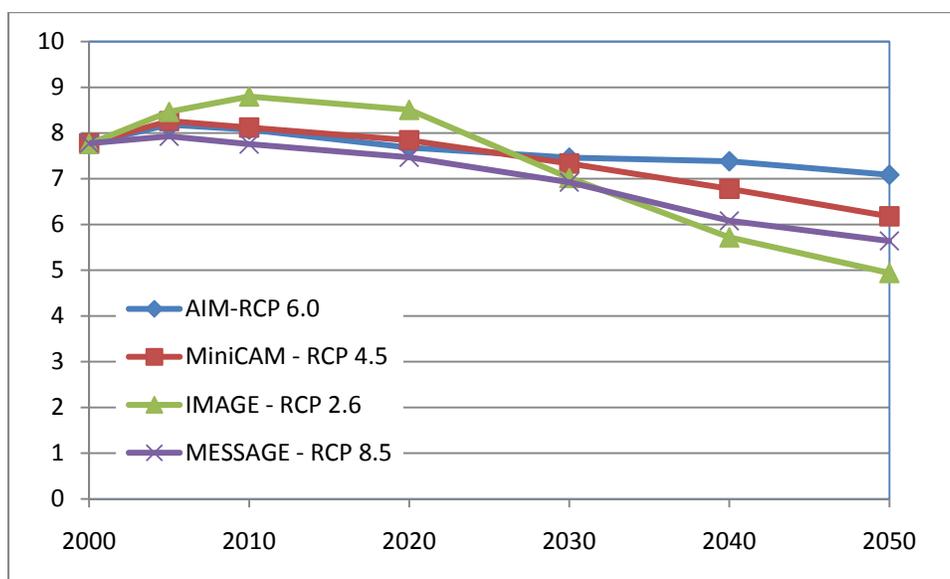


Figure 6-1. Black Carbon Emissions Growth, 2000-2030 under IPCC A1B Scenario. Top: 2000-2030 Black Carbon Emissions Growth Factors by Sector for Selected World Regions (from IPCC A1B scenario). Bottom: 2000-2030 Black Carbon and Carbon Monoxide Emissions Growth Factors for Transportation (Mobile) Sector in Specific Regions. Emissions in sectors with a growth factor less than one (see dark line, added) will decline. (Source: Jacobson and Streets, 2009)

1 projected to grow in several world regions but decline in others, as illustrated by growth factors greater
 2 than or less than one, respectively (Jacobson and Streets, 2009). In developed countries, the majority
 3 of the emissions reductions in the transportation sector are projected to result from implementation of
 4 Euro V and similar standards that lead to the use of diesel particulate filters (DPFs) in the diesel fleet.
 5 However, other studies have indicated that emissions from shipping in the Arctic region may increase
 6 due to the retreat of Arctic sea ice opening up new shipping routes and increased economic activity in
 7 that region (Corbett et al., 2010).

8 In its most recent work, the IPCC has also developed four “Representative Concentration
 9 Pathways” for use as a consistent set of emissions inputs for projecting future climate change. These
 10 four pathways (Figure 6-3) are defined by the total radiative forcing resulting from each pathway in
 11 2100, including GHGs and other forcing agents, and range from 2.6 to 8.5 Wm⁻². Global BC emissions in
 12 all four pathways peak in 2005 or 2010, are 8 to 20% below 2010 levels by 2030, and continue
 13 decreasing for the rest of the century to about half of 2010 levels. Emissions for the RCP pathways are
 14 reported in combinations of five regions and four sectors on the website¹ (though the underlying,
 15 gridded dataset is further disaggregated).



16
 17 **Figure 6-3. Future Emissions of BC under IPCC Representative Concentration Pathways, 2000-**
 18 **2050 (Gg/year)**

19 Notes:
 20 RCP2.6 (RCP 3-PD) – van Vuuren et al., 2007
 21 RCP 4.5 – Clarke et al., 2007; Smith and Wigley, 2006; and Wise et al., 2009
 22 RCP 6.0 – Fujino et al., 2006; and Hijioka et al. 2008
 23 RCP 8.5 – Riahi et al., 2007

24

¹ <http://www.iiasa.ac.at/web-apps/tnt/RcpDb/dsd?Action=htmlpage&page=welcme#>

1 Consistent with the findings of the academic studies, under certain pathways there are a few
2 region and sector combinations whose BC emissions do not peak until 2020 or 2030. Two out of the
3 four RCP pathways show near-term increases in BC emissions from all sectors in Asia, the Middle East,
4 and Africa, and for all regions some of the pathways show increases in open burning emissions (from
5 both deforestation and agricultural burning). Because of the potential for increases in open burning, OC
6 emissions are not projected to decline as quickly as BC emissions.

7 BC emissions in the U.S. are projected to decline, but this trend will be driven largely by
8 reductions in mobile diesel emissions, as discussed in detail in Chapter 7. The limited EPA modeling
9 inventories that project emissions into the future (year 2020) indicate that direct PM_{2.5} emissions from
10 industrial sources are not expected to decline significantly in the next decade, and emissions from fossil
11 fuel combustion will only decline about 20% by 2020 (U.S. EPA, 2006). Because of the small size of
12 anticipated reductions in direct PM_{2.5} emissions from these categories, projected BC emissions changes
13 are also small and unlikely to affect U.S. BC emissions trends in the future in the absence of additional
14 control requirements. Open biomass burning, the second largest source category in the United States,
15 exhibits significant year-to-year variability in emissions, and it is difficult to predict future year
16 emissions. However, it should be noted that emissions in this category may grow significantly in the
17 future if climate change results in increased wildfires, as predicted in many scenarios (Wiedinmyer,
18 2010).

19 Projected future emissions reductions will not happen in the U.S. or elsewhere in the absence of
20 continued policies to encourage adoption of DPFs in the mobile sector, continued economic
21 development leading to a shift away from traditional cookstoves, and other environmental and
22 economic developments. As noted, there are also several sectors and regions, such as transport
23 emissions in developing nations and open biomass burning emissions globally, where emissions are not
24 projected to peak for another decade or two. Given the array of available control technologies and
25 strategies, as outlined in the next several chapters of this report, it is possible to make larger and more
26 rapid reductions in BC emissions globally than current baseline estimates project.

27 Some countries have already begun looking at these possibilities. For example, the Arctic
28 Council countries (Canada, Denmark, Finland, Iceland, Norway, Russia, Sweden, and the United States)
29 formed a special Task Force on Short-Lived Climate Forcers in 2009 to consider whether additional or
30 accelerated mitigation strategies may be needed to address warming in the Arctic region. Noting that
31 emissions in the Arctic region from sources other than land-based transport—particularly residential
32 heating, agricultural and forest burning, and marine shipping—will likely remain the same or increase
33 without new measures, the Task Force has recommended that “Arctic Council nations individually and
34 collectively work to implement some early actions to reduce black carbon.” Which measures would be
35 implemented, and the impact of such measures on total BC emissions from Arctic Council nations,
36 remains undetermined, but could influence future emissions trajectories.

1 **6.5 Climate Benefits of Reducing Black Carbon Emissions**

2 As discussed in Chapter 2, the nature and distribution of BC and its mechanisms of action mean
3 it can have important direct and indirect effects on climate that differ from those of GHG. These effects
4 are not limited to those derived from radiative forcing on a global scale, but include altered clouds,
5 ‘dimming,’ changes in the vertical structure of the atmosphere, and resultant changes in precipitation.
6 Deposited BC can result in disproportionate warming in areas covered by snow and ice, which is greatest
7 near source regions (e.g. the Himalayas) but still significant in the Arctic.

8 On the presumption that reducing BC emissions could mitigate some of these effects, several
9 groups have developed preliminary analyses of global strategies to reduce BC emissions (e.g. Cofala et
10 al., 2006; Rypdal et al., 2009; Baron et al., 2009; Kandlikar et al., 2009). These assessments have
11 generally found the largest BC reductions at lower costs in Asia, which has high emissions from
12 residential cooking and heating as well as poorly controlled transportation and small industrial sources.
13 Some also included programs to reduce biomass burning in Africa and South America. BC strategies
14 ranged from improved combustion to add-on particle controls. The actual benefits to global or regional
15 climate resulting from such strategies can, however, vary significantly with the nature of the co-
16 pollutants emitted from different sources, their location, and the nature of the controls.

17 Although the kinds of uncertainties in inventories, understanding of atmospheric processing and
18 interactions, and other factors discussed in Chapter 2 limit our ability to provide quantitative prediction
19 of the array of potential climate benefit of strategies to reduce BC from various sources, several
20 investigators have used global climate models to examine reductions of BC, OC and in some cases
21 associated GHG from fossil, biofuel, and biomass sector sources. Most of these have focused on the
22 effect of global reductions on radiative forcing or temperature. The results are generally in line with
23 expectations based on the BC to OC ratios from the three major sectors (see Section 2.6.1). The largest
24 and most consistent negative forcing or cooling is found with reductions in emissions from fossil fuel
25 sources with high BC/OC including all ‘fossil soot and gases’ (Jacobson, 2010), on- and off-road land
26 transport (Unger et al., 2009; Unger et al., 2010), and diesels (Bauer et al., 2010). By contrast, a
27 combined assessment of multiple models of global reductions in open biomass burning (low BC/OC)
28 found a small but net negative global average forcing due to the larger amount of cooling from OC (Kopp
29 and Mauzerall, 2010). The results vary with region (Jacobson, 2004), however, and recent research on
30 BC deposition in the Arctic and elsewhere suggest biomass burning may be contributing significantly to
31 melting and warming in such regions (Stohl et al., 2007; Flanner et al., 2007, 2009; Hegg et al. 2009;
32 Koch et al., 2007). Important uncertainties include the extent to which BrC directly emitted and
33 formed downwind of biomass burning contributes to absorption (Magi et al., 2009) and the fraction of
34 open burning that is not sustainable (i.e. increases CO₂ emissions over time) (Jacobs, 2004).

35 The latter uncertainties also apply to assessing the benefits of reducing residential combustion
36 of solid biofuels, where recent modeling results are more mixed. Jacobson (2010) and Unger et al., 2010
37 found reducing BC, OC, and GHG gas emissions from biofuels resulted in net cooling. On the other hand,
38 Bauer et al. (2010) find reductions of BC and OC from biofuels resulted in a net warming and Chen et al.
39 (2010) found a net warming for BC/OC reductions in both biofuels and fossil fuels. The latter two

1 studies did not include the snow albedo effect or associated GHG reductions, but this cannot fully
2 explain the results. Instead, while all of these studies use advanced approaches for evaluating aerosol-
3 cloud interactions, the different treatments in the latter studies resulted in larger estimates for the
4 indirect effects of BC and OC, particularly for Chen et al. As noted in Chapter 2, the contribution of BC
5 sources to indirect effects remains one of the most important uncertainties in evaluating strategies.

6 The modeling studies summarized above used somewhat arbitrary reductions to estimate the
7 sign and potential magnitude of changes in forcing or temperature of major reductions in BC and related
8 emissions from significant source categories. The most comprehensive assessment to date or more
9 realistic strategies is the current draft analysis by the United Nations Environment Program (UNEP),
10 which included BC reduction measures with those for ozone (methane) to provide an integrated analysis
11 of climate and health benefits. Using an integrated modeling approach addressing a range of co-emitted
12 pollutants, the draft UNEP/WMO Assessment identified a small number of emissions reduction
13 measures. This basket of measures included both BC reduction measures and methane reduction
14 measures. For BC, the assessment modeled the impact of both “technical measures,” such as improving
15 coke ovens and brick kilns and increasing use of diesel particulate filters, and “non-technical measures,”
16 such as eliminating high-emitting vehicles, banning open burning of agricultural waste, and eliminating
17 biomass cookstoves in developing countries. In the climate simulations, the authors addressed the
18 uncertainties in indirect and direct effect of BC and OC effect by using a range of values from the
19 literature.

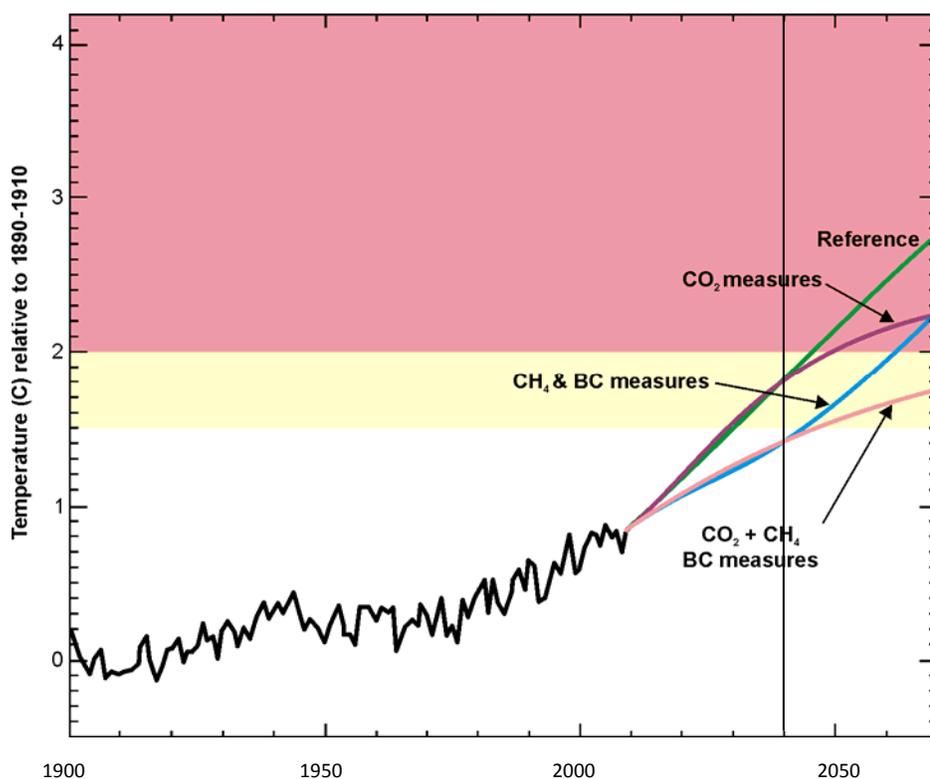
20 The modeled effectiveness of the basket of measures over the near term is shown in Figure 6-4.
21 As the figure illustrates, the reductions in CH₄ and BC combined produce a noticeable impact on near-
22 term warming as compared to the reference case or CO₂ measures by themselves. The analysis showed
23 that even aggressive CO₂ reductions may not keep climate change from approaching 2° C by mid-
24 century.² At the same time, it is important to note that the benefits of reducing BC and CH₄ are
25 insufficient to avert warming over the long term. Reducing short-lived climate forcers now, while
26 neglecting to achieve aggressive CO₂ reductions, may not keep temperatures from reaching the 2° C
27 mark in 2070 and beyond. These results, and those from other studies on the temporal aspects of
28 reducing BC and other short-lived forcers, underscore the scientific rationale for addressing long-lived
29 GHGs and short-lived forcers like BC as two distinct, complementary programs that act on different time
30 scales.

31 These results, and those from other studies on the temporal aspects of reducing BC and other
32 short-lived forcers, underscore the scientific rationale for addressing long-lived GHGs and BC
33 simultaneously as two complementary strategies to address global warming and other effects of climate
34 change.

35

² An increase in global mean temperatures of 2° C since preindustrial times was adopted as an international target under the UN’s Copenhagen Accord in December 2009.

1



2

3

Figure 6-4. Observed Deviation of Temperature to 2009 and Projections under Various Scenarios (UNEP, 2011 draft). Immediate implementation of the identified BC and CH₄ measures, together with measures to reduce CO₂ emissions, would greatly improve the chances of keeping Earth's temperature increase to less than 2°C relative to pre-industrial levels. The bulk of the benefits of CH₄ and BC measures are realized by 2040 (vertical line).

4

5

6

7

8

9

10

11

12

13

14

15

16

17

18

19

20

21

22

1 Recent findings of the Arctic Council Task Force on Short-Lived Climate Forcers suggest that
2 mitigating in- or near-Arctic sources will have a greater Arctic climate impact than the size of these
3 sources alone would indicate, with important seasonal and spatial variations. Specifically, in its March
4 2011 draft *Progress Report and Recommendations for Ministers* (Arctic Council, 2011), the Task Force
5 noted that:

6 [In the Arctic, the potential for ... offsetting effects from non-black carbon aerosols is weaker. Over highly
7 reflective surfaces such as ice and snow in the Arctic, the same substances that might cool the climate in
8 other regions may cause warming since they are still darker than ice and snow. This warming impact is
9 magnified when black carbon physically deposits on snow or ice. Emissions closer to the Arctic have a
10 greater chance of depositing, and thus appear to have greater impact per unit of emission.

11 The Task Force highlighted the importance of sectors such as land-based transportation, open biomass
12 burning, residential heating, and marine shipping in the Arctic, but also noted that emissions sources
13 outside of Arctic Council nations are important for Arctic climate change, partly because of the volume
14 of these emissions.

15 As discussed in Section 2.6.5, Menon et al. (2010), modeling the impacts of estimated increases
16 in BC between 1990 and 2000, found that Indian fossil/biofuel in particular may be responsible for some
17 of the observed patterns and trends in snow and ice melting and precipitation in the Himalayan region.
18 Such changes may have significant implications for water supply in the region. While a number of
19 studies have suggested BC and associated emissions may play a role in reduced monsoon rains, current
20 modeling capabilities do not provide a basis for reliable quantitative assessments of the extent to which
21 emissions reductions might reverse observed changes in precipitation.

22 Other work has examined the relationship between long-term trends in aerosol emissions and
23 regional temperatures (Shindell and Faluvegi, 2009), but few studies have examined the climate benefits
24 of specific particle control programs. A recent study of particular relevance examined the results from
25 California's laws to reduce particle pollution, in particular those regulating diesel emissions. The study
26 found that these rules reduced atmospheric concentrations of BC with a measurable impact on radiative
27 forcing. Modeled results indicate that the decrease in BC in California has led to a cooling of 1.4 W m^{-2}
28 ($\pm 60\%$) (Bahadur et al., 2011). So, while uncertainties remain, as outlined in previous chapters,
29 emerging research suggests that targeting emission reductions from key sectors can have measurable
30 benefits for climate.

31 **6.6 Public Health and Welfare Benefits of Reducing Black Carbon Emissions**

32 Reductions in BC emissions will have significant public health and welfare benefits. Since
33 controls generally affect the entire emissions mixture, BC mitigation approaches will generally reduce
34 total $\text{PM}_{2.5}$ mass, not just BC. Furthermore, while net climate impacts depend on the countervailing
35 effects of reductions in cooling aerosols that result from controls applied for BC, no such tradeoffs exist
36 for public health and the environment. All reductions in $\text{PM}_{2.5}$ that result from controls aimed at
37 reducing BC emissions are likely to produce public health benefits. The adverse health and welfare
38 impacts of all $\text{PM}_{2.5}$ are well documented in the scientific literature, previously discussed in Chapter 3

1 and in EPA's recent PM ISA (U.S. EPA 2009). The public health and welfare benefits achieved via
2 programs to reduce ambient PM_{2.5} have been quantified and monetized, providing greater confidence
3 that controls aimed at BC will produce clearly identifiable and very sizeable benefits. Emerging studies
4 suggest that BC-targeted strategies will provide similar benefits. Thus, while there may still be
5 significant uncertainty about some of the climate impacts and benefits of different BC mitigation
6 strategies, there is far greater certainty that such strategies will provide sizeable benefits for public
7 health and the environment.

8 Because the drivers for climate and health impacts of BC emissions differ, mitigation strategies
9 that achieve the greatest near-term climate benefits may not yield the greatest co-benefits for public
10 health, and vice versa. To identify mitigation strategies that optimize near-term climate and health
11 benefits, future studies should assess the full impacts of individual economic sectors and mitigation
12 measures on climate and health, accounting for the full mixture of emissions and both outdoor and
13 indoor exposure.

14 **6.6.1 Global Health Benefits**

15 The largest potential public health and environmental benefits are achievable internationally,
16 particularly in South and East Asia, where both pollution and population are high. For the small number
17 of international BC health studies currently available measuring the benefits and costs of possible
18 mitigation strategies, the estimated public health benefits alone (without counting any other climate or
19 environmental benefits) exceed the estimated costs of controls for many measures. This result suggests
20 that these reduction measures will be advantageous for society independent of the level of climate
21 benefits achieved.

22 Studies have clearly demonstrated that exposure to outdoor and indoor PM_{2.5}, including BC, is
23 associated with a large number of premature deaths internationally. In 2004, the WHO estimated that
24 *outdoor* urban PM_{2.5} was responsible for 800,000 premature deaths worldwide each year (WHO 2004).
25 More recently, Anenberg et al. (2010) estimated about 3.7 million global premature deaths annually due
26 to outdoor anthropogenic PM_{2.5} using a global atmospheric model to isolate the total anthropogenic
27 contribution to PM_{2.5} concentrations (calculated as the difference between simulated present-day
28 concentrations in 2000 and preindustrial concentrations in 1860) with full spatial coverage including
29 both urban and rural populations. The WHO also estimates that *indoor* PM exposure is associated with
30 1.5 million annual premature deaths worldwide (WHO 2006). New estimates of the global burden of
31 both *outdoor and indoor* air pollution on premature mortality are forthcoming from the Global Burden
32 of Diseases, Injuries, and Risk Factors Study (Institute for Health Metrics and Evaluation 2010,
33 www.globalburden.org).

34 As discussed previously, a growing body of literature examines the climate impacts of BC
35 emissions and their mitigation. However, the associated health impacts have been studied less
36 extensively. Jacobson (2010) estimated that biofuel combustion causes eight times more premature
37 deaths globally than fossil fuel combustion, largely because biofuel combustion occurs mainly in very
38 populated regions of the world. Saikawa et al. (2009) also estimated substantial benefits of BC

1 reductions in China. Studies examining surface BC concentrations globally find that concentrations are
2 highest over East Asia, South Asia, and Southeast Asia (e.g. Koch et al. 2009), the greatest population
3 centers of the world. The co-location of BC concentration with population likely translates into a
4 substantial impact on global public health, and potentially large health benefits of mitigation.

5 Since only a few studies examine BC health impacts, EPA led a study to better understand the
6 impacts of BC emissions on air quality and human health (Anenberg et al. in preparation). This study
7 used a global atmospheric model to simulate the difference in surface PM_{2.5} concentrations between a
8 present day (2002) base case and control simulations where anthropogenic BC emissions are halved
9 globally, in each major world region individually (Table 6-1), and from the three main BC-emitting
10 economic sectors individually (residential, industrial, and transportation). The study also examined a
11 case where global anthropogenic BC and OC emissions are halved together since BC and OC are co-
12 emitted and likely to both be affected by mitigation measures. Consistent with other studies examining
13 the health impacts of PM_{2.5}, Anenberg et al. (2010) used epidemiologically-derived concentration-
14 response functions from the latest re-analysis of the American Cancer Society Study (Krewski et al. 2009)
15 to translate PM_{2.5} concentration changes to mortality impacts, assuming all PM_{2.5} components and
16 mixtures are equally potent in causing premature death (see Chapter 3).

17 The Anenberg, et al. study estimated that halving anthropogenic BC emissions globally would
18 avoid 157,000 (95% confidence interval, 120,000-194,000) annual premature deaths worldwide. Over
19 80% of these premature deaths are reduced in East Asia (53%) and South Asia (31%), where large
20 populations are exposed to high concentrations (Figure 6-2). Halving anthropogenic emissions in each
21 major world region individually demonstrated that the vast majority of avoided deaths from halving BC
22 occur within the source region. Thus, the model results suggest that the contribution of BC emissions
23 from other regions to surface air quality and health within a region is very small. Per unit emission, the
24 mortality impact of BC emissions is 50% larger for South Asia than for East Asia (Figure 6-3). This is likely
25 because emissions changes in East Asia have smaller impacts on concentrations and because baseline
26 mortality rates are higher in South Asia.

27 This study also analyzed the surface air quality and health impacts of halving BC emissions
28 globally from the residential, industrial, and transportation sectors individually. Together, these sectors
29 contribute over 90% of global anthropogenic BC emissions. Globally, halving residential, industrial, and
30 transportation emissions contributed 46%, 35%, and 15% of the avoided deaths from halving all
31 anthropogenic BC emissions, respectively. These contributions are 1.3, 1.2, and 0.6 times each sector's
32 portion of global BC emissions, owing to the degree of co-location with population globally. Avoided
33 deaths are likely underestimated for the residential sector since indoor exposure was excluded.

34 Figure 6-6 shows that while the industrial and residential sectors in East Asia have the greatest
35 BC emissions ("mitigation potential"), all three sectors in South Asia have the greatest mortality impacts
36 per unit of emissions ("mitigation efficiency"). Outside of South Asia and East Asia, mitigation efficiency
37 is greatest for the Former Soviet Union, Southeast Asia/Australia, and Europe, while mitigation potential
38 is greatest for the residential sector in Africa/Middle East and for the transportation sector in Europe
39 and North America.

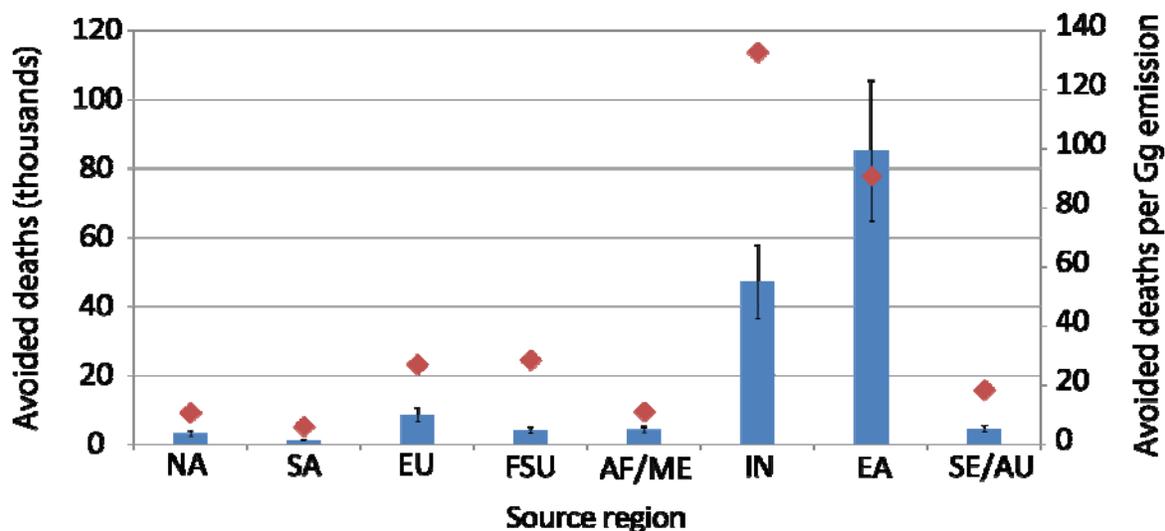
1 Finally, halving global anthropogenic OC emissions along with BC is estimated to result in eight
 2 times more avoided premature deaths annually than halving BC alone. This approach would prevent
 3 over one million premature deaths globally and about 16,000 deaths in the United States annually. This
 4 suggests that estimating reductions in BC alone greatly underestimates the full air pollution-related
 5 mortality benefits of BC mitigation, which in practice would be expected to reduce both OC and BC.

6 These study results suggest that the health benefits of mitigating BC emissions are likely to be
 7 substantial. Further, BC mitigation efforts are likely to be more effective at reducing mortality in some
 8 regions than others, largely driven by population exposure. Although the coarse grid resolution (~170
 9 km on a side) used by Anenberg et al. (in preparation) was unable to capture fine-scale spatial gradients
 10 in population and concentration, emissions from different sectors result in different exposure patterns.
 11 Therefore, the health response to controlling emissions from different regions and from different source
 12 sectors is likely to vary. Finer scale models can be used to investigate how different mitigation strategies
 13 impact health within individual world regions.

Abbreviation	Region
NA	North America
SA	South America
EU	Europe
FSU	Former Soviet Union
AF/ME	Africa/Middle East
IN	South Asia (India)
EA	East Asia (China)
SE/AU	Southeast Asia/Australia

14
 15
 16
 17

Table 6-1. Regional Definitions used by Anenberg et al. (in preparation)

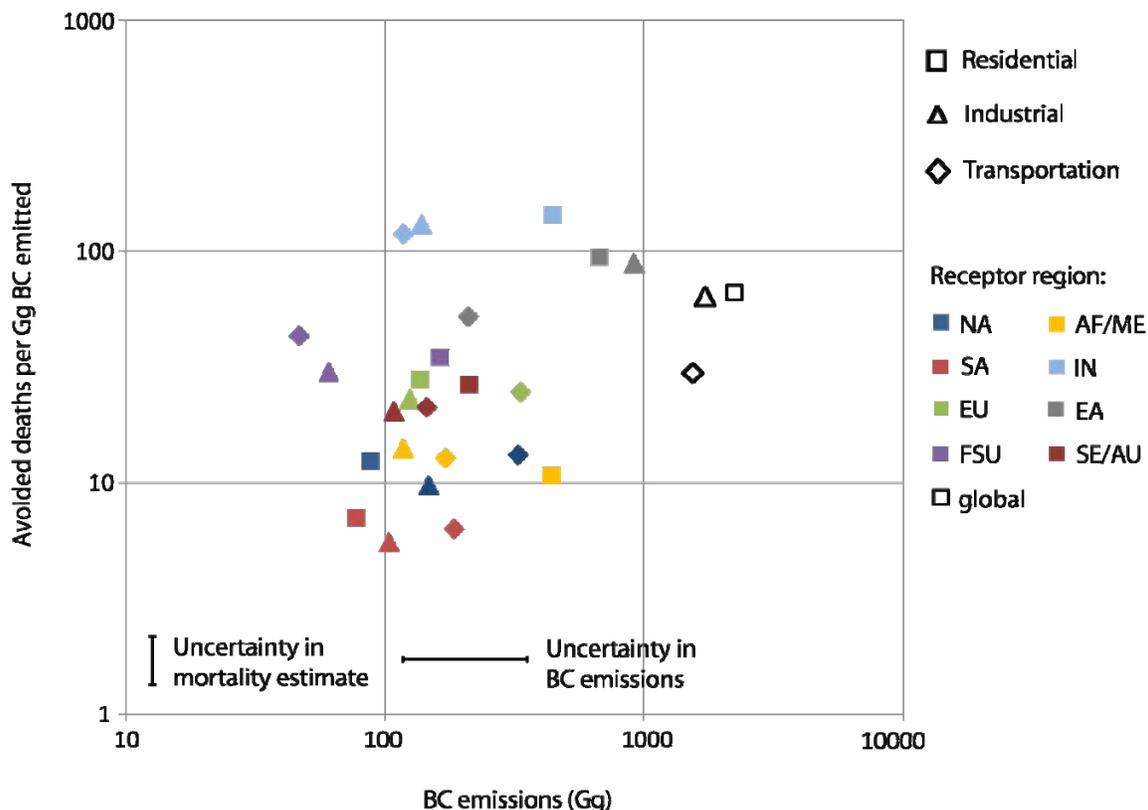


18
 19
 20
 21
 22

Figure 6-5. Estimated Global Mortality Benefits of Black Carbon Reductions. Global annual avoided premature cardiopulmonary and lung cancer deaths (thousands; blue bars) and avoided premature deaths per Gg BC emissions reduced (red diamonds), for halving anthropogenic BC emissions in each source region relative to the base case (Anenberg et al. in preparation).

- 1 Notes:
 2 1. Confidence intervals (95%) reflect uncertainty in the CRF only.

3



4
 5 **Figure 6-6. Annual Avoided Premature Cardiopulmonary and Lung Cancer Deaths Per Unit BC Emissions**
 6 **Reduced vs. Total BC Emissions (Gg) for Particular Source Sectors within Each Region** (Anenberg et al. in
 7 preparation).

- 8 Notes:
 9 1. Avoided deaths are estimated in the three simulations where global emissions in each sector are halved, and shown
 10 for each receptor region; these deaths are compared with emissions from each region, assuming that deaths from
 11 inter-regional transport are negligible.

12 Uncertainty in the mortality estimates is calculated from the uncertainty in the CRF only (~22%
 13 and 56% from mean for cardiopulmonary and lung cancer mortality). Uncertainty in BC emissions is
 14 assumed to be a factor of 2 from the central estimate (Bond et al., 2004; 2007). Since these
 15 uncertainties are factor differences from the central estimate, they are identical for each data point.
 16 Only a few other studies have examined the health benefits of specific international BC mitigation
 17 measures. The recently released draft UNEP assessment (UNEP, 2011, under review) provides the most
 18 comprehensive analysis to date of the benefits of BC mitigation measures, including both the technical
 19 and non-technical measures described above. These technical and non-technical measures together are
 20 estimated to reduce millions of premature PM_{2.5}-related deaths worldwide annually, based on 2030
 21 population and emissions. Consistent with the results of Anenberg et al. (in preparation), over 80% of
 22 the benefits occur in Asia. The study also found that the substantial health benefits of the joint air

1 quality/climate mitigation measures examined occur regardless of whether measures to reduce long-
2 lived GHG have been implemented.

3 In addition to the UNEP study and Anenberg et al. (in preparation), there is a small but emerging
4 body of literature assessing the global health benefits of PM_{2.5} emission reductions. The results from
5 additional global and international studies are summarized in Appendix 3. Many of these studies
6 estimate the avoided premature mortalities associated with reductions in BC and other constituents,
7 while other studies attempt to compare the costs and benefits of potential mitigation strategies. These
8 studies indicate that a large number of premature deaths can be avoided annually by undertaking
9 strategies to reduce BC emissions (Anenberg, et al., in preparation; Wilkinson, 2009; Saikawa et al.,
10 2009; Jacobson, 2010). The studies that include a benefit-cost comparison show that estimated human
11 health benefits significantly exceed the estimated costs for certain BC mitigation strategies (Smith et al.,
12 2008; Baron et al., 2009; and Kandlikar et al., 2009). Thus, such reductions appear advantageous to
13 society independent of the level of climate benefits achieved. This is particularly true of sources
14 associated with high human health exposures, such as cookstoves (which are often used indoors in
15 confined spaces) and sources (such as vehicles) located in densely populated areas. Strategies that
16 reduce emissions from these sources can achieve widespread health benefits; in the case of cookstoves,
17 such benefits accrue especially to women and children, who experience the highest exposures and
18 therefore are most at risk from cookstove emissions.

19 **6.6.2 Health and Welfare Benefits in the United States**

20 Historically, the United States has been quite successful in achieving significant PM_{2.5} reductions
21 through programs such as the PM_{2.5} NAAQS and a variety of mobile source rulemakings yielding large
22 human health and welfare benefits. EPA's analyses have consistently suggested that the benefits
23 outweigh the costs of these domestic emissions control programs by a wide margin. Further reductions
24 in ambient PM_{2.5}, including BC and other constituents, are likely to yield additional public health benefits
25 in the United States. However, the magnitude of the PM_{2.5} reductions achieved to date in the United
26 States means that the remaining mitigation potential is not as high as it is in countries that have yet to
27 implement significant PM control programs.

28 Analyses conducted to date indicate that PM_{2.5} control programs in the United States yield
29 significant health and environmental benefits.³ At present EPA is able to quantify and monetize a
30 number of human health benefit categories such as avoided mortalities, avoided respiratory and cardiac
31 events, and other health effects, but many non-climate welfare categories are un-quantified with
32 monetization typically limited to visibility improvements in specific geographic U.S. locations. In the
33 illustrative regulatory impact analysis conducted by EPA in 2006 for the revised PM_{2.5} NAAQS, benefits
34 are estimated to be more than 3 times higher than costs in 2020 (U.S. EPA, 2006). Benefits for the
35 nonroad diesel rule are estimated to exceed costs by more than 47 times in 2030 (U.S. EPA, 2004).

³ A list of PM benefits that are currently quantifiable and can be evaluated in monetary terms is provided in the Regulatory Impact Analysis for PM_{2.5} National Ambient Air Quality Standard (U.S. EPA, 2006).

1 Additional rules with direct PM reductions benefit-to-cost ratios are shown in Table 6-2 below.⁴ It is
 2 relevant to note that the rules listed reflect not only direct PM reductions but typically reductions in PM
 3 precursors and hazardous air pollutants.

4 EPA has separately estimated benefits per ton of PM_{2.5} reduced from various U.S. source
 5 categories (Fann, Fulcher and Hubbell 2009). The benefit-per-ton estimates found in Fann, et al. (2009)
 6 reflect a specific set of key assumptions and input data. As EPA updates these underlying assumptions
 7 to reflect the scientific literature, the benefit-per-ton estimates are re-estimated and are available at:
 8 <http://www.epa.gov/air/benmap/bpt.html>. These methods provide a useful “shorthand approach” for
 9 assessing potential benefits that may flow from different mitigation strategies on a broad scale. For
 10 directly emitted PM_{2.5} (including BC) from all sources, these benefits (on average) range from \$210,000
 11 to \$820,000 per ton of PM_{2.5} reduced in 2015 (2006\$). While EPA has not separately estimated the
 12 benefits per ton for BC reductions specifically, Table 6-3 illustrates the results for reductions in direct
 13 carbonaceous emissions generally (i.e. BC + OC) for 2015, 2020, and 2030. It is clear that controls on all
 14 sources of direct PM_{2.5} can produce substantial public health benefits in the United States; furthermore,
 15 these benefits are 7 to 300 times greater than the benefits per ton estimated for reductions of other PM
 16 precursors (Fann et al. 2009), indicating that controls on direct PM_{2.5} may be particularly effective for
 17 protecting public health.

18 **Table 6-2. List of Benefits, Cost and Benefit to Cost Ratios for US Rules with Direct PM Reductions^{1,2}**

Rule (by Sector)	Annual Benefits	Annual Costs	Benefit/Cost Ratio	Benefit Year
Transportation				
(billions of dollars)				
Light Duty Tier 2	\$25	\$5.3	4.7	2030
Heavy Duty 2007	70	4.2	16.7	2030
Nonroad Diesel Tier 4	80	1.7	47.1	2030
Locomotive & Marine Diesel	11	0.7	15.7	2030
Ocean Vessel Strategy	107	3.1	34.5	2030
Stationary Sources				
2006 PM NAAQS ³	\$17	\$5.40	3.1	2020
Cement NESHAP ⁴	\$7.4-\$18	\$.93-\$.95	8-18.9	2013
Stationary Spark Ignition RICE NESHAP ⁴	\$510-\$1200	\$253	2-4.7	2013
Stationary Compression Ignition Engine NESHAP ⁴	\$940-\$2,300	\$373	2.5-6.2	2013
¹ Rules include a combination of direct PM _{2.5} and PM precursor reductions ² 3% discount rate used for benefit estimates ³ Estimates of benefits and costs for the PM NAAQS are illustrative since individual States will make the decisions about actual control strategies implemented to comply with the NAAQS ⁴ Millions of dollars annually, year dollars vary with each rule but are consistent for each rule.				

19

20

⁴ These estimated benefit-to-cost ratios relate to reductions in not only direct PM_{2.5} but also in other controlled co-pollutants. EPA did not estimate the costs and benefits of controls on direct PM_{2.5} or specific constituents separately.

1 **Table 6-3. Direct PM_{2.5} National Average Benefits Per Ton Estimates by Source Category for the United**
 2 **States (3% discount rate)¹**

	Monetized Benefit Per Ton in 2015	Monetized Benefit Per Ton in 2020	Monetized Benefit Per Ton in 2030
PM Benefits Per Ton (thousands of 2006 dollars)			
Area Source			
Pope et al.	\$340	\$370	\$450
Laden et al.	820	910	1,100
Mobile Source			
Pope et al.	260	280	340
Laden et al.	630	700	830
EGU and Non-EGU			
Pope et al.	210	230	270
Laden et al.	520	570	660

3 ¹ Source PM_{2.5} Benefit Per Ton Estimates. <http://www.epa.gov/air/benmap/bpt.html> (data accessed February 2011) and Fann,
 4 Fulcher, Hubbell 2009 methodology. In this analysis carbon refers to directly emitted carbonaceous particles. These are U.S.
 5 national average estimates, and these estimates may vary for different geographic locations in the country.

6 **6.7 Key Factors to Consider in Pursuing BC Emissions Reductions**

7 Current policies, if fully implemented, are expected to achieve significant reductions in BC
 8 emissions in the coming decades, with significant regional variability. However, these reductions will be
 9 gradual, and even after they are fully realized, substantial BC emissions will remain in some sectors and
 10 regions. Thus, there is a great deal of interest in identifying strategies that can be employed to achieve
 11 further—or faster—reductions in BC emissions in different regions and sectors. Though there are some
 12 remaining uncertainties regarding the most efficient and effective mitigation pathway(s), considering
 13 the following key factors could help improve the likelihood that selected mitigation strategies will
 14 achieve substantial public health benefits and reduce the rate of near-term warming:

- 15 • PM mitigation strategies that focus on sources known to emit large amounts of BC—
 16 especially those with a high ratio of BC to OC, like diesel emissions—will maximize climate
 17 benefits.
- 18 • BC reductions depend on mitigation activities that reduce *directly emitted* PM_{2.5}. Strategies
 19 that focus on reducing SO₂ and NO_x emissions to control secondarily formed PM will not
 20 result in substantial decreases in BC emissions.
- 21 • BC is a regional pollutant. Therefore, it is important to evaluate the benefits of emissions
 22 reductions in terms of regional impacts, not just in terms of global averages. Considering
 23 the location and timing of emissions and accounting for co-emissions will improve the
 24 likelihood that mitigation strategies will be beneficial for both climate and public health.
 - 25 ○ Some of the largest climate benefits of BC-focused control strategies may come
 26 from reducing emissions that affect the Arctic, the Hindu Kush-Himalayan-Tibetan
 27 Plateau, and other ice- and snow-covered regions.

- 1 ○ In addition, large climate benefits may also result from reductions in regions that
2 have both high BC emissions and large populations, such as East Asia, South Asia,
3 and urban areas throughout the world. Reductions in these regions can provide
4 benefits for climate, health, and precipitation.
- 5 • The effect of BC emissions reductions on human health are a function of changing exposure
6 and the size of the affected population. The largest human health benefits from BC-focused
7 control strategies will occur locally near the emissions source and where exposure affects a
8 large population.
- 9 • Cost benefit analysis conducted to support mitigation decisions should incorporate public
10 health and welfare benefits as well as climate benefits. Such analysis can fully inform
11 decision makers regarding available choices, but can be complicated by uncertainties in
12 valuing the climate impacts. Nevertheless, public health benefits alone are so large that
13 they are likely to overshadow the net cost-benefit analysis of any BC reduction strategy,
14 regardless of the climate impacts.

15 **6.8 Overview of Main Mitigation Options**

16 The control strategies described in detail in the next four chapters reflect the range of existing
17 mitigation programs that affect BC emissions. The mobile and stationary source reductions that have
18 been achieved indicate that currently available control technologies and strategies can achieve
19 impressive reductions in BC emissions. For other source categories, existing mitigation options are
20 more limited or are more poorly characterized. For many types of sources, BC is a small portion of the
21 total direct PM_{2.5} emissions, and there is limited information regarding the impact of available control
22 strategies on BC and other individual components of the PM_{2.5} mixture. Research suggests, for example,
23 that some models of improved cookstoves may not substantially reduce BC emissions (or may even
24 increase them). For stationary sources, too, additional research is needed to determine whether newer
25 add-on pollution controls are leading to further BC reductions. For open biomass burning, large
26 uncertainties remain about the composition and volume of emissions resulting from different types of
27 fires. Though these fires are known to vary significantly depending on the type of fuel(s) involved and
28 the local conditions under which they occur, there is relatively little information about the impact of
29 various control strategies on BC and BrC emissions resulting from these fires.

30 **6.8.1 Black Carbon Mitigation in the United States**

31 As discussed above, substantial reductions in U.S. BC emissions are expected to be achieved by
32 2030, largely due to current or planned controls on mobile diesel engines. Other source categories,
33 including stationary source emissions, residential wood combustion, and open biomass burning, have
34 more limited mitigation potential due to smaller remaining emissions in these categories, or limits on
35 the availability and implementation feasibility of additional, effective BC control strategies. The
36 following chapters of this report address in greater detail what is known about control technologies,

1 costs and implementation for BC emissions in the United States. Some of the key messages can be
2 summarized as follows:

- 3 • The United States has already enacted stringent standards for *new mobile source engines*
4 that are projected to reduce BC emissions by 84% between 2005 and 2030 as the existing
5 fleet is replaced by these new engines. Though the standards do not dictate the use of a
6 specific technology, the mobile source reductions will be achieved mostly from DPFs on new
7 diesel engines, in conjunction with ultra low sulfur diesel fuel. EPA has estimated the cost of
8 controlling PM_{2.5} from diesel engines via new engine requirements at about \$11,000-
9 12,000/ton.
- 10 • For the existing 11 million *in-use mobile diesel engines* in the United States, programs such
11 as EPA's National Clean Diesel Campaign and the SmartWay Transport Partnership Program
12 can help achieve additional emissions reductions through retrofits. DPFs can reduce PM_{2.5}
13 emissions by up to 99%, at a cost of \$5,000 to \$15,000 for passive DPFs, and \$20,000 to
14 \$50,000 for active DPF systems. However, not all engines are good candidates for DPFs
15 because of old age or poor maintenance.
- 16 • *Stationary source BC emissions* in the United States have declined dramatically in the last
17 century; remaining emissions constitute 8% of the inventory and come primarily from coal
18 combustion (utilities, industrial/commercial boilers, other industrial processes) and
19 stationary diesel engines. Available control technologies and strategies include direct PM_{2.5}
20 reduction technologies such as fabric filters (also known as baghouses), electrostatic
21 precipitators, and DPFs. These strategies range in cost from as little as \$35 per ton PM_{2.5} to
22 over \$20,000 per ton PM_{2.5}, depending on the source category. They may also involve
23 millions of dollars in initial capital costs.
- 24 • *Residential wood combustion (RWC)* represents a small portion (3%) of the U.S. BC
25 inventory, but mitigation opportunities are available. In part because seasonal use of these
26 sources is concentrated in northern areas, reducing emissions may reduce deposition on
27 snow and ice. EPA already has established emissions standards for new residential wood
28 stoves, and is working to revise and expand these standards to include all residential wood
29 heaters, including hydronic heaters, furnaces, and fireplaces as well as stoves. Mitigation
30 strategies for RWC sources include providing alternatives to wood, replacing inefficient units
31 (wood stoves, hydronic heaters) with newer, cleaner units through voluntary or subsidized
32 change-out programs, or retrofitting existing units to enable use of alternative fuels such as
33 natural gas (fireplaces). New EPA-certified wood stoves have a cost of about \$3000 per ton
34 PM_{2.5}, while gas fireplace inserts average \$1500 per ton PM_{2.5}.
- 35 • *Open biomass burning*, including wildfires, prescribed fires and agricultural burning,
36 accounts for a significant portion (35%) of the U.S. BC inventory (with wildfire contributing
37 about 68% to that total). However, data on the percent of land area affected by different
38 types of burning is very limited, and emissions inventories are highly uncertain. At this time,
39 little is known about how specific measures would impact climate, both globally and

1 regionally. Developing appropriate mitigation is highly dependent on a number of variables,
2 including timing and location of burning, resource management objectives, vegetation type,
3 and available resources. The costs of mitigation measures are uncertain and potentially
4 high, as they depend on various site-specific factors.

5 **6.8.2 Black Carbon Mitigation around the Globe**

6 In contrast to remaining emissions in the United States, significant international BC mitigation
7 opportunities are available in the residential and industrial sectors, although mobile sources also remain
8 a major international source of BC. This is a reflection of both differences in the major emitting source
9 categories within the inventory, and the particular control technologies and opportunities available for
10 deployment internationally. It is important to note also that there are significant differences in control
11 opportunities between developed and developing countries. Other developed countries have emissions
12 patterns and control programs that are similar to the United States, though the timing of planned
13 emissions reductions may vary. Developing countries have a higher concentration of emissions in the
14 residential and industrial sector, but the growth of the mobile source sector in these countries may lead
15 to an increase in their overall BC emissions and a shift in the relative importance of specific BC emitting
16 sources over the next several decades. More information on these source categories is available in the
17 following sector mitigation chapters, though cost, emissions and other relevant data are not as widely
18 available internationally.

- 19 ○ *Residential cookstoves* represent one of the most promising opportunities internationally, in
20 large part because of the enormous public health benefits that could result from cooking with
21 cleaner fuels and stoves. Approximately 3 billion people worldwide cook their food or heat their
22 homes by burning biomass or coal in rudimentary stoves or open fires. The resulting exposures
23 to cookstove emissions lead to 2 million deaths each year, making cookstoves the fifth worst
24 overall health risk factor in poor developing countries. BC emissions from cookstoves are
25 estimated to account for approximately 27% of the global inventory, though these estimates are
26 uncertain and likely accompanied by substantial co-emissions of OC. While the world-wide
27 stove market is approximately 500-800 million households, current programs likely replace only
28 approximately 5-10 million improved stoves per year. Significant expansion of such programs
29 would be necessary to achieve large-scale climate and health benefits. A wide range of
30 improved stove technologies is available, but the potential climate and health benefits vary
31 substantially by technology and fuel. The costs range from \$8-\$100+ per stove. Improved
32 cookstove technologies all face important supply, cost, performance, usability, marketability and
33 other barriers that have impeded progress in the past; however, a number of factors point to
34 much greater potential to achieve large-scale success in this sector today.
- 35 ○ Emissions from *stationary sources* represent 14.5% of the global inventory; major sources
36 include brick kilns, coke ovens (largely from iron/steel production), electric utilities, and oil and
37 gas flaring. Little is known about many of these sources and their emissions, though preliminary
38 data does suggest that in some countries, these facilities are uncontrolled. Nevertheless, as

1 indicated by Anenberg and the UNEP Assessment, mitigation opportunities in this sector may be
2 substantial.

- 3 ○ For *mobile sources*, which account for approximately 18% of the global inventory, both new
4 engine standards and retrofits of existing engines/vehicles may help reduce BC emissions in the
5 future. Other countries have begun the phase-in of emissions standards and ULSD fuel, which is
6 a prerequisite for the proper functioning of DPFs. However, these requirements lag behind in
7 some regions, as do on-the-ground deployment of DPFs and ULSD. As a result, there remains
8 significant opportunity internationally to accelerate the deployment of clean engines and fuels.
- 9 ○ *Open biomass burning* is the largest source of BC emissions globally, accounting for over 40% of
10 the inventory. This includes emissions from agricultural burning, prescribed fires, and wildfires,
11 which together affect 340 million hectares/year. However, emissions of OC are seven times
12 higher than BC emissions from this sector, and better and more complete emissions inventory
13 data are needed to characterize impacts of biomass burning and evaluate the effectiveness of
14 mitigation measures at reducing BC. Successful implementation of mitigation approaches in
15 world regions where biomass burning is widespread will require training in proper burning
16 techniques and tools to ensure effective and appropriate use of prescribed fire.

(This page intentionally left blank)

7. Mitigation Approaches for Mobile Sources

7.1 Summary of Key Messages

- In the United States, mobile sources accounted for 52% of total BC emissions (69% of non-wildfire emissions) in 2005, approximately 90% of which came from diesel vehicles or engines. On a global basis, mobile sources are responsible for approximately 20% of the BC emissions, with total emissions and percentage attributable to mobile sources both significantly lower in developing countries.
- Mobile source BC emissions in developed countries have been declining rapidly since the 1990s due to regulations on PM emissions from new engines, and substantial further emissions reductions are expected by 2030 and beyond. Such regulations have been effective in reducing emissions of BC from on-road vehicles (mainly diesel trucks), and nonroad diesel engines, locomotives, and commercial marine vessels, particularly in the United States and Europe.
 - In the United States, new engine requirements have resulted in a 30% reduction in BC emissions from mobile sources between 1990 and 2005. As vehicles and engines meeting new regulations are phased into the fleet, a further 84% reduction in BC emissions from mobile sources is projected from 2005 to 2030, leading to a total decline of 89% in BC emissions between 1990 and 2030.
 - Most of these reductions are concentrated in the diesel fleet, and can be achieved via application of diesel particulate filters (DPFs) combined with ultra low sulfur diesel fuel. DPFs typically eliminate more than 90% of diesel PM and can reduce BC by as much as 99%.
 - The cost of controlling PM_{2.5} from most types of diesel engines is about \$11,000-\$12,000/ton based on prior EPA rulemakings.
 - Internationally, other countries have and are continuing to adopt emission standards (including those for diesel engines with ultra low sulfur fuel) similar to EPA emission standards. However, standards for new engine lag behind in some regions.
- There are approximately 11 million on-highway and nonroad diesel engines currently in operation in the United States, and many of these engines will remain in operation for the next 20 to 30 years.

- 1 • Currently available, cost-effective diesel retrofit strategies can reduce harmful emissions from in-use
2 engines substantially.
 - 3 ○ DPFs can reduce PM emissions by up to 99%, at a cost of \$5,000 to \$15,000 for passive DPFs,
4 and \$20,000 to \$50,000 for active DPF systems. However, not all engines are good
5 candidates for DPFs because of old age or poor maintenance.
 - 6 ○ The National Clean Diesel Campaign and the SmartWay Transport Partnership Program are
7 EPA's two primary programs responsible for reducing emissions from in-use vehicles and
8 equipment. NCDC has provided over \$460 million in grant funds and the SmartWay
9 program has enrolled over 2600 partners to support reductions in diesel emissions.
 - 10 ○ Other strategies to reduce emissions from existing engines include improved fleet
11 maintenance practices; engine repower, upgrades, or reflash; cleaner fuels; fuel economy
12 improvements; idle reduction programs; and shifts in mode of transportation.
 - 13 ○ Internationally, retrofit programs present significant financial and logistical challenges. This
14 is particularly true in developing countries, where infrastructure is lacking to assist with
15 vehicle registration, inspection and maintenance programs, technology certification/
16 verification programs, and application of readily available technologies. Vehicles in these
17 regions tend to be older and less well-maintained than in developed countries, and the
18 availability of low-sulfur diesel fuel is limited. In addition, the costs of DPFs may be
19 prohibitive for some countries.

20

21 **7.2 Introduction**

22 A number of PM_{2.5} control strategies have proven successful in reducing BC emissions from
23 mobile sources, which represent one of the most important categories of BC¹ emissions globally,
24 especially within developed countries (see Chapter 4). The two principal strategies include: (1)
25 emissions standards for new vehicles and engines, with emissions reductions occurring as the vehicle
26 and engine fleet turns over, and (2) controls or strategies that reduce emissions from existing engines,
27 such as diesel retrofits. In this section, these two major strategies are explored, with emphasis on
28 describing the anticipated impact of these approaches on emissions by 2030. It is important to note

¹ As mentioned in Chapter 5, optical measurements of BC are limited and vary depending on measurement technique. Measurements of elemental carbon (EC) by thermal optical methods are more widespread and consistent; mobile source emissions inventories and information about control strategies for mobile sources usually involve EC measurements. To ensure consistency in this report, however, the term BC is used throughout.

1 that these strategies are complementary, and can be employed simultaneously. The joint application of
2 new engine standards and controls on in-use engines has been very successful in both the United States
3 and Europe in reducing direct PM emissions—including BC—from mobile sources.

4 Existing programs provide important insights into achievable emissions reductions, costs, and
5 implementation challenges for new and existing vehicles/engines in the mobile sector. Emphasis is
6 placed on programs and strategies which have proven successful in the United States, including both
7 new vehicle/engine standards and programs addressing in-use diesels such as EPA’s National Clean
8 Diesel Campaign, the SmartWay Transport Partnership Program, and California’s mandatory diesel
9 retrofit program. The section discusses the impact of these approaches on current and anticipated
10 future emissions levels, and describes the specific control technologies and strategies involved, along
11 with the cost of these approaches. A close examination of such strategies may offer insights into
12 applicability of such strategies elsewhere.

13 The main technology for diesels reducing black carbon emissions is the catalyzed diesel
14 particulate filter (DPF) discussed later in this section. It is important to note that since DPFs are
15 poisoned by fuels with high sulfur content, mitigation of mobile source BC emissions depends on the
16 availability and widespread use of ultra low-sulfur fuels (15 ppm sulfur). Typically, the low-sulfur diesel
17 fuel is in the market place about the same time that the DPFs are introduced, although some countries,
18 particularly in the developing world, may introduce low-sulfur fuel before adopting stringent PM
19 emission standards. The timing of ultra low-sulfur fuel availability in different world regions is discussed
20 in this section, and in further detail in Appendix 4.

21 **7.3 Emissions Trajectories for Mobile Sources**

22 As discussed in Chapter 4, mobile sources remain the dominant emitters of BC in developed
23 countries. In the United States, for example, mobile sources were responsible for about 52% of BC
24 emissions in 2005, almost all of which (90%) came from diesel vehicles or engines. If wildfire emissions
25 are excluded, then mobile sources account for 69% of the 2005 domestic inventory. On a global basis,
26 mobile sources are responsible for approximately 20% of the BC (Bond et al, 2004) with total emissions
27 and percentage attributable to mobile sources both significantly lower in developing countries. A
28 number of studies have projected that these emissions are likely to increase globally in the future,
29 largely due to growth in the transportation sector in developing countries (Streets et al., 2004; Jacobson
30 and Streets, 2009) (see Chapter 6). However, mobile source BC emissions in developed countries have
31 been declining rapidly since the 1990s. Regulations on (PM) emissions from new engines, particularly in
32 the United States and Europe, have been effective in reducing emissions of BC from on-road vehicles
33 (mainly diesel trucks), and nonroad diesel engines, locomotives, and commercial marine vessels,
34 although Europe has not currently adopted stringent locomotive and commercial marine standards as
35 the United States has. Substantial emissions reductions are expected over the next two decades and
36 more.

37 In the United States, new engine requirements have resulted in a 30% reduction in BC emissions
38 from mobile sources between 1990 and 2005. As vehicles and engines meeting new regulations are

1 phased into the fleet, a further 84% reduction in BC emissions from mobile sources is projected from
2 2005 to 2030, leading to a total decline of 89% in BC emissions between 1990 and 2030 as shown in
3 Table 7-1 . Most of these reductions are concentrated in the diesel fleet. For example, from 1990-2005,
4 there was a 30% decline in BC emissions from diesel trucks. Due to new regulations, a further 95%
5 decline is projected in diesel truck BC emissions by 2030 (97% total decline since 1990). Other
6 categories of diesel engines, such as nonroad diesels (e.g., agricultural, construction equipment),
7 commercial marine diesels, and locomotives are also projected to have major declines (75-90%) in BC
8 emissions from 2005 to 2030 in the United States. BC emissions from gasoline vehicles and nonroad
9 gasoline engines, which are much smaller sources of BC, are projected to decline by 80% during 1990-
10 2030 time period, with a 23% reduction occurring from 2005-2030 with most of that reduction occurring
11 for onroad gasoline vehicles largely due to the use of catalyts.^{2,3}

12 Considering only the emissions from United States mobile sources occurring north of the 40th
13 parallel in 2005, EPA estimates there will be a substantial decline of approximately 83% in these
14 emissions by 2030 as well. The domestic emissions reductions are slightly higher north of the 40th
15 parallel due to expected more rapid vehicle/nonroad equipment turnover than in the country as a
16 whole. However, these estimates do not reflect potential future increases in emissions from marine
17 freight transport that may occur under future climate scenarios. The total or seasonal loss of Arctic sea
18 ice may result in new marine trade routes through the Arctic. Such developments could potentially
19 result in greater emissions in the Arctic, with greater potential for deposition on remaining ice. United
20 States emissions inventories currently contain no projections of these potential future emissions in the
21 Arctic area.

22 Table 7-1 below shows the emission reductions in black carbon (as well as PM_{2.5} and organic
23 carbon) going from 1990 through 2030 for various mobile source sectors which are discussed in the
24 following sections. Also, Figure 7-1 shows the reductions in BC graphically from 1990 through 2030.

25

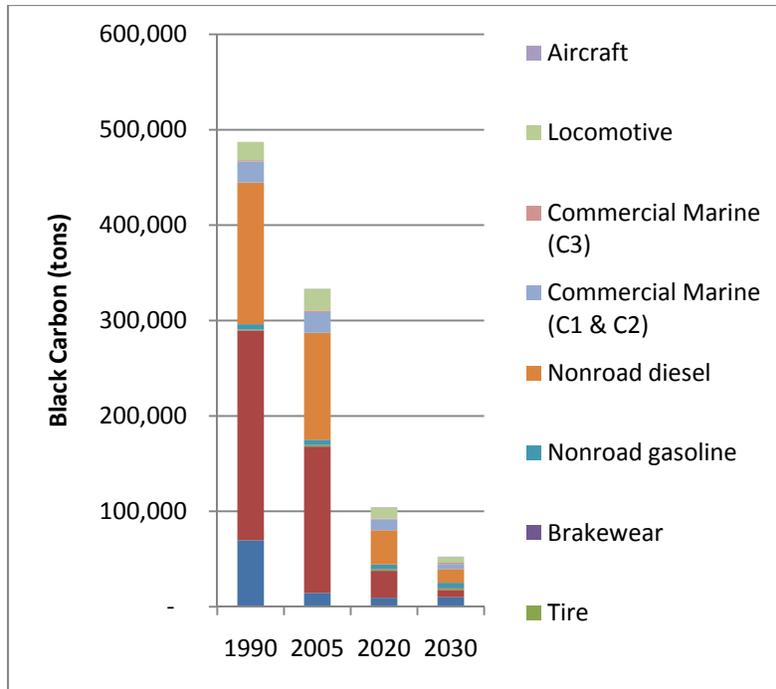
² Unlike the reductions for diesels, the reductions in BC from gasoline engines occurred due to regulation of other pollutants (such as hydrocarbons [HC], carbon monoxide [CO], and oxides of nitrogen [NO_x]) rather than regulation of PM itself. In general, BC emissions from gasoline vehicles and engines have been less studied than those from diesel engines.

³ Tire and brake wear are also considered to be mobile sources. Emissions from these categories in the United States increased from 1990 to 2030 due to increases in vehicle miles traveled (VMT). Tire and brake wear are relatively minor sources of BC compared to exhaust emissions (i.e., less than 1% of the total in 1990 but 4% in 2030) although they are larger from a PM standpoint. Importantly, BC accounts for 22% of PM emissions from tire wear. At present, there are no EPA emission standards for either tire or brake wear PM emissions.

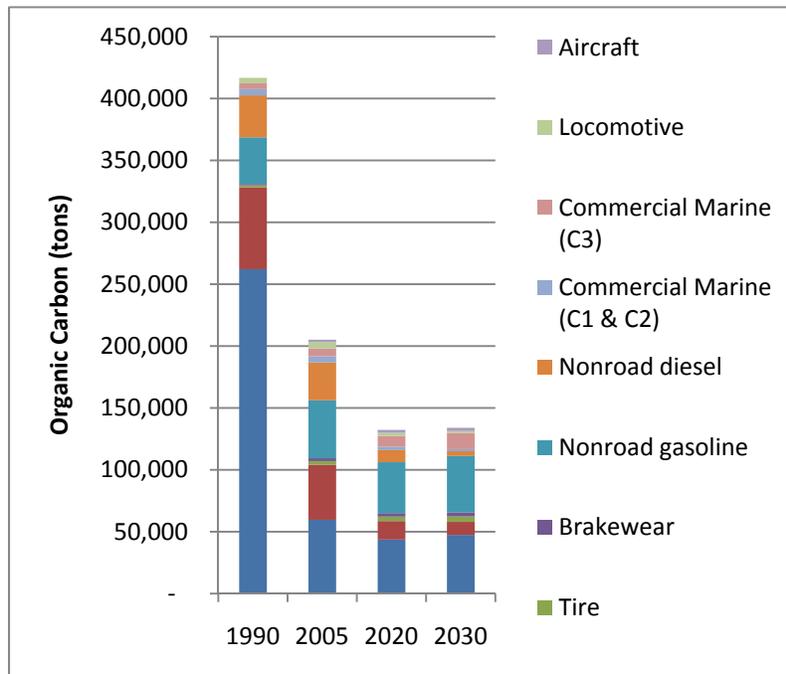
Table 7-1: Mobile Source BC, OC, and PM2.5 Emissions 1990-2030 (short tons).

Source Category	Year				% change	
	1990	2005	2020	2030	1990--->2005	2005--->2030
<u>ELEMENTAL CARBON</u>						
Onroad gasoline	69,629	14,510	9,538	10,027	-79%	-31%
Onroad diesel	219,958	153,477	28,175	7,615	-30%	-95%
Tire	809	1,198	1,435	1,720	48%	44%
Brakewear	290	475	569	682	64%	44%
Nonroad gasoline	5,420	5,444	4,702	5,174	0%	-5%
Nonroad diesel	148,537	112,058	35,718	14,240	-25%	-87%
Commercial Marine (C1 & C2)	22,122	21,652	11,595	5,440	-2%	-75%
Commercial Marine (C3)	1,262	1,681	864	1,306	33%	-22%
Locomotive	19,317	22,495	11,349	5,684	16%	-75%
Aircraft*	283	410	457	553	45%	35%
Total	487,627	335,405	106,423	54,471	-31%	-84%
<u>ORGANIC CARBON</u>						
Onroad gasoline	262,065	59,657	43,711	47,421	-77%	-21%
Onroad diesel	66,056	44,423	14,883	10,580	-33%	-76%
Tire	1,734	3,060	3,678	4,407	76%	44%
Brakewear	1,191	2,321	2,790	3,343	95%	44%
Nonroad gasoline	37,613	46,734	41,137	45,424	24%	-3%
Nonroad diesel	33,872	30,618	9,759	3,891	-10%	-87%
Commercial Marine (C1 & C2)	5,045	4,937	2,772	1,710	-2%	-65%
Commercial Marine (C3)	4,734	6,303	8,644	13,060	33%	107%
Locomotive	4,405	5,130	2,659	1,507	16%	-71%
Aircraft*	1,372	1,988	2,217	2,682	45%	35%
Total	418,087	205,172	132,252	134,025	-51%	-35%
<u>DIRECT PM2.5</u>						
Onroad gasoline	335,205	75,924	54,682	59,106	-77%	-22%
Onroad diesel	290,478	208,473	43,698	18,765	-28%	-91%
Tire	3,678	5,325	6,450	7,727	45%	45%
Brakewear	11,129	17,801	21,559	25,830	60%	45%
Nonroad gasoline	54,198	55,834	49,000	54,078	3%	-3%
Nonroad diesel	192,905	145,289	46,310	18,463	-25%	-87%
Commercial Marine (C1 & C2)	28,730	28,119	15,789	9,741	-2%	-65%
Commercial Marine (C3)	42,082	56,028	14,407	21,767	33%	-61%
Locomotive	25,087	30,910	15,145	8,584	23%	-72%
Aircraft*	2178	3,156	3,519	4,257	45%	35%
Total	985,671	626,859	270,559	228,318	-36%	-64%

* landings and take offs only; #s not available for in flight emissions

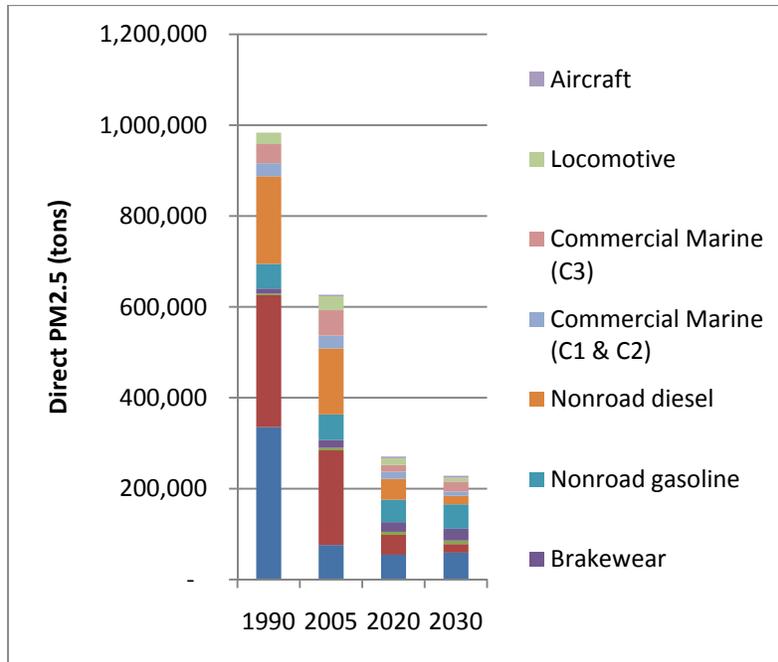


1

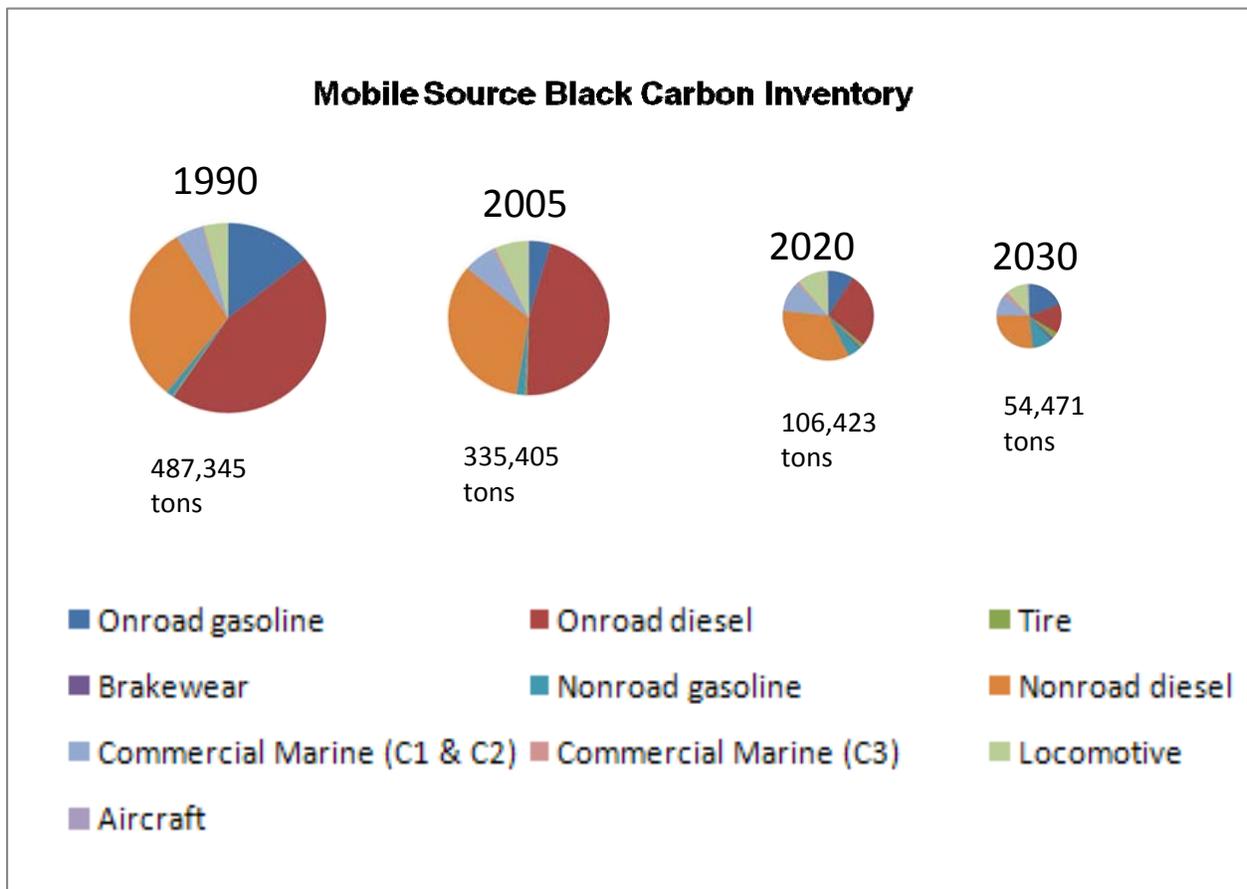


2

3



1



2

3

4

Figure 7-1. Estimated Changes in Emissions of Black Carbon, Organic Carbon, and Direct PM_{2.5} from Mobile Sources in the United States, 1990-2030.

1 **7.4 New Engine Standards in the United States**

2 In the United States, PM emissions standards for new mobile source engines are being phased in
 3 across different sectors between 2007 and 2020. These standards will lead to the large reductions in
 4 mobile source emissions of BC discussed above.⁴ The realized reductions depend on the rate of fleet
 5 turnover—i.e. the rate at which older vehicles and engines are replaced with new vehicles that comply
 6 with the latest emissions standards. The rate of fleet turnover depends heavily on the type of vehicle or
 7 engine, with on-road engines such as passenger cars and light-duty trucks being replaced more
 8 frequently than some other types of mobile sources, such as nonroad equipment. The state of California
 9 has its own diesel PM standards as promulgated by the California Air Resources Board (CARB). These
 10 standards are, in general, similar if not identical to the Federal standards. CARB also has its own
 11 gasoline PM standards. A detailed list of the mobile source PM standards is contained in Appendix 5.

12 **7.4.1 On-road and Nonroad Diesel Engines**

13 Diesel PM, as it exits the engine, is 70-80% BC for the pre-2007 model year diesel trucks and
 14 current diesel nonroad engines. The main source of diesel PM has traditionally been heavy-duty diesel
 15 trucks with gross vehicle weights from 8,501 to 80,000 lbs. The first standards controlling diesel PM for
 16 on-road engines were standards for visible smoke (which has some correlation with PM) effective with
 17 the 1970 model year followed by increasingly stringent PM mass standards starting with the 1988 model
 18 year. For the 2007 vehicle (engine) model year, stringent emission standards of 0.01 g/BHP-hr (grams
 19 per brake/horsepower/hour – a standard unit for emissions from heavy-duty mobile source engines)
 20 became effective for heavy-duty diesel engines, which represents over 99% control from a pre-control
 21 diesel engine in the 1970 time frame.⁵

22 As a result of these standards, BC emissions have been dramatically or even preferentially
 23 reduced as the major PM constituent.⁶ To meet these stringent PM standards, virtually all new on-
 24 highway diesels in the United States, beginning with the 2007 model year have been equipped with

⁴ EPA models the cumulative reductions for each category of mobile sources attributable to all past and current standards promulgated for that category rather than modeling the reduction for a particular standard.

⁵ EPA's emissions standards for heavy-duty diesel trucks have always been engine standards since the same engine can be used in a wide variety of truck chassis bodies with many of these bodies manufactured by companies different from those who manufacture the engines. For light-duty vehicles and trucks (trucks up to 8,500 lbs gross vehicle weight), the emission standards in g/mile apply to the car/truck itself.

⁶ Ultrafine particles from pre-2007 diesel engines generally comprise primarily of BC, OC, metals, and sulfates. DPFs preferentially reduce BC, OC, and metals. Also, the use of ultra low sulfur diesel fuel reduces total sulfate emissions (and emissions of ultrafine sulfate PM). Recent work shows that DPFs reduce particle number (an indicator of ultrafines or nanoparticles) by up to 90-99% based on emissions characterization with four 2007 heavy duty diesel engines. See Khalek et al., 2009.

1 diesel particulate filters (DPFs). DPFs typically eliminate more than 90% of diesel PM and can reduce BC
2 by as much as 99%. The type of DPFs typically used on new model year vehicles are called “wall flow”
3 filters with a catalyst coated on a ceramic monolith with the exhaust flowing through the filter walls
4 trapping the PM and allowing the exhaust gases to flow through. The trapped PM is then oxidized by
5 reaction with compounds such as oxygen and nitrogen dioxide on the catalyst surface. This technology
6 preferentially removes solid particles, such as BC. BC emissions from the heavy-duty diesel truck fleet
7 have been reduced by 30% from 1990-2005, and EPA projects that the application of DPFs will result in a
8 further 95% reduction by 2030, from 153,477 tons to 7,615 tons.

9 Corresponding national PM emissions standards of 0.01 g/mile took effect for U.S. passenger
10 cars (and light-duty trucks) from 2004-2006. These “Tier 2” standards apply to both gasoline and diesel
11 light-duty vehicles, although there are very few diesel passenger cars in the United States, unlike in
12 Europe where diesel passenger vehicles are used extensively.

13 Nonroad diesel engines also emit a significant amount of BC. EPA first set emission standards
14 for PM for these engines beginning in 1996. Recent rules issued in 2004, to be effective with the 2012
15 calendar year for newly manufactured engines, will result in widespread use of DPFs with dramatic
16 reductions (~ 99% from a pre-control engine) in PM and BC. These standards will be fully phased in
17 around 2015 for new model year nonroad diesel engines but will be phased into the fleet some years
18 later with fleet turnover. EPA’s latest version of the NONROAD model estimates all of these regulations,
19 including those resulting in use of DPFs, will result in an 87% decrease in emissions between 2005 and
20 2030, from 112,058 tons of BC in 2005 to 14,240 tons in 2030, despite substantial expected growth in
21 use of these engines over this time period. Cumulatively, this will be a 90% decrease from 1990 to 2030.

22 As mentioned briefly in the introduction to this chapter, an important prerequisite for the
23 application of DPFs is a switch to low-sulfur fuel. Low-sulfur fuel is needed, and has been required in the
24 United States by regulation, to preserve catalytic activity of the DPF, which is poisoned by sulfur. EPA
25 first regulated sulfur content in on-road diesel fuel to 500 ppm in 1993, resulting in typical fuel sulfur
26 levels of about 300 ppm. Prior to that, sulfur level in on-road diesel fuel was about 2,000 ppm. In 2006,
27 the sulfur level was limited to 15 ppm for on-road diesel fuel and has been reduced gradually in nonroad
28 diesel fuel, first to 500 ppm in 2007 for all categories except ocean-going vessels, and, starting in 2010,
29 to 15 ppm for most categories. In the case of locomotive and marine diesel fuel, this second step will
30 occur in 2012. Thus, all highway diesel vehicles and nonroad engines in the United States must now or
31 will soon operate on “ultra-low sulfur diesel” (ULSD). Typical in-use fuel sulfur levels are about 7 ppm.

32 It is important to note that the net climate impact of the application of DPFs will be offset
33 somewhat by the necessary co-emissions reductions in sulfate, which is reflecting (cooling).⁷ Also, while

⁷ The 15 ppm sulfur limit greatly reduces SO_x emissions, some of which convert to sulfate in the ambient air. For exhaust emissions of sulfates, the situation is more complicated since a typical conversion rate of SO₂ to sulfate for diesel engines without DPFs is about 2% but increases to about 50% for vehicles/engines with DPFs. Due to the dramatic reduction in diesel

1 diesel PM from pre-2007 engines has a high level of BC in PM, it also has some OC (about 22%), which is
 2 also greatly reduced by the DPF in later model years. The net climate impact of the application of DPFs
 3 will be affected by these reductions in OC emissions. Still, given the predominance of BC in diesel
 4 exhaust (70-80%), emissions reductions from this source category have a strong likelihood of providing
 5 climate benefits.

6 The EPA nonroad diesel rule⁸ issued in 2004 provides an aggregate cost estimate for controlling
 7 PM emissions using DPFs on new engines of about \$11,000-12,000 per ton. This cost figure includes the
 8 additional cost of ULSD fuel, engine costs, and equipment costs. As shown in Table 7-2, similar cost
 9 estimates were developed in 2001 for the Heavy-Duty Diesel Rule for on-road⁹ and the 2008 rule
 10 controlling emissions from locomotive and marine diesel engines.¹⁰

11 **Table 7-2. Cost Estimates for Particulate Matter Controls on New Diesel Engines**, based on recent U.S.
 12 EPA rulemakings

Rule	Estimated Cost per Ton PM _{2.5} Reduced	
	NPV, 3% rate	NPV, 7% rate
Heavy-Duty Diesel Rule (2001)	\$11,791	\$13,607
Nonroad Diesel Rule (2004)	\$11,200	\$11,800
Locomotive/Marine Rule (2008)	\$8,440	\$9,620

13
 14 It is important to note that the controls applied under these regulations affect multiple pollutants, not
 15 just BC. Furthermore, the analyses conducted during the 2001-2008 time frame utilized the best cost
 16 information available at that time, as well as emission reductions (total tons reduced) based on EPA's

fuel sulfur, there is still some reduction in sulfate emissions from vehicles/engines with DPFs and 7 ppm diesel fuel sulfur versus vehicles/engines without DPFs using fuel meeting the 500 ppm limit, which results in a typical sulfur level of 200-300 ppm. A 50% conversion of SO_x to sulfate with the typical 7 ppm fuel sulfur level results in less exhaust sulfate (about 35% less) than from an older pre-trap diesel using fuel with the 200-300 ppm sulfur limit.

⁸ *Control of Emissions of Air Pollution from Nonroad Diesel Engines and Fuel*. Federal Register: June 29, 2004 (Volume 69, Number 124). See specifically, *Final Regulatory Analysis: Control of Emissions from Nonroad Diesel Engines*, EPA420-R-04-007, Chapter 8, Table 8.7.1, page 33, May 2004 (<http://www.epa.gov/nonroaddiesel/2004fr.htm#ria>).

⁹ *Control of Air Pollution from New Motor Vehicles: Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements, Final Rule*. Federal Register: January 18, 2001 (Volume 66, Number 12). This rule applies to 2007 and later model-year heavy duty diesel on-road engines. See specifically, *Regulatory Impact Analysis: Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements*; Chapter VI, Table VI F-4, page VI-19, January 2001 (<http://www.epa.gov/otaq/highway-diesel/regs/ria-vi.pdf>).

¹⁰ *Control of Emissions of Air Pollution from Locomotive Engines and Marine Compression-Ignition Engines Less than 30 Liters per Cylinder; Republication*. Federal Register: June 30, 2008 (Volume 73, Number 126). See specifically, *Regulatory Impact Analysis: Control of Emissions of Air Pollution from Locomotive Engines and Marine Compression Ignition Engines Less than 30 Liters Per Cylinder*; Chapter 5, Table 5-67, page 5-98, June 2008 (<http://www.epa.gov/oms/regs/nonroad/420r08001a.pdf>).

1 then-current emission models. Since then, the emission models have changed; however, in the absence
2 of new analysis, the \$11,000-12,000 cost/ton is the best available EPA information for control of diesel
3 PM from newly manufactured on-road vehicles and nonroad engines.

4 **7.4.2 On-road and Nonroad Gasoline Engines**

5 On-road gasoline PM emissions have decreased dramatically over the years, especially with the
6 use of catalysts and unleaded gasoline starting with the 1975 model year vehicles. For example, PM
7 emissions for a typical car using leaded gasoline in 1970 were about 0.3 g/mile compared to emissions
8 from current vehicles with unleaded fuel of about 0.001 g/mile, a reduction of over 99% (Coordinating
9 Research Council, 2008). While BC emissions were not usually measured in the PM from cars in the
10 1970s, some limited measurements suggest that BC made up about 10-20% of the PM at that time,
11 compared to about 20% of PM mass in 2005. Thus, the per-vehicle PM reductions since 1970 have
12 resulted in a substantial reduction in BC emissions. Most of this BC comes under “rich” operating
13 conditions (where there is insufficient air for full combustion such as during cold-start or high load
14 conditions). EPA’s most recent modeling indicates that BC emissions from on-road gasoline engines
15 have declined 79% since 1990, from 69,629 in 1990 tons to 14,510 tons BC in 2005, and will decline a
16 further 30% by 2030 (to 10,027 tons).

17 Under the Tier 2 exhaust regulations mentioned above for light duty vehicles (passenger cars
18 and light-duty trucks), EPA set a PM emissions standard for both gasoline and diesel vehicles at 0.01
19 g/mile starting in 2004, with full phase-in for all light-duty vehicles (including light-duty trucks) in model
20 year 2009. When the Tier 2 rules were promulgated, EPA estimated that a total of 36,000 tons of PM_{2.5}
21 would be reduced in the year 2030 from these standards (versus not having these standards) using the
22 emissions models available at that time (U.S. EPA, 1999). Prior exhaust standards from the 1990s and
23 earlier also have helped reduce PM: while these regulations do not limit PM directly, they resulted in
24 better control of air/fuel ratio and improved catalyst formulations to meet HC, CO, and NO_x emissions
25 standards, all of which affected PM emissions levels. Because the regulations were targeted at other
26 pollutants, however, EPA has not calculated a cost for the resulting PM reductions specifically.

27 It should be noted that most new vehicles now emit below the Tier 2 PM standard by a factor of
28 10. However, recently a relatively new technology, gasoline direct injection (GDI), is being utilized for a
29 number of reasons such as improved fuel economy and performance. GDI engines differ from
30 conventional fuel injected engines in that the fuel is injected directly into the cylinder (like in a diesel
31 engine) rather than at the intake port. GDI vehicles are expected to constitute a major part of the new
32 vehicle fleet in the coming years and may be 90% of new vehicle sales in model year 2016. The specific
33 technology for injecting and guiding the gasoline spray into the engine coupled with the catalyst has a
34 major impact on the magnitude of the PM emissions. Some vehicles with GDI have higher PM emissions
35 (ranging from 0.002-0.020 g/mile versus a typical level of 0.001 g/mile for currently produced new
36 vehicles). Initial studies by the EPA and California Air Resources Board (CARB) suggest that “wall guided”
37 GDI results in higher BC emissions than more advanced “spray guided” GDI. However, other GDI
38 vehicles have emissions that are as low as conventional Tier 2 vehicles. EPA is now working on proposing
39 Tier 3 standards for LDGV vehicles which may address PM emissions from gasoline-fueled vehicles.

1 CARB has issued a preliminary discussion paper discussing the option of tightening the PM mass
2 standard effective for the 2014 model year (California Air Resources Board, 2010). The present CARB
3 PM standard (LEV II) is 0.01g/mile which is also the EPA emission standard. The possible standard
4 presented in the discussion paper is 0.006 g/mile in 2014 and 0.003 g/mile in 2017. CARB had also
5 considered a standard specifically for black carbon but announced at its November 2010 LEV III (Low
6 Emission Vehicle) workshop that it would not set such a standard. CARB is still considering though a
7 standard for PM number. Nonroad gasoline engines are either 2-stroke engines (where lubricating oil is
8 mixed into and burned with the gasoline) or 4-stroke engines. Such engines are used in a wide variety of
9 equipment including lawn/garden and recreational marine engines. These also emit significant PM
10 mass, especially the 2-stroke engines. EPA estimates that BC emissions from nonroad gasoline engines
11 will decline approximately 5% (from 5,444 tons to 5,174 tons) between 2005 and 2030, largely due to
12 changes needed to meet standards for volatile organic compound (VOC), CO, and NO_x emissions
13 standards being applied to several categories of nonroad gasoline engines which will also reduce PM.
14 Current information, which needs to be updated, used in EPA air quality modeling suggests that BC is
15 approximately 10% of PM mass with the same number being used for both 2-stroke and 4-stroke
16 engines. However, PM emissions from nonroad gasoline engines, particularly the 2-stroke engines, have
17 been characterized far less thoroughly than emissions from on-road gasoline vehicles, and EPA's
18 estimates of BC emissions are highly uncertain. EPA places a high priority on obtaining better BC
19 emission data from both 2-stroke and 4-stroke nonroad gasoline engines.

20 **7.4.3 Other Mobile Sources – Commercial Marine Vessels, Locomotives, and Aircraft**

21 Locomotives have used diesel (diesel electric) engines predominantly since the 1950s. EPA has
22 implemented several tiers of emission standards for PM for these engines with the most recent set of
23 standards to be effective in 2015. These newest standards will result in the use of DPFs on new
24 locomotives which preferentially reduce BC. In addition, national emission standards require that older
25 locomotives that are remanufactured must be certified to more stringent emission standards than their
26 prior certification level.

27 Commercial marine vessels are classified as C1, C2, and C3 based on engine size. C1 marine
28 engines are similar in size (less than 5 l/cylinder) to those used in construction/farm equipment. C2
29 marine engines (between 5-30 l/cylinder) are similar to locomotive diesels. The C3 engines (greater
30 than 30 l/cylinder vessels) are similar to those used in some power plants and are used in ocean-going
31 vessels. The most recent set of emission standards for these engines will result in most new C1 and C2
32 commercial marine engines having DPFs starting in 2014. For these engines, there will be a dramatic
33 drop in PM emissions and an even more dramatic drop in BC emissions. Like locomotives, older marine
34 diesel engines must be certified to more stringent emission standards upon remanufacturing, compared
35 to their previous certification level. The level of the standards to which these remanufactured engines
36 must be certified varies depending on engine type and year of manufacture for the original engine.

37 PM emissions from C3 engines comprise mainly sulfate (about 75%) and relatively little BC (less
38 than 1%). Due to recent work with the International Maritime Organization (IMO), there will be large
39 reductions in the higher sulfur level of the fuel (largely bunker diesel fuel composed of especially high

1 molecular weight, even solid, hydrocarbon compounds) used in these engines (see Appendix 4). As this
2 sulfur level is reduced, PM will be greatly reduced but BC levels are expected to stay roughly the same
3 on a per-vessel basis and will constitute a larger percentage of the PM emissions. There is some
4 increase in BC emissions from 2005 due to an increase in usage of these vessels. Though C3 marine is
5 responsible for less than 1,000 tons of BC emissions for the entire US, there is some concern that
6 emissions from these vessels could have disproportionate impact on the Arctic, especially if Arctic
7 marine traffic increases as shipping lanes open due to ice melt in the region. Additional BC emissions
8 data and modeling/deposition studies are needed to clarify the impact of C3 marine vessels.

9 There has been limited research into the BC emissions from aircraft. Additional
10 characterization of aircraft emissions would help to better understand BC emissions from aircraft
11 although there is sufficient information to develop a PM and an initial BC and OC inventory.

12 In general, therefore, additional emissions information for commercial vessels, locomotives and
13 aircraft would improve characterization of BC, since present data are limited, and it is difficult to
14 determine how much BC will be reduced by the PM standards affecting these sources.

15 **7.5 New Engine Standards Internationally**

16 Heavy-duty on-road diesel vehicles represent the predominant mobile source of BC in most
17 areas of the world, although nonroad diesel (and locomotives and commercial marine) can also be
18 significant. Given the importance of diesel engines internationally, use of DPFs to reduce PM_{2.5} will also
19 result in large reductions in BC from the global mobile source sector. Some countries have already
20 made significant progress in this area and have introduced diesel PM standards (mainly for on-road
21 vehicles) which effectively reduce BC. While broad-scale application of DPFs is an attractive option to
22 reduce global emissions, this is dependent on simultaneous use of ULSD fuel. Many other developed
23 countries in Europe and Asia have already adopted low-sulfur fuel requirements. As a result, BC
24 emissions from mobile sources are declining in many regions, especially in Europe and Japan. However,
25 many developing countries have not yet switched to low-sulfur fuel, and PM emissions controls are less
26 common. Each of these issues is discussed further, below.

27 **7.5.1 International Regulations of Diesel Fuel Sulfur Levels**

28 As noted above, the availability and widespread use of low-sulfur fuels is a critical prerequisite
29 to effective BC control from mobile sources. Like the United States, Canada, Japan, and the European
30 Union, have adopted strict control on on-road diesel fuel sulfur levels, and many other countries have
31 also adopted regulatory standards for reducing sulfur levels in on-road diesel fuel to levels needed to
32 enable low-emission vehicle technologies. In other regions, however, reductions in the sulfur content of
33 fuel lag behind. This effectively constrains BC emissions reductions in these countries, since higher
34 sulfur fuels prevent DPFs from functioning properly, even if they were applied.

35 The United Nations Environment Programme's (UNEP) Partnership for Clean Fuels and Vehicles
36 (PCFV) founded at the World Summit on Sustainable Development in 2002 promotes low sulfur fuels

1 and cleaner vehicle standards and technologies. This partnership has over 100 members from the oil
2 and gas industry, engine and retrofit manufacturers, government agencies, and environmental NGOs.
3 Currently, the PCFV is conducting a low sulfur campaign with a call for global adoption of 50 ppm sulfur
4 gasoline and diesel. The implementation of 50 ppm sulfur programs would allow countries to begin to
5 deploy DPFs, which would produce significant reductions in PM_{2.5} and BC. However, the U.S. EPA
6 believes a further reduction to sulfur levels at or below 15 ppm is needed for DPFs to function for their
7 intended lifetime. Further detail on the diesel sulfur reduction activities of countries outside the US,
8 Canada, Japan, and the European Union is provided in Appendix 4. Most of the actions underway in
9 other countries focus on fuels for on-road vehicles. Sulfur limits for nonroad diesel fuel are also needed
10 on an international basis to facilitate BC control. It is important to note that the cost to provide the
11 ULSD fuel will vary from one country to another depending on fuel supplies and refinery capabilities.
12 Thus, while the benefits of low sulfur fuels and advanced emission control technologies far outweigh the
13 costs, the often substantial upfront costs of upgrading existing refineries presents a challenge for many
14 governments.

15 The global community has also been working to reduce the sulfur content of fuels used in
16 marine vessels. Currently, the IMO has established requirements for the sulfur content of bunker type
17 fuel used in C3 marine vessels on both a global basis and for an Emission Control Area (ECA) in specific
18 target years (U.S. EPA, 2010, RIA C3 Marine). However, these requirements are designed to reduce
19 sulfate emissions, rather than to enable use of DPFs, and even the cleanest fuel on this schedule (1,000
20 ppm sulfur within the ECA by 2015) would not enable use of DPFs (see Appendix 4).

21 **7.5.2 Standards for New Engines outside the United States**

22 Many other countries have adopted PM emission standards for new engines. Most of these
23 standards affect on-road engines, and the rigor of these standards and the time for phase-in of new
24 engine requirements differs significantly among countries. In general, developed countries have
25 adopted standards sooner and have mandated more rapid phase-in schedules than developing
26 countries. Canada generally adopts U.S. motor vehicle standards directly following US implementation,
27 thus similar percentage reductions in BC can be expected from similar engine categories in Canada.
28 European and Japanese diesel PM standards have been reducing steadily over the last decade and are
29 achieving BC reductions similar to those in the United States. In the next few years, the level of the
30 standards will be such that DPFs will be used on almost all new on-road European and Japanese diesel
31 engines.

32 In Europe, DPFs were first applied to light-duty diesels; these requirements are relatively recent,
33 with the latest standards, known as Euro 5, becoming effective in 2009. Nonroad diesels will start to
34 phase in DPFs starting with what are termed Stage IIIB standards in 2011. The nonroad reductions will

1 be followed by Euro 6 on-road heavy-duty diesel standards which will require DPFs on all new trucks
2 starting in 2013. Likewise locomotive engines will have DPFs by 2011.¹¹

3 Other countries have adopted or proposed heavy-duty engine emission standards equivalent to
4 earlier U.S. or Euro emission standards. In the Americas, these countries include Argentina, Brazil, Chile,
5 Mexico, and Peru. In the western Pacific and Asia, these countries include China, India, the Republic of
6 Korea, Singapore, and Thailand. China is following the European emission standard progression with
7 some time lag; however, China has not yet implemented low sulfur fuel nationwide to enable
8 widespread use of DPFs. In Europe outside of the European Union, Russia and Turkey have adopted
9 earlier Euro standards. These countries are making progress in reducing BC emissions from heavy-duty
10 vehicles. In addressing the future impact of possible standards, it is important to account for both the
11 vehicle/engine standards and growth in the number of vehicles/engines as well as increases in usage
12 (such as vehicle miles traveled).

13 Relatively little is known about the costs of DPFs in other countries. However, it is expected that
14 the costs for DPFs should not differ greatly from costs in the United States. More details on diesel PM
15 emission standards in other countries are discussed in Appendix 6. It is important to note that few
16 countries have pursued standards for nonroad diesels such as construction and farm equipment,
17 locomotives, and commercial marine vessels (categories 1 and 2). Such standards, which already exist in
18 the United States, may offer a mitigation opportunity internationally.

19 **7.6 Mitigation Approaches for In-use Mobile Sources in the United States**

20 Though emissions standards for new engines will reduce emissions over time, existing engines
21 can remain in use for a long time (20 to 30 years) (U.S. Census Bureau, 2004). Opportunities to control
22 BC emissions from in-use vehicles center almost exclusively on diesel engines. Despite EPA's diesel
23 engine and fuel standards taking effect over the next decade for new engines, in-use diesel engines will
24 continue to emit large amounts of PM and BC, as well as other pollutants such as NO_x, before they are
25 replaced. For this reason, strategies to reduce emissions from in-use engines have received a great deal
26 of attention. EPA estimates that in-use mitigation strategies can be applied to 11 million of the on-
27 highway and nonroad engines now in the U.S. diesel fleet.

28 A variety of cost-effective strategies are available to reduce substantially harmful emissions
29 from in-use vehicles. As used by EPA, the term diesel retrofit includes any technology or system that
30 achieves emission reductions beyond that required by the EPA regulations at the time of new engine

¹¹ The European standards use the PMP (particle measurement program) methodology with a thermal denuder before the PM is measured which removes much of the organic PM and, thus, PM as measured by the European test procedure has less organics than that measured by the US test procedure. This is an important distinction for PM control and may affect the control technology used, which could affect BC reductions.

1 certification. Diesel retrofit projects include the replacement of high-emitting vehicles/equipment with
2 cleaner vehicles/equipment, repowering or engine replacement, rebuilding the engine to a cleaner
3 standard, installation of advanced emissions control after-treatment technologies such as DPFs, or the
4 use of a cleaner fuel (U.S. EPA, 2006 “Diesel Retrofit”).

5 The BC mitigation potential of diesel retrofits applied to existing engines depends on several
6 factors, including engine application (vehicle or equipment type), engine age, engine size, and engine
7 condition (maintenance) and remaining engine life. One or more of these factors will dictate the
8 suitability of a mitigation strategy. Some engines, whether because of old age, poor maintenance or
9 duty cycle, are not able to be retrofitted with DPFs. Engines with limited remaining life or low usage
10 rates are not good candidates for retrofits when cost-effectiveness is considered. It can also be
11 technically infeasible to replace an old engine with a new one in many cases because of insufficient
12 space in the original vehicle or piece of equipment. For some of these vehicles, truck replacement with
13 scrappage of the original vehicle, may be the only viable option to reduce BC emissions. It is also
14 possible for 10%-15% of the vehicles in a typical fleet to emit 50% or more of each major exhaust
15 pollutant due to malfunctioning engine parts (National Academies Press, 2001). This is an important
16 consideration in developing mitigation strategies.

17 The National Clean Diesel Campaign and the SmartWay Transport Partnership Program are
18 EPA’s two primary programs responsible for reducing emissions from in-use diesel vehicles and
19 equipment. These programs support the testing and deployment of numerous technologies and
20 strategies to reduce emissions from in-use diesel engines, including BC, and can provide immediate
21 reductions. These programs are described in more detail below, following a discussion of key retrofit
22 technologies and approaches for reducing emissions from in-use vehicles and engines.

23 **7.6.1 Available Retrofit Technologies and Strategies for In-use Engines**

24 **7.6.1.1 After-treatment Devices**

25 Typically, after-treatment diesel retrofit involves the installation of an emission control device to
26 remove emissions from the engine exhaust. This type of retrofit can be very effective at reducing PM
27 emissions, eliminating up to 99 % of BC in some cases. Some examples of diesel retrofit devices that
28 reduce BC include partial flow filters and DPFs.¹² EPA and CARB adhere to rigorous verification
29 processes to evaluate the performance and reliability of available retrofit technologies. These processes
30 evaluate the emission reduction performance of retrofit technologies, including their durability, and
31 identify engine operating criteria and conditions that must exist for these technologies to achieve those

¹² See http://www.meca.org/cs/root/diesel_retrofit_subsite/what_is_retrofit/what_is_retrofit.

1 reductions. Federal funding under the National Clean Diesel Campaign requires recipients to use EPA or
2 CARB-verified diesel retrofit technologies for clean diesel projects.

3 As previously mentioned, DPFs are wall-flow exhaust after-treatment devices that are effective
4 at significantly reducing diesel PM emissions by 85% to 90% and BC emissions by up to 99%. Because BC
5 exits the engine in solid particle form, DPFs can reduce BC up to 99%. The small amount of PM that does
6 penetrate a DPF is composed of mainly sulfate and OC. DPFs typically use a porous ceramic, cordierite
7 substrate, or metallic filter to physically trap PM and remove it from the exhaust stream. The collected
8 PM is oxidized primarily to CO₂ and water vapor during filter regeneration. Regeneration can be passive
9 (via a catalyst) or active (via a heat source) and is necessary to keep the filter from plugging and
10 rendering the engine inoperative. Regular engine maintenance is essential to DPF performance.

11 Passive regeneration occurs when exhaust gas temperatures are high enough to initiate
12 combustion of the accumulated PM in the DPF, usually in the presence of a catalyst, but without added
13 fuel, heat, or driver action. Active regeneration may require driver action and/or sources of fuel or heat
14 to raise the DPF temperature sufficiently to combust accumulated PM. Active DPFs may be necessary
15 for lower engine temperature applications, such as lower speed urban and suburban driving; otherwise
16 the DPF may become plugged due to an accumulation of PM.

17 For large, on-highway trucks, retrofitting passive DPFs generally costs between \$8,000 to
18 \$15,000, including installation, depending on engine size, filter technology and installation
19 requirements. Active DPF systems can cost \$20,000 for a heavy duty diesel truck and up to \$50,000 for
20 a large piece of nonroad equipment. Vehicle inspection, data logging, and backpressure monitoring
21 systems are required with each installation; these costs are typically included in the cost of the DPF
22 (NCDC Technical Bulletin, 2006).¹³

23 *Partial Diesel Particulate Filters* [MECA] PDPFs provide moderate (around 30% to 50%)
24 reduction of PM from diesel exhaust. However, at this time there is no consensus and limited test data
25 on the effectiveness of PDPFs to reduce BC. PDPFs typically employ structures to briefly retain particles
26 for oxidation, structures to promote air turbulence and particle impaction, and catalysts to oxidize diesel
27 particles. Partial flow filters are capable of oxidizing the soluble organic fraction of diesel exhaust and
28 likely some BC. As of October 2010, only three PDPF technologies were verified by CARB (none by EPA),
29 and these were only verified for transport refrigeration units (TRU). These devices cost about \$4000-
30 \$8000 per unit.

¹³ These cost estimates are from NCDC's Cost Effectiveness Paper 2006, updated to 2010 dollars.

1 **7.6.1.2 Other Retrofit Strategies**

2 A variety of other strategies can also reduce emissions from in-use vehicles. While the precise
3 impact of such strategies on BC emissions can be more difficult to quantify than application of an after-
4 treatment device, these strategies may substantially reduce emissions, while improving fuel economy
5 and extending engine life.

6 *Improved Fleet Maintenance Practices:* Since a small percentage of vehicles in a given fleet may
7 be responsible for a majority of the fleet’s emissions, one of the first steps for reducing emissions is to
8 take an inventory and inspect vehicles and equipment. This information may be used to identify
9 vehicles in need of repair and find candidates for other mitigation options. Repair of poorly operating
10 engines typically decreases emissions and improves fuel economy. Furthermore, regularly performed
11 maintenance will extend the life of vehicles and equipment (PCV, 2009). For example, many
12 manufacturers prescribe that engines be rebuilt after accumulating a set number of hours of use. An
13 engine rebuild involves replacing some old parts and cleaning/machining durable parts to original
14 factory specifications. In some cases, an aftertreatment technology could be installed at the time of
15 engine rebuild. This would save time since the vehicle or equipment would not need to be removed
16 from service any longer than prescribed for normal maintenance.

17 *Engine Repower, Upgrades, or Reflash:* Significant emissions reductions can be achieved by
18 repowering, upgrading, or reflashing a diesel engine. Engine repowering (i.e., replacing the engine, but
19 not the entire vehicle) is straightforward, and the benefits are easily quantified. For example, when an
20 uncontrolled engine is taken out of service and replaced with a new engine, the emissions benefits are
21 determined from the difference in emissions levels of each engine. The cost of replacing a vehicle or
22 piece of equipment is much higher than replacing just the engine. However, not all vehicles/equipment
23 can be repowered. New engines are not always compatible with the original vehicle/equipment.

24 An alternative to vehicle/equipment replacement and engine repower is “engine upgrade”. An
25 engine upgrade is the process by which parts of an in-use engine are replaced with newer components,
26 resulting in lower emissions. Engine upgrades are normally sold as kits from an engine manufacturer
27 and include newer mechanical parts, and, for electronically controlled engines, changes to the computer
28 program that controls the engine. This is known as a reflash, and it can change the mix of pollutants in
29 the exhaust stream (e.g., by changing the injection timing). Engine upgrades, including reflashes, are
30 generally less expensive than replacing an entire engine, but they are only available for specific engines.
31 Thus, implementation is limited by the number of upgradable engines currently in service.

32

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
27

[TEXT BOX]

Local Retrofit Projects in the United States

Agricultural Vehicle Repowers

The Air Pollution Control District in San Joaquin Valley received \$2 million to repower 30 agricultural vehicles with new engines that meet or exceed EPA’s Tier 3 diesel emission standards. Using ARRA funds, EPA awarded this project because of its long-term economic and immediate health benefits for the community. The repowered engines are expected to reduce emissions of NO_x by 33.61 tons and PM by more than 1 ton annually.

Locomotive Repower

The Railroad Research Foundation was awarded \$2.9 million to repower 4 locomotives that operate as switchers in rail yards in Baton Rouge, Louisiana. The original locomotives were built with 3,500 horsepower engines in 1985 and 1986, and the new engines meet or exceed EPA Tier 2 locomotive engine emission standards. Tier 2 locomotive emissions are one-third those from Tier 0 locomotives.

Shore Power

Massachusetts Port Authority was awarded \$400,000 to install shore-side electric power to ships, with a 9-unit shore connection system serving 18 berths in South Boston. Most vessels dock at the pier 100 to 300 days per year, and typically run diesel generators for 10 to 14 hours to provide cabin heat, generate power to unload fish, and supply electricity for other needs. The new on-shore power hook-ups are projected to reduce PM emissions by 96 %.

Construction Retrofits

New Jersey Department of Environmental Protection (NJDEP) was awarded \$1.73 million to pay for the cost and installation of pollution control devices on various construction vehicles used in New Jersey. Funding under this program has allowed NJDEP to implement Phase 2 of its existing New Jersey Clean Construction Program to retrofit non-road equipment used on publicly funded construction projects. The retrofits are projected to reduce PM by 3.8 tons annually.

[END TEXT BOX]

Cleaner fuels can lead to BC reductions via multiple pathways. As previously stated, ULSD fuel is necessary for diesel particulate filters and other aftertreatment technologies to be effective. Fuel options such as compressed natural gas (CNG), liquefied natural gas (LNG), ethanol, and hydrogen can yield substantial reductions in PM and BC. However, this requires installation of engines and fuel systems compatible with these fuels as well as infrastructure to facilitate storage and delivery of the fuels. Many U.S. urban fleets of heavy-duty vehicles have shifted their diesel-fueled vehicles to those fueled with CNG. Transit buses and solid waste collection vehicles are among those fueled with CNG. Recently, a number of drayage trucks in Southern California’s Port of Los Angeles and Port of Long Beach have been converted from diesel to LNG.

Another form of fuel switching is electrification. As previously stated in this report, power plant supplied electricity has extremely low emissions of BC. If mobile sources can be powered by electricity, BC emissions can be reduced. One example of this is cold-ironing (shore power) at seaports, which

1 allows marine vessels shut down their engines and run normal operations by plugging into electrical
 2 connections at docks. When a vessel is at berth, it typically runs its auxiliary diesel engines to provide
 3 power for normal operations (referred to as hotelling). For example, CARB has estimated that 1.8 tons
 4 per day of diesel PM was emitted by approximately 2,000 hotelling ocean-going vessels in California in
 5 2006 (Technical Support Document: Initial Statement of Reasons for The Proposed Rulemaking [TSD]).
 6 Hotelling emissions can be dramatically reduced if the vessel uses “shore power” electricity while at
 7 port. It should be noted that emissions of other pollutants should be considered when pursuing this and
 8 other alternative fuels/energy sources. For example, electrification shifts the emissions from the mobile
 9 source to the power plant.

10 *Fuel economy improvements* may yield reductions in BC. Some fuel savings devices, such as low-
 11 rolling-resistance tires and aerodynamic technologies (e.g., trailer gap reducers, trailer boat tails, and
 12 trailer side skirts) reduce fuel use with little change to engine operation. These fuel saving devices likely
 13 result in PM reductions; however, additional research is needed to quantify the emission reductions.
 14 Hybrid vehicles are potential technologies for CO₂ reductions, but further research is necessary to
 15 determine the extent of PM or BC reductions.

16 *Idle reduction:* Long-duration idling of truck and locomotive engines consumes over 1 billion
 17 gallons of diesel fuel annually resulting in 5,000 tons of PM, a significant fraction of which is BC (i.e., 15-
 18 40%). It is important to consider that while BC is a significant fraction of overall diesel PM, BC/PM ratios
 19 differ during idling. The reduction in PM due to idling has definite health benefits, and the reduction in
 20 fuel use results in reduced CO₂ emissions and, in turn, climate benefits. However, the net climate
 21 benefit due to reduction in idling PM is less understood. Furthermore, idling increases fuel and engine
 22 maintenance costs, shortens engine life, increases driver exposure to air pollution, and creates elevated
 23 noise levels. Idle reduction programs and technologies are already prevalent in the US. They serve as
 24 one of the simplest and lowest cost methods to reduce emissions from engines. Because reducing idling
 25 reduces engine operation, emissions of all pollutants are lower. Strategies for reducing idling include
 26 both operational and technological methods. Examples of on-board truck technologies include:

- 27 – Automatic engine shut-off devices programmed to shut down the engine after a preset time
 28 limit
- 29 – Direct-fired heaters to eliminate idling used to heat the cab
- 30 – Auxiliary power units (APU) or generators to provide power for cab comfort at rest stops
 31 and eliminate the need to run the truck engine
- 32 – Battery or alternatively powered heating and air conditioning units

33 Off-board technologies include truck stop electrification, which provide conditioned air and electricity to
 34 truck cabs for accessory loads while at a truck stop. These systems also may provide telephone, cable
 35 TV, and internet access. A majority of U.S. states and many municipalities have anti-idling regulation in
 36 place to limit idling of vehicles (American Transportation Research Institute, 2010).

1 *Transportation modal shift:* Transportation of certain goods can be altered to reduce BC
2 emissions and increase efficiency. Specifically, a shift from trucks to rail or to sea and inland waterways
3 can reduce diesel truck PM emissions and alleviate traffic congestion (Barth and Tadi, 1998; Winebrake
4 et al., 2008). It is important to note that modal shifts can result in localized increases in emissions
5 where goods movement is concentrated, such as ports and rail yards. While the percentage of BC in
6 total locomotive PM emissions is roughly equal to that of diesel trucks (72-78%), diesel engines under
7 idle or low load, such as occur in intermodal freight terminals, emit PM with a smaller fraction of BC
8 (approximately 15-40%). In addition, ship emissions can exhibit very different characteristics from truck
9 or locomotive engines, particularly emissions from slow-speed engines used in ocean-going vessels
10 (Category 3) burning residual (bunker) fuel. As described in Chapter 4 on inventories, recent studies
11 have reported BC to be a minor fraction of PM from Category 3 marine engines. However, these data
12 are limited to a few studies. Further research is needed in order to better characterize ship emissions
13 and to better understand the effects of modal shifts on BC emissions.

14 Some diesel *retrofit technologies* were designed to reduce other pollutants, such as NO_x and
15 hydrocarbons, and do not significantly impact BC emissions. Such technologies include:

- 16 • Diesel oxidation catalysts [NCDC Technical Bulletin] (DOCs) provide minimal BC reductions.
17 DOCs are exhaust after-treatment devices that reduce PM, HC and CO emissions from diesel
18 engines and are widely used as a retrofit technology because of their simplicity, relative low
19 cost, and limited maintenance requirements. DOCs verified by EPA and CARB are typically
20 effective at reducing PM by 20 to 40 %, though the PM removed by DOCs is composed largely of
21 organic carbon that comes from unburned fuel and oil. DOCs are not an effective mitigation
22 strategy for BC reductions.
- 23 • Closed Crankcase Ventilation Systems [NCDC Website] (CCVS) provide negligible BC reductions.
24 In many diesel engines, crankcase emissions or "blow-by" emissions are released directly into
25 the atmosphere through the "road draft tube." Closed Crankcase Ventilation (CCV) devices
26 capture and return the oil in blow-by gas to the crankcase, directing HC and toxics to the intake
27 system for re-combustion instead of emitting them into the air.
- 28 • Selective Catalytic Reduction [Dieselnet.com] (SCR) systems inject a reducing agent such as
29 diesel exhaust fluid (DEF), a urea solution, into the exhaust stream where it reacts with a
30 catalyst to reduce NO_x emissions. Most 2010 and newer on-road diesel engines come equipped
31 with an SCR system and SCRs are also available as after-treatment retrofits. SCR systems require
32 periodic refilling of the reductant and may also be used with a catalyzed DPF to reduce PM
33 emissions. Coupling engine design techniques that lead to a reduction of BC through a low PM
34 engine strategy with a NO_x after-treatment control device such as an SCR has been an approach
35 used in Europe. SCR systems, which are effective in reducing NO_x by 60 to 80%, can provide
36 potential BC reductions when the engine fuel injection timing is changed for lower PM and
37 higher NO_x emissions.

1 **7.6.2 Cost Effectiveness of Retrofits**

2 In 2006, EPA published a report on the cost effectiveness of heavy-duty diesel engine retrofits
3 (EPA420-S-06-002). The analysis presented in that report, which was based on data collected from
4 2004-2005, estimated the cost effectiveness of installing a passive DPF on a Class 8 truck to be \$12,100-
5 \$44,100 per ton of PM_{2.5} reduced. Model year 1994 and newer class 8 trucks employed in long-haul
6 operation are generally good candidates for DPFs.

7 In 2009, EPA published a Report to Congress, *Highlights of the Diesel Emission Reduction*
8 *Program*, which provides information on the overall cost-effectiveness of various diesel emission
9 reduction strategies funded under the Diesel Emission Reduction Act. The Report estimates that the
10 average cost-effectiveness of the DERA projects funded in 2008 ranged from \$9,000 to \$27,700 per ton
11 of PM 2.5. According to this analysis, which is currently being updated, diesel retrofit strategies
12 compare favorably with other emission reduction strategies used to attain national ambient air quality
13 standards that range from \$1,000 to \$20,000 and as high as \$100,000 per ton of PM_{2.5} on an annualized
14 basis. However, most diesel retrofit strategies are less cost-effective than regulatory programs designed
15 to set PM emissions standards for new diesel engines, such as the emissions standards for 2007 and
16 later model year heavy-duty highway engines.

17 **7.6.3 Applicability of Diesel Retrofits**

18 The ability to install diesel retrofits on different diesel vehicles and equipment depends on a
19 number of factors. Not all engine types are equally well suited to retrofit strategies; for others (e.g.,
20 bulldozers), long engine lifetime may make retrofits the only feasible option. The on-highway diesel
21 vehicles in the United States are mostly heavy-duty trucks. The 2002 Census indicated that most
22 trucking companies are small businesses that own only one to three trucks. Smaller businesses are less
23 able than large businesses to absorb capital costs associated with emissions reductions from diesel
24 engines.

25 The nonroad engine and vehicle category includes a diverse range of equipment from
26 lawnmowers to marine and locomotive engines to construction machinery. Each category has specific
27 needs and challenges. Construction equipment, for example, is often much more expensive with longer
28 useful lives than on-highway vehicles. This adds complexity when considering mitigation. Vehicle
29 replacement is difficult for large construction equipment due to their high costs. In addition, repower
30 options are only available for certain types of construction machines due to space limitations in the
31 engine compartment.

32 Currently, PM mitigation strategies for marine and locomotive engines are limited. No DPFs are
33 verified or certified by federal or state agencies for these engines. Therefore, upgrading/replacing
34 engines and fuel switching are currently the two most viable mitigation strategies for these engines.
35 Fuel switching could also include the use of shore power for larger marine vessels, which eliminates
36 local PM emissions while ships are at port. New emission reduction technologies are being developed to

1 reduce locomotive and marine emissions. For example, marine engine upgrade kits have been
2 implemented with funding support from the EPA Emerging Technologies Program.¹⁴

3 **7.6.4 Experience with Diesel Emissions Reduction Programs in the United States**

4 Federal, State, and local agencies have demonstrated substantial capacity, funded in large part
5 through the Diesel Emission Reduction Act Program, to develop and implement diesel emissions
6 reduction programs. Collectively, these agencies, in partnership with environmental and industry
7 stakeholders, have built a strong foundation for the testing, verification and implementation of new
8 technologies and strategies. Many of these programs provide funding or other incentives for voluntary
9 diesel retrofits, engine replacements, or idle reductions. These programs include EPA's National Clean
10 Diesel Campaign (NCDC) and the SmartWay Transport Partnership; FHWA's Congestion Mitigation and
11 Air Quality (CMAQ) Improvement Program; the Texas Emissions Reduction Plan (TERP), and California's
12 Carl Moyer Memorial Air Quality Standards Attainment Program.

13 **7.6.4.1 National Clean Diesel Campaign (NCDC)**

14 The National Clean Diesel Campaign (NCDC) is a partnership that aims to accelerate the
15 implementation of emissions control strategies in the existing fleet through approaches such as
16 retrofitting, repairing, replacing, repowering, and scrappage of diesel vehicles and equipment; reducing
17 idling; and switching to cleaner fuels.

18

19 [TEXT BOX]

20

EPA Diesel Emissions Reduction Program

21 The Diesel Emissions Reduction Act Program (DERA) may serve as one of the best avenues and foundations for
22 reducing BC emissions in the United States (U.S. EPA, 2009). The Energy Policy Act of 2005 (title VII, Subtitle G,
23 Sections 791-797) provides EPA with grant and loan authority to promote diesel emission reductions from the
24 existing in-use fleet in the United States and authorizes appropriations of up to \$200 million per year to the Agency
25 under the DERA provisions for FY2007 through FY2011. Congress appropriated \$169.2 million in funding under this
26 statute in FY 2008 through FY 2010. In addition, the American Recovery and Reinvestment Act of 2009 allotted the
27 National Clean Diesel Campaign \$300 million. The Diesel Emissions Reduction Act of 2010 was signed into law by
28 President Barack Obama in January 2011. This law authorizes DERA for \$100 million per year from FY2012 through
29 FY2016.

30 DERA offers a funding vehicle for immediate BC reductions within the in-use fleet. The first year of DERA funding
31 reduced emissions from more than 14,000 diesel-powered highway vehicles and pieces of nonroad equipment.
32 DERA funding supported a wide range of verified technologies, cleaner fuels, and certified engine configurations,

¹⁴ See <http://www.epa.gov/cleandiesel/projects/proj-emerge.htm>.

1 such as repowers, replacements, idle-reduction technologies, biodiesel, and retrofit devices such as DPFs. DERA
2 funding also supported diesel programs in state governments.

3 The diesel emission reductions resulting from the FY2008 grants for PM will total approximately 2,200 tons by
4 2031 which translates to 1,540 tons of BC reductions, assuming 70% of PM is BC. The health benefits will range
5 from a net present value of \$580 million to \$1.4 billion, including an estimated 95 to 240 avoided premature
6 deaths.

7 In 2008-2010, EPA received an overwhelmingly positive response in its Request for Proposals (RFPs) under the
8 National Clean Diesel Funding Assistance Program, which receives just over half of the annual funds appropriated
9 for DERA. The Agency received 594 applications nationwide requesting more than \$665 million. Thus, applicants
10 requested \$7 for every \$1 available in clean diesel funding. In addition, the Agency received 607 applications
11 nationwide requesting more than \$1,648 million for the solicitation under the Recovery Act. The total amount of
12 matching funds offered by applicants is generally greater than the amount of funds requested.

13 While the DERA program achieved significant emission reductions from in-use diesel engines in its first year, a
14 large number of high-emitting engines remain currently in use. Further opportunities to build on the experience
15 gained to date are widespread. In moving forward in the program, two challenges remain. First, there are too few
16 verified technologies for nonroad and marine engines and older diesel trucks, limiting the extent of achievable
17 emission reductions. Second new incentives are needed to retire the oldest and dirtiest engines. The nonroad
18 market is complicated by the number and diversity of nonroad equipment types, the range of horsepower and
19 engine types involved, and the varying usage and duty cycles of the equipment.

20 Through the DERA Program, EPA is committed to reducing diesel emissions, including BC, targeting current
21 nonattainment areas where clean diesel strategies can assist in meeting local emission reduction goals, and
22 provide assistance to state and local governments in developing their own clean diesel programs. In addition, EPA
23 continues to provide high-quality data to states that depend on the performance of diesel emission reduction
24 strategies in their air quality plans, through in-use testing—confirming the performance of verified technologies in
25 the field—and working cooperatively with industry groups, engine manufacturers, and state agencies such as CARB
26 to expand the list of clean diesel technology options for partners. The DERA Program aims to continue to develop
27 innovative financing approaches for stretching federal dollars to maximize diesel emission reductions, and to
28 develop timely educational materials to build awareness of clean diesel opportunities.

29 Overall, the DERA Program has set the foundation for opportunities for further BC reductions from in-use diesel
30 engines. The first years of the program offer lessons learned for greater implementation of BC-reducing
31 technologies and illustrate the successes of an innovative and incentive-based program that works in partnership
32 with existing EPA rules.

33 [END TEXT BOX]

34

35 **7.6.4.2 SmartWay**

36 In 2004 EPA launched its SmartWay Transport Partnership. SmartWay is an innovative
37 collaboration between EPA and the freight sector that is designed to improve energy efficiency, reduce
38 greenhouse gas and air pollutant emissions, and improve energy security, by accelerating the adoption
39 of environmentally clean and fuel efficient transportation options. Typically, SmartWay projects
40 combine fuel-saving technologies with emission control technologies; some technologies—like idle
41 reduction or newer truck replacements—do both. SmartWay includes programs to test and verify fuel-
42 saving equipment and vehicles; develop innovative finance strategies to promote retrofitting or
43 accelerated replacement of older vehicles and equipment; and develop tools and methods to assess and

1 track emissions from SmartWay partners. Companies that participate in SmartWay Transport programs
2 save money, reduce fuel consumption, and are recognized for their social responsibility and leadership.
3 SmartWay tracks fuel savings, reductions in greenhouse gas emissions, reductions in smog-forming NO_x
4 emissions, and reductions in PM. SmartWay promotes a number of technologies that directly reduce
5 emissions of PM. These include idle reduction, accelerated vehicle replacement and emission control
6 retrofits.

7 While a wide variety of technologies exist to reduce fuel costs for trucking companies, many
8 companies lack the up-front investment capital to benefit from them. The SmartWay Finance program,
9 funded by DERA, aims to accelerate the deployment of energy efficiency and emission control
10 technologies by helping vehicle/equipment owners overcome financial obstacles. Since 2008, the
11 SmartWay Finance program has awarded over \$30 million to help small trucking companies reduce fuel
12 costs and emissions. These innovative loans help small trucking firms reduce PM emissions, and lower
13 their fuel costs by purchasing newer used trucks equipped with idling and emissions reduction
14 technologies.

15 **7.6.4.3 Congestion Mitigation and Air Quality Improvement Program (U.S. DOT)**

16 The Congestion Mitigation and Air Quality (CMAQ) Improvement Program, jointly administered
17 by the U.S. Department of Transportation's Federal Highway Administration (FHWA) and the Federal
18 Transit Administration (FTA) provides roughly \$1.7 billion in annual funding for a variety of emission
19 reduction projects including transit, traffic signalization, bicycle/pedestrian facilities, demand
20 management, and diesel retrofit projects. According to the most recent data available, between 2005
21 and 2007, approximately \$285 million of CMAQ funds were spend on diesel retrofits. New priority for
22 the funding of diesel retrofit projects was established by Congress with the Safe, Accountable, Flexible,
23 Efficient Transportation Equity Act: A Legacy for Users (SAFETEA-LU) in 2005.

24 The allocation of CMAQ funds is managed by the state DOTs. CMAQ aims to implement projects
25 that will help areas attain or maintain the NAAQS. Diesel retrofits are more cost-effective in reducing
26 PM than other typical CMAQ projects, such as traffic signal optimization (Diesel Technology Forum,
27 2006, 2007).

28 **7.6.4.4 State Programs**

29 *Mandatory retrofits:* The state of California has enacted legislation to require in-use heavy duty
30 diesel fleets to meet minimum emission standards. The legislation is implemented through CARB and
31 applies to many sectors, including both on-highway and nonroad diesel engines. Most of the regulations
32 require accelerated fleet turnover, which includes repowering or retiring vehicles, or requiring best
33 available control technology (BACT) to be installed on diesel engines. Almost all on-highway heavy-duty
34 diesel vehicles, including buses, drayage trucks, and class VIII trucks will be required to reduce diesel
35 emissions.

36 Several states have passed legislation similar to California's. New Jersey has instituted a
37 mandatory retrofit program requiring owners of diesel vehicles to retrofit with best available retrofit

1 technology (BART). The state reimburses vehicle owners/operators for all expenses. New York has also
2 instituted a mandatory retrofit program that applies to all heavy-duty state-owned and contractor
3 vehicles.

4 *Incentive programs:* The state of California's Carl Moyer Memorial Air Quality Standards
5 Attainment Program provides incentive grants for cleaner-than-required engines, equipment, and other
6 sources of pollution providing early or extra emission reductions. The program started in 1998 and has
7 funded hundreds of millions of dollars worth of projects since its inception. California voters also passed
8 Proposition 1B in 2006, which allocated \$1 billion to reduce air pollutant emissions from freight
9 emissions along California's trade corridors. Both of these incentive funding programs rank applicants
10 based on cost effectiveness (e.g., \$/ton). Carl Moyer funds cannot be used to fund compliance with
11 state or federal laws. Thus, funding opportunities are becoming limited due to California's
12 implementation of regulations affecting most categories of mobile sources.

13 The Texas Emissions Reduction Plan (TERP), a program of the Texas Commission on
14 Environmental Quality (TCEQ) provides financial incentives to eligible individuals, businesses or local
15 governments to reduce emissions from polluting vehicles and equipment in the state of Texas. TERP has
16 provided over \$797 million since 2002, affecting over 12,500 diesel engines with engine/vehicle
17 replacement as one of the key clean diesel strategies. Though this incentive program focuses more
18 heavily on NO_x, there is still an opportunity for manufacturers to develop both NO_x and PM combination
19 technology strategies for BC reductions, through the New Technology Research and Development
20 Program (NTRD) which encourages and supports the research, development, and commercialization of
21 technologies that reduce pollution in Texas (Texas Emission Reduction Plan website).

22 **7.7 Mitigation Approaches for In-use Mobile Engines Internationally**

23 There are millions of large diesel-powered vehicles throughout the world, including buses, heavy
24 duty trucks, off-road vehicles, locomotives, and marine vessels. The exact size of the international diesel
25 fleet is not easily characterized. Some countries are similar to the United States in one or more of the
26 following: vehicle registration, inspection and maintenance programs, availability of low-sulfur fuel,
27 technology certification/ verification programs, and readily available technologies. However, many
28 (mainly developing) nations have little to none of this infrastructure in place. Furthermore, developing
29 countries tend to have older and less well-maintained engines and vehicles than developed countries,
30 and the availability of low-sulfur diesel fuel is limited. Therefore, many engines in developing countries
31 are not good candidates for tailpipe control strategies like passive DPFs. In addition, the costs of DPFs
32 may be prohibitive for some countries. Most retrofit programs around the world (including in the
33 United States) have relied heavily on government funding, which presents a significant financial
34 challenge.

35 EPA has often advised other nations and supported international demonstration projects in an
36 effort to transfer information and technologies to those that seek to reduce emissions from mobile
37 sources. Additionally, EPA's diesel retrofit experts have advised and participated in several pilot retrofit
38 projects where diesel trucks and buses were fitted with various exhaust after-treatment devices. Low-

1 sulfur diesel was obtained for the projects in most cases. The projects have shown generally that, if
2 appropriate fuel is provided and engine maintenance is addressed, that partial flow filters and DPFs are
3 viable options to reduce PM (and thus BC) on some vehicles. Following a relatively small EPA supported
4 pilot project in Beijing in 2006, city authorities went on to retrofit more than 6000 vehicles with active
5 DPFs prior to the Beijing Olympics. That number is now above 8,000 and growing. EPA has also assisted
6 in retrofit projects in Mexico City, Bangkok, Santiago, and Pune (India).

(This page intentionally left blank)

8. Mitigation Approaches for Stationary Sources

8.1 Summary of Key messages

- There has been a dramatic decline in BC emissions from industry in developed countries over the last century. Stationary sources in the United States now account for only 8% of the U.S. BC inventory; sources include industrial, commercial, and institutional (ICI) boilers; power plants; industrial processes such as cement manufacturing; and stationary diesel engines.
- Internationally, emissions from stationary sources account for about 14% of the global inventory, with highest emissions in China, the former USSR, India, and central/South America. Main sources are brick kilns, coke ovens (largely from iron/steel production), and industrial boilers.
- Available control technologies and strategies include direct PM_{2.5} reduction technologies such as fabric filters (baghouses), electrostatic precipitators (ESPs), and diesel particulate filters (DPFs). Once installed, these strategies range in cost-effectiveness from as little as \$35/ton PM_{2.5} to \$500/ton PM_{2.5} or more, depending on the source category. However, they also may involve tens of millions in initial capital costs. Additional source testing data is needed to clarify the efficiency of these controls for removing BC specifically.
- Internationally, emissions from a number of source categories may grow as countries industrialize. Reducing emissions from smaller, inefficient facilities may require phasing out or replacing the entire unit, while larger facilities can apply many of the existing PM filter technologies already in commercial use. However, both of these options may be associated with substantial cost and implementation difficulties.

8.2 Introduction

Emissions of BC from stationary sources¹ generally represent a smaller portion of current global inventories than mobile sources and other source categories. As mentioned in Chapter 4, this is due in large part to a significant decline in industrial BC emissions from developed countries over the past century. These reductions have been achieved through improved combustion, shifts in fuel use, and application of control technologies to limit direct PM emissions. Although some uncertainty remains

¹ The term “stationary sources” as used in this chapter refers to large and small industrial or combustion operations. It does not include residential fuel combustion for heating or cooking.

1 regarding the exact efficiency of these control techniques for reducing the BC fraction of PM_{2.5}
2 emissions, that uncertainty does not change the conclusion that emissions of BC from U.S. stationary
3 sources are relatively modest in comparison to other key sectors of the national inventory. In contrast,
4 stationary sources represent a larger fraction of international inventories, and in some regions these
5 sources are key contributors to overall direct PM_{2.5} emissions which adversely affect public health and
6 the environment. Thus, continued mitigation of stationary source BC emissions domestically and
7 internationally will lead to improved public health and will also provide climate co-benefits.

8 There are a number of relatively well-developed control technologies that have successfully
9 been applied to reduce direct PM_{2.5} (including BC) from stationary sources. This section discusses PM
10 control technologies and strategies that are applicable to BC mitigation from domestic and international
11 stationary sources. Where possible, it provides information about the applicability, performance, and
12 costs of these approaches. Since these control technologies are well-established, much of this
13 information is drawn from EPA and other control technology guidance documents developed for PM
14 mitigation purposes.

15 **8.3 Emissions from Key Stationary Source Categories**

16 The combustion of fossil fuels such as coal or oil is often the primary source of BC emissions at
17 an industrial facility. In the United States and other developed countries, stationary source emissions of
18 BC have been reduced substantially from historic levels. As discussed in Chapter 4, current emissions
19 from stationary sources (including both “industrial sources” and “fossil fuel combustion” categories in
20 the U.S. inventory) account for roughly 8% of the U.S. BC inventory (see Table 4-2). These emissions
21 come from industrial, commercial, and institutional (ICI) boilers; power plants; and other types of
22 industrial sources, such as cement manufacturing or stationary diesel engines used for many purposes
23 including irrigation or oil and gas extraction.

24 Stationary sources account for a slightly higher percentage (14%) of total worldwide BC
25 emissions, and almost 25% of BC emissions from contained combustion (i.e. sources other than open
26 biomass burning) (see Table 4-2 and Bond et al., 2004). In certain developing world regions, such as
27 China and India, stationary sources represent a very significant percentage of the BC inventory. The
28 regions with the highest percentage of “contained” BC emitted from industry and power generation are
29 China, the former USSR, India, and central/South America (Zhang et al., 2007). Key source categories
30 include brick kilns, coke production / iron and steel production, and industrial boilers. As discussed in
31 Chapter 6, however, BC emissions from industrial sources are expected to decline worldwide under
32 most scenarios. This decline is anticipated to occur in developing countries as well as developed
33 countries.

34 In the United States, direct emissions of PM and BC from stationary sources have been reduced
35 significantly due to improved combustion efficiencies in industrial operations and implementation of
36 federal and state clean air regulations over the past several decades. This declining emissions trend is
37 expected to continue as further reductions will be needed to meet revised air quality standards and
38 mitigate adverse effects on public health and the environment. Projections indicate that direct PM

1 emissions from stationary sources are expected to decline by about 20% between 2005 and 2020 (PM
2 NAAQS Regulatory Impact Analysis). For example, sources in nonattainment areas will be required to
3 implement emissions reduction strategies to help areas attain the 1997, 2006, and any future revisions
4 to the PM_{2.5} NAAQS. Certain facilities will also be required to comply with revisions to maximum
5 achievable control technology (MACT) and new source performance standards (NSPS) for specific key
6 categories. These standards will lead to control of some sources that currently do not have any PM
7 controls; they will also lead to improved levels of control for certain sources that already have PM
8 controls. However, in an overall sense, near-term BC emission reductions from domestic stationary
9 sources are expected to be modest when compared to expected reductions in other sectors, such as the
10 mobile source category.

11 In general, stationary sources burning coal dominate the U.S. BC inventory for stationary
12 sources. However, many of these sources have high combustion efficiencies and have already applied
13 substantial emissions controls: for example, nearly all large coal-fired EGUs have electrostatic
14 precipitators (ESPs) or fabric filters for PM control. Estimates by the U.S. Department of Energy indicate
15 that 76% of fossil-fuel steam-electric generating units in the United States (1,194 of 1,568) have some
16 form of PM control—and those that do not are likely to be fueled by natural gas (U.S. Energy
17 Information Administration, 2010). More than 80% of these sources operate ESPs, while about 14%
18 have fabric filters. These control technologies are described further, below.

19 ICI boilers are a wide-ranging category of combustion units that collectively can burn a wide
20 variety of combustible fuels, including coal, oil, natural gas, and biomass. There are thousands of ICI
21 boilers across the country, varying in size from a few million Btu/hr for small commercial or industrial
22 units to over 10 million Btu/hr for large boilers. Their operations range from intermittent to near-steady
23 state. Most large units are covered under new regulations that include stringent standards for PM,
24 mercury, and certain hazardous air pollutants.² EPA has projected that the new emissions limits
25 applicable to major source boilers and process heaters will reduce PM_{2.5} emissions from these sources
26 by 47,000 tons by 2014.

27 Stationary engines burning diesel fuels also account for substantial BC emissions. These engines
28 are similar to mobile diesel engines and typically use the same fuels, but they can also operate using
29 natural gas or heavier fuel oil grades than mobile diesel engines. They are used to perform a range of
30 different tasks, such as pumping water or oil through pipelines, operating equipment in remote
31 locations, or providing backup power generation.

32 Many other categories of industrial sources emit relatively low amounts of BC. In the current
33 U.S. inventory, the “natural gas combustion” sector appears to have substantial BC emissions, but this is
34 likely due to severe constraints on the data used to generate these estimates.³ Given our knowledge of

² The final rule was signed on February 21, 2011, but has not yet been published in the Federal Register. For more information, see: <http://www.epa.gov/airquality/combustion/docs/20110221mboilersfs.pdf>.

³ The current AP-42 emissions factor for BC from natural gas combustion is considered to be highly questionable. Bond et al. (2006) indicated significantly lower emissions factors for industrial natural gas combustion than that published in AP-42. Bond reported an emission factor of 0.004±0.004 g PM per kg fuel, two orders of magnitude lower than the 0.21 g/kg found in AP-42.

1 the utility and major source boiler inventory and the mechanisms of BC formation, EPA does not believe
2 that there are significant BC emissions from natural gas combustion sources with good combustion
3 practices. It is recommended that additional source testing and research be conducted to improve
4 current emission factors associated with natural gas combustion. It is also recommended that additional
5 source testing and research be conducted on the related category of oil and gas flaring (see additional
6 discussion in section 8.X below).

7 Another category of note is use of biomass for power and steam generation. While wood-fired
8 boilers are currently a fairly small part of the U.S. inventory, there is the possibility that more stationary
9 sources may increase their use of biomass as a fuel source with the intention of reducing their carbon
10 footprint. To the extent that sustainable biomass becomes a more common source of fuel, BC emissions
11 could rise in absolute terms if not effectively controlled. Fortunately, effective technologies are already
12 available on the market that can control emissions from these sources, as described below.

13 **8.4 Available Control Technologies for Stationary Sources**

14 This section provides an overview of the main technologies for reducing $PM_{2.5}$ emissions from
15 stationary sources. Several post-combustion PM control technologies have been in operation for many
16 years and have been demonstrated to be quite effective in reducing $PM_{2.5}$. These technologies are also
17 considered to be relatively effective at controlling BC because BC is a filterable component of $PM_{2.5}$.
18 Many studies to date have assumed that $PM_{2.5}$ control technologies will reduce similar fractions of $PM_{2.5}$
19 and BC mass. However, it has also been recognized that reduction efficiency declines to some extent as
20 particle size decreases (and BC particles are commonly smaller than 1 micrometer in diameter). For this
21 reason, it is recommended that additional source testing and research be conducted on stationary
22 sources to better understand control efficiencies for BC and to develop improved emission factors for
23 specific source categories.

24 The two most effective control technologies for $PM_{2.5}$ (and therefore for BC) are fabric filters
25 (sometimes called baghouses) and electrostatic precipitators (ESPs). Although there are other
26 technologies used to reduce emissions of PM (such as cyclones and Venturi scrubbers), they are often
27 designed to control larger particles (PM_{10} and larger), and therefore are considered to be less effective
28 in terms of BC mitigation. EPA provides a thorough overview of the principles of operation, design
29 variations, applicability, performance, and associated costs of fabric filters and electrostatic precipitators
30 (ESPs) in the 2002 EPA Air Pollution Control Cost Manual (see U.S. EPA, 2002, Chapters 1 and 3).

31 **8.4.1 Fabric Filters**

32 A fabric filter unit consists of one or more isolated compartments containing rows of fabric bags
33 in the form of round, flat, or shaped tubes, or pleated cartridges. Particle-laden gas passes up (usually)
34 along the surface of the bags then radially through the fabric. Particles are retained on the upstream
35 face of the bags, and the cleaned gas stream is vented to the atmosphere. The filter is operated
36 cyclically, alternating between relatively long periods of filtering and short periods of cleaning. During
37 cleaning, dust that has accumulated on the bags is removed from the fabric surface and deposited in a

1 hopper for subsequent disposal. Fabric filters are not recommended for boilers burning oil because
2 particles from oil combustion are sticky and tend to clog the filter.

3 A properly designed and well run baghouse will generally have extremely high particle collection
4 efficiencies (i.e., greater than 99.9%). Baghouses are particularly effective for collecting fine particles
5 from power generation and a range of industrial facilities. For example, tests of bag houses on two
6 utility boilers (Broadway and Cass, 1975; Cass and Broadway, 1976) showed efficiencies of 99.8 % for
7 particles 10 μm in diameter and larger and 99.6% to 99.9% for particles 2.5 μm in diameter and smaller.
8 Studies have shown that collection efficiencies greater than 99% can be achieved for particles less than
9 1 μm in diameter (NESCAUM, 2005; Buonicore and Davis, 1992). A recent report for the U.S. Forest
10 Service on the applicability of different PM emissions control technologies to small wood-fired boilers
11 found that mechanical collectors, such as multicyclones, were only modestly effective in reducing PM
12 emissions, with an average of about 15% control efficiency. In the Forest Service study, fabric filters
13 achieved 74% reduction of $\text{PM}_{2.5}$, even with some of the uncontrolled flue gas circumventing the
14 baghouse (Hinckley and Doshi, 2010).

15 **8.4.2 Electrostatic Precipitators**

16 An ESP is a particle control device that uses an electrical charge to move the particles out of the
17 flowing gas stream and onto collector plates. ESPs typically achieve greater than 99% particle removal
18 efficiency (depending upon the design parameters chosen), although the removal efficiency varies with
19 particle size. Smaller particles are more easily carried by the gas stream, and therefore the ESP
20 collection efficiency for very fine particles like BC is typically lower than the efficiency for larger particles
21 (i.e. greater than 1 micrometer in diameter). Appropriately designed ESPs are effective at removing
22 particles from sources operating at high temperatures and having large volumes of gas. They can be
23 used on any PM source that emits BC. Sources such as biomass combustors that also generate
24 significant levels of condensable PM may benefit from wet ESP designs, in which the collector surfaces
25 are washed with water (either continuously or intermittently) to clean the particles from the collectors.
26 Tubular ESPs are most commonly used for operations where the PM is either wet or sticky (U.S. EPA,
27 2002, Chapter 1). Typical applications include sulfuric acid plants, coke oven by-product gas cleaning
28 (tar removal), and recently in iron and steel sinter plants. Because wood combustion systems in
29 particular can produce PM that is sticky, tubular ESPs may be appropriate for use in small systems for
30 reduction of PM and BC.

31 To address the lower ESP collection efficiency on submicrometer particles, a hybrid PM
32 collection system can be employed. Some designs place the baghouse downstream of an existing ESP to
33 improve overall collection efficiency. Others integrate the ESP and baghouse components. This type of
34 system can achieve 99.99% control of all particle sizes from 0.01 to 50 micrometers (Zhu, 2003).

35 A relatively new technology known as an agglomerator can also be used in conjunction with a
36 control device (such as an ESP) in utility or industrial applications. This technology is installed in the high
37 velocity ductwork leading to the control device. It pretreats the dust particles prior to entering the
38 device, agglomerating small and large particles together, thereby making it easier for the control device

1 to collect the larger particles. It has been shown to improve the ESP collection efficiency of very fine
2 particles (less than 1 micrometer in size) by 75-90% (Truce and Wilkison, 2008).⁴ There are a number of
3 commercial installations of the Indigo Agglomerator™ technology in place; most installations are
4 upstream of an ESP, but this technology has also been successfully operated in conjunction with a fabric
5 filter or wet scrubber.

6 Wet particulate scrubbers are generally not appropriate for control of BC. Collection efficiencies
7 for wet scrubbers vary with the particle size distribution of the waste gas stream. In general, collection
8 efficiency for submicrometer particles is much lower than for ESPs or fabric filter systems.
9 Submicrometer particle collection efficiencies for wet scrubbers typically are on the order of 50% or less,
10 although cyclonic wet scrubbers may be able to remove as much as 75% of submicrometer particle mass
11 (U.S. EPA, 2002, Chapter 2).

12 **8.4.3 Diesel Particulate Filters and Oxidation Catalysts**

13 There are more than a million stationary diesel engines in use today and together these sources
14 have substantial emissions of PM and NO_x. For most diesel engines, BC is a significant component of
15 untreated exhaust; these emissions can be reduced through diesel particle filter (DPF) technology.

16 As described earlier in this chapter, DPFs were originally developed for mobile engine
17 applications. They include variations such as diesel particle traps and catalytic and noncatalytic soot
18 oxidation systems. These units typically involve mechanical filtering of soot particles and a mechanism
19 for oxidation of the soot to CO₂. This second step is sometimes referred to as regeneration, and
20 eliminates the need for collecting and disposing of the captured particles. Catalysts are used to enhance
21 the oxidation process. Depending upon the design and operation of the DPFs, removal efficiencies of
22 between 40% and 99% can be achieved (van Setten, et al., 2001).

23 To ensure optimal performance of DPFs and to avoid poisoning of the catalyst, the diesel engine
24 should burn fuel with low sulfur content. DPFs have been identified by the California Air Resources
25 Board (CARB) as a verified technology for stationary engines serving prime and emergency standby
26 generators and pumps.⁵

27 EPA issued new source performance standards in July 2006 (71 FR 39153) for new compression
28 ignition (CI) stationary internal combustion engines. These standards implemented new restrictions on
29 emissions of PM, NO_x, VOC, and CO as well as new limits on the level of sulfur permitted in diesel fuel.
30 In June 2010, additional revisions were proposed for engines with a displacement of 10 to 30 liters per
31 cylinder – to the same stringent levels required by EPA's regulations for similar size nonroad diesel
32 engines used in marine applications. This rule is scheduled to be finalized in 2011.

⁴ See also: http://www.indigotechnologies.com.au/agg_overview.php.

⁵ A summary of CARB verified diesel emission control strategies is located at: <http://www.arb.ca.gov/diesel/verdev/vt/cvt.htm>.

1 **8.5 Cost-Effectiveness of PM Control Technologies**

2 The cost-effectiveness of an emissions control device for a stationary source is often expressed
3 in terms of dollars per ton of pollutant reduced. Factored into this amount are capital costs (amortized
4 over several years) for design and installation of the control equipment, and annual costs for operating
5 and maintaining the equipment. Because many emission standards for stationary sources to date have
6 included emission limits for total filterable PM (as opposed to PM_{2.5} or black carbon), many of the cost-
7 effectiveness values found in published reports are expressed in terms of the cost per ton of reducing
8 total PM. It has been noted earlier that we assume that BC emissions will be reduced with the operation
9 of a fabric filter or ESP on a stationary source. However, it is acknowledged that actual control
10 efficiencies for capturing submicrometer BC particles are uncertain, and they are likely to be somewhat
11 lower than the assumed control efficiencies for total PM or PM_{2.5}. For this reason, additional research
12 and source testing is needed to develop improved measurement techniques and development of robust
13 emission factors for specific source categories.

14 The effectiveness of a given control technology used for a specific source category will depend
15 not only upon the performance of the particular technology, but also upon the level of control that is
16 already in place. For instance, most large coal combustion sources, such as EGUs, are likely to be well
17 controlled to comply with prior PM emission standards. In contrast, smaller and older coal combustion
18 units that have not been subject to similar emission standards may demonstrate greater cost
19 effectiveness because installation of the same technology will remove a greater mass of PM (including
20 PM_{2.5} and BC) compared to a well-controlled EGU. Therefore, some sources that have been completely
21 exempt from PM control because of their age, small size, or limited operation (such as certain distillate
22 oil or coal combustion systems) may present favorable mitigation opportunities. Thus, a reasonable and
23 cost-effective mitigation strategy requires detailed knowledge of the sources and their emissions, on
24 both a per-source basis and across the full population of those sources.

25 The 2002 EPA Air Pollution Control Cost Manual and several related 2003 control technology
26 fact sheets provide typical cost effectiveness ranges for PM reduction by fabric filters and ESPs. The
27 cost-effectiveness range identified for a fabric filter was \$37 to \$337 per ton; for an ESP it was \$35 to
28 \$520 per ton.

29 Table 8-1 presents information on PM control cost ranges as adapted from multiple sources
30 referenced in a 2009 NESCAUM report for ICI boilers (NESCAUM, 2009). Capital costs can vary
31 significantly depending on the source specific characteristics. For some large utility boilers (500
32 megawatts), a fabric filter can require an investment on the order of \$70 to 105 million (NESCAUM,
33 2010). Cost-effectiveness values per ton are commonly higher for oil combustion units than for those
34 burning coal or wood. In general, however, PM control technologies are well-established and they
35 provide significant public health benefits, particularly for communities located close to emission sources.

36 It should be noted that control of PM and BC results in millions of pounds of particulates being
37 captured and disposed of as solid waste; and in some cases it is discharged as part of wastewater

1 discharges. The problems associated with disposal of coal combustion residues are well recognized, and
 2 it will be important to manage such wastes effectively in the future.

3

4 **Table 8-1. PM Control Costs for ICI Boilers.** (NESCAUM, 2009)

Fuel Type	Technology	PM Reduction Potential	Size of ICI Boiler (mmBTU/hr)	Capital Costs (million)	Cost Effectiveness (\$/ton removed)	Reference
Coal	Dry ESP	90-99%	250	\$3 - 40	\$156 – 1300	MACTEC, 2005
Coal	Fabric Filter	90-99%	250	\$2 - 23	\$423 – 1006	MACTEC, 2005
Oil	Dry ESP	90-99%	250	\$2 - 22	\$2328 – 21,009	MACTEC, 2005
Wood	ESP	99.5%	Medium		\$203 – 292	STAPPA, 2006
Wood	Fabric Filter	99.5%	Medium		\$147 – 249	STAPPA, 2006

5

6

7 **8.6 Mitigation Approaches Other than PM Control Technologies**

8 **8.6.1 Process Modification/Optimization**

9 As a product of the combustion process, BC can be reduced by approaches other than direct
 10 reduction using PM control devices. Process modification and/or optimization can be an effective
 11 means of reducing PM emissions. Some general examples of process optimization include reducing the
 12 frequency of mass transfer operations, improving operational efficiency, and the proper use of dust
 13 collection devices at the point of generation.

14 Cost values for these approaches are difficult to estimate, but are often negative. Often, steps
 15 to improve operational efficiency require only investments in instrumentation or operator training to
 16 yield on-going reductions in fuel consumption and emissions of all pollutants, including BC. Changes in
 17 fuel can be more expensive and are incurred over the entire period in which they are used, but are
 18 dependent upon fluctuating and often highly local market conditions. However, for many situations,
 19 particularly smaller boilers for which the costs of control technology investment and operation would
 20 make up a significant fraction of the system's initial and operating cost, conversion to fuels (such as
 21 natural gas) that generate lower PM and BC can be less expensive than use of post-combustion control
 22 equipment.

23 One specific example technique to reduce PM emissions for existing boilers is a boiler tune-up.
 24 Fuel usage can be reduced by improving the combustion efficiency of the boiler. At best, boilers may be
 25 85% efficient and untuned boilers may have combustion efficiencies of 60% or lower. As combustion
 26 efficiency decreases, fuel usage increases to maintain energy output resulting in increased emissions.
 27 Lower efficiency also results in formation of PM constituents like BC that are formed from incomplete

1 combustion of the fuel. The objective of good combustion is to release all the energy in the fuel while
2 minimizing losses from combustion imperfections and excess air. A tune-up can make a significant
3 difference in energy consumption and emission levels.

4 **8.6.2 Fuel Substitution and Source Reduction Approaches for PM**

5 The type of fuel and process has a great impact on PM emissions from combustion. Coal, oil,
6 and natural gas are the most common fuels used. Of these fuels, coal combustion generally results in
7 the highest PM emissions. As noted earlier, increased use of biomass fuels may also lead to higher BC
8 emissions, unless suitable control techniques are applied.

9 Fuel substitution can be an effective means of reducing PM emissions for many industrial fuel
10 combustion processes that generate process heat or electricity. Switching to fuels that generate lower
11 levels of BC per btu can be a viable alternative. However, there are several factors to consider when
12 evaluating whether fuel switching is the best option. When the age of a boiler or space constraints
13 make add-on control technology not cost effective, fuel switching may be an alternative. Capital
14 investment is usually small when compared to that of control devices. In addition, fuel switching can
15 lead to cleaner and safer unit operation. However, the lower capital cost must be weighed against a
16 change in fuel prices, such as would be incurred by switching from coal to natural gas. The actual cost of
17 converting to a different fuel that reduces BC emissions must account for the cost of installing the
18 necessary fuel feed systems, fuel price differential, and changes in non-fuel maintenance and
19 operational costs.

20 Fuel substitution for the purpose of reducing BC emissions can also reduce emissions of SO₂,
21 NO_x, and CO, depending upon the characteristics of the original and replacement fuels. A common
22 conversion is from coal to natural gas, with coal to distillate oil an appropriate alternative. Switching
23 from distillate oil to natural gas is also a possible approach for reducing BC emissions, but the reductions
24 in BC for such a change will be less than when switching from coal. Where sulfur contents of the original
25 coal or distillate oil are high, SO₂ emission reductions may not be significant when converting from coal
26 to distillate oil. Switching to natural gas from either fuel will likely result in significant SO₂ emission
27 reductions.

28 When considering fuel switching to reduce BC effects on climate, the impact on CO₂ emissions is
29 obviously a consideration. Conversion from coal to gas or wood will reduce CO₂ as well as BC. A further
30 alternative may be to switch to a biomass-based fuel oil. The bulk of liquid biofuels appropriate for use
31 in boilers is in the form of biodiesel, although there have been some evaluations of other biomass-based
32 fuel oils developed specifically for use in boilers (Partanen and Allen, 2005; Adams et al., 2002).

33 **8.7 Mitigation Approaches for Stationary Sources Internationally**

34 As discussed earlier in this section, stationary source emissions of PM are generally considered
35 to be well-controlled in most developed countries due to the operation of common control technologies
36 such as fabric filters and ESPs. The picture is different in developing countries, where a number of

1 specific industrial source categories have been identified as important contributors to BC emissions. The
2 source categories of concern vary by country and regions of the world. Mitigation opportunities exist in
3 these countries and regions because known control technologies exist and have been demonstrated to
4 be effective. This section will address the source categories that have been identified in the emission
5 inventories as being major contributors to BC emissions and for which known control technologies exist.
6 The source categories are brick kilns, coke production / iron and steel production, power generation and
7 industrial boilers, and oil and gas flaring.

8 **8.7.1 Brick Kilns**

9 Brick and masonry production in many developing countries (such as China, India, Bangladesh,
10 Vietnam, Nepal, and Pakistan) has increased in recent decades in response to growing urbanization and
11 increasing demand for construction materials. Currently, brick production is estimated to be growing at
12 a rate of 4% per year.⁶

13 Conventional brick kilns (such as bull's trench, clamp, and intermittent downdraught kilns)
14 generally are operated by small-scale ventures in rural areas, often with poor conditions for workers
15 (French, 2007; Gupta, 2003). Low-quality coal and firewood are common fuels used in brick-making; in
16 some cases, even waste fuels such as used tires are employed. These kiln designs have inefficient
17 combustion, leading to high emissions of both greenhouse gases and PM (and associated local air
18 pollution health effects). The inefficient operation of these kilns also leads to high fuel costs, and kiln
19 operations have been found to contribute to localized deforestation when cheap firewood is harvested
20 in lieu of purchasing more expensive coal to use as fuel.

21 The most basic BC mitigation technique is the replacement of inefficient kilns with kilns having
22 improved energy efficient designs, such as the vertical shaft brick kiln (VSBK), the tunnel kiln, or the
23 hybrid Hoffman kiln (HHK). These kilns generally require less than 50% of the fuel needed for a
24 conventional kiln (UNDP, 2007) and have been estimated to reduce PM emissions proportionally. Bond
25 and Sun estimated that reducing emissions by switching to a more efficient kiln design can be cost-
26 effective, in the range of \$5.5 to \$11 per ton of CO₂-equivalent (based on 20-year GWP) (Bond and Sun,
27 2005). China has taken steps over the past decade to promote the transition to the more efficient HHK
28 in many areas. In Bangladesh, the United Nations Development Program initiated a \$25 million project
29 in 2010 to implement 15 energy-efficient kiln demonstration projects over the next five years (UNDP,
30 2010).

31 Under a "business as usual" scenario, global BC emissions from brick kilns are expected to
32 decline by about 11% (428 to 381 Gg) over the 2005-2030 time period, reflecting a gradual introduction
33 of more efficient kilns. However, the technical mitigation potential for this sector exceeds this projected
34 reduction. Given the rapid rate of urbanization projected for coming decades in many countries, and
35 the high fuel cost and significant health and climate impacts associated with uncontrolled brick kilns,

⁶ See <http://www.resourceefficientbricks.org/background.php>.

1 appropriate policy options, technical assistance, and financial incentives could be considered to
2 accelerate the transition to more efficient brick kilns.

3 **8.7.2 Coke Production / Iron and Steel Production**

4 Coke is a key input used in the production of iron and steel. In the coking process, coal is heated
5 to very high temperatures for up to 36 hours in an airless furnace, and volatile carbonaceous gases are
6 driven off. Modern plants minimize emissions by capturing the coke oven gas and using it in a separate
7 chemical recovery process where it is refined into by-products and usually burned for heat production.
8 However, some small-scale plants located in developing regions or countries with economies in
9 transition still do not capture the carbonaceous emissions from coke production. These plants
10 represent potential BC mitigation opportunities.

11 The global demand for coke and steel has increased significantly in the past two decades. Bond
12 et al developed global emissions estimates for BC from coke production of 380,000 Gg, based on 1996
13 data (about 8% of estimated global “contained” emissions). It is acknowledged that this estimate was
14 highly uncertain due to the lack of information regarding the number of polluting “beehive” or
15 “indigenous” plants currently in operation globally. In the late 1990s, China was considered to have the
16 largest coke production capacity of any country by far; and it continues to be responsible for more than
17 60% of global coke production (based on 2008 data) and more than a third of global steel production.
18 Most coke production in China is conducted by state-owned enterprises. However, in 2004 it was
19 estimated that smaller township and village enterprises operating less capital intensive “indigenous”
20 plants were responsible for about 15% of the coke production in China; it is assumed that these smaller
21 but uncontrolled plants are responsible for a majority of BC emissions from the industry (Dukan, 2010;
22 Polenske and McMichael, 2002).

23 One mitigation option to reduce BC from coke plants is simply to phase out smaller uncontrolled
24 operations. China has initiated policies to phase out certain plants with uncontrolled emissions, but the
25 portion of the industry that has shut down or consolidated, and the extent to which emissions have
26 dropped to date is not well characterized. Another mitigation option is to retrofit the plant to be able to
27 operate an electrostatic precipitator which would reduce PM_{2.5} emissions by 95% or more. BC emissions
28 associated with larger coking operations could also be reduced via implementation of an energy
29 recycling program to recover waste heat from the very high temperature coking process. The recovered
30 heat would be converted to steam and used to power a generator which in turn would help provide
31 electricity needed for plant operations, reducing the total amount of coal needed run the plant and the
32 associated BC emissions (Polenske and McMichael, 2002).

33 Coke production and the iron and steel industry are important contributors to BC emissions in
34 other regions as well. Bluestein et al. noted that emissions from uncontrolled blast furnaces in the
35 former Soviet republics may have the potential to contribute to BC levels in the Arctic (Bluestein et al.,
36 2008). However, additional information is needed to improve our global inventories of BC from coke
37 production. To the extent that existing sources in China and other coke producing nations are not
38 recapturing exhaust gases, advanced technologies are readily available to reduce emissions significantly.

1 **8.7.3 Power Generation and Industrial Boilers**

2 Internationally, certain sources of power generation (especially smaller power plants, industrial boilers,
3 and stationary diesel engines) continue to operate without effective PM_{2.5} controls and represent
4 opportunities for mitigation. In 2001, it was estimated that 20% of the power plants in China were
5 operating without effective PM controls yet were responsible for 62% of the total PM_{2.5} emissions from
6 power plants. In addition, many industrial boilers in China are known to operate only wet scrubbers and
7 cyclones, which are effective in capturing larger particles but have low fine-particle removal efficiencies
8 (Zhang, et al., 2007). In regions of the world where electricity from the grid is unreliable or not
9 available, there is a substantial reliance on stationary diesel generators for power. For example, it is
10 estimated that diesel generators in India account for as much as 17% of total power generation (USAID,
11 2010, April). Diesel generators are also widely used in the Arctic region and contribute to BC deposition
12 locally (Quinn et al., 2008). Since control technologies are available to control emissions from these
13 sources effectively, additional emissions reductions could be achieved. However, further investigation is
14 needed to determine the cost-effectiveness of control options in specific locations.

15 **8.7.4 Oil and Gas Flaring**

16 Natural gas is a byproduct of the oil extraction process and it is often treated as a waste gas and
17 disposed of rather than captured for economic use. When not captured, it is either directly vented to
18 the atmosphere or it is burned through a process called flaring. The combustion process during flaring
19 can be inefficient and characterized by a distinct dark-colored, sooty plume. Oil and gas flaring and
20 venting leads to significant emissions of greenhouse gases (especially methane) and a variety of other
21 air pollutants, including BC, hydrocarbons and toxic air pollutants. Flaring can lead to significant health
22 impacts on nearby communities. BC emissions from flaring are of particular concern if they can impact
23 areas of snow and ice in the Arctic region.

24 Global estimates of pollutant emissions from flaring and venting are still quite uncertain. It has
25 been estimated that globally the natural gas wasted due to flaring is about 5% of the total annual
26 natural gas consumption. In 2002, the World Bank started the Global Gas Flaring Reduction initiative.
27 Many countries are now self-reporting flaring and venting data. NOAA has also developed
28 methodologies to estimate flaring activity through the use of satellite remote sensing data. Based on
29 this information, the countries with the highest estimated levels of flaring are Nigeria, Russia, Iran, Iraq,
30 and Angola (Buzcu-Guven et al., 2010). More work is needed to improve estimates of BC emissions from
31 flaring. Mitigation of venting and flaring activities will be an ongoing challenge for the future. Reducing
32 BC (and methane) emissions from flaring and venting activities would require expanded efforts to make
33 use of the natural gas for power generation on site or to capture the gas so that it can be distributed and
34 marketed. There are clear economic incentives for this. EPA is working with a number of governmental
35 and private partners to address these issues through the Global Methane Initiative.

1 **8.8 Technical and Research Needs**

2 Emissions of BC from industrial sources, both domestic and international, currently represent a
3 modest percentage of total BC emissions. In some regions, such as Asia, industrial emissions are more
4 significant (almost a quarter of “contained” emissions) (USAID, 2010 (April)). It is expected that over the
5 next two decades, global emissions from the industrial category will become a greater percentage of
6 “contained” global BC emissions as reductions occur in other sectors. The reduction of BC and PM_{2.5}
7 from industrial categories can be very cost-effective when considering the substantial health benefits
8 they provide to local populations, in addition to broader climate benefits. Studies in the US show that
9 reductions of PM_{2.5} from ICI boilers may yield benefits estimated at \$230,000 to \$560,000 per ton
10 (2008 dollars) (U.S. EPA, 2010 draft RIA for ICI boilers). For these reasons, controlling BC emissions from
11 industrial sources should remain a part of any overall BC reduction strategy.

12 While this is the case, the characterization of emission factors and emission inventories for key
13 sectors are recognized by many experts to be uncertain, and there is an important need to improve
14 PM_{2.5} and BC emission factors for industrial sectors. In some cases, only a few source tests may provide
15 the basis for many emission factors. The difficulty of measuring BC emissions that remain after control
16 devices have already treated the exhaust emissions lies in the difficulty of measuring emission rates at
17 the source of emissions. BC emissions are extremely difficult to measure under real-world and field
18 conditions. Our prior experience with PM control clearly indicates that some ultra-fine particles (BC and
19 OC) are being captured in control devices for larger particles, but reduction efficiencies of control
20 devices are generally considered to be lower for sub-micrometer BC particles than for total PM. To what
21 extent is not well documented (Streets et al., 2001).

22 To help develop improved PM and BC emission factors, global inventories, and future year
23 projections, additional source test information needs to be collected and evaluated for priority
24 categories, both in the United States and abroad. This research would quantify the before and after
25 measurements of pollution to quantify the emissions of BC that pass through existing control devices
26 into the ambient air, establish improved emission factors for different source categories, and assess the
27 engineering modifications that can be made to these control techniques that enhance their BC capture
28 capability. This could be facilitated by additional funding for technical and research programs, and
29 greater collaboration between EPA, state governments, industry groups, academic institutions, and
30 governments from other countries. Bilateral and multilateral assistance programs can also play an
31 important role in evaluating the cost-effectiveness of BC control measures in priority world regions.

(This page intentionally left blank)

9. Mitigation Approaches for Residential Heating and Cooking

9.1 Summary of Key Messages

- In the developed world, residential combustion is a small but potentially important source of BC emissions. There are clear health benefits of reducing residential wood smoke both indoors and outdoors. The climate impacts depend on the relative proportion of OC emissions, location of emissions (over ice/snow) and the type of wood-burning appliances used. Upgrading old wood stoves in areas with snow and ice to cleaner burning appliances appears to be the most effective strategy to reduce BC and OC from RWC.
 - U.S. residential wood combustion (RWC) is approximately 3% of the domestic BC inventory. Residential wood smoke contains PM_{2.5}, air toxic pollutants (e.g., benzene), methane, CO₂, OC, BC, and BrC.
 - EPA is currently working to establish new source performance standards for all types of residential wood heaters, including hydronic heaters, furnaces, and wood stoves.
 - Mitigation strategies for RWC sources have generally focused on either replacing inefficient units (wood stoves, hydronic heaters) with newer, cleaner units through voluntary or subsidized changeout programs, or retrofitting existing units to enable use of alternative fuels such as natural gas (fireplaces). New EPA-certified wood stoves have a cost-effectiveness of about \$3000/ton PM_{2.5}, while gas fireplace inserts average \$1500/ton PM_{2.5}.
 - The Arctic Council Task Force on Short-Lived Climate Forcers has identified wood stoves and boilers as a key mitigation opportunity for Arctic nations. The Task Force has recommended countries consider measures such as emissions standards, change-out programs, and retrofits to reduce BC from wood stoves, boilers, and fireplaces.
- In the developing world, about 3 billion people depend on rudimentary stoves or open fires fueled by solid fuels (e.g., wood, dung, coal, charcoal, crop residues) for residential cooking and heating. This number is expected to increase in the coming decades. Cleaner cooking solutions have the potential to provide huge public health benefits, and may be particularly important for reducing regional climate impacts in sensitive regions such as the Hindu Kush-Himalayan-Tibetan region.
 - Exposure to cookstove emissions leads to an estimated 2 million deaths each year and ranks as one of the five worst overall health risk factors in poor developing countries. Reductions

- 1 in these emissions likely represent the largest public health opportunity among all the
2 sectors considered in this report.
- 3 ○ The BC climate impacts from cookstoves are likely to be strong in a regional scope, and
4 additional source testing and modeling is needed to clarify the composition of emissions
5 from these sources and their net climate impact.
 - 6 ○ Cookstove mitigation activity today is hard to quantify definitively: while a preliminary
7 count by the Partnership for Clean Indoor Air indicated PCIA Partners reported selling ~ 2.5
8 million stoves in 2010, it is likely that 5-10 million “improved” stoves are sold each year by
9 commercial entities. In addition, there is not reliable data on the quality or performance of
10 many of these stoves. The full market of stoves is on the order of 500-800 million homes (3
11 billion people); thus, significant expansion of current clean cookstove programs would be
12 necessary to achieve large-scale climate and health benefits.
 - 13 ○ Many improved cooking solutions exist, but all face important supply, cost, performance,
14 usability, marketability and/or other barriers that make large-scale progress very difficult.
15 The potential climate and health benefits vary substantially by technology and fuel.
 - 16 ■ The performance hierarchy for improved cooking solutions appears to be as follows,
17 in generally decreasing order for both costs and emissions performance: 1)
18 electricity; 2) clean fuels such as LPG or ethanol; 3) advanced biomass stoves (e.g.,
19 forced air fan or gasifier stoves); and 4) rocket stoves. For all solid fuel stoves (3 and
20 4), processing the fuel into pellets or briquettes allows for greatly improved
21 combustion.
 - 22 ■ Biogas may be the cleanest, most climate-neutral (renewable) cooking solution
23 suitable for large-scale use; solar stoves are ultimately the cleanest solution, but
24 have not yet demonstrated an ability to reach large scales of dissemination.
 - 25 ○ A number of recent developments – including the growth of promising businesses and a
26 variety of business models; innovations in stove design, testing, and monitoring; carbon
27 financing; emerging research quantifying the health benefits of improved stoves; and new
28 country-based and global efforts to address these risks – have created a real opportunity to
29 achieve clean cooking solutions at a global scale.
 - 30 ○ The recently launched Global Alliance for Clean Cookstoves (GACC) (led by the United
31 Nations Foundation, with significant U.S. government participation) represents an enormous
32 opportunity to build on existing successes and rapidly increase the use of clean cooking
33 solutions. The Alliance will build on the EPA-led Partnership for Clean Indoor Air with more
34 than 450 Partners working in 115 countries, as well as several major efforts of leaders in this
35 field (e.g., Shell Foundation, GIZ, SNV, United Nations agencies).

- 1 ○ GACC’s goal is to disseminate 100 million clean cooking solutions by 2020. Achieving this
2 scale of progress will not be easy – it will require significant investments, demand a
3 coordinated global approach, and need to be based primarily on sustainable commercial
4 businesses that produce high-quality stoves that meet local users’ needs.

5 **9.2 Introduction**

6 Household energy use represents an extremely important source of BC emissions worldwide,
7 accounting for 25% of the total global BC inventory. In developed countries, most of these emissions are
8 associated with residential wood combustion (RWC), generally for heating. Total emissions from RWC in
9 developed countries are estimated at about 4% of the total global inventory (311 Gg) and 16% of total
10 residential emissions worldwide. In developing countries, emissions from residential combustion are
11 more often linked to widespread use of small stoves for cooking and/or heating. These cook stoves
12 utilize a wider range of fuels, including coal, natural gas, and dung as well as wood, charcoal, and other
13 biomass-related fuels. The emissions from residential cook stoves represent a much larger fraction of
14 the global inventory, accounting for 21% of total global BC emissions (1635 Gg) and 84% of emissions
15 from residential sources worldwide. The variety of sources and fuels within the residential category,
16 and the significant differences between developed and developing countries make this sector among
17 the most challenging from a mitigation perspective. However, given the vast number of people
18 dependent upon residential sources for everyday needs, such as heating and cooking, this sector also
19 represents one of the biggest opportunities for public health improvements through reductions of BC
20 and overall PM_{2.5}.

21 This section is divided into two parts. First, it presents information regarding available
22 mitigation approaches for residential wood combustion in the United States and other developed
23 countries. There are a number of cost-effective, advanced mitigation technologies that are well known
24 and easily deployed; the biggest challenge remains one of implementation and outreach. The section
25 then examines the technologies and approaches available for reducing emissions from the residential
26 sector in developing countries, where the scale of the problem is much broader, the range of sources
27 and fuels more complicated, and the challenges to effective implementation much larger. It describes
28 the advanced cookstove technologies that are currently available and their costs, and considers the
29 emissions reduction potential if these technologies were adopted on a large scale.

30 **9.3 Residential Wood Combustion in Developed Countries**

31 There are an estimated 29 million wood-burning fireplaces, over 12 million wood stoves and
32 hundreds of thousands of hydronic heaters (also known as outdoor wood boilers) throughout the United
33 States. Emissions from these appliances contain PM_{2.5}, toxic air pollutants, and other pollutants that can
34 adversely impact health and climate. The majority of these emissions come from old, inefficient wood
35 stoves built before 1990. Wintertime wood smoke emissions contribute to PM_{2.5} nonattainment and
36 localized problems in many areas in the United States. For this reason alone, replacing inefficient wood
37 stoves and educating wood burners on proper burn practices and stove operation are important
38 strategies for reaching domestic air quality goals. In fact, there is far greater certainty about the public

1 health benefits of reducing residential wood smoke emissions, both indoors and outdoors, than about
2 the net climate impacts, especially in light of the high level of OC emissions from these sources.

3 **9.3.1 Emissions from Residential Wood Combustion**

4 Incomplete combustion of wood results in emissions of fine and ultrafine particles, including BC,
5 BrC and other non-light absorbing OC particles. Inorganic materials, such as potassium, are also present
6 in lesser quantities as part of the mix of emitted particles. In the United States, RWC contributes over
7 350,000 tons of PM_{2.5} throughout the country—mostly during the winter months. Of this,
8 approximately 21,000 tons is BC, which is about 3% of total U.S. BC emissions. The key emitting source
9 categories that comprise RWC are wood stoves, manufactured and masonry fireplaces, hydronic
10 heaters, and indoor furnaces. The 2005 PM_{2.5} inventory shows that cord wood stoves contribute about
11 52%, fireplaces 16%, hydronic heaters 16%, indoor furnaces 11% and pellet stoves and chimineas (free-
12 standing outdoor fireplaces) the remaining 5%. Since 2005, the popularity and use of outdoor hydronic
13 heaters has grown. As a result the emissions from these units are growing and are of particular concern
14 to many areas, like the Northeast and Midwest.

15 In addition to PM_{2.5} and BC, wood smoke contains toxic air pollutants such as benzene and
16 formaldehyde, as well as methane (CH₄), CO, and CO₂. Nationally, RWC accounts for 44% of polycyclic
17 organic matter (POM) emissions and 62% of the 7-polycyclic aromatic hydrocarbons (PAHs), which are
18 classified as probable human carcinogens (NATA, 2005). All of these pollutants are products of
19 incomplete combustion (PIC). These emissions are the direct consequence of poor appliance design and
20 improper owner operation (e.g., using unseasoned wood) leading to incomplete combustion of the fuel.

21 OC emissions from RWC generally far exceed the BC emissions, making the BC/OC ratio
22 relatively small. However, different wood burning appliances combust wood in varying ways, resulting
23 in different BC/OC ratios. In general, wood stoves have higher BC/OC ratios than fireplaces (see Figure
24 9-1), and also represent a significantly larger percentage of the PM_{2.5} emissions inventory. The type of
25 wood burned also affects the amount of BC and OC emissions.

26 Despite the relatively low BC/OC ration from RWC in general, it is important to consider the
27 location of these emissions. While OC emissions are generally considered to have a cooling effect, OC
28 emissions over areas with snow/ice may be less reflective than OC over dark surfaces, and may even
29 have a slight warming effect (see Flanner et al., 2007). Significantly, the vast majority of residential
30 wood smoke emissions occur during the winter months; the highest percentage of wood stove use is in
31 the upper Midwest (e.g., Michigan), the Northeast (e.g., Maine), and the mountainous areas of the
32 Pacific Northwest (e.g., Washington), where snow is present a good portion of the winter months.

33

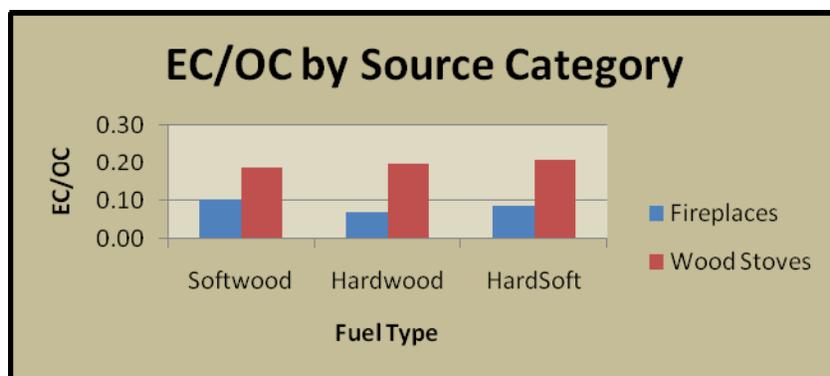


Figure 9-1. BC/OC Emission Ratios by Source Category and Fuel Type.

9.3.2 Approaches for Controlling Emissions from RWC

Mitigation of RWC $PM_{2.5}$ emissions generally involves increasing the combustion efficiency of the source. Wood burning appliances with lower combustion efficiencies tend to have higher emissions of most pollutants than do those with higher efficiencies. Due to design, conventional wood stoves, most fireplaces, and outdoor hydronic heaters do not burn wood efficiently or cleanly. Mitigation strategies for RWC sources have generally focused on either replacing inefficient units (such as wood stoves and hydronic heaters) with newer, cleaner units through voluntary or subsidized changeout programs, or retrofitting existing units (such as fireplaces) to enable use of alternative fuels like natural gas. The United States has been working to establish emissions standards for certain RWC sources, but it takes time for such programs to become effective, as they depend on the turnover in existing units. This is discussed more fully below.

To achieve the cleanest and most efficient combustion, the appliance needs to reach and maintain a sufficiently high temperature for all the necessary reactions to occur; adequate time for those reactions; and enough turbulence to ensure oxygen is available when and where it is needed. EPA-certified wood stoves, wood pellet stoves and Phase-2 qualified outdoor wood-fired hydronic heaters¹ are typically designed to increase temperature in the firebox, allow for adequate outside air to mix long enough for more complete combustion. The importance of the combustion conditions within these home-heating appliances, and the wood species used as fuel, in determining the composition of the resulting wood smoke is reflected by the observed variability in measured BC/OC ratios discussed above.

In general, greater combustion efficiency leads to reductions in the mass of direct PM emissions, including BC, as well as reductions in emissions of the gas-phase pollutants such as CO, CH_4 , and the volatile PAHs. For example, in a recent EPA study comparing a New Source Performance Standard (NSPS)-certified wood stove to a traditional zero clearance fireplace, the total PAH emission factor was found to be up to twice as high for the fireplace as for the more efficient stove burning the same oak fuel (Hays, et al., 2003). The same can be observed for other pollutants depending on appliance type,

¹ For a list of such appliances, see <http://www.epa.gov/burnwise/owhlist.html>.

1 wood species, moisture content, and so forth. A more efficient appliance also burns less wood for the
2 same heat output, leading to additional emissions reductions.

3 **9.3.3 Emissions Standards for New Wood-burning Units**

4 EPA has authority to establish NSPS emissions standards for new RWC sources, such as
5 fireplaces, woodstoves, and hydronic heaters. These standards establish manufacturing requirements
6 to limit emissions from new units. Such standards can be updated over time as new technologies
7 become available. Since 1988, EPA has regulated PM_{2.5} emissions from new residential wood heaters
8 sold in the United States. The Residential Wood Heaters NSPS (also referred to as the wood stove NSPS)
9 defines a wood heater as an enclosed, wood burning appliance capable of and intended for space
10 heating or domestic water heating that meets specific criteria, including an air-to-fuel ratio in the
11 combustion chamber averaging less than 35-to-1; a usable firebox volume of less than 0.57 cubic meters
12 (20 cubic feet); a minimum burn rate of less than 5 kg/hr (11 lb/hr) tested by at an accredited
13 laboratory; and a maximum weight of 800 kg (1,760 lb). Many types of sources are exempt from the
14 existing NSPS, including:

- 15 • Wood heaters used solely for research and development purposes
- 16 • Wood heaters manufactured for export (partially exempt)
- 17 • Coal-only heaters
- 18 • Open masonry fireplaces constructed on site
- 19 • Boilers
- 20 • Furnaces
- 21 • Cookstoves

22
23 The Residential Wood Heaters NSPS is unusual in that it applies to mass-produced consumer
24 items and compliance for model lines can be certified “pre-sale” by the manufacturers. A traditional
25 NSPS approach that imposes emissions standards and then requires a unit-specific compliance
26 demonstration would have been very costly and inefficient. Therefore, the NSPS was designed to allow
27 manufacturers of wood heaters to avoid having each unit tested by allowing, as an alternative, a
28 certification program that is used to test representative wood heaters on a model line basis. Once a
29 model unit is certified, all of the individual units within the model line are subject to similar labeling and
30 operational requirements.

31 EPA is currently in the process of revising the Residential Wood Heaters NSPS. Specifically, the
32 Agency is considering tightening the air pollution emission limits, adding limits for all pellet stoves,
33 reducing the exemptions, and adding regulations for more source categories, including hydronic heaters
34 and furnaces. EPA expects to propose appropriate revisions by June 2011, and finalize revisions in 2012.
35 The tightening of the wood heater NSPS has the potential to help reduce future residential wood
36 burning emission throughout the United States.

1 **9.3.4 Mitigation Opportunities for In-Use RWC Sources**

2 A fundamental limitation of the standards for new sources discussed above is that they cannot
3 influence emissions from units that were purchased prior to establishment of the NSPS. It can take a
4 long time for NSPS to actually reduce emissions, depending on the rate of replacement of existing
5 units—and in many cases, these units can remain in service for decades. Thus, alternative mitigation
6 strategies are needed to reduce emissions from existing sources.

7 In 2004, a panel convened by the National Academies of Science made several
8 recommendations to the EPA for improving air quality management in the United States. One of their
9 recommendations was to develop and support programs to address residential wood smoke. Since
10 2005, EPA has developed a residential wood smoke reduction initiative that has various components to
11 support state, local, and tribal communities in addressing their wood smoke challenges. This initiative
12 focuses on ensuring that wood burning is as clean and efficient as possible to help reduce emissions of
13 harmful pollutants, the amount of fuel used, and the risk of chimney fires from creosote that builds up
14 due to incomplete combustion. In general, these programs were developed to reduce PM_{2.5} and toxic
15 air pollutants, but can be employed to help reduce BC and other GHG (e.g., CH₄ and CO₂) from RWC.
16 The initiative has the following key components:

17 **9.3.4.1 Great American Wood Stove Changeout Program**

18 The hearth industry estimates that of the 12 million wood stoves in U.S. homes today, 75% are
19 wood stoves built before 1990. EPA is working with the hearth products industry and others to help
20 state, local, and tribal agencies create campaigns to promote replacement of old wood stoves and
21 wood-burning fireplaces with new, cleaner-burning and more energy efficient appliances. Programs
22 vary from one community to another, with some areas focusing on changing out old wood stoves and
23 others on retrofitting open fireplaces with cleaner burning options (e.g., gas stoves). The campaigns are
24 typically led by local government or non-profit organizations at the county or regional level.

25 Residents of participating communities generally receive incentives such as cash rebates, low/no
26 interest loans and discounts to replace their old, conventional wood stoves and fireplace inserts with
27 cleaner-burning, more efficient EPA-certified gas, pellet, electric, wood stoves and fireplaces or even
28 geothermal heat pumps. A new EPA-certified wood stove, new flue, and professional installation cost,
29 on average, \$3,500. Some areas have provided cash incentives to low-income participants only, while
30 others have provided incentives to everyone in the community. The local agency leading the
31 replacement program will sometimes include weatherization programs which insulate homes to help
32 reduce heat loss and reduce fuel consumption. Households that participate in these programs are
33 required to surrender their old wood stoves to be recycled.

34 Some of the benefits of replacing inefficient wood stoves include:

- 35 • Reduction in PM_{2.5} and toxic (e.g., benzeno(a)pyrene) air pollutants by 70%

- 1 • Reduction in indoor PM_{2.5} emissions by 70% according to University of Montana²
- 2 • Improvement in energy efficiency by 50%, using one-third less wood
- 3 • Reduction in CH₄, BC, and CO₂ from improved combustion efficiency and use of less fuel
- 4 wood

5
6 A variety of examples of state and local efforts to reduce emissions from older appliances are available
7 at EPA's Burn Wise website: <http://www.epa.gov/burnwise/casestudies.html>.

EPA estimates that every 1,000 old wood stoves changed out to cleaner burning hearth appliances will result in annual pollution reductions of:

- 815 tons of CO₂
- 53 tons of methane (CH₄)
- 27 tons of PM_{2.5}
- 4 tons of toxic pollutants
- 14 tons of OC
- 1.6 tons of BC

*Numbers generated using EPA's Wood Stove and Fireplace emissions calculator:

<http://www.epa.gov/burnwise/resources.html> and EPA's [speciation profile data base](#).

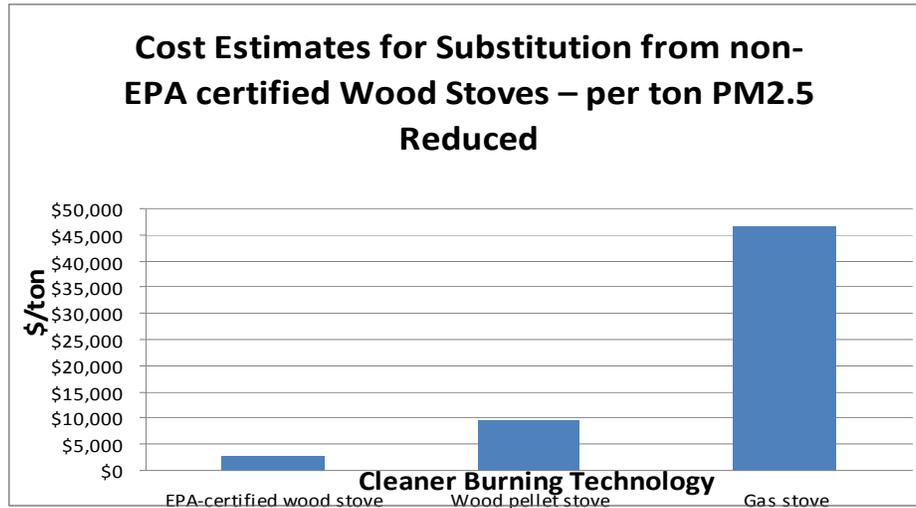
8
9 EPA's wood stove changeout effort has focused primarily on counties at or near nonattainment
10 for PM_{2.5} where wood smoke is an important local source. EPA estimates that through 2010, the Great
11 American Woodstove Changeout Program has helped replace nearly 18,000 wood stoves and fireplaces
12 in 50 areas. From 2010 on, this program is anticipated to reduce an estimated annual total of 300 tons
13 of PM_{2.5} and 50 tons of hazardous air pollutants (HAPs), providing approximately \$110 to \$270 million in
14 estimated annual health benefits.³

15 The best available cost-effectiveness information on residential wood smoke mitigation comes
16 from a Mid-Atlantic Regional Air Management Association document called *Control Analysis and*
17 *Documentation for Residential Wood Combustion in the MANE-VU Region (2006)*. This document
18 focused on the costs of total PM_{2.5} mitigation. This analysis indicates that the cost per ton of PM_{2.5}
19 emissions reduced from wood stove changeouts and fireplace retrofits is relatively low compared to
20 many other PM_{2.5} controls. Figures 9-2 and 9-3 provide PM_{2.5} cost-effectiveness estimates which vary
21 depending on the type of wood burning appliance being replaced (old wood stove vs. open fireplace)
22 and on the replacement technology (e.g., EPA-certified wood stove vs. wood pellet).

² For more information, see: <http://www.ncbi.nlm.nih.gov/pubmed/18665872>

³ Based on national average benefit-per-ton estimates derived from Fann et al. (2009), these estimates do not reflect local variability in population density, meteorology, exposure, baseline health insurance rates, or other local factors that might lead to higher or lower benefits.

1

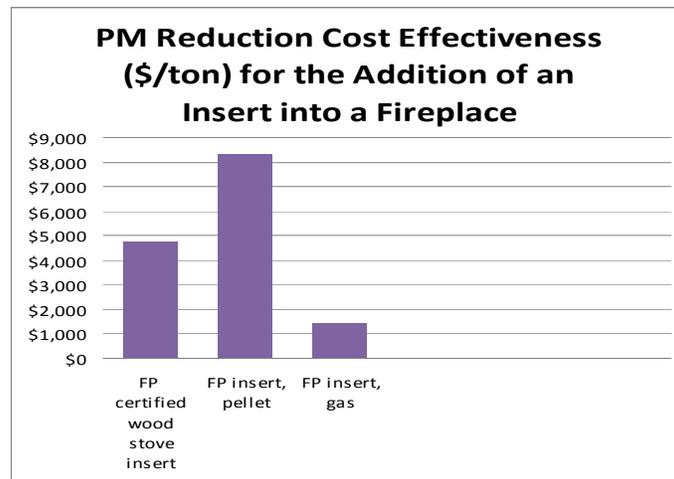


2

3

4

Figure 9-2. Cost Per Ton PM_{2.5} Reduced for Replacing Non-EPA-Certified Wood Stove with EPA-Certified Woodstove



5

6

Figure 9-3. Cost Per Ton PM_{2.5} Reduced (\$/Ton) for the Addition of an Insert into a Fireplace.

7

9.3.4.2 Outdoor Wood-Fired Hydronic Heater Program

8

9

10

11

12

13

14

15

In 2007, EPA initiated a partnership to reduce emissions from new outdoor wood-fired hydronic heaters. This program is aimed largely at areas with PM_{2.5} air quality problems. EPA has worked with industry to reach agreement on voluntary performance levels for new heaters to bring them to market faster than feasible under regulation. Similar to the wood stove changeout program there are potential climate change, air quality, and energy efficiency benefits with this program. The program is structured in two phases: under Phase 1, qualified new units were 70 % cleaner than existing units and, under Phase 2, which began in October 2008, new units are required to be 90 % cleaner than existing units. EPA has now expanded the program to include indoor models and hydronic heaters that are fueled by

1 other kinds of solid biomass (e.g., wood pellets). Manufacturers may not use Phase 1 labels after March
2 31, 2010. As of 2010, nearly 8,400 EPA-qualified units had been sold; 24 manufacturing partners had
3 agreed to produce units 70%-90% cleaner; and 22 models had been placed on the market, reducing an
4 estimated annual 4,770 tons of PM_{2.5} emissions and providing approximately \$1.7 billion to \$4.2 billion
5 in estimated annual health benefits.

6 **9.3.4.3 New Construction Wood-Burning Fireplace Program**

7 The EPA voluntary Wood-burning Fireplace Program is modeled after the Hydronic Heater
8 Program and helps reduce wood smoke emissions growth in areas with PM_{2.5} air-quality problems. The
9 two-phase program covers new installation of low mass (i.e., pre-manufactured) and masonry fireplaces
10 and is expected to drive technology improvements much sooner than possible through regulation. The
11 program qualifies models achieving a Phase 1 (34% reduction) or a Phase 2 (54% reduction) PM_{2.5}
12 emission level. EPA has worked closely with the hearth products industry to develop this program;
13 however, growth in the program has been hampered by the slowdown in new home construction in the
14 United States.

15 **9.3.5 Additional Regulatory Approaches to Limiting Wood Smoke Emissions**

16 A variety of regulatory programs, including wood burning curtailment programs and
17 requirements to remove old stoves upon resale of a home, have proven effective in helping to address
18 wood smoke.

19 Wood Burning Curtailment Programs: One of the most effective ways a community can reduce wood
20 smoke is by developing a mandatory curtailment program or institute “burn bans.” Some communities
21 implement both a voluntary and mandatory curtailment program depending on the severity of their
22 problem. Curtailment programs often have two stages with Stage 1 allowing EPA-certified wood stoves
23 to operate and Stage 2 banning all wood burning appliances, unless it is the homeowner’s only source of
24 heat. Although curtailment programs are not always popular with the public, this measure can be highly
25 effective at reducing wood smoke. As an example, the Sacramento Air Quality Management District’s
26 Stage 2 program, implemented in 2008-2009, reduced PM_{2.5} levels by 12 µg/m³. The cost effectiveness
27 was estimated to be approximately \$6,000 - \$10,500 per ton of PM_{2.5} (Sacramento Metropolitan Air
28 Quality Management District, 2009). To increase the likelihood of success, curtailment programs should
29 include a forecasting and public notification system. In addition, an enforcement component is also
30 important to ensure the public takes the program seriously.

31 Removal of Old Wood Stove Upon Re-Sale of a Home: Old wood stoves are usually made of metal,
32 weigh 250 to 500 pounds, last for decades, and can continue to pollute for just as long. As a result,
33 homeowners are less likely to replace old stoves with a new, cleaner burning technology or remove the
34 old stove especially if they are not using it. To help get these old stoves “off-line,” the state of Oregon
35 and some local communities in other states have required the removal and destruction of old wood

1 stoves upon the resale of a home. Specifically, this requirement has proven very effective in locations
2 like Mammoth Lakes, CA and Washoe County, NV.⁴

3

Education and Outreach: EPA's Burn Wise Campaign

Perhaps one of the biggest opportunities to reduce wood smoke emissions, including BC, lies in the hands of those who burn wood, regardless of the type of appliance they own. How wood stoves are operated and what is burned are as important as the type of stove used. EPA has heard from state, local, and tribal governments and from the public that even people who own an EPA-certified wood stove are often times burning "green" unseasoned wood, trash, and/or improperly operating their appliance resulting in high wood smoke emissions.

In October 2009, EPA launched an education campaign called Burn Wise to promote responsible wood-burning and to educate users on the connection between what they burn, how they burn, and the impacts on their health and the environment. The campaign provides a website (www.epa.gov/burnwise), fact sheets, posters, and public service announcements. EPA has coordinated with the hearth products industry, chimney sweeps (Chimney Safety Institute of America) and other partners on the development and implementation of the campaign.

Getting people to change their habits and behaviors, including their wood burning practices is typically not a trivial or inexpensive task. Equally challenging is measuring the effectiveness of social marketing or education campaigns like Burn Wise. However, EPA does believe the benefits, particularly the public health benefits are worth it and that some methods are more effective than others. For example, Environment Canada implemented a "Burn It Smart" campaign that included conducting community based workshops. The workshops were targeted in areas with government officials believed heating with wood was very common. Even though they did not calculate emission reductions, a follow-up survey of 174 people indicated that:

- 73% percent of the respondents said the workshops brought about positive change on how they burned wood
- 34% have updated their wood burning appliances, 90 % of those chose EPA-approved appliances
- 41% of those surveyed have changed out or intend to change out their old wood burning appliances for cleaner technology

4

5

9.3.6 Wood Smoke Reduction Resource Guide

7 In October 2009, EPA released a resource guide called *Strategies for Reducing Residential Wood*
8 *Smoke* that was written for state, local, and tribal air pollution control officials so they would have a
9 comprehensive list of strategies to help reduce wood smoke from residential heating. The guide
10 provides education and outreach tools, information on regulatory approaches (e.g., burn bans) to
11 reduce wood smoke, as well as voluntary programs to change out old, inefficient wood stoves and
12 fireplaces. It also notes the upcoming wood heater NSPS has the potential to help reduce future
13 residential wood burning emission throughout the United States. Several state and local communities
14 have effectively implemented residential wood smoke control strategies and have significantly reduced

⁴ For more information, see: <http://www.gbuapcd.org/rulesandregulations/PDF/Reg4.pdf>.

1 harmful wood smoke pollution. For example, Lincoln County, MT and Sacramento Metropolitan Air
2 Quality Management District have encouraged comprehensive wood smoke reduction strategies to help
3 these areas clean the air and protect public health.

4 **9.4 Residential Cookstoves in Developing Countries**

5 More than 3 billion people worldwide cook their food or heat their homes by burning biomass
6 (e.g., wood, dung, crop residues, and charcoal) or coal in polluting and inefficient traditional stoves (IEA,
7 2010: 9). BC emissions from these sources are estimated to account for 21% of the total global
8 inventory. This use of solid fuels also represents a significant part of energy use in developing regions—
9 including nearly 50% of total primary energy supply in Africa, and about 27% in India (IEA, 2009: 654,
10 648). Use of biomass and waste in developing nations—nearly all of which is for household cooking and
11 heating—accounts for about 60% of global renewable energy use (IEA, 2009, Annex A). About 82% of
12 those who rely on traditional biomass fuels for cooking live in rural areas; however, in Sub-Saharan
13 Africa, nearly 60% of people living in urban areas also rely on biomass (IEA, 2010: 20).

14 As discussed in Chapter 3, several decades of research document the significant risks to public
15 health associated with traditional cookstoves. Exposure to cookstove emissions leads to an estimated 2
16 million deaths each year and ranks as one of the five worst overall health risk factors in poor developing
17 countries. Emissions from cookstove use have been linked to adverse respiratory, cardiovascular,
18 neonatal, and cancer outcomes and to cancer (Smith et al., 2004). About 30% of the global mortality
19 associated with exposure to cookstove smoke occurs in Africa, and about a quarter each in India and
20 China. Recently, the contribution of this source category to emissions of BC and other aerosols has been
21 the focus of growing interest, especially in terms of impacts on sensitive regions such as the Himalayas.
22 However, there remains significant uncertainty about the extent of BC emissions from cookstoves, and
23 the effect of those emissions on climate. Given the complex emissions mixture resulting from cookstove
24 use, further study is needed to pinpoint the most beneficial strategies for reducing BC emissions from
25 this source. Unquestionably, however, this sector represents the area of largest potential public health
26 benefit of any of the sectors considered in this report. Mitigation of emissions from cookstoves offers a
27 tremendous opportunity to protect health, improve livelihoods, and promote economic development—
28 particularly for women and children. For this reason alone, irrespective of the additional climate
29 benefits that may potentially be achieved, mitigation of cookstove emissions is a pressing priority.

30 Mitigating BC emissions from cookstoves depends first on identifying technologies that are
31 proven effective in reducing BC emissions, and second on disseminating these technologies on a large
32 scale. As discussed below, not all improved stoves reduce BC emissions, and while some improved
33 technologies are emerging, no advanced stoves that burn solid fuel have yet been adopted on a broad
34 scale (though LPG has been widely disseminated as a clean cooking fuel, and China (see below)
35 implemented a very large earlier stoves program using intermediate scale stoves). The problem is
36 complicated by the fact that both the impacts of cookstoves and the solutions are regionally dependent.
37 Specifically, the extent of achievable BC reductions, and the impact of those reductions, will depend on
38 the type of stove, the type of fuel used, and the location of emissions. Improved cookstoves and fuels
39 must satisfy the needs of local users, enabling them to cook local foods at the time and in the manner

1 they prefer, using the fuels that are available and affordable. Given the array of different technologies
2 and fuels currently in use, and the sheer number of sources involved, mitigation of BC emissions from
3 cookstoves represents an enormous challenge. However, given their significant contribution to the
4 global inventory of emissions, and the increasing availability of cost-effective and locally appropriate
5 solutions, this sector represents one of the most promising for mitigation of BC internationally.

6 **9.4.1 Emissions from Cookstoves**

7 Currently, residential cookstoves contribute approximately 21% of the global BC emissions
8 inventory, with emissions concentrated in China, India, and Africa. Dependence on traditional biomass
9 fuels is highly correlated with poverty; countries with higher household income also tend to have a
10 higher share of modern fuels for residential consumption. While the percentage of people relying on
11 traditional biomass fuels for basic household energy needs is expected to decrease in most areas over
12 the coming decades, the aggregate number of people relying on biomass for cooking and heating is
13 expected to increase by 100 million people by 2030 due to population growth (IEA, 2010: 9). IEA
14 projects that the fastest shift toward modern fuels will occur in India, and the slowest shift will occur in
15 Sub-Saharan Africa (IEA, 2010: 21). The impact of these changes on emissions is still unclear: as
16 discussed in Chapter 6, under most scenarios, residential emissions are projected to decline significantly
17 by 2030 and further still by 2050 (Streets, et al. 2004). However, the rate of decline will depend on rates
18 of adoption of cleaner fuels and cooking technologies described below, and some regions may
19 experience near-term increases in emissions.

20 **9.4.2 Technologies and Approaches for Controlling Emissions from Cookstoves**

21 Because cooking is such a variable, individual-specific activity, there are many complexities
22 related to achieving reductions in BC emissions from improved cookstoves. The type of fuel and its
23 moisture content, the type of stove, the purpose for which it is used (heating vs. cooking), and the
24 manner in which the stove is tended all affect composition of emissions (MacCarty et al., 2008). Cooking
25 practices vary both daily and seasonally due to variation in available foods and fuels, and variation in
26 fuel quality. Additionally, there may be significant variation in the efficiency and durability of stoves,
27 even those that are mass produced.

28 In the past, “improved” stoves typically meant low-cost, locally made stoves aimed at improving
29 efficiency and reducing fuel use. A primary motivation for the use of improved stoves was to reduce
30 demand for fuel wood, thereby reducing pressures on forests as well as the time spent by women and
31 children gathering fuel (Graham et al., 2005; Winrock, 2005). However, not all such stoves functioned as
32 intended. For example, stoves that have a large amount of heated mass, such as the Lorena stove, may
33 remove smoke with a chimney, prevent burns, and help warm a house, but may not save fuel compared
34 with an open fire (USAID 2007).

35 Over the last ten years, a new suite of improved stoves has been introduced to the marketplace.
36 As a group, these improved stoves are designed to be efficient and clean (as well as safe), and utilize a
37 variety of different technologies and fuels. Most are produced locally for the nearby market, while

1 there are a few that are mass produced internationally and can be shipped anywhere in the world. The
2 stoves span a wide range of cost, durability, and performance, and are designed for different types of
3 staple foods. Importantly, however, these stoves are generally designed to reduce fuel use and
4 emissions of PM_{2.5} and CO (as proxies for the broader suite of emissions from these stoves). Few of the
5 stoves currently on the market were designed to reduce BC specifically (the new Turbococina stove is an
6 exception). In laboratory settings, most of these stoves achieve PM_{2.5} reductions of 40% to 70%.
7 Preliminary (unpublished) results from EPA tests indicate the non-advanced stoves may not substantially
8 reduce BC emissions, but some forced-draft (or “fan”) stoves significantly reduce BC emissions
9 compared to the open fire. Field testing has begun to demonstrate whether stoves perform as well in
10 actual (real-world) conditions as in the laboratory, but much more such testing is needed, as well as
11 additional research and development in stove design to determine if the stoves are reducing BC in
12 addition to total PM_{2.5}.

13 Among the new technologies now on the market there are a few advanced solutions that reduce
14 PM_{2.5} by 90% to 95%; the limited lab testing performed on these stoves to date indicates that they
15 reduce BC by a similar percentage. These solutions include advanced forced-draft stoves and “gasifier”
16 stoves that use various solid biomass fuels (including wood, pellets, crop residues, etc.); biogas stoves;
17 and liquid-fuel stoves that burn ethanol, plant oil, or other biomass fuels. Some stoves can now convert
18 waste heat to electricity to drive a fan; this in turn enables excellent emissions performance (including
19 BC emissions reductions) without the need for access to electricity.⁵ Some of these stoves are being
20 further tested for emissions in both the lab and the field. These new stove technologies have the
21 potential to reduce emissions from cookstoves nearly to the levels of clean fuels such as LPG (Wilkinson
22 et al., 2009), but many require specific and/or highly processed fuels, which increases the cost
23 (Venkataraman et al., 2010). Cost is lower for forced-draft stoves without electricity generation, but
24 battery-powered fan stoves require intermittent access to electricity.

25 While the basic outlines of lab and field tests have been in place for decades, it is only in the
26 past five years that organizations funding household energy interventions have begun requiring
27 emissions pre-testing, or that performance benchmarks (even informal ones) have been established.
28 The lack of international standards remains a limitation, but recent testing in both lab and field settings
29 (see below) demonstrate that this new generation of stoves is achieving real and measurable results.

30 Based on available performance and cost data, currently available technologies exhibit a wide
31 range of performance. These options include:

- 32 • *Electric Stoves:* Cooking with electricity produces zero emissions within a household, and
33 therefore is highly effective at reducing personal exposures of stove users. However, from a
34 broader public health or climate perspective, emissions associated with the increase in power

⁵ These stoves may also soon be able to reliably generate enough electricity to be used for other purposes (e.g., lights or cell phones), which could increase consumer demand. The change in emissions with the new stove would depend in part on the extent to which overall stove usage increased due to demand for these extra services (Venkataraman, et al. 2010).

1 production must also be considered. The ongoing electricity costs for these stoves can vary
2 substantially by region.

- 3 • *Gas and Liquid Fuels:* Switching from solid fuels to gaseous or liquid fuels is often the easiest
4 means of dramatically lowering emissions from cooking. In laboratory testing, the Aprovecho
5 Research Center (Aprovecho) found that using liquid petroleum gas (LPG) decreased the amount
6 of energy used by 69%, the mass of fuel used by 89%, particle emissions by over 99%, CO
7 emissions by 98%, and time to boil by 40%, as compared to cooking over an open fire (MacCarty
8 et al., 2010). Field research in Guatemala showed that LPG stoves could reduce 24-hr PM_{2.5}
9 concentrations by over 90% (Naeher et al, 2000). Liquid fuels such as ethanol, kerosene, and
10 plant oils are also options. Aprovecho's lab tests found that cooking with ethanol or kerosene
11 decreased the mass of fuel used by 75% and 82% and particle emissions by over 99% for each;
12 CO emissions by 92% and 87% (MacCarty, et al., 2010). Biogas derived from waste biomass is
13 potentially as clean as LPG, but it is renewably derived (reducing CO₂ impacts) and requires no
14 distribution infrastructure. Emissions testing of biogas stoves to date suggests that these stoves
15 perform significantly better than solid fuel/stove combinations with regard to emissions of
16 methane, CO, VOC, and CO₂ (Smith et al., 2000). Plant oils are another liquid fuel being used
17 today for cooking, but EPA is not aware of any published, independent testing results for stoves
18 using these fuels. Preliminary (unpublished) results from recent EPA tests of one plant oil stove
19 indicate substantially lower emissions of BC compared to the open fire. Stoves using gas and
20 liquid fuels involve an upfront cost of \$5 to \$50 per stove, as well as an ongoing cost for the fuel
21 that varies substantially by region, fuel, and changing economic conditions. LPG stoves can also
22 require significant deposits on the cylinders, another serious barrier for the very poor. It is also
23 important to note that poorly made kerosene stoves in particular pose safety concerns,
24 including the potential for severe burns and injury associated with accidental fires (Peck, et al.,
25 2008).

- 26 • *Processed Solid Fuels:* For much of the developing world, the advanced solutions described
27 above may be unavailable or simply too costly to use. Stoves utilizing processed solid fuels in
28 the form of charcoal, pellets, prepared wood, and briquettes, may be more accessible, and
29 these can also represent very clean solutions. However, like the clean fuels noted above, using
30 processed fuels also involves an ongoing operating cost, which may serve as a barrier for these
31 solutions, especially in regions where fuel wood can be collected free of charge. However, in
32 markets where fuel is purchased, stoves that increase combustion efficiency by 50% are often
33 the easiest stoves to market, since the consumer can expect a quick payback period on the
34 initial investment.

35 Charcoal is the most common processed solid fuel used today. Lab tests of charcoal stoves for
36 climate forcing emissions found that these stoves—relative to an open fire—reduced the BC/OC
37 ratio somewhat, and reduced total particles by about two-thirds (MacCarty, et al., 2008).
38 Aprovecho has tested many charcoal stoves for PM_{2.5}, CO, and fuel use, finding that PM_{2.5}
39 emissions were 90% lower than for a 3-stone fire and fuel use savings ranged from 45% to 65%.
40 Most charcoal stoves cut time to boil, though only modestly. However, CO emissions increased

1 for all stoves except one (MacCarty, et al., 2010). In 2007, EPA tested two charcoal stoves and
2 found that relative to a 3-stone fire, PM_{2.5} emissions from the charcoal stoves fell by over 90%
3 from a hot start but increased when operated from a raw, cold start, and both stoves increased
4 CO emissions (Jetter and Kariher, 2008). It is important to note that the laboratory emissions
5 tests do not account for emissions in the charcoal production process, which is highly inefficient
6 and polluting, with significant net climate impacts (Bailis, et al., 2005).

7 Creating pellets from biomass or briquettes from either coal or biomass can lead to substantial
8 improvements in efficiency and emissions when pellets are burned in well-designed stoves. The
9 Oorja stove (developed by BP and now owned by First Energy of India) is an example of a very
10 clean-burning pellet stove—in this case the pellets are made from crop residues by a partner
11 company. More than 400,000 Oorja stoves have been sold and between 250,000 and 350,000
12 are in use every day. However, given the cost of pellets, this stove competes with LPG. Other
13 examples include project-based work that have developed briquettes from waste biomass (Haiti,
14 Ghana, and Uganda), stoves designed to burn pellets made from locally available waste biomass
15 (West Africa and elsewhere), and a stove that burns rice hulls (Philippines), though EPA is not
16 aware of any examples where this work has been carried to a large scale. With regard to coal
17 cooking, laboratory measurements indicate that the combination of using improved stoves with
18 processed coal briquettes could have a dramatic impact on aerosol emissions. Zhi et al
19 measured reductions in particles of 63%—with OC decreasing 61% and BC decreasing 98%. This
20 reduced the BC/OC ratio by about 97%, from 0.49 to 0.016 (Zhi, et al., 2009).

- 21 • *Advanced Biomass Stoves:* There are two types of advanced biomass stoves that can achieve
22 high levels of performance: forced draft and gasifier stoves. Gasifier stoves can be forced-draft
23 or natural draft. These stoves can burn processed or raw biomass, though it is likely the case
24 (field testing data forthcoming) that those using processed fuels will perform better in the field,
25 since processed fuels eliminate a major variable in real-world use of the stoves. It is also likely
26 that lab and field test results will be more consistent for stoves that burn processed fuels. Lab
27 testing of advanced biomass stoves to date generally confirms that these advanced biomass
28 stoves can achieve remarkable emissions reductions—up to 93% lower than traditional stoves
29 (Venkataraman et al., 2010). One study found that these stoves achieved substantial reductions
30 in both overall particles and BC specifically, with the fan stove significantly reducing particle
31 emissions and the gasifier stoves reducing total particles by about two-thirds (as well as
32 reducing the BC:OC ratio). The study also showed that the fan stove was able to reduce time to
33 boil, at least under the lab conditions (MacCarty et al., 2008).

34 Under Aprovecho's broader lab testing, forced draft fan stoves all reduced (relative to a 3-stone
35 fire) fuel use (by 37% to 63%), CO emissions (in all cases by over 85%), PM_{2.5} emissions (from 82-
36 98+%), and time to boil (11% to 65%). Similarly, the gasifier stoves tested by Aprovecho saved
37 on fuel use, reduced CO emissions, achieved dramatic reductions in particle emissions (with one
38 exception), and cut the time to boil, though generally all to a lesser extent than the fan stoves
39 (MacCarty et al., 2010). In EPA's 2007 testing, the one advanced fan stove tested (Philips) had

1 the best overall performance and the lowest pollutant emissions, reducing emissions of key
2 pollutants such as PM_{2.5} and CO by about 90%. Notably, of the wood burning stoves tested, this
3 stove was also the one that required the least attention to operate (Jetter and Kariher, 2008).

4 It is important to note that while very promising in terms of performance, these models are still
5 in the research and development stage and that very few have been introduced in the market
6 today. These stoves are typically more costly than other biomass stoves, currently costing in the
7 range of \$25-100 per unit (plus any processed fuel costs, which can be substantial⁶), though
8 prototypes for newer (but also less durable) versions have been developed that manufacturers
9 estimate will cost closer to \$30 at full production.

- 10 • *Rocket Stoves:* Where advanced stoves are not widely available in the marketplace, or are not
11 affordable, rocket type stoves are typically the most efficient and clean wood-burning
12 alternative. Rocket stoves have a combustion chamber designed to allow for better mixing of
13 combustion gases and higher combustion temperatures, which substantially reduces emissions
14 without relying on electricity or other sophisticated components. MacCarty et al. (2008) found
15 that the rocket stove reduced total particle emissions by about 40%, but that nearly all of the
16 emissions reductions were of organic matter; BC emissions for this stove did not decrease (and
17 thus the BC:OC fraction increased dramatically). The study also showed that the rocket stove
18 was able to reduce time to boil, though to a lesser extent than an advanced fan stove.

19 Aprovecho tested a wide variety of rocket stoves and found that all but two saved on fuel use
20 (26% to 51% savings relative to a 3-stone fire). All rocket stoves cut CO emissions by 70% or
21 more, while performance on PM_{2.5} emissions varied much more widely (one actually increased
22 emissions), with 60% of those tested achieving reductions of over 50%. Some of the rocket
23 stoves actually increased the time to boil, though most cut it modestly (MacCarty et al., 2010).
24 In EPA's 2007 testing, several non-advanced wood stoves were tested and results varied
25 depending on the design and the stage of operation. Generally, emissions were lower than the
26 3-stone fire, with faster times to boil. For example, the UCODEA wood stove—now called
27 Ugastove—reduced PM_{2.5} and CO emissions by 48% to 65% when operated at high power, and
28 35% to 50% at low power (Jetter and Kariher, 2008).

29 USAID recently conducted extensive field testing of five non-advanced biomass stoves in the
30 Dadaab refugee settlements in Kenya. They tested for fuel use, time to boil, and several user
31 preferences—but did not test for BC or any other emissions—and concluded that “all five tested
32 stoves outperformed the open fire, requiring significantly less fuel to cook the test meal....with
33 savings ranging from 32% to 65” (USAID, 2010). Additional testing of two manufactured rocket
34 stoves by Columbia University researchers demonstrated “substantial and statistically significant
35 fuel savings relative to the three-stone fire” (38% and 46% on average, for the two stoves), but
36 further stressed that fuel savings is just one factor that affects suitability of any given stove in a

⁶ For example, the pellets for the Oorja stove in India cost roughly 7 Rupees (~15¢) per kilogram of pellets.

1 particular community. Other relevant factors include stove size, ease of use, and cooking time
2 (Adkins et al., 2010).

3 Several studies have measured changes in indoor concentrations of PM_{2.5} (but not BC) in
4 kitchens in Latin America due to the transition from a traditional open fire to the use of a griddle
5 stove (known in Latin America as a *plancha* stove)—a raised wood-burning stove with a
6 chimney, typically designed with a flat griddle to make tortillas. Naeher et al. (2000) reported
7 reductions in 24-hour PM_{2.5} concentrations of over 80%, and reported earlier measurements
8 that achieved reductions ranging from 57% to 82%. Masera et al. found that CO and PM_{2.5}
9 concentrations in the kitchens using a so-called *Patsari* stove were reduced by 66% and 67%,
10 respectively, compared to traditional cooking methods (Masera, et al., 2007). Johnson et al.
11 (2007) further reported that while *Patsari* stoves reduce overall particulate emissions in homes
12 (including net BC emissions), the BC/OC ratio went up, making the net warming implication
13 more ambiguous. McCracken et al. measured personal exposures (always less than reductions
14 in indoor air concentrations since individuals do not spend all of their time in kitchens) and
15 reported reductions in daily average exposure to PM_{2.5} of over 60% (McCracken, et al., 2007).

16 These stoves typically cost anywhere from \$8-\$100 per unit, depending on the design, quality of
17 materials, performance, use of a chimney, use of a metallic *plancha* (for making tortillas), and
18 durability. Certain models of these stoves have combustion chambers that might also be used
19 to build quality-controlled mud stoves—the combustion chambers themselves may cost as little
20 as \$4 to produce.

- 21 • *Simple Stoves*: Aprovecho test results for a wide variety of simple stoves without a rocket or
22 other improved combustion chamber indicated that the performance of these stoves varies
23 enormously, with only two of seven tested achieving meaningful emissions and fuels use
24 reductions. Most achieved some fuel savings, but increased particle emissions (MacCarty et al.,
25 2010). These stoves typically cost only \$2 to \$10 per unit, but may last only a few months due
26 to use of less durable materials and lower quality construction.
- 27 • *Solar Cookers*: Solar cookstoves are emissions free, and thus the cleanest solution. However,
28 the constraints of current solar cookers are significant: they have limited use in the early
29 morning, late afternoon, or on cloudy or rainy days; they can greatly increase cooking time; and
30 they are not suitable for cooking many foods. For this reason, the potential for current solar
31 cookers is best thought of as part of an integrated solution. EPA is not aware of any example of
32 solar cookers (which range in cost from \$20 to \$75 per stove, including the pot) being adopted
33 at a large scale in a given region. However, additional advances, such as improvements in
34 energy storage capacity, it is conceivable that solar stoves could be an effective tool for this field
35 in the future.
- 36 • *Behavioral and Structural Solutions*: Many behavioral and structural steps can be taken to
37 reduce human exposures to cookstoves smoke. These include cooking outdoors, keeping
38 children away from cooking stoves, adding ventilation to the kitchen, preparing fuel (drying and

1 cutting to a smaller size), tending stoves more carefully, lighting stoves with improved
2 techniques, or requiring stoves to have chimneys. Each of these solutions will diminish
3 immediate human exposures to cookstove smoke, and are thus to be encouraged as much as
4 possible, though the net benefit to human health may be tempered non-trivially by worsened
5 ambient air quality when chimney stoves are the intervention. For the core purposes of this
6 report, it is also critical to note that some of these solutions will have little impact on BC
7 emissions or related climate impacts, while others (such as preparing the fuel, tending stoves
8 more carefully, and using improved lighting techniques), may reduce climate forcing emissions.

9 **9.4.3 Programmatic Considerations for Cookstove Mitigation**

10 As this extended discussion of currently available technologies indicates, there are a number of
11 promising opportunities in the cookstove field. Advanced stoves can provide dramatic improvements in
12 public health, and may also offer opportunities to reduce BC emissions. However, with the important
13 exception of widespread adoption of LPG as a cooking fuel, the current scale of total stove replacements
14 is limited, and the number of advanced stoves deployed as part of these programs is very small.

15 There have been many efforts to bring improved cookstoves to different parts of the world,
16 ranging from large-scale government efforts in both China and India to countless small non-
17 governmental organization-led efforts in communities across the globe. These efforts have had varying
18 degrees of success. By far the most successful effort historically in terms of level of penetration of
19 improved stoves was China's National Improved Stove Program (NISP), introduced by China's Ministry of
20 Agriculture in the 1980s. The NISP targeted 860 of China's 2,126 counties, and the government statistics
21 indicate that from 1982 to 1992, 129 million improved stoves had been installed in rural households
22 (Graham, et al., 2005). Gradually, the Chinese government shifted to focus on supporting stove
23 manufacturers (Sinton, et al., 2004), and follow-on programs increased total penetration to close to
24 200 million households (Graham, et al., 2005). This program was primarily designed to reduce fuel use
25 (as related to its contribution to deforestation). Thus, while the use of chimneys allowed China to lower
26 indoor pollution somewhat, they were not able to reduce overall air pollution and GHG emissions
27 (Wilkinson, et al., 2009). It is not clear to what extent, if any, this effort may have had on BC emissions.

28 In 1983, the government of India launched its National Program of Improved Chuhlas (NPIC).
29 Over the next 17 years, the program introduced about 32 million improved biomass stoves to rural
30 households around the country (Barnes and Kumar, 2002). While results varied substantially from region
31 to region, "A 1995-96 survey conducted by the National Council of Applied Economic Research (NCAER)
32 in 18 states indicated that 71% of the cook stoves were in working order and 60% were in use" (Sinha,
33 2002). Like the Chinese program, India's NPIC was designed to lower demand for fuel wood. The
34 removal of indoor smoke was a secondary priority (Winrock, 2005). The NPIC has several shortcomings
35 that limited its long-term success, including poorly designed subsidies, poorly designed stoves
36 developed without user input, poor maintenance programs, and—in most regions—no commercial basis
37 for sustained results (much greater success resulted where a commercial model was followed) (Winrock,
38 2005). In spite of its shortcomings, India's earlier program remains—after China's NISP—the largest
39 cookstove program ever implemented (Barnes and Kumar, 2002).

1 Indonesia, Nepal, Mexico, and Peru have launched national stove programs, while many other
2 countries are actively working in this field. In December 2009, the government of India announced that
3 it would launch a new National Biomass Cookstove Initiative to build on India's earlier national program,
4 but be based almost entirely on a commercial business model in close cooperation with leading
5 manufacturers of clean stoves and fuels in India. India will also seek to catalyze further stove and fuel
6 innovations, for example via a global stove design prize.

7 The United States has been an active participant in the effort to address the many health risks
8 associated with traditional cookstoves. At the 2002 World Summit on Sustainable Development, U.S.
9 EPA brought together leaders from the government, private, academic, and non-governmental sectors
10 to launch the Partnership for Clean Indoor Air (PCIA). Through 2010, key PCIA Partners have reported
11 helping 6.6 million households adopt clean cooking and heating practices, reducing harmful exposures
12 for more than 30 million people. PCIA has found that to succeed with sustainable household energy and
13 health programs in developing countries requires focusing on four essential elements, including meeting
14 social and behavioral needs of users; developing market-based solutions; improving technology design
15 and performance; and monitoring impacts of interventions.

16 Over time, the scale and pace of cookstove replacements have been increasing worldwide.
17 Preliminary 2010 estimates from PCIA based on partial reporting from its network of more than 460
18 partners indicate that partners sold 2.5 million stoves in 2010 (this figure will increase as more partners
19 report their results). Based on the latest survey results, PCIA Partners are more than doubling their
20 stove sales every other year. This does not include the internal Chinese stove market and independent
21 manufacturers that make and sell different versions of the so-called "Jiko" charcoal stove across Africa.
22 Including these sales, the total number could be as high as 5 million to 10 million stoves per year, though
23 there is not reliable international data on the quality or performance of many of these stoves. Despite
24 this progress, the total impact of the cookstove replacements to date has been small, given that the
25 total stove market is on the order of 500 million to 800 million homes.

26 In addition to the design and fuel innovations noted above, a number of recent developments
27 point to a much greater potential for reaching large scale progress in the cookstove sector. These
28 include:

- 29 • *Growth of Existing Businesses and Business Models:* An increasing number of businesses are
30 manufacturing and/or selling improved stoves and fuels, utilizing a wide range of business
31 models. These models include NGOs working to catalyze local businesses around a common and
32 tested stove design (e.g., GERES/ Cambodia's local partners just sold their 1-millionth stove);
33 working to develop local businesses to make and sell artisanal stoves (e.g. GIZ's global efforts to
34 provide over 4 million homes with improved stoves over the past 5 years); a local factory selling
35 directly (e.g., HELPS/Guatemala's grew over 500% in two years to sell a projected 95,000 in
36 2010); international manufactures with local distributors (e.g., a partnership between the
37 Aprovecho Research Center in Oregon on design, Shengzhou Stove Manufactures in China, and
38 Colorado-based EnviroFit International on sales); and, major corporations building their business
39 in emerging markets (e.g., Bosch-Siemens).

- 1 • *New Scalable Technologies:* Many of the stoves noted above represent a new suite of stove
2 technologies that are well designed and durable, and for which extensive emissions testing has
3 been conducted. Such stoves could be mass produced, which would improve the scalability of
4 these solutions (Venkataraman, et al., 2010).
- 5 • *Carbon Financing:* Cookstove businesses are increasingly leveraging carbon financing in both
6 the formal and voluntary markets to provide capital and increase public awareness. The
7 financing arrangements vary substantially, but typically yield about 0.5 to 2 tons of CO₂-e per
8 stove per year for improved wood and charcoal stoves, and up to 3 to 5 tons of CO₂-e per stove
9 per year for improved coal stoves. Importantly, however, these credits are based on GHG
10 (mostly CO₂) emissions reductions, as measured by reductions in fuel use during in-field tests.
11 Additional work would be required to establish credits for BC reductions. Carbon financing is
12 already transforming financing of cookstove efforts into more rigorous financial transactions
13 with rigor and accountability for stoves sold, stove performance in the field, and stove utilization
14 over time. The high transactions costs involved in obtaining project approval also incentivize
15 large-scale projects and encourage the continued use of approved stoves for many years to
16 generate ongoing credits. Impact investing is a separate, but important opportunity to bring
17 social capital investments to this field, and examples of this tool applied to the cookstove field
18 are beginning to emerge.
- 19 • *New Testing and Monitoring Tools:* The demand for rigorous monitoring for carbon and other
20 financing, research, and other needs has also led to the development of less expensive and
21 more effective monitoring technologies that greatly improve our ability to measure and
22 interpret field results. These include relatively inexpensive PM_{2.5} monitors, BC monitors,
23 personal exposure monitors for CO and PM_{2.5}, portable stove emissions testing hoods, stove use
24 monitors, and cell-phone based wireless monitoring tools.

25 In spite of this progress, achieving large-scale adoption of clean cooking solutions will not be
26 easy, and many remaining barriers must be addressed. A recent World Bank study has summarized
27 some of the key challenges, emphasizing the need for a range of stoves that meet users' needs, with
28 demonstrated ability to reduce fuel use and indoor smoke, while maintaining durability and safety. The
29 report also notes that successful programs require functioning commercial markets. Innovative
30 financing techniques and well-constructed monitoring and evaluation programs were other tools
31 highlighted as critical to success in reaching the poor (World Bank, 2010). Other major considerations
32 include:

- 33 • *Institutional Barriers:* Such barriers include the lack of accepted international standards for
34 different stove-fuel combinations, the lack of independent stove testing facilities in market
35 places around the world, and the lack of health guidelines regarding what interim targets on
36 what is considered a "clean" stove.
- 37 • *Cost:* The cost of improved stoves and fuels alone pose a major challenge for many households.
38 Additional financial barriers include tariffs and duties to import stoves, the large investment

1 needed to take a prototype stove to mass production, the cost and difficulty of developing
2 distribution chains in target markets, the high transactional costs of carbon financing, and the
3 costs of managing an inventory for a widely fluctuating market during business start-up.
4 Separate financing tools are needed make advanced stoves affordable for the poorest
5 populations.

- 6 • *Social Barriers:* Improved stoves have not always been designed with the needs and social
7 practices of end users in mind. By extensively testing prototype stoves with users, commercial
8 businesses have been able to lessen these risks.

- 9 • *Global Leadership:* Coordination and cross-disciplinary leadership is needed to pursue
10 integrated solutions that address each of the climate, health, gender, forestry, energy,
11 agricultural, and other dimensions of the cookstove issue. In the past decade, several new
12 efforts have emerged that have brought new focus to the health and climate risks of cookstoves,
13 and new rigor to solutions to these risks. These include the U.S. EPA-led PCIA, the Shell
14 Foundation’s Breathing Space program, GIZ’s HERA program, and SNV’s global biogas efforts, as
15 well as more isolated investments by the World Bank, USAID, and several agencies focused on
16 refugee camps (e.g., United Nations High Commissioner for Refugees and World Food
17 Programme).

18 In September 2010, the United Nations Foundation and nineteen founding partners launched
19 the Global Alliance for Clean Cookstoves. This new Alliance is a new public-private initiative whose
20 mission is “to save lives, improve livelihoods, empower women, and combat climate change by creating
21 a thriving global market for clean and efficient household cooking solutions.” The Alliance will work
22 closely with private, non-governmental, UN and other partners to expand efforts to address the global
23 and local barriers that have limited the scope of cookstove replacements. The Alliance has set an
24 interim goal of having 100 million new homes adopt clean and safe cooking solutions by 2020. The U.S.
25 government is a leading partner to the Alliance, with the U.S. Department of State leading diplomatic
26 outreach and several agencies (EPA, HHS (including the National Institutes of Health and the Centers for
27 Disease Control and Prevention), DOE, and USAID) contributing substantially to the applied research
28 agenda of the Alliance.

29 Solutions on this scale are needed to resolve the tremendous human health and environmental
30 burden—including the climate impacts—of traditional cookstove use. As the above discussions indicate,
31 large scale success in this field may be within reach. Substantial reductions in BC on the order of 90% to
32 95% per household likely depend on switching to cleaner fuels or advanced biomass stoves. Such highly
33 efficient, clean stoves help meet multiple goals, including fuel efficiency, health protection, low climate
34 impacts, and reduction of outdoor pollution (Venkataraman, et al., 2010).

1 Currently, simple unimproved stoves dominate the marketplace. Most current improved stove
2 sales are of the intermediate variety – rocket stoves or other solutions that achieve important health⁷
3 and fuel use benefits, but will not achieve the large health and BC benefits sought. As the Alliance
4 advances towards its interim goal of reaching 100 million homes, solutions will need to evolve towards
5 cleaner fuels and more advanced stoves so as to ensure that substantial public health and BC benefits
6 are achieved. Additional research and innovations to bring these very clean solutions to massive
7 populations are needed to move as rapidly as possible to achieve the health and climate benefits that
8 advanced stoves can bring to families and the environment.

⁷ As head of the Department of Environmental Health Engineering at Sri Ramachandra University in Chennai, India Kalpana Balakrishnan has said, “[These] existing improved stoves have to go some way before they can meet a health-based standard, but they are much, much better than the traditional stoves we have now” (Adler, 2010).

(This page intentionally left blank)

10. Mitigation Approaches for Open Biomass Burning

10.1 Summary of Key Messages

- Open biomass burning is the largest source of BC emissions globally, affecting 340 million hectares/year. However, total emissions of OC are seven times higher than total BC emissions from this sector, and better and more complete emissions inventory data are needed to characterize the impacts of open biomass burning and evaluate the effectiveness of mitigation measures for reducing BC emissions.
 - Wildfire accounts for a large portion of BC emissions from open biomass burning: in the United States, for example, wildfires account for 68% of BC emissions from open biomass burning.
 - The regions of the world responsible for the majority of BC emissions from open biomass burning are Africa, Asia, and South America, with significant contributions from Russia/Central Asia and North America. There is large interannual and regional variability in these emissions.
 - BC emissions from open biomass burning (predominately from widespread agricultural burning and large wildfires occurring in the northern latitudes) have been tied to reduced snow and ice albedo in the Arctic.
- Certain emissions reductions techniques may yield reductions in BC emissions from open biomass burning; however, most of these techniques were developed to reduce total PM_{2.5} emissions from fires and there is still substantial uncertainty about their effectiveness for reducing BC emissions specifically, especially given diverse, site-specific burning conditions.
- Appropriate mitigation measures depend on the timing and location of burning, resource management objectives, vegetation type, and available resources. It is important to note that fire plays an important ecological role in many ecosystems, and prescribed burning is one of the basic tools utilized to achieve multiple land-management objectives in fire-dependent ecosystems.
- Successful implementation of mitigation approaches in world regions where biomass burning is widespread will require training in proper burning techniques and tools to ensure effective use of prescribed fire.

1 **10.2 Introduction**

2 This section presents currently available information regarding mitigation efforts and techniques
3 that may help reduce particle emissions from open biomass burning (agricultural burning, prescribed
4 burning, and wildfires). The effectiveness of these controls on emissions of BC and OC (including brown
5 carbon) requires further study. In addition, given the importance of planned fire as a land management
6 tool, there are important tradeoffs that must be considered in evaluating mitigation options for open
7 biomass burning.

8 **10.3 Emissions from Open Biomass Burning**

9 Open biomass burning, as discussed in this report, encompasses three main categories of
10 burning: agricultural burning, prescribed burning, and wildfire.¹ Table 10-1 describes each type of open
11 biomass burning, the land types on which they may occur, and examples of typical resource
12 management objectives each burning type is designed to achieve. In some cases, there are slight
13 differences in how these terms apply to domestic and international burning practices.

14 The Joint Research Centre of the European Commission estimates that 350 million hectares (865
15 million acres) of land were affected by fire, worldwide, in 2000 (FAO, 2007). However, given the lack of
16 an international standard for fire terminology and the lack of consistent data reporting and collection, it
17 is not possible to distinguish among the fractions of land area that were subject to agricultural versus
18 prescribed burning or wildfire (FAO, 2007). Generally, the mass of BC emitted from open biomass
19 burning will depend on the size and duration of the fire, fuel type, fuel conditions, fire phase, and the
20 meteorological conditions on the day of the burn. The emissions estimates presented in Chapter 4
21 indicate that open biomass burning represents a potentially large, though poorly quantified portion of
22 the U.S. BC emissions inventory. As with the international fire emissions inventories, available data are
23 limited regarding the percentage of land area affected by different types of burning. It is also important
24 to note that emissions of OC are seven times higher than BC emissions from this sector. Preliminary
25 research suggests that the OC fraction may be dominated by BrC, which also absorbs light. More
26 focused research is needed to clarify the composition and quantity of emissions from different types of
27 fires.

¹ Categories of contained biomass combustion, including residential heating and cooking and industrial biomass combustion, are addressed in previous chapters.

1 **Table 10-1. Types of Open Biomass Burning.** (U.S. EPA, 1998)

Type of burning	Description	Land type	Typical resource management objective(s)
Agricultural	The planned burning of vegetative debris from agricultural operations. (Domestic)	Forestland, cropland, rangeland, grassland, wetlands	Restore and/or maintain fire-dependent ecosystems; control weeds, pests, and disease; manage lands for endangered species; promote various vegetation responses; reduce fuel loading to reduce catastrophic wildfire risk; improve crop yield; control invasive species; facilitate crop rotation; remove crop residue
	The use of fire as a method of clearing land for agricultural use or pastureland. (International)	Forestland, rangeland, grassland, wetlands	Conversion of land into cropland or pastureland
Prescribed	The planned burning of vegetation under controlled conditions to accomplish predetermined natural resource management objectives. Conducted within the limits of a fire plan and prescription that describes the acceptable range of weather, moisture, fuel, fire behavior parameters, and the ignition method to achieve the desired effects.	Forestland, rangeland, grassland, wetlands	Restore and/or maintain fire-dependent ecosystems; control weeds, pests, and disease; manage lands for endangered species; promote various vegetation responses; reduce fuel loading to reduce catastrophic wildfire risk
Wildfire	An unplanned, unwanted wildland fire (such as a fire caused by lightning), unauthorized human-caused fires (such as arson or acts of carelessness by campers), or escaped prescribed burn projects (escaped control due to unforeseen circumstances)	Forestland, rangeland, grassland, wetlands	Fire suppression or other appropriate management response

2

3 As the estimates in Chapter 4 indicate, open biomass burning is the largest BC source in Africa,
 4 Central and South America, and Asia, and is one of the largest sources of BC in Russia/Central Asia (the
 5 former USSR) and North America. However, there is considerable variation in the type of open burning
 6 that dominates in different regions. Fires in sub-Saharan Africa are primarily due to slash-and-burn
 7 practices for clearing agricultural sites, burning of crop residues, escaped planned burning, acts of
 8 carelessness, and arson (FAO, 2007). The primary causes of fire in Central and South America include
 9 large-scale conversion of moist tropical forest to rangeland and agriculture, arson, negligence, and
 10 hunting (FAO, 2007). Available information suggests that the majority of fires in China and other East
 11 Asian countries are uncontrolled wildfires, typically caused during land conversion, or by arson and acts
 12 of carelessness (FAO, 2007). Prescribed burning is used to some degree in China to reduce catastrophic
 13 wildfire risk (Morgan, 2009). In India, and other South and Southeast Asian countries, fire emissions
 14 stem from agricultural burning, rangeland clearing, escaped planned burning, or acts of carelessness
 15 (FAO, 2007). Agricultural burning in Kazakhstan, southern Russia, Central and Eastern Europe is a

1 seasonal occurrence, typically starting at the end of April and lasting for a few weeks (Warneke et al.
2 2009; Stohl et al. 2007). Wildfires in Russia (Siberia) are primarily caused by lightning, escaped planned
3 burning, or acts of carelessness (FAO, 2007), and occur from late April throughout the summer
4 (Warneke et al. 2009; Generoso et al., 2007). Russia experiences many smoldering fires in drained or
5 dry peatlands that burn for long periods and produce large quantities of smoke (FAO, 2007). In the Far
6 East and southern Siberian portions of Russia, extensive prescribed burning of the grasslands has been
7 used in the spring to reduce highly flammable surface fuels (FAO, 2007).

8 As described in Chapters 2 and 4, there is strong evidence to suggest that emissions from fires in
9 one world region can significantly impact other world regions through transport and deposition
10 processes. Reduced snow and ice albedo, and increased rates of melting in the Arctic, the Himalayas,
11 and other snow and ice-covered regions of the world are major impacts of BC deposition, with
12 implications for freshwater resources in regions dependent on snow-fed or glacier-fed water systems.
13 Most of the BC that reaches the Arctic has been traced to sources to sources in the Northern mid-
14 latitudes (AMAP, 2009), with open biomass burning as one of the largest of the sources. A primary
15 determinant of the downwind impact of a large fire on snow and ice-covered regions is the height to
16 which the plume rises, i.e., its injection height. Fire plumes observed by satellite between 1978 and
17 2009 have shown that more dense wildfire plumes rose to the level of the free troposphere, i.e., 8 km,
18 where long-range transport can occur more readily, over North America than over Australia, or Russia
19 and Northeast Asia (Guan et al., 2010). This difference has been attributed to the type of wildfire that
20 dominates in North America, i.e., boreal crown fires² that are large and very high in temperature. In
21 general, between 5 and 28% of the plumes from large wildfires in North America rise into the free
22 troposphere (Val Martin et al., 2010).

23 Current emissions projections suggest that direct PM emissions from open biomass burning will
24 continue to dominate global BC inventories. In addition, several major climate change science
25 assessments have concluded that large, catastrophic wildfires will likely increase in frequency over the
26 next several decades because of climate warming (Field et al., 2007; Ryan et al., 2008; Wiedermeier,
27 2010; Larkin 2010). General climate warming encourages wildfires by extending the summer period that
28 dries fuels and promoting easier ignition and faster spread (Field et al., 2007). Earlier spring snowmelt
29 has led to longer growing seasons and drought, especially at higher elevations where the increase in
30 wildfire activity has been greatest (Field et al., 2007). Increased temperature in the future will likely
31 extend fire seasons throughout the western United States, with more wildfires occurring both earlier
32 and later than is currently typical, and will increase the total area burned in some regions (Field, et al.,
33 2007). Within Arctic regions, climate change is expected to shift the treeline northward, with forests
34 replacing a significant portion of land that is currently tundra and tundra vegetation moving into
35 currently unvegetated polar deserts (ACIA, 2004). Changes in Arctic climate are also expected to
36 increase the frequency, severity, and duration of wildfires in boreal forests and dry peat lands,

² Crown fires occur in the tops of trees and are spread more quickly than ground fires. Boreal forests are generally defined as those occurring at high northern latitudes across North America and Eurasia, below the Arctic tundra.

1 particularly after melting of permafrost (ACIA, 2004; Schneider et al. 2007). These climate-related
2 changes in wildfire location, duration, and frequency will affect both BC and OC emissions.

3 **10.4 Fire as a Resource Management Tool**

4 Both natural and prescribed fires play an important ecological role across the globe, benefiting
5 those plant and animal species that depend upon natural fires for propagation, habitat restoration, and
6 reproduction. Most North American plant communities evolved with recurring fire and are dependent
7 on recurring fire for maintenance. The natural fire return interval (i.e., estimates of how often fires
8 would naturally occur without human intervention) may vary from one to two years for prairies, three to
9 seven years for some long-needle pine species, 30-50 years for species such as California chaparral, and
10 over 100 years for species such as Lodgepole pine and coastal Douglas-fir.

11 Natural fires also reduce fuel load and tree density, helping to reduce the risk of catastrophic
12 wildfires. In many parts of the United States, historical land management practices during the late 19th
13 and early 20th centuries (e.g., fire suppression, logging, and livestock grazing) have altered the natural
14 fire regime, changed forest structure, and led to heavy fuel accumulation in forests. This, in turn, has
15 increased the size of wildfires and total area burned (Miller et al., 2009; Noss et al., 2006; Allen et al,
16 2002; McKelvey 1996). Accumulated fuel loads will likely continue to affect the size and frequency of
17 large wildfires in the coming decades.

18 In the United States, prescribed burning is one of the basic tools relied upon by land owners and
19 managers to achieve multiple management objectives in fire-dependent ecosystems. When one
20 management objective is to maintain a fire-dependent ecosystem, the effects of fire cannot be
21 duplicated by other tools. Prescribed fire can also be used to reduce heavy fuel loads, which has the
22 benefit of helping to prevent catastrophic wildfires.

23 The following section includes an outline of strategies that have been used for conducting
24 prescribed and agricultural burning in a manner that protects air quality by reducing smoke emissions,
25 and managing burning conditions to protect downwind populations. In addition, the importance of fire
26 prevention is discussed. These methods may also be applied with the goal of reducing BC emissions,
27 overall, and/or the goal of reducing downwind deposition of BC on snow and ice. As will be discussed,
28 the techniques listed may be more useful in some ecosystems than in others. Further study is needed to
29 identify appropriate strategies to apply under each circumstance.

30 **10.5 Smoke Mitigation Technologies and Approaches in the United States**

31 Appropriate mitigation of BC from open biomass burning would depend on the type, timing, and
32 location of burning and must balance multiple objectives including resource management, climate
33 protection, and health protection. Currently available literature is extremely limited regarding the
34 effectiveness of any given mitigation practices for reducing BC emissions from the three general types of
35 burning. More research is needed to better understand the efficacy, potential unintended

1 consequences, and cumulative effects arising from the implementation of any proposed mitigation
2 techniques.

3 As a starting point, however, it is appropriate to consider how approaches currently used to
4 manage the air quality impacts of open biomass burning may be applicable to BC. Most U.S. domestic
5 policies and programs at the local, state, and federal level focus on protecting air quality and public
6 health by managing smoke and minimizing PM emissions. There are two basic approaches that are
7 commonly applied to manage the air quality impacts from open biomass burning: (1) use techniques
8 that reduce the emissions produced for a given area; and/or (2) redistribute the emissions through
9 meteorological scheduling and by sharing the airshed (Ottmar et al., 2001).

10 One common approach in the United States for limiting the impacts of open biomass burning is
11 the development and application of smoke management programs. The Interim Air Quality Policy on
12 Wildland and Prescribed Fires (U.S. EPA, 1998)³ recognizes the role fire plays as a resource management
13 tool. The policy addresses wildland and prescribed burning managed for resource benefits on public,
14 tribal, and privately-owned wildlands. The policy integrates two public policy goals: (1) to allow fire to
15 function, as nearly as possible, in its natural role in maintaining healthy wildland ecosystems and, (2) to
16 protect public health and welfare by mitigating the impacts of fire emissions on air quality and visibility.
17 The policy encourages state and tribal authorities to adopt and implement smoke management
18 programs to mitigate the public health and welfare impacts from prescribed fires and promote
19 communication and coordination of prescribed burning among land owners.

20 A smoke management program establishes a basic framework of procedures and requirements
21 for planning and managing smoke from prescribed fires. It is typically developed by a state/tribal agency
22 with cooperation and participation by various stakeholders (e.g., public/private land owners/managers,
23 the public). If a state/tribe determines that a smoke management program is needed, they may choose
24 to develop a program using an array of smoke management practices/emission reduction techniques
25 that they believe will prevent air quality violations and address visibility impairment.⁴ A smoke
26 management program can range from a purely voluntary program to a program where prescribed fires
27 are regulated by a permitting authority that analyzes meteorological conditions and air quality
28 considerations and authorizes burning by time of day, fire location/ size and anticipated duration. The
29 more-structured program may include enforceable requirements on who may burn and when burning
30 may occur.

31 The basic elements of a smoke management program include guidelines or requirements
32 regarding authorization to burn, coordination and scheduling, and air quality assessment (U.S. EPA,

³ As discussed in EPA's 2007 Final Rule on the Treatment of Data Influenced by Exceptional Events (72 *Federal Register* 13560), the Interim Air Quality Policy on Wildland and Prescribed Fires ("Interim Fire Policy") is currently under revision.

⁴ EPA intends to include guidance on the use of basic smoke management practices in the revised Fire Policy when it is finalized.

1 1998). In cases where burn plans are developed, these generally focus on (1) Actions to minimize
2 emissions (emission reduction techniques); (2) Evaluation of predicted smoke dispersion; (3) Public
3 notification; (4) Contingency measures to reduce exposure; and (5) Fire monitoring and plume
4 dispersion characteristics. In addition, smoke management programs frequently lay out guidelines or
5 requirements for recordkeeping and reporting; public education and awareness; surveillance and
6 enforcement; and program evaluation.

7 In developing a smoke management program, authorities have a number of options available for
8 reducing emissions, e.g. emissions reduction techniques (ERTs), and for managing smoke, that can be
9 applied under different circumstances. It is important to note, however, that decisions regarding the
10 appropriate use of different techniques are influenced by a number of considerations—including but not
11 limited to air quality impacts, water quality impacts, Endangered Species Act requirements, and basic
12 resource management objectives. The following section provides an overview of the current practices
13 employed for mitigating air quality impacts.

14 **10.5.1. Emissions Reduction Techniques**

15 Emissions reduction techniques may offer the benefits of reduced BC emissions and reduced
16 downwind impacts related to BC deposition on snow and ice. However, there is still substantial
17 uncertainty about the applicability and effectiveness of these emissions reduction techniques for
18 reducing BC under diverse, site-specific burning conditions. The appropriateness of a given mitigation
19 practice and its effectiveness at reducing PM_{2.5} and/or BC will depend on the type of fuel being burned
20 (e.g., crop residue or forest), the management objectives of the burn, and the seasonal timing and
21 geographic location of the burn. An additional consideration is that open biomass burning occurs on
22 land under various ownership (i.e., federal, state, tribal, and private), which affects management
23 decisions and the types of burning practices implemented on those lands. Currently available literature
24 identifies a number of current fire management practices to address air quality impacts of PM emissions
25 from agricultural and prescribed burning; they are listed below.

26 **10.5.1.1. Agricultural Burning PM Mitigation Techniques**

- 27 • Reduce the number of acres burned
 - 28 ○ Reduce burning through conservation tillage, soil incorporation, or collecting and
 - 29 hauling crop residues to central processing sites (WRAP, 2002).
 - 30 ○ Apply alternate year burning which involves alternating open field burning with various
 - 31 methods of mechanical removal techniques. The period may involve burning every
 - 32 other year or every third year (U.S. EPA, 1992).
- 33 • Increase combustion efficiency
 - 34 ○ Use bale/stack for agricultural residue. The bale/stack burning technique is designed
 - 35 to increase the fire efficiency by stacking or baling the fuel before burning. Burning in
 - 36 piles or stacks tends to foster more complete combustion, thereby reducing PM
 - 37 emissions. This control is applicable to field burning where the entire field would be set
 - 38 on fire, and can be applied to all crop types (AirControlNet v4.1).

- 1 ○ Propane flammers are an alternative to open field burning.
- 2 ○ Use backing fires (“backburning”). Flaming combustion is cleaner than smoldering
- 3 combustion. Backburning ensures more fuel is consumed in the flaming phase (Ottmar
- 4 et al, 2001).
- 5 • Reduce fuel loadings
- 6 ○ Remove straw/stubble before the burn.
- 7 • Change burn timing from early spring to either winter or summer to reduce higher impact of BC
- 8 on snow/ice. Quinn et al. (2008) suggest that this technique may be especially important for
- 9 mitigating climate impacts in the Arctic, to reduce springtime deposition when the snow and
- 10 icepack is large. Applicability of this technique will be limited by the type of crop, the resource
- 11 objectives sought, and biological and operational constraints.
- 12 • Convert Land Use
- 13 ○ Convert from a crop that requires burning to a crop that does not.
- 14 ○ Convert land to non-agricultural use.
- 15 • Educate Farmers
- 16 ○ Provide training to farmers on proper burning techniques that reduce emissions.

17 **10.5.1.2. Prescribed Burning PM Mitigation**

- 18 • Reduce the area burned
- 19 ○ Use mosaic burning. Landscapes often contain a variety of fuel types that are non-
- 20 continuous and vary in fuel moisture content. Prescribed fire prescriptions and lighting
- 21 patterns can be assigned to use this fuel and fuel moisture non-homogeneity to mimic a
- 22 natural wildfire and create patches of unburned areas or burn only selected fuels
- 23 (Ottmar et al., 2001).
- 24 • Reduce fuel consumed (Ottmar et al., 2001)
- 25 ○ Burn fuel when moisture content is high. Fuel consumption and smoldering can be
- 26 minimized by burning under conditions of high fuel moisture of duff, litter, and large
- 27 woody fuels.
- 28 ○ Conduct burns before precipitation. Scheduling a prescribed burn before a precipitation
- 29 event may limit the consumption of large woody material, snags, stumps, and/or
- 30 organic ground matter.
- 31 • Reduce fuel loadings (Ottmar et al., 2001)
- 32 ○ Burn outside the growing season, burn after timber harvest, and burn frequently.
- 33 Prescribed burning at appropriate times can help reduce the size and magnitude of
- 34 wildfires.
- 35 ○ Expand the use of biomass. Harvesting and selling or trading the biomass is one
- 36 alternative to prescribed burn. Woody biomass can be used in various industries such
- 37 as pulp and paper, methanol production, and garden bedding. This alternative is most
- 38 applicable in areas that have large diameter woody biomass and the biomass is plentiful

1 and accessible so as to make biomass utilization economically viable. Small-diameter
 2 biomass can be used as posts, poles, or tree stakes. Neary and Zieroth (2007)
 3 documented a successful USDA Forest Service project in Arizona to remove and sell
 4 small-diameter trees for use in small power plants that burn wood fuel pellets. Biomass
 5 can also be pyrolyzed to produce biochar, a fine-grained charcoal, for use as a soil
 6 amendment (i.e., to improve physical properties of the soil, such as water retention,
 7 permeability, water infiltration, drainage, aeration and structure) (www.biochar.org).

8 ○ Use other fuel treatments such as mechanical treatments/removal. Mechanical
 9 treatments may be appropriate when management objectives are to reduce fuel density
 10 to reduce a wildfire hazard, or to remove logging waste materials (slash) to prepare a
 11 site for replanting or natural regeneration. On-site chipping or crushing of woody
 12 material, removal of slash for off-site burning or biomass utilization, whole tree
 13 harvesting, and yarding (pulling out) of unmerchantable material may accomplish these
 14 goals. Mechanical treatments are normally limited to accessible areas, terrain that is
 15 not excessively rough, slopes of 40% or less, sites that are not wet, areas not designated
 16 as national parks or wilderness, areas not protected for threatened and endangered
 17 species, and areas without cultural or paleological resources.

18 ○ Use chemical treatments. When the management objective is to preclude, reduce, or
 19 remove live vegetation and/or specific plant species from a site, chemical treatments
 20 may be appropriate tools. However, other potential environmental impacts caused by
 21 applying chemicals must also be considered.

22 ○ Use animal grazers. Increasing grazing by sheep, cattle, or goats before burning on
 23 rangelands and other lands can reduce grassy or brushy fuels prior to burning, and can
 24 help reduce burn frequency.

25 ● Increase combustion efficiency

26 ○ Use mass-ignition techniques that produce short-duration fires (e.g., aerial ignition).
 27 Mass ignition can shorten the duration of the smoldering phase and reduce the amount
 28 of fuel consumed.

29 ○ Use backing fires (see above).

30 ○ Burn piles or windrows. Fuels concentrated into clean and dry piles or windrows
 31 generate greater heat and burn more efficiently.

32 ○ Use air curtain incinerators, which are large metal containers or pits with a powerful fan
 33 device to force additional oxygen into the fire, to produce a very hot and efficient fire
 34 with very little smoke. Air curtain incinerators offer a useful alternative to current fuel
 35 reduction and disposal methods, providing the benefits of producing lower smoke
 36 emissions compared to pile or broadcast burning; burning a greater variety, amount,
 37 and size of materials from dead to green vegetation; reducing fire risk; operating with
 38 fewer restrictions in weather and burn conditions; and containing burn area to a specific
 39 site (<http://www.fs.fed.us/eng/pubs/html/05511303/05511303.html>)

40 ● Education for Resource Managers

41 ○ Train resource managers on proper burning techniques to reduce emissions.

42

1 Currently available literature is extremely limited regarding the cost of reducing BC emissions
2 from agricultural and prescribed fire. Many of the PM emission reduction techniques described above
3 require substantial infrastructure and resource investment (e.g., roads, machinery, etc.) or the existence
4 of a market for biomass utilization products (e.g., wood pellets or biochar). The availability of the
5 required infrastructure, resources, and markets will vary across the country, making the cost of potential
6 mitigation options highly uncertain and dependent on the technique(s) and the site-specific
7 environmental conditions in which the technique(s) are applied. A recent study (Sarofim et al. 2010)
8 surveyed currently available literature to develop rough cost estimates for the major categories of PM
9 emission reduction techniques described above (i.e., increase combustion efficiency, reduce fuel
10 consumed, reduce fuel loadings, and reduce the area burned).⁵ The authors found that these
11 techniques are on the whole likely to be quite expensive for the amount of BC reduced, although there
12 may be potential for lower cost mitigation approaches in locations where markets for biomass utilization
13 exist.

14 **10.5.2. Fire Prevention Techniques**

15 While wildfires are part of the natural functioning of many ecosystems, increasing fuel loads
16 within the United States over the past century have made wildfires harder to control and more
17 expensive to suppress. In addition, wildfires often pose a dangerous threat to the lives and property of
18 civilians and firefighters. Fire prevention techniques can be effective in helping to prevent unwanted
19 human caused fires. Efforts by the U.S. Forest Service and other resource management agencies are
20 currently underway to turn fire suppression programs into more proactive fire management programs
21 that effectively apply fire prevention and hazardous fuels reduction techniques, extensive public
22 education, and law enforcement (National Interagency Fire Center, 2011).

23 Fire prevention approaches involve a combination of engineering, education, and enforcement.⁶
24 Education strategies often represent low-cost approaches for preventing unwanted fires. Such
25 strategies must include clear planning and communications with regard to subjects such as fire-prone
26 areas where access is closed or restricted; appropriate use of campfires, smoking, and fireworks; and
27 managing the burning of trash and debris. Raising public awareness through education and outreach,
28 including utilizing media such as newspapers, radio, and television, is also important. Such educational
29 campaigns can be highly effective in preventing unwanted fires: the U.S. Forest Service's long-standing
30 Smokey Bear campaign is among the most successful fire prevention awareness and education
31 campaigns ever conducted (National Wildfire Coordinating Group, 2007).

⁵ The authors calculated unit emissions reductions of the various mitigation options using emissions factors in tonnes of BC/OC per kilogram of dry matter burned. Because these emissions factors vary according to the particular crop/ecosystem burned and the phase of burning (e.g., flaming or smoldering), there was a range of values each open biomass burning source category. Sarofim et al. (2010) used the median (when multiple data points were available) or the midpoint (when only two data points were available) of the range.

⁶ Additional information on each of these strategies is available on the National Wildfire Coordinating Group's publications page at <http://www.nwccg.gov/teams/wfewt/products.htm>.

1 **10.6 Mitigation Technologies and Approaches Globally**

2 As discussed in Chapter 2, a number of recent studies have pointed to the importance of
3 reducing international BC emissions from open biomass burning to alleviate effects on the Arctic, the
4 Himalayas, and other key snow and ice-covered regions. Many of the mitigation techniques and
5 approaches described above could also be applied internationally, and such strategies could provide
6 important climate benefits. However, the practical mitigation options available on the ground in
7 different regions are limited for a number of reasons. Critical barriers to implementing mitigation
8 measures internationally fall within three areas: (1) weak governance (e.g., requisite laws and policies at
9 all levels of government to authorize and enforce fire management practices); (2) lack of local capacity
10 (e.g., requisite funding, training, equipment, and human resources to implement fire management); and
11 (3) lack of support infrastructure (e.g., roads and other infrastructure to access rural areas prone to
12 wildfire, monitoring and early warning systems to detect and track fires).

13 According to the FAO (2007), many African countries particularly in sub-Saharan Africa have no
14 central government fire management policy, and there is a widespread lack of support infrastructure,
15 funding, equipment, and adequately-trained human resources for fire management. While most
16 countries in Central, South and Southeast Asia have a government fire policy, limited funding resources
17 restrict their ability to establish or maintain effective fire management programs (FAO, 2007). According
18 to Morgan (2009), the Association of Southeast Asian Nations instituted a “zero burning” policy in 1999,
19 but it has been largely ineffective. China, Japan, and South Korea have advanced fire detection systems,
20 including the use of remote sensing (Morgan, 2009), but often at the local level, villages and
21 communities lack resources, adequate training, and professional expertise to control large wildfires
22 (FAO, 2007). In many countries in South America, illegal burning even on state-protected lands is
23 widespread due to the absence of enforcement and criminal penalties (FAO, 2007). Russia, on the other
24 hand, has well-defined laws regulating forest burning practices, but lacks strong enforcement (FAO,
25 2007).

26 Given these challenges, addressing fundamental barriers to implementation may be just as or
27 more important than identifying and promoting more technological forms of mitigation such as specific
28 burning techniques. Capacity-building efforts may include building basic fire management
29 infrastructure, strengthening governance structures to create and enforce fire policies, and developing
30 economic alternatives to slash-and-burn agriculture. Fire prevention education for the general public
31 and training for workers in the agricultural and forestry sectors in the controlled use of fire will also be
32 important.

33 There is relatively little information regarding costs of open biomass burning mitigation
34 internationally. Mitigation costs will vary according to country, and will likely be higher in developing
35 countries due to more extensive barriers to implementation as described above. These costs will
36 depend on local environmental conditions, ecosystem type, fire management capacity, and support
37 infrastructure. Costs may also vary within individual countries, according to locality, because authority
38 and responsibility for fire management is often decentralized and is left up to local or regional
39 authorities (FAO, 2007).

1 To address the impact of open biomass burning internationally, the United States has recently
2 initiated research efforts and other international cooperative activities to evaluate and reduce BC
3 emissions from open biomass burning in and around the Arctic. The U.S. State Department is
4 coordinating a \$5 million Arctic Black Carbon Initiative that will fund a number of activities, including a
5 project by the U.S. Department of Agriculture (USDA) to address biomass burning emissions in Eurasia.
6 USDA’s multi-agency program contains the following components (USDA, 2010):

- 7 ○ Research Activities: USDA scientists (led by the U.S. Forest Service and Agricultural Research
8 Service) will seek to improve estimation of emission and transport of BC from agricultural burning
9 and forest fires by quantifying spatial and temporal patterns of these emission in Eurasia and
10 conducting an assessment of long-range transport of BC from fires in Russia and adjoining regions to
11 the Arctic. The research will identify meteorological conditions and potential source locations for
12 Arctic transport of smoke and analyze agronomic practices in Eurasia to identify opportunities for
13 reduced use of agricultural burning.

- 14 ○ Technical Exchange and Other Cooperative Activities: The U.S. Forest Service and Foreign
15 Agricultural Service will implement technical exchanges and cooperation between U.S. and Russian
16 experts on BC, agricultural burning, and fire management. These efforts will support training
17 activities and the development and implementation of innovative local-level “pilot” programs
18 designed to illustrate strategies and practices that could be more broadly applied to reduced any
19 negative environmental impacts of agricultural and forest fires. Key issues include interagency
20 cooperation on fire management, fire budgets, and GIS and remote sensing. USDA will also facilitate
21 public-private partnerships to develop local-level fire wardens and fire brigades in Russia and
22 outreach to farmers in Russia to increase awareness of approaches to reduce BC emissions from
23 agricultural burning.

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
27
28
29
30
31
32

11. Metrics for Comparing Black Carbon Impacts to Impacts of Other Climate Forcers

11.1 Summary of Key Messages

- Due in large part to the difference in lifetime between BC and CO₂, the relative weight given to BC as compared to CO₂ (or other climate forcers) is very sensitive to the formulation of the metric used to make the comparison. There is currently no single metric (e.g., GWP) that is widely accepted by the science and research community for this purpose.
- There are several metrics that have been applied to the well-mixed GHGs with respect to different types of impacts, especially the GWP (Global Warming Potential) and GTP (Global Temperature Potential). These metrics can be applied to BC, but with difficulty due to important differences between BC and GHGs. Recently, new metrics designed specifically for short-lived climate forcers like BC have been developed, including the SFP (Specific Forcing Pulse) and STRE (Surface Temperature Response per unit continuous Emission).
 - Carbon mass ratios (e.g., OC/BC ratios) provide a simpler way to help prioritize among mitigation options based on a very rough indication of potential climate impacts. However, such ratios are fairly crude and serve only as a rough guide for which sectors or emissions sources may provide the greatest opportunity for climate benefits relative to other sources.
- There is significant controversy regarding the use of metrics for direct comparisons between the long-lived gases and the short-lived particles.
 - There are a number of factors that should be considered when deciding which metric to use, or whether comparisons between BC and CO₂ are useful given the policy question. These include: the time scale (20 years, 100 years, or more), the nature of the impact (radiative forcing, temperature, or more holistic damages), the inclusion of different processes (indirect effects, snow albedo changes, co-emissions), and whether sources and impacts should be calculated regionally or globally.
- Explicit tradeoffs between BC and CO₂ may not be appropriate, depending on policy goals. BC mitigation offers an opportunity to address key climate effects (including melting, precipitation and dimming as well as radiative forcing) and to slow the rate of climate change. However, BC reductions cannot substitute for reductions in long-lived GHGs, which are essential for mitigating climate change in the long run.

11.2 Introduction to Metrics

This section introduces the concept of using metrics for comparing BC-related impacts to those of other climate forcers. It explains some of the approaches to developing metrics and provides a comparison of common metrics used for GHGs and for BC. This section concludes with a discussion of the most salient limitations associated with specific metrics and with using metrics in general.

The goal of a *metric*, as used in this report, is to quantify the impact of a pollutant relative to a common baseline. Such metrics can be used to compare between two or more climate forcers, e.g., CO₂ versus methane, or to estimate the climate effects of different emissions sources (or mitigation measures). Metrics that enable comparisons among pollutants or sources based on common denominators can also be used for the implementation of comprehensive and cost-effective policies in a decentralized manner (for example, in a market-based climate program) so that multi-pollutant emitters can compose mitigation strategies (Forster, 2007).

Climate metrics are often defined relative to a baseline pollutant (usually CO₂) and focus on a particular climate impact (such as radiative forcing or temperature) that would be altered due to a change in emissions. For example, in EPA's annual *Inventory of U.S. Greenhouse Gas Emissions and Sinks*, the "global warming potential" (GWP) metric is used to convert all GHGs into "CO₂-equivalent" units. Importantly, metrics such as GWP have been used as an exchange rate in multi-gas emissions policies and frameworks (IPCC, 2009). The key assumption when developing a metric is that two or more climate forcers are comparable or exchangeable given the policy goal, i.e., one pound of apples may be comparable to or exchangeable with one pound of oranges if the goal is not to overload a truck, but not if the goal is to make apple cider. Therefore, when used as an exchange rate in multi-pollutant emissions framework, a metric allows substitution between climate forcers which are presumed to be equivalent for the policy goals (Forster, 2007).

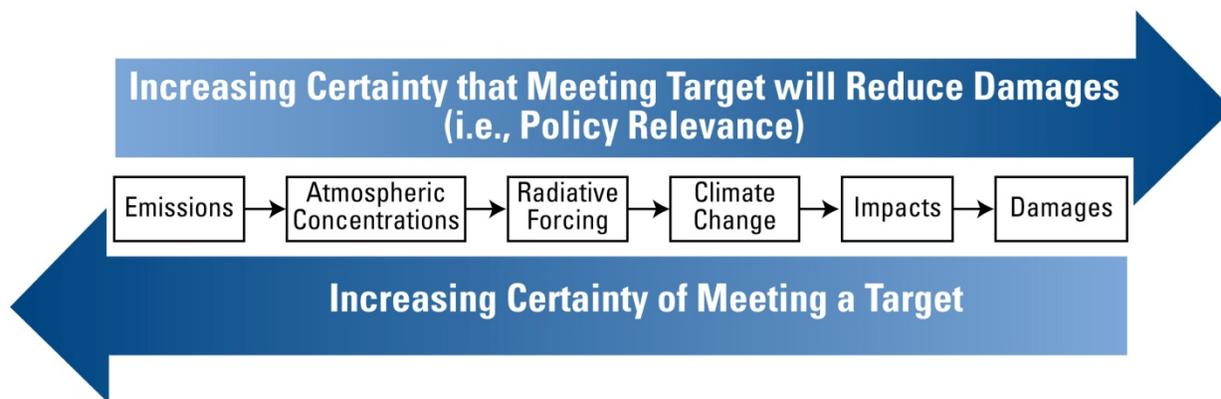
Metrics can also be used to prioritize among mitigation measures designed to control emissions of similar compounds from different sources. As described in Chapter 2, aerosols are composed of a numerous components, both warming (BC) and cooling. A metric can aggregate these effects in order to determine the relative contribution of a given source or measure. One of the key metrics in this regard is the use of carbon mass ratios such as "OC/BC". The potential utility of such ratios, as compared to other types of climate metrics for BC, is discussed further below.

11.3 Metrics along the Cause and Effect Chain

For both BC and GHGs, there is a cause and effect chain starting with anthropogenic emissions and leading to changes in concentrations, radiative forcing, physical climatic changes, and impacts on human and natural systems (Figure 11-1). Some of the links in this cause and effect chain may be simultaneous rather than sequential. For example, atmospheric loading of aerosols affects dimming and precipitation directly, without being mediated through radiative forcing. Nor is the chain always unidirectional. Climatic changes can lead to changes in atmospheric concentrations of climate-forcing pollutants (for example, changes in precipitation will change aerosol lifetimes) or even emissions of

1 those pollutants (for example, changes in temperature affects fossil fuel consumption for heating and
 2 cooling needs, which affects emissions of particles and precursors). There are uncertainties at each
 3 stage of the cause and effect chain, and these uncertainties compound over multiple steps of the chain.
 4 The uncertainties for BC are generally larger at all stages of the causal chain compared to the long lived
 5 GHGs (for reasons discussed in this and other chapters of this report).

6



7

8 **Figure 11-1. Cause-Effect Chain from Emissions to Climate Change, Impacts, And Damages** (adapted
 9 from Fuglestedt et al. 2003). The arrows indicate that a policy could focus on different elements along
 10 the causal chain, and depending on whether the policy focuses on the emissions or damages end of the
 11 chain can determine the certainty of meeting the stated policy target versus the certainty of reducing
 12 damages at issue.

13 Within the climate change field, metrics have been calculated for changes in radiative forcing,
 14 global mean temperature, and monetized damages. The closer the metric is to the emissions end of the
 15 chain, the less uncertainty there is in how to calculate the metric – it is easier to determine how a
 16 change in emissions will change concentrations than it is to determine how a change in emissions will
 17 change temperature (a calculation which requires several intermediate steps). Additionally, the further
 18 along the chain, the more physical systems (and economic systems) need to be included in order to
 19 calculate the metric. However, if a reduction in damages is considered the ultimate objective of the
 20 policy, then a metric that focuses explicitly on impacts or damages best represents that objective. Since
 21 the economic value of damages (expressed in dollars) is one of the easiest metrics for the public to
 22 understand, there has been a great deal of interest recently in calculating the monetary value of climate
 23 change impacts associated with different pollutants (see Chapter 2.7). The choice of a metric can be
 24 considered in part a choice about how to allocate uncertainty between how to calculate the metric and
 25 the representativeness of the metric for the ultimate impacts of interest.

26 Fuglestedt (2009) identified the following considerations for developing a metric for climate
 27 forcers (see Table 11-1 for examples of how commonly use metrics address these considerations):

- 28 1. What climate impact is of interest for the policy being considered?
- 29 2. What climate forcer will be used as the baseline for comparison?

- 1 3. What is the temporal frame for emissions? Is it an instantaneous pulse or a sustained change in
- 2 emissions?
- 3 4. What is the temporal frame for the impact? 10 years, 50 years, 100 years? Is the impact
- 4 considered only at the end point of the time frame, or integrated over the period?
- 5 5. Does the metric address the magnitude of change or the rate of change or both?
- 6 6. What is the spatial dimension of the metric for both emissions and impacts? Is it global or
- 7 regional?
- 8 7. What economic considerations should be taken into account? How are damages in the far future
- 9 weighed compared to damages in the near term?

10 First, the climate impact must be identified because the effectiveness of a given metric is dependent on
 11 the primary policy goal. Considerations 2 through 7 are then framed by the selected climate impact.
 12 This is important because choosing an inappropriate metric could lead to policy decisions that ultimately
 13 result in undesirable climate or economic impacts.

14 **11.4 Commonly-Used Metrics for GHGs**

15 Article 2 of the United Nation Framework Convention on Climate Change (UNFCCC) calls for a
 16 policy that addresses the magnitude and the rate of climate change as well as the cost effectiveness of
 17 controlling emissions (IPCC, 2009). Therefore, appropriate metrics could cover either the physical or
 18 economic dimensions of climate change, or both. A number of metrics have been developed and
 19 refined for application to CO₂ and other long-lived GHGs. These metrics are summarized in Table 11-1
 20 and described further below. Their potential applicability to BC is considered in the next section. Note
 21 that the last two metrics listed in the table (SFP and STRE) were developed specifically for application to
 22 short-lived climate forcers like BC, and are discussed only in section 11.5, below.

23 **Table 11-1. Examples of Commonly Used Metrics for GHGs.**

Metric Type	Climate Impact	Baseline Forcer	Emissions type	Spatial Scale	Includes rate of change?
GWP (Global Warming Potential)	Integrated radiative forcing	CO ₂	Pulse	Global	No
GTP (Global Temperature Potential)- sustained	Temperature	CO ₂	Sustained	Global	No
GTP-pulse	Temperature	CO ₂	Pulse	Global	No
Cost-effectiveness metrics (e.g. Manne & Richels, Global Cost Potential)	Mainly temperature	CO ₂ or \$ value	Optimal emissions calculation	Global	Optional
Value of Damages (e.g., Social Cost of Carbon, Global Damage Potential)	Range of climate damages	\$ value	Pulse	Global	Limited
SFP (Specific Forcing Pulse)	Energy	None	Pulse	Global or regional	No

STRE (Surface Temperature Response per unit continuous Emission)	Temperature	CO ₂	Sustained	Global	No
--	-------------	-----------------	-----------	--------	----

1 **11.4.1 Global Warming Potential**

2 To date, the most widely established and well-defined metric is the global warming potential
3 (GWP). The definition of the GWP by the IPCC, 2007 is,

4 “An index, based upon radiative properties of well-mixed *greenhouse gases*, measuring
5 the *radiative forcing* of a unit mass of a given well-mixed greenhouse gas in the present-
6 day *atmosphere* integrated over a chosen time horizon, relative to that of *carbon*
7 *dioxide*. The GWP represents the combined effect of the differing times these gases
8 remain in the atmosphere and their relative effectiveness in absorbing outgoing *thermal*
9 *infrared radiation*. The *Kyoto Protocol* is based on GWPs from pulse emissions over a
10 100-year time frame.”

11 The identified climate impact the GWP addresses is radiative forcing. The climate forcer used as
12 the baseline is CO₂ (e.g. GWP = 1). The temporal frame for emissions is a pulse. The GWP provides the
13 magnitude, but not the rate of change, of the integrated radiative forcing over a given time frame. The
14 time frame is usually 100 years, but in addition 20 year and 500 year GWPs are sometimes presented to
15 show how GWPs would differ if short-term or long-term impacts are given more weight. The GWP
16 captures the global average change in radiative forcing. Finally, the GWP, which addresses only
17 radiative forcing, a physical metric, and does not take into account any economic dimension.

18 There have been a number of criticisms of the GWP in the peer-reviewed literature (O’Neill
19 2000, Shine 2009). Despite these criticisms, at the time of the Kyoto Protocol in 1997, the GWP was
20 adopted as the metric used in climate negotiation. While acknowledging that there are shortcomings
21 involved in using GWPs even for comparisons among the long-lived gases, a recent IPCC Expert Meeting
22 on the topic found that GWPs were still a useful measure for these gases (IPCC, 2009). It remains the
23 most accepted metric due to simplicity, the small number of input parameters, the relative ease of the
24 calculation, and a lower level of uncertainty compared to some alternatives (Shine et al., 2005). The
25 GWPs as calculated by the IPCC Second Assessment Report (Schimel et al., 1996) remain the standard
26 GWPs used for the official U.S. GHG emissions inventory compiled annually by EPA.¹

27 **11.4.2 Global Temperature Potential**

28 One alternative metric that has received recent attention is the global temperature potential
29 (GTP). The GTP is also a physical metric that compares the global average temperature change at a
30 given point in time resulting from equal mass of emissions of two climate forcers (Shine et al., 2005).
31 The GTP moves one step further down the cause and effect chain and addresses a climate response to

¹ See <http://www.epa.gov/climatechange/emissions/usgginventory.html>.

1 radiative forcing, the global-mean surface temperature change. The GTP therefore includes more
2 physical processes, such as the heat exchange between the atmosphere and ocean, than the GWP. This
3 also introduces more uncertainty to the metric. In addition, while the GWP represents the integrated
4 radiative forcing of a pulse of emission over a given time period, the GTP is evaluated at a given point in
5 time (IPCC, 2009). There are two versions of the GTP: one which involves the effects of a pulse of
6 emissions, and the other involves a sustained reduction of emissions. The sustained version of the GTP
7 results in comparative values between different gases that are similar to the values calculated using
8 GWPs. The pulse version of the GTP, by contrast, leads to longer-lived gases being given more relative
9 weight because a pulse of a short-lived gas has very little impact on temperatures many years in the
10 future. Like GWPs, the GTP can be calculated over a variety of timescales, with 20, 100, and 500 being
11 the timescales most commonly presented. There are advantages and disadvantages to using either the
12 GWP or a GTP, and they may each address different policy goals and may be more relevant to different
13 climate forcers and time frames, depending upon the policy need. To date, however, the GTP has not
14 been used in any official application.

15 **11.4.3 Cost-effectiveness Metrics**

16 Manne and Richels (2001) examined relative tradeoffs between different gases that vary over
17 time and are calculated to optimally achieve a given target using a computer model that included
18 economic considerations. Similarly, the Global Cost Potential (GCP), compares the relative marginal
19 abatement costs for two climate forcers when a given climate change target is achieved at least cost
20 (IPCC, 2009). These approaches define a temperature or radiative forcing target and calculate the
21 relative (or absolute) dollar value that should be imposed on different gases in order not to exceed that
22 target.

23 **11.4.4 GHG Metrics for Measuring Economic Impacts**

24 Two metrics, the Global Damage Potential (GDP) and the social cost of a pollutant, involve
25 monetization of the damages of climate change (see detailed discussion in Chapter 2.7). The social cost
26 calculation has most commonly been used for CO₂ alone, where it is referred to as the Social Cost of
27 Carbon (SCC). Even where risks and impacts can be identified and even quantified with physical metrics,
28 it may be difficult to monetize these risks and impacts (e.g., such as ecosystem damage or the potential
29 to increase the probability of an extreme weather event) such that an accurate cost-benefit comparison
30 could be undertaken. The GDP compares the relative damage resulting from an equal mass of emissions
31 of two climate forcers (IPCC, 2009). Both the GDP and the social cost calculation depend on the physical
32 aspects of the climate system as well as the economic linkages between climate change impacts and the
33 economy (IPCC, 2009). Therefore, the GDP and the social cost require calculations of the entire cause
34 and effect chain, but as a result contain a large amount of uncertainty.

35

11.5 Applicability of Climate Metrics to Black Carbon

This section discusses the use of well-established metrics such as the GWP and GTP as they relate to BC emissions and identifies alternative metrics that may be more relevant to BC. As introduced in Chapter 2, BC influences the climate differently than the warming effects of greenhouse gases. These differences have important implications for identifying appropriate metrics to compare climate impacts (and reductions). **Table 11-2** (reprinted from Chapter 2) compares some of BC's climate attributes and effects to those of CO₂.

Table 11-2. Comparison of BC to CO₂ on the Basis of Key Properties that Influence the Climate

	BC	CO ₂
Atmospheric lifetime	Days to weeks	100+ years (some stays for millennia)
Distribution of atmospheric concentrations	Highly variable both geographically and temporally, correlating with emission sources	Generally uniform across globe
Direct radiative properties	Absorbs all wavelengths of solar radiation	Absorbs only thermal infrared radiation
Global mean radiative forcing (IPCC) (see section 2.6)	0.34±0.25 W m ⁻² direct forcing 0.1±0.1 W m ⁻² (snow/ice albedo forcing)	1.66±0.17 W m ⁻²
Cloud interactions	Multiple cloud interactions that can lead to warming or cooling, as well as effects on precipitation	No direct cloud interactions
Surface albedo effects	Contributes to accelerated melting of snow/ice and reduces reflectivity by darkening snow and ice, enhancing climate warming	No direct surface albedo effects
Contribution to current global warming	Likely 3 rd largest contributor (after CO ₂ and CH ₄), but large uncertainty	Largest contributor

As discussed in detail below, the significant differences between BC and CO₂ make applying the metrics described in the previous section difficult and, for some purposes, wholly inappropriate. One of the most essential factors to consider is that BC is most clearly related to short-term climate impacts, and is principally a regional pollutant. The lifetime of BC (weeks) is much shorter than the mixing time of the atmosphere (1 to 2 years), so the climate impacts of BC depend on where and when it is emitted. In comparison, the shortest-lived gas in the Kyoto basket has a lifetime longer than one year, and the majority of the Kyoto gases have lifetimes ranging from decades to millennia. In addition, the variations in atmospheric concentrations of BC between regions, contrasting with the well-mixed nature of most GHGs, has not been captured in most metrics to date. Thus, focusing on long-term, global average radiative forcing impacts—the frame of reference for long-lived GHGs—may lead to distorted policy

1 decisions about BC. Conversely, focusing on short-term or regional impacts may be inappropriate for
2 decisions involving long-lived GHGs.

3 There are four physical metrics that have been commonly used to compare BC to other
4 substances: GWPs, GTPs, Specific Forcing Pulses (SFPs, described below), and Surface Temperature
5 Response per Unit Continuous Emissions (STRE, also described below).

6 **11.5.1 Global Warming Potential**

7 While a GWP can be calculated for BC, there are reasons that GWPs may be less applicable due
8 to the different nature of BC in terms of various physical properties and the fact that unlike GHGs, BC is
9 not well mixed in the atmosphere. However, because GWPs are the most commonly used, and only
10 official, metric in climate policy discussions, many studies have calculated GWPs for BC. One-hundred-
11 year GWPs for BC in the literature range from 330 to 2,240: e.g., 330 to 2,240 tons of CO₂ would be
12 required to produce the same integrated radiative effect over 100 years as one ton of BC. Some of the
13 range in these estimates results from inclusion of different and uncertain indirect and snow effects, use
14 of a different CO₂ lifetime for the baseline, or recognition of the dependence of a GWP for BC on
15 emission location.

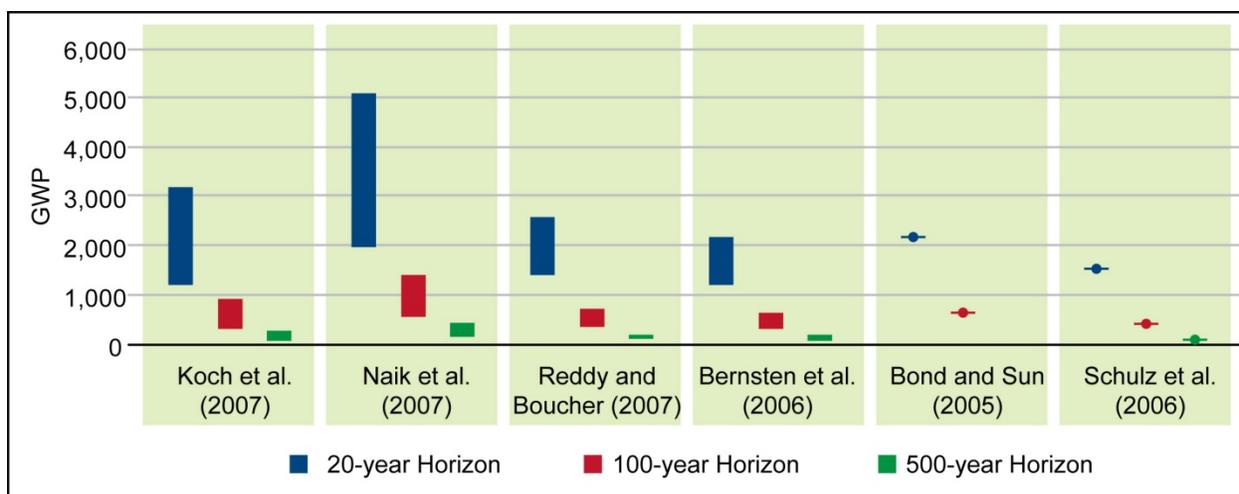
16 Using time periods other than 100 years has also been explored for GWPs. Those who are
17 concerned with near-term impacts (such as Arctic ice retreat) sometimes suggest 20-year GWPs as more
18 appropriate for short-lived forcings such as BC (CATF, 2009). Jacobson (2007) estimates a 20-year GWP
19 for BC of 4470. However, for those concerned about the long-term problems of climate change, even
20 100-year GWPs may be considered too short (Intergovernmental Panel on Climate Change, 2009a).
21 Because BC is a short-lived species, the shorter the policy relevant time horizon considered, the greater
22 the relative importance of BC compared to CO₂ (and vice versa: the longer the relevant time horizon,
23 the less important BC is compared to CO₂). If the focus is on achieving immediate climate benefits
24 within a 10-20 year time period, GWP₂₀ provides a more realistic picture of the impact of reductions in
25 different species in the near-term. On the other hand, if the concern is to identify measures that will
26 help avert climate change at a broad scale, over a longer time frame, as the problem is generally
27 conceptualized, a 20-year time horizon is insufficient and GWP₁₀₀ is a more relevant metric.

28 **11.5.2 Global Temperature Potential**

29 GTPs, as described previously, evaluate the impact on temperature at a given time. However,
30 there has been a few different approaches about how GTP is applied with respect to how the emissions
31 are reduced and how the impacts are calculated. Boucher and Reddy (2008) use a short, pulse-like (1-
32 year) reduction of emissions and find that the 100-year GTPs are about a factor of 7 smaller than the
33 corresponding GWPs. Berntsen et al. (2006) reduced BC emissions for a 20-year time span
34 (approximately the lifetime of a given investment in abatement technology) and found that the 100-year
35 GTP of BC was about 120 to 230 (i.e., reducing 120 to 230 tons of CO₂ has the same impact on
36 temperatures in 100 years as reducing 1 ton of BC).

1 Several papers have recently summarized different BC GWP and GTP estimates (Sarofim et al.,
 2 2010; CARB, 2010; and Fuglesvedt, 2010). However, of the studies surveyed by these three papers only
 3 Hansen et al. considered indirect cloud interactions of BC, which could lead to reductions of the GWP
 4 estimates because of the potential cooling effects of indirect radiative effects of BC. Only a few included
 5 estimates for metrics of co-emitted OC. If co-emissions and indirect effects are not included, then any
 6 metric will likely overestimate the globally averaged climate benefits of reducing BC.

7 Figure 11-2, based on Fuglesvedt (2010), summarizes a number of studies that attempted to develop
 8 metrics for comparing CO₂ and BC. The GWP values in the Y axis of the figure refer to the number of
 9 tons of CO₂ emissions which are calculated to be equivalent to one ton of BC emissions based on the
 10 particular metric. This figure shows how the GWP metric depends on the time horizon used (20 year to
 11 500 years). Additionally, for the first four studies, the range of values results from a dependence of the
 12 GWP on the region in which the emission occurs. The difference between the studies results from
 13 differences in the climate models used to link the emissions to the warming or temperature change.
 14 Figure 11-3 shows a similar analysis from Fuglesvedt (2010) which evaluates the equivalent GTP for
 15 these different models.



16
 17 **Figure 11-2. Ranges and Point Estimates for Regional Estimates of GWP Values for One-Year Pulse**
 18 **Emissions of Black Carbon for Different Time Horizons.** Note that the first four studies referenced
 19 evaluated GWP values for different sets of regions; Bond and Sun (2005) and Schulz et al. (2006) produced
 20 global estimates only.

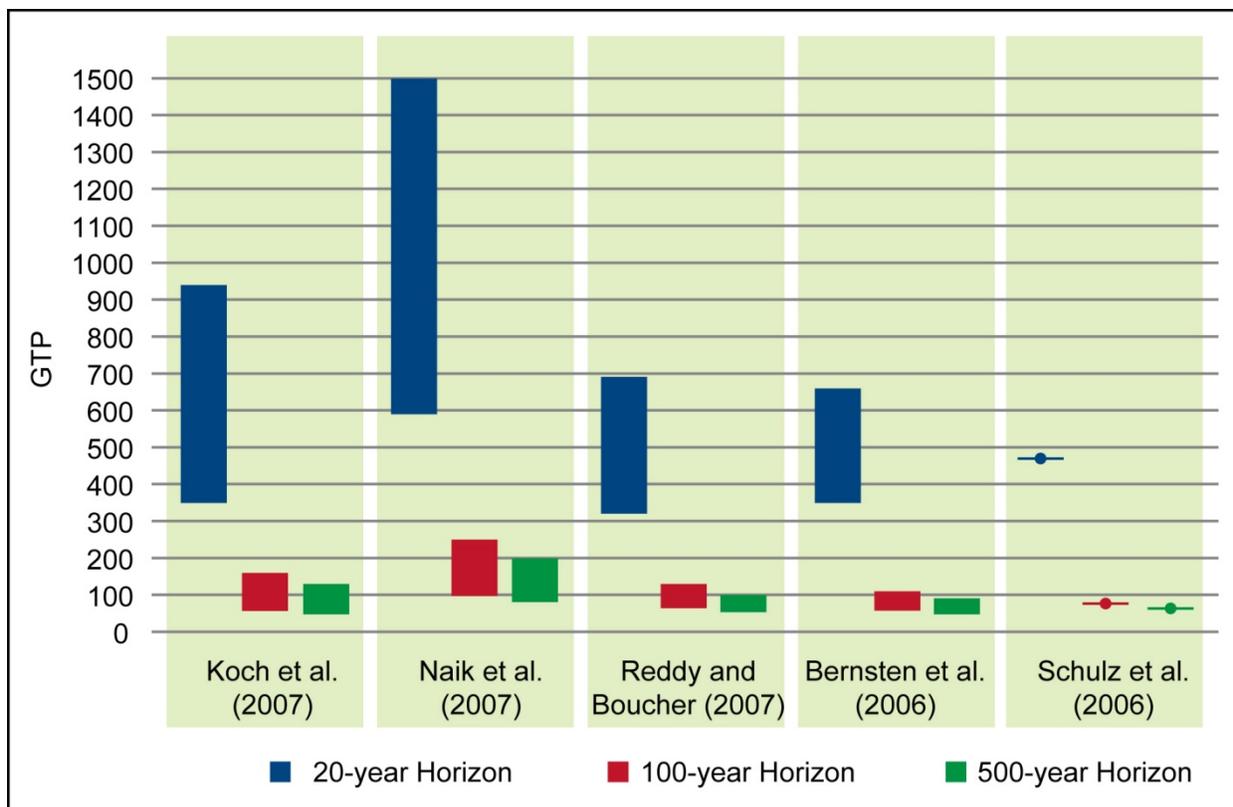


Figure 11-3. Ranges and Point Estimates for Regional Estimates of GTP Values for One-Year Pulse Emissions of Black Carbon for Different Time Horizons. Note that the five studies referenced evaluated GTP values for different sets of regions; Schulz et al. (2006) produced global estimates only.

Fuglesvedt (2010) shows that the metric for comparing BC to CO₂ can range from a ton of BC being equivalent to 48 tons of CO₂ based on a 100-year GTP (which measures the temperature change 100 years after a pulse of emissions) to 4,900 tons of CO₂ based on a 20-year GWP (which integrates the total radiative forcing impact of a pulse of emissions over the 20-year time span). The variation between GWPs or GTPs for emissions from different locations demonstrates how variability in convective properties, exposure to sunlight, and different surface albedos can cause the effect of a given unit of emissions of BC to vary and therefore for the GWP or GTP calculated for emissions from different locations to vary. Given a specific timescale, metric, and computer model, the two figures show that this dependence on emissions location can lead to changes in GWP or GTP by up to factor of three. Such dependence on emissions location for long-lived GHGs does not come into play when calculating their GWPs.

Sarofim (2010) also summarized a number of studies, and further analyzed how the GWP estimate depended on inclusion of either fossil fuel organic carbon co-emissions or snow albedo impacts. Sarofim (2010) found that inclusion of these processes can change the value of the metric by about a factor of two. Other effects that were not quantified in the paper, but that can lead to significant differences between model estimates of GWPs, are the inclusion of indirect effects on clouds,

1 or the assessment of a larger range of sectors and co-emission types. Additionally, because most
2 metrics use CO₂ as a baseline forcer, the use of different carbon cycle models can significantly influence
3 the metric values for black carbon. Some researchers may report metric values in carbon equivalents,
4 rather than CO₂ equivalents, which leads to a factor of 3.7 difference.

5 **11.5.3 Specific Forcing Pulse**

6 A new metric has been proposed by Bond et al. (2011) to quantify climate warming or cooling
7 from short-lived substances (lifetimes of less than four months): the Specific Forcing Pulse (SFP). This
8 metric is based on the amount of energy added to the Earth system by a given mass of the pollutant.
9 The rationale for developing this new metric was that short-lived substances contribute energy on
10 timescales that are short compared to time scales of mitigation efforts, and therefore can be considered
11 to be “pulses.” Bond et al. (2011) find that the SFP of the direct effect of BC is 1.03 ± 0.52 GJ/g, and with
12 the snow albedo effect included is 1.15 ± 0.53 GJ/g. They also find that the SFP for OC is -0.064 (from -
13 0.02 to -0.13) GJ/g, which leads to a conclusion that for direct forcing only, a ratio of about 15:1 for OC
14 to BC is close to climate neutral; however, this does not include cloud indirect effects or co-emissions of
15 substances other than OC. Bond et al. also find that the SFP varies by 45% depending on where the BC is
16 emitted. While the paper notes that fundamental differences in temporal and spatial scales raise
17 concerns about equating the impacts of GHGs and short-lived aerosols, they do use the SFP to calculate
18 a GWP for the direct effect of BC of 740 ± 370 , for both the direct and the snow albedo effect of BC of
19 830 ± 440 , and for organic matter of -46 (from -18 to -92).

20 **11.5.4 Surface Temperature Response per Unit Continuous Emission**

21 Another new metric, the Surface Temperature Response per unit continuous Emission (STRE)
22 has been proposed by Jacobson (2010). The STRE is similar to the sustained version of the GTP.
23 Jacobson found that the STRE (which he compares to GWPs) for BC on the 100 year time scale is 2900 to
24 4600 for BC in fossil fuel soot and 1060-2020 for BC in solid-biofuel soot. The uncertainty ranges
25 presented by Jacobson depend on his assumption that CO₂ will decay exponentially with either a 30- or
26 a 50-year lifetime. A more sophisticated carbon cycle model or the Bern carbon cycle approximation
27 from the IPPC (which is a sum of 4 exponentials rather than a single exponential as in the Jacobson
28 calculations) would have likely resulted in a lower STRE. Jacobson also presents estimates of the
29 combined BC plus OC STRE, finding that the STRE for emissions of BC+POC from fossil soot ranges from
30 1200 to 1900 and for emissions from biofuel soot the STRE ranges from 190 to 360.

31 **11.5.5 Economic Valuation Metrics**

32 Economic valuation approaches for BC that focus on valuing climate damages from a
33 comprehensive, societal standpoint are discussed in detail in Chapter 2.7. For reasons discussed in that
34 section, techniques used to value the climate damages associated with long-lived GHGs are not directly
35 transferrable to BC or other short-lived forcers. In fact, most such approaches have focused exclusively
36 on valuing the climate impacts of CO₂, and may not even be transferrable to other GHGs. Additional

1 work is needed to design approaches to valuing climate impacts of BC directly, and to incorporate those
2 approaches into metrics comparable to the social cost of carbon (SCC).

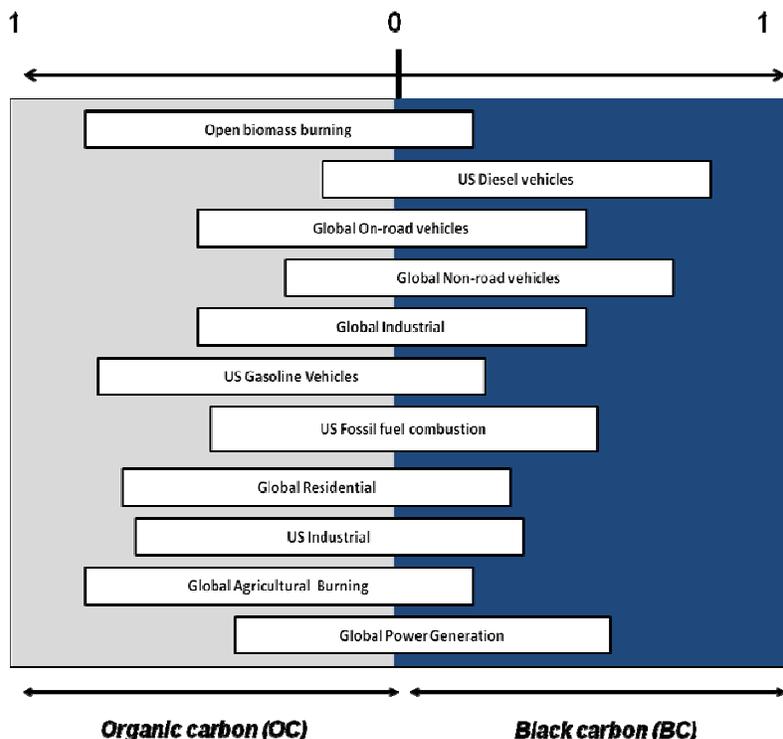
3 **11.5.6 Carbon Mass Ratios**

4 One of the key uses of metrics in a policy context is to help prioritize mitigation measures based
5 on their reductions of substances with similar characteristics. In theory, any of the metrics presented in
6 this report can be used to calculate a value for BC and for all the co-emissions for a single source to
7 determine the net value of that source or measure for mitigation. However, this type of approach
8 requires complicated modeling, and there are significant uncertainties. These uncertainties are
9 compounded by the variability of BC forcing effects depending on the region and season of emission,
10 and also the tight coupling to co-emissions of other (often cooling) chemical species (see Chapter 2 for a
11 more complete description of these effects). One simplified approach involves using carbon mass ratios
12 such as OC/BC ratios to provide a rough guide to prioritizing measures.

13 As described in Chapter 4, emissions from different categories of sources have characteristic
14 chemical profiles. The ratio between the measured mass of OC and BC in these emissions, denoted
15 OC:BC or OC/BC, is sometimes used as a metric to identify and compare sources. Other commonly used
16 ratios include OC/EC, and OM/BC, where OM represents the total mass of organic matter. Chapter 4 of
17 this report provides OC/BC and BC/PM_{2.5} ratios for a number of source categories. Figure 11–4
18 illustrates the variation in emissions profiles among sources by comparing relative OC/BC ratios. As
19 shown in the figure, emissions from diesel vehicles have more BC relative to other constituents than
20 emissions in other source categories.

21 Based on an assumption that OC will primarily reflect light and thereby induce a negative
22 radiative forcing (cooling) effect in the atmosphere, and that BC will primarily absorb light and induce a
23 positive forcing (warming) effect, the OC/BC ratio has been used to rank the net warming potential of
24 individual source categories. Emission sources with low OC/BC ratios are generally thought to have the
25 largest potential to warm the climate, though there is no agreement within the scientific literature
26 about how to interpret specific ratios.

27



1

2 **Figure 11-4. OC (left) and BC (right) Emissions from Key U.S. and Global Emissions Source Categories,**
 3 **Expressed as a Fraction of Total Carbon (OC + BC) Emissions from that Category.**

4

5 Several factors limit the value of the OC/BC ratio as a metric for estimating the climate forcing
 6 impact of a combustion source. The assumption that OC scatters solar radiation only neglects
 7 consideration of BrC and its potential warming effect. Assuming that all particle OC is scattering may
 8 underestimate the positive forcing (warming) impact of a given source. Another important limitation on
 9 the use of the OC/BC (or OC/EC) ratio for ranking combustion sources according to climate warming
 10 impact is that an emission plume will contain other climate-relevant pollutants. Sulfates, nitrates and
 11 secondary organic aerosols (SOA) or their precursors, sulfur dioxide (SO₂), NO_x, and VOCs, form
 12 additional light-scattering material within the plume. Ramana et al. (2010) note that the extent of BC-
 13 induced warming depends on the concentration of both sulfate and OC. The authors examine the
 14 potential climate benefits from controlling fossil fuel soot vs. biomass burning soot based on the ratio of
 15 BC to sulfate. Further, the aging process described in Chapter 2 induces optical changes in an emitted
 16 particle mixture, including coating of BC particles, leading to enhanced light absorption (Novakov, 2007;
 17 Lack et al., 2010). These effects are not captured by an OC/BC ratio. Finally, many analyses that employ
 18 OC/BC thresholds for “net warming effects” do not take into account other effects, such as effects on
 19 precipitation and all the indirect effects related to particle-induced changes in clouds; these are
 20 discussed in Chapter 2.

21 Despite the many limitations of OC/BC ratios, they continue to be used to prioritize
 22 carbonaceous aerosol mitigation options. For the reasons stated above, these ratios should serve only

1 as a approximate indicator of potential radiative effects of categories of emissions sources: the specific
2 circumstances or policy goals should override generic OC/BC rankings in cases where, for example,
3 emissions are affecting the Arctic, since even mixtures that contain more reflective aerosols can lead to
4 warming over such light-colored surfaces. In addition, OC/BC ratios are irrelevant to effects that are
5 shared among BC and other aerosols. This includes precipitation or dimming effects, and impacts on
6 public health. For these types of effects, mitigation strategies that reduce direct PM_{2.5} emissions or
7 overall ambient PM_{2.5} concentrations will provide the largest benefits, and the ratio of BC to other
8 constituents is far less important.

9 **11.6 Using Metrics in the Context of Climate Policy Decisions**

10 The choice of a metric depends greatly on the policy goal. No single metric can be used to
11 accurately address all the consequences of emissions of all the different climate forcers. There are
12 multiple reasons to compare BC to other short-lived and long-lived climate substances, including offsets,
13 credit trading, evaluation of net effects of a mitigation option, or illustrative analyses. However, all of
14 the differences between BC and the well-mixed gases must be considered. The appropriate metric to
15 use depends on factors such as: the time scale (20 years, 100 years, or more), the nature of the impact
16 (radiative forcing, temperature, or more holistic damages), concern over different processes (indirect
17 effects, snow albedo changes, co-emissions), and whether sources and impacts should be calculated
18 regionally or globally. It is important to note that different climate models will yield different results
19 even if the same metric definition is chosen. For example, a ton of BC has been calculated as equivalent
20 to anywhere from 48 tons of CO₂ to 4,600 tons of CO₂ in different analyses.

21 There is currently no single metric widely accepted by the research and policy community for
22 comparing BC and long-lived GHGs. In fact, some question whether and when such comparisons are
23 useful. For example, there are concerns that some such comparisons may not capture the different
24 weights placed on near term and long-term climate change. Certainly, the appropriateness of the
25 comparison depends on the policy question at hand, and the differences in lifetime, uncertainties, co-
26 emissions, modes of interaction with the climate system, and non-climatic effects such as human health
27 should be evaluated when choosing a metric. This section highlights how these differences affect the
28 metric choice.

29 **11.6.1 Incorporating Consideration of Lifetime**

30 The difference in lifetime between BC and GHGs raises concerns about prioritizing short-term vs.
31 long-term impacts, given that different metrics can place different values on short-term and long-term
32 effects. For gases with very different lifetimes, this leads to large differences in the metric values. As
33 the earlier discussion of specific metrics illustrated, there are large differences between the 20-year and
34 100-year GWP (or the sustained GTP) as well as large differences between a sustained GTP and a pulse
35 GTP. BC reductions can contribute significantly to near-term rate of change and other climate impacts,
36 but BC reductions today have much smaller effects on temperatures in 100 years. Thus, BC emissions
37 reductions that come at the expense of an increase in CO₂ emissions would result in short-term cooling

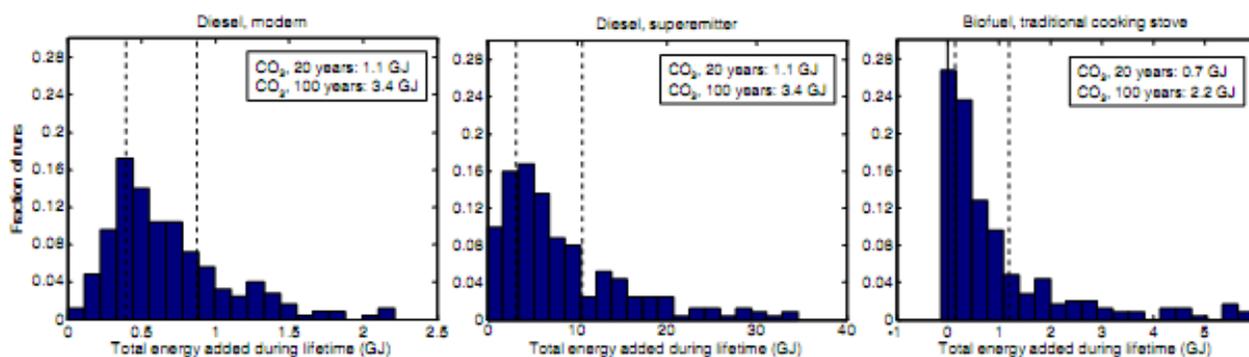
1 but an additional commitment to long-term radiative forcing due to the life time of CO₂ in the
2 atmosphere (Daniel et al., 2009).

3 The tradeoff between capturing short-term and long-term impacts is a policy question. Much
4 like the original choice of 100 years for the GWP was a policy compromise between long-term and short-
5 term impacts, policymakers need to decide whether using a GWP or GTP metric is an acceptable
6 compromise given a desire to compare BC and the long-lived GHGs. The answer partly depends on how
7 the metric is used to inform the policy decision. The NRC has warned against delaying CO₂ reductions in
8 favor of short-lived forcer mitigation, suggesting that CO₂ emissions control and control of short-term
9 forcing agents could be thought of instead as “two separate control knobs that affect entirely distinct
10 aspects of the Earth’s climate” (NRC 2010). The results of the draft UNEP assessment discussed in
11 Chapter 6 suggest that the two strategies are complementary and should be pursued simultaneously,
12 with BC reductions forming part of a larger strategy for short-lived climate forcers and in conjunction
13 with slower acting CO₂ programs. Such an approach could incorporate separate metrics for short-lived
14 and long-lived species. One metric would be appropriate for guiding global emissions of climate forcers
15 to achieve stabilization of GHG concentrations in the long-term, while another metric would focus on
16 mitigating near-term warming and could be used to guide regional emissions reductions in short-lived
17 climate forcers to reduce the impacts on regional forcing, precipitation, and ice/snow melt. The
18 complexity of climate change may also indicate that pursuing a multi-basket metric approach would best
19 capture the variety of spatial, temporal and uncertain features (IPCC, 2009).

20 While studies performed to date are limited because most do not include the full set of aerosol
21 interaction effects, co-emissions, or other uncertainties, taken as a whole studies to date suggest that
22 reductions of BC —if sustained over many decades —can serve as a complement to near-term GHG
23 reductions (Grieshop 2009 and Kopp and Mauzerall 2010). For example, Grieshop et al. (2009) used a
24 valuation that “one ton of black carbon causes about 600 times the warming of one ton of carbon
25 dioxide over a period of 100 years” in order to state that eliminating present-day emissions of BC over
26 the next 50 years would have “an approximately equivalent climate mitigation effect to removing 25 Gt
27 C from the atmosphere over the same period.” 25 Gt was chosen because it equals one “wedge” from
28 the Pacala and Socolow (2004) study that identified large-scale mitigation options over the next 50
29 years. However, this study did not do any calculations to compare the short-term and long-term effects
30 of implementing a black carbon wedge rather than an additional greenhouse gas wedge, did not
31 examine co-emissions, and did not take into account cloud interaction effects.

32 In addition, implementing aerosol mitigation measures for BC-rich sources can yield more
33 cooling over the short term (10-20 years) than eliminating CO₂ emissions from those sources (Bond
34 2007, Jacobson 2005, Sarofim 2010). Sarofim (2010), for example, addressed one specific mitigation
35 option (retrofits of some U.S. diesel vehicles) and showed that the CO₂ equivalent reductions calculated
36 by using a GWP would lead to radiative forcing reduction from black carbon mitigation peaking in the
37 year that the vehicles are retrofitted and dropping to almost zero after 20 years as the retrofitted
38 vehicles are retired. In contrast the radiative forcing reduction from the CO₂ equivalent mitigation
39 calculated using GWPs peaks about a decade after the start of the mitigation period at only a tenth of

1 the BC peak, but at the end of the century the radiative forcing reduction is still more than half of what
 2 it was at that peak. In addition, Bond (2007) examined emissions from multiple source types and
 3 compared the integrated forcing from those sources over 20 years for carbonaceous aerosols (both OC
 4 and BC) to the integrated forcing from CO₂ (an approach similar conceptually to using GWP weightings).
 5 The study showed that the aerosol emissions resulting from burning 1 kg of fuel in a super-emitting
 6 diesel vehicle has more than a 90% chance of contributing more total forcing than CO₂ from that source
 7 over a 20 year timeframe, and even for a normal (pre-2007) diesel, the aerosol emissions resulting from
 8 burning 1 kg of fuel are likely to contribute more than half as much warming as the CO₂ emissions over
 9 20 years (see Figure 11-5). This study did not account for the indirect effects of aerosols or snow albedo
 10 effects. Jacobson (2005) did include co-emissions and more cloud interactions, and still found that
 11 diesel vehicles warmed climate more than gasoline vehicles for 13-54 years, because the higher BC
 12 emissions from diesel vehicles outweighed the lower CO₂ emissions over that timeframe.



13
 14 **Figure 11-5. Integrated Forcing by Aerosols Emitted from Burning 1 Kg of Fuel from Different Sources,**
 15 based on results of 250 Monte Carlo simulations. (Note scale differences) (Bond 2007, Figure 6)

16 A different approach avoids the limitations of choosing a single metric to compare emissions of
 17 BC and CO₂, and instead investigates how reductions of BC over the entire century would change the
 18 difficulty of meeting radiative forcing targets. Kopp and Mauzerall (2010) calculated the optimal CO₂
 19 emissions pathways in order to meet a 2.21 W m⁻² target in 2100. They found that meeting this target
 20 required 50% reductions of CO₂ by about 2050. However, if this target were tightened to accommodate
 21 the positive radiative forcing from carbonaceous aerosols (both OC and BC) from contained combustion
 22 source (fossil fuels and biofuels), then the 50% reduction of CO₂ would need to occur 1 to 15 years
 23 earlier, depending on the assumptions about carbonaceous aerosol emissions pathways and forcing
 24 strength. Therefore, rather than assessing the benefits of BC reductions in the near future like the
 25 previous studies, this study assesses the radiative benefits of BC reductions at the end of the century
 26 and then translates those benefits into near term CO₂ emissions targets. This study included both co-
 27 emissions and an estimate of indirect effects.

28 11.6.2 Considering the Full Range of BC Effects

29 As discussed in Chapter 2, BC is associated with complex indirect effects and a number of
 30 hydrological effects that are distinct from the impacts of long-lived GHGs. Capturing these additional

1 effects in a single global metric is challenging; however, the current GWP metric continues to see
2 widespread use despite already not capturing the ecosystem effects of CO₂-driven ocean acidification or
3 the health and agricultural impacts of methane-induced ozone production.

4 Additionally, for most GHGs, relative radiative forcing is a reasonable approximation of
5 temperature impacts: a given W m⁻² of CO₂ has similar impacts to a W m⁻² of N₂O. By contrast, BC
6 forcing includes a combination of surface dimming and absorption of both ingoing and outgoing
7 radiation at many wavelengths, while GHGs mainly absorb outgoing thermal infrared radiation. As
8 discussed in Chapter 2, the temperature change resulting from a given W m⁻² of forcing from the snow
9 albedo effect may be much greater than the temperature change resulting from a W m⁻² of CO₂ forcing,
10 whereas the result of forcing from BC direct effects may depend on the pattern of BC loading. Inclusion
11 of the cloud effects of BC makes this metric even more uncertain.

12 Finally, the top of the atmosphere (TOA) radiative forcing changes (or even the resulting
13 temperature effects) of BC do not capture the direct (not radiative forcing mediated) effects of BC on
14 the water cycle nor effects such as surface dimming.

15 Further complicating the use of existing metrics for BC are the significant remaining
16 uncertainties in estimates of BC forcing, especially regarding the indirect cloud effects. A hypothetical
17 consequence of the globally averaged nature of common metrics is that the right mix of BC and OC
18 emissions might have no net radiative forcing impacts and yet still have significant impacts on
19 precipitation, dimming, and snow melt – as well as possibly a regional pattern of warming and cooling
20 despite a net zero effect on a global scale. This regional pattern of warming and cooling could lead to
21 equity and social justice issues additional to those present from climate change on the global scale
22 (whether due to black carbon or well-mixed GHGs).

23 This uncertainty can be compared to the uncertainty in forcing from changes in well-mixed GHG
24 concentrations of only 10% of 2.63 W/m² (Forster et al. 2007). Therefore, there is a discrepancy in the
25 magnitude of the uncertainty involved in any calculation of net effects of BC emissions compared to a
26 similar calculation for the net effect of GHG emissions.

27 **11.6.3 Considering the Impact of Co-emissions from Different Sectors**

28 Reduction measures for BC usually lead to reductions in emissions of other cooling aerosols.
29 Most BC mitigation studies, including many of the metrics-related studies discussed in this section, do
30 not account for reductions in light-scattering co-emissions and potential alterations to indirect cooling
31 effects. This contributes to uncertainty about the suitability of substituting near-term BC mitigation for
32 near-term CO₂ mitigation for the purposes of reducing the long-term warming, even when GWPs can be
33 calculated to compare the impact of emissions reductions in BC and CO₂.

34 The issue of co-emissions also exists for reductions of CO₂ and other GHGs, for example when
35 burning sulfur-rich fuels because sulfate has a cooling effect. However, the different timescales of
36 climate effects, and the ability to reduce co-emissions separately with end-of-pipe technologies, makes

1 it possible to reduce GHGs and co-emissions from a given source somewhat independently. For BC, the
2 co-emissions act on the same timescale, are often reduced by the same measures, and often have
3 comparable magnitude of climate effects.

4 **11.6.4 Inclusion of Health and Other Non-climate Effects in Metrics**

5 As discussed in Chapter 3, BC also has important non-climate effects, including significant direct
6 impacts on human health, as well as visibility impairment and other welfare effects. These effects are
7 already addressed by current air quality legislation, specifically regulations promulgated by EPA under
8 the Clean Air Act, but there may still be co-benefit opportunities. The recent draft UNEP report (UNEP
9 draft Summary for Decision Makers, 2011) addresses the simultaneous direct health and climate
10 benefits of a possible set of BC reduction measures. Radiative forcing or temperature-based metrics will
11 not capture these effects, but their existence should be included in determining broader policy designs.

12. Conclusions and Research Recommendations

This report serves primarily as a review and synthesis of available scientific and technical information on BC. This information includes published studies, emissions inventories, and observational data regarding ambient concentrations and source-specific BC emissions. Taken together, this evidence suggests that continued reductions in BC emissions can provide important benefits for both climate and public health. Mitigation of BC thus offers a clear opportunity: carefully designed programs that consider the full air pollution mixture (including BC, OC, and other co-pollutants) can slow near-term climate change while simultaneously achieving lasting public health benefits. Furthermore, currently available control technologies and mitigation approaches have already been shown to be effective in reducing BC emissions, often at quite reasonable costs. These mitigation approaches could be utilized to achieve further BC reductions. In the U.S. and Europe, additional mitigation of BC emissions is already expected to occur over the coming decades as existing regulations are implemented. These same approaches could help reduce emissions in developing countries, although some source categories, such as improved stoves for residential heating and cooking, will require tailored solutions designed to address specific needs and challenges.

12.1 Conclusions

The detailed discussions presented in earlier chapters of this report lead to several major conclusions. First, ***BC and other light-absorbing particles exert a powerful influence over the earth's climate, especially at the regional scale.*** There is no question that BC is a powerful absorber: it can absorb a million times more energy than CO₂ per unit mass. Due to this strong absorption potential, existing studies suggest that BC is the second or third largest contributor to warming after CO₂ and possibly methane. Regionally, the effects of BC on warming and melting are especially strong in sensitive regions such as the Arctic and the Himalayas. BC contributes to warming by directly absorbing incoming and outgoing radiation, and also darkens snow and ice, which reduces the reflection of light back to space and accelerates melting. The indirect effects of BC through interactions with clouds are more uncertain; the net impact of these indirect effects is likely cooling, which partially offsets the direct and snow/ice albedo effects. BC has also been linked to surface dimming and changes in precipitation patterns, impacts which have large environmental consequences.

Second, ***BC is different from long-lived GHGs like CO₂ both in the variety of mechanisms by which it affects climate and its short atmospheric lifetime.*** In addition to causing warming, BC directly impacts snow and ice, surface dimming, and precipitation patterns, and it has been associated with Atmospheric Brown Clouds (ABCs), the disruption of the monsoon, and glacial retreat. The GHGs like CO₂ that act primarily by trapping outgoing infrared radiation from the earth's surface do not have as wide a range of direct effects on climate as BC. Furthermore, BC is short-lived in the atmosphere and is primarily a regional pollutant, unlike the globally well-mixed GHGs. BC's short atmospheric lifetime of

1 days to weeks, combined with its strong warming potential, means that its radiative effects are nearly
2 immediate. Furthermore, these impacts are geographically and temporally concentrated. This has
3 several implications: first, reductions in BC emissions have the potential to produce climate benefits
4 immediately; second, globally averaged metrics (such as global radiative forcing) are not well suited to
5 evaluating the effects of BC; and third, timing and location will be critically important in designing
6 mitigation strategies.

7 Third, while there remains uncertainty about costs of BC mitigation and the impact of co-
8 emitted pollutants, ***mitigating BC can make a difference in the short term for climate, at least in***
9 ***sensitive regions***. Appropriate mitigation strategies will involve targeted local/regional actions that vary
10 between locations. Sources that affect sensitive regions (i.e., emissions reaching ice and snow-covered
11 regions, the Arctic, and the Himalayas) and sources that are rich in BC (e.g., diesel vehicles) offer the
12 clearest opportunities for climate benefits. However, even BC sources that have high emissions of co-
13 pollutants such as OC and sulfate (themselves considered cooling agents because they reflect sunlight)
14 may be appropriate mitigation targets regionally. Pursuing these reductions offers the possibility of
15 slowing the rate of climate change, and reducing its overall magnitude. While BC mitigation cannot
16 substitute for long-lived GHG reductions, which are essential for mitigating climate change in the long
17 run, BC reductions may be an effective climate mitigation strategy in the near term.

18 Fourth, ***BC mitigation strategies are likely to provide substantial public health and***
19 ***environmental benefits***. The potential public health and non-climate environmental benefits (e.g.,
20 visibility, deposition) associated with mitigation of BC and other co-pollutants are very significant, both
21 domestically and globally. Due to the extensive literature on health and welfare effects of PM generally,
22 there is relative certainty that BC reductions will provide substantial public benefits. In many cases, these
23 benefits are likely to exceed the costs of mitigation. This helps reduce uncertainty about whether to
24 pursue BC reductions, since the public health benefits alone may justify pursuing mitigation measures in
25 a number of sectors. While it is not yet possible to determine precisely the net global climate impact of
26 investments in certain mitigation strategies, such as cookstove replacement programs, the benefits of
27 these programs for public health outweigh any remaining uncertainties. Considered as local or regional
28 strategies, these investments are likely to be win-win for climate and public health.

29 Fifth, ***careful targeting of mitigation programs is essential for both public health and climate***
30 ***purposes***. Both source mix and location must be considered in designing mitigation strategies. Since
31 the effects of BC reductions are likely to be concentrated regionally, the benefits of mitigation depend
32 on how specific populations and environments are affected. The emissions reductions most beneficial
33 for climate will not necessarily coincide with the reductions that have the largest public health impact.
34 The largest public health benefits from BC-focused control strategies will occur locally near the
35 emissions source and where exposure affects a large population, while the climate benefits will depend
36 more heavily on the extent to which those emissions are reaching climate-sensitive areas such as alpine
37 regions. Thus, “generic” BC reduction strategies that are not aimed at specific sources in specific
38 locations will not be as effective for either climate or public health purposes as strategies that focus on
39 the emissions affecting either populations or sensitive climate endpoints most directly. While there may

1 be considerable overlap in emissions reductions strategies that are co-beneficial for climate and public
2 health, it is important to consider this explicitly in evaluating different strategies and control options.

3 Sixth, ***the sequence of policies is important for ensuring maximum benefits***. This is true in a
4 couple of ways. In some cases, application of BC control technologies or substitution of low emissions
5 alternatives depends on other variables falling into place first. For example, DPFs on mobile sources will
6 only function properly with low-sulfur fuels; thus, programs to ensure the widespread availability of low-
7 sulfur fuel must precede new-engine emissions standards or DPF retrofits. Similarly, clean cookstove
8 programs relying on processed fuels will only be effective if they ensure reliable fuel supply and account
9 for the ongoing costs associated with those fuels. Thus, in many cases BC mitigation efforts depend on
10 bundled policies which need to be thought about in concert. Also, sequencing of BC reduction efforts
11 involves making choices regarding which sources should be highest priority for mitigation. All sources
12 emitting BC emit a mixture of pollutants; emissions from some sources are more BC-rich than others.
13 Emissions from mobile diesel engines, for example, are typically about 75% BC. These engines also tend
14 to be concentrated where people live, i.e. in urban areas, so that prioritizing them from a mitigation
15 standpoint ensures both significant public health benefits and also a greater likelihood of climate
16 benefit. The composition of the emissions mixture should be considered, along with the location of
17 emissions and the cost of controls, in identifying top-tier mitigation options.

18 Finally, ***there is a strong need for additional quantitative analysis examining the climate,***
19 ***public health, and environmental impacts of specific control strategies***. Designing effective programs
20 for BC mitigation requires a more refined understanding of impacts on specific endpoints and in specific
21 regions. This is especially true for climate: since BC mitigation strategies largely focus on PM_{2.5}
22 reductions and may affect other components of the emissions mixture as well (e.g. methane, CO₂), it is
23 critical to understand the net impact of the strategy, including co-pollutant reductions. The *climate*
24 impacts of PM_{2.5} control programs implemented to date are not well understood. Re-orienting those
25 programs to ensure maximum reductions in BC and other light-absorbing PM would require thorough
26 analysis of the potential climate benefits of such strategies, as well as any public health implications. In
27 particular, a lot more methodological support is needed to develop approaches for quantifying climate
28 benefits at the regional scale.

29 **12.2 High Priority Research Needs**

30 There are a number of high priority research topics that could help advance efforts to control BC
31 emissions and reduce key remaining uncertainties. Based on the scientific and technical information
32 reviewed for this report, EPA concludes that priority should be given to research in the following areas:

33 **1. *Standardized definitions and improved instrumentation and measurement techniques for light-*** 34 ***absorbing PM, coupled with expanded observations.***

35 In order to accurately assess the impacts of BC (and co-pollutant) emissions, it is essential to
36 have a clear understanding of the optical properties of atmospheric aerosols and be able to trace those
37 to emissions from specific sources. Precise and consistent definitions and measurements of BC and
38 other carbonaceous aerosols are needed to ensure accurate assessment of BC emissions, climate and

1 public health impacts, and mitigation options. Additional research is needed to improve
2 instrumentation and measurement techniques to quantify accurately the light-absorption properties of
3 BC, BrC, and other aerosols; to utilize harmonized measurement and reference methods to standardize
4 definitions of BC, BrC and other compounds; and to use refined measurement techniques to collect
5 more data on light-absorption capacity of emissions from specific sources. Additional representative
6 source measurements to better characterize BC emissions by source category, fuel type and combustion
7 conditions can help improve emissions inventories and reduce modeling uncertainties.

8 It is equally important to expand the observational record for BC, including observations of
9 atmospheric concentrations of BC and BC deposition. Existing measurements of BC are sparse in both
10 spatial and temporal coverage, even in countries with more advanced monitoring programs such as the
11 United States. An expanded observational record based on standardized measurement techniques and
12 instruments could provide important information about BC transport, vertical distribution, atmospheric
13 interactions, and deposition. Such data could be used to inform climate models and verify impacts.

14 **2. *Continued investigation of basic microphysical and atmospheric processes affecting BC and other***
15 ***aerosol species to facilitate improvements in modeling and monitoring of BC.***

16 Many of the basic microphysical and atmospheric processes that BC and other aerosol species
17 undergo are not very well understood. This includes the mixing of BC with other aerosol species, the
18 atmospheric aging of BC and how aging affects BC's climatic and health impacts, and interactions with
19 cloud droplets and the hydrologic cycle in general. Incomplete understanding of these basic properties
20 limits the scientific community's ability to model BC in the atmosphere and estimate its impacts.
21 Improvements in our understanding of these basic properties through controlled experiments and
22 atmospheric observations could improve climate models, and could also inform ongoing efforts to
23 investigate the health effects of PM constituents, including BC.

24 **3. *Improving global, regional, and domestic emissions inventories with more laboratory and field***
25 ***data on activity levels, operating conditions, and technological configurations, coupled with better***
26 ***estimation techniques for current and future emissions.***

27 Given the diversity and ubiquity of sources of BC, accurately measuring and tracking emissions
28 of BC and its co-pollutants from specific sources is a very difficult undertaking. Emissions inventories in
29 the United States and other developed countries account for most source categories, but considerable
30 uncertainty remains, especially regarding emissions from nonroad mobile sources (aircraft, locomotives,
31 ocean-going vessels); newer technology on-road diesel/gasoline vehicles; high-emitting vehicles; and
32 vehicles operating at low temperatures. In addition, emissions from key industrial sources, flaring, and
33 residential heating remain poorly characterized.

34 Uncertainties are larger for BC inventories in developing countries and globally. For these
35 inventories, priorities include better characterization of emissions from residential cookstoves, in-use
36 mobile sources, small fires, smaller industrial sources such as brick kilns, and flaring emissions. For
37 sources such as cookstoves, improved characterization depends critically on field-based measurements
38 of emissions from in-use sources. In addition, usage patterns need to be reviewed to ensure that
39 appropriate "activity" levels are applied to emission factors to arrive at final emissions estimates.

1 Finally, fuller incorporation of regional inventories into global inventories could improve country- and
2 region- specific emissions estimates.

3 Quantifying and reducing the uncertainties in global, regional, and domestic emissions
4 inventories requires collecting source-specific emissions data from lab or field-based measurements and
5 gathering information on activity levels, operating conditions, and technological configurations. For
6 hard-to reach areas, improvements in estimation techniques could significantly improve global and
7 regional inventories. Systematic collection and sharing of emissions data and meta-data is important
8 for both scientific and policy purposes.

9 **4. Focused investigations of the role of brown carbon (BrC).**

10 The role of BrC is important for determining the potential climate benefit of mitigating sources
11 with high OC emissions. Several aspects of the BrC issue warrant research. First, multi-wavelength
12 measurements are needed to separate and characterize BrC and BC. Reporting column data by
13 wavelength may aid model-observation comparison as BC and BrC differ in terms of peak absorption.
14 Second, BrC should be incorporated into climate models, and the impact of BrC on net forcing estimated
15 independently from the impact of BC. Atmospheric observations of BrC and experimental
16 methodologies for determining BrC emissions are also needed, in conjunction with the improvements in
17 BrC measurements described above.

18 **5. More detailed analysis of the climate and health benefits of controlling BC from sources of specific**
19 **types or in specific locations.**

20 The ability to link specific BC sources in specific regions to climate and human health impacts,
21 all the way through the causal chain, needs improvement. Improved characterization of BC control
22 strategies and their net impact on radiative forcing, as influenced by location, will help ensure maximum
23 climate benefits. This depends in large part on improved emissions characterization and measurement,
24 as described above, but also more refined modeling techniques capable of evaluating regional or local
25 scale impacts. Greater attention should be paid to the location of the proposed change in emissions,
26 especially for near-Arctic or near-Himalayan emissions. Similarly, the links between non-radiative
27 impacts of BC, such as changes in rain, snow, and water resources, and specific source classes or regions
28 have not yet been well established; the ability to relate non-radiative effects to aerosol (and precursor)
29 emissions needs further development.

30 Health impacts will also vary by source type and location. Further analysis is needed to help
31 identify emissions reductions with maximum co-benefits for public health and climate. The ongoing
32 research on the health impacts of components of particulate matter, such as BC, is also very important
33 for understanding what specific parts of the ambient PM mix are responsible for certain health
34 endpoints.

35 **6. Refinement of climate metrics specific to BC and other short-lived climate forcers.**

36 Some of the fundamental assumptions that go into the calculation of policy-relevant metrics for
37 long-lived GHGs do not apply to BC; therefore, it is difficult to apply metrics developed for GHGs to BC
38 and other short-lived forcers. Though “alternative” metrics have been proposed for BC, none is yet

1 standardly utilized. Appropriately tailored metrics for BC are needed in order to quantify and
2 communicate BC's impacts and properly characterize the costs and benefits of BC mitigation. Improved
3 metrics could incorporate non-radiative impacts of BC, such as impacts on precipitation and public
4 health. Developing methods to quantify the benefits of BC mitigation on both climate and health would
5 encourage policy decisions that factor in climate and health considerations simultaneously, within a
6 unified framework. Analysis is also needed to examine how utilizing alternative metrics would affect
7 policy priorities and preferred mitigation options.

8 **7. Analysis of Key Uncertainties**

9 Systematic analysis of key remaining uncertainties and technical gaps regarding BC could help
10 prioritize future research and investment by clarifying which of these factors exert the largest influence
11 on modeled outcomes. Such analysis would involve both: (1) Model intercomparison of BC radiative
12 forcing and climate impacts between global and regional models, along with comparisons to ambient
13 measurements (including remote sensing and tracer-based analyses); and (2) Sensitivity analysis of the
14 factors influencing models' representation of (a) the net effect of a given mitigation measure,
15 considering all co-emitted pollutants; and (b) the overall global and regional contribution of BC and OC
16 to radiative forcing and temperature change.

17 Global and regional models give a different range of predictions of BC's RF and climate impacts
18 due to different model configurations, parameterizations and assumptions (model resolution, chemical
19 and physical processes, aging/mixing, deposition, etc.). A comparison of model results and diagnostic
20 analysis will reduce remaining uncertainties regarding BC impacts. Additional sensitivity analysis should
21 consider the importance of: (a) emissions inventories utilized (including type and magnitude of co-
22 emissions represented); (b) model representation of transport and vertical distribution of emissions,
23 aging and mixing of particles, radiative properties of particles, and particle interactions with clouds and
24 snow; and (c) the use of observational data to constrain model results and emissions estimates.

1 **Appendix 1. Ambient and Emissions Measurement** 2 **of Black Carbon**

3 **A1.1 Introduction**

4 Measurements of black carbon and other PM constituents are critical to understanding the
5 climate impacts of these substances, as well as evaluating human health and environmental effects.
6 These measurements serve as important inputs to air quality forecasting and climate models, source
7 apportionment models, and emissions inventories. Deposition measurements are also needed to judge
8 impacts on snow and ice.

9 Observational data for BC comes from two main sources: ambient measurements and source-
10 based emissions measurements. These measurements involve both sample collection and sample
11 analysis procedures, with each step having important impacts on reported measurements. Most
12 estimates of BC are based on thermal-optical and filter-based optical techniques, which classify the
13 measured quantity as apparent elemental carbon (ECa) and apparent black carbon (BCa). While the
14 terms “black carbon” and “elemental carbon” are frequently used as labels for quantities produced, the
15 addition of the term “apparent” clarifies that these are considered to be estimates of BC concentrations.
16 The appendix describes the most common sample analysis methods (thermal-optical and optical), the
17 types of instruments that can be used for these methods, and key limitations in current measurement
18 methods, approaches, and instruments. This appendix also describes the key sources of ECa and BCa
19 measurement data in the United States, in terms of the types of ambient data collected and the
20 information gathered from testing of both stationary and mobile sources. Next, the appendix describes
21 key applications of source-testing data, particularly for constructing U.S. emissions inventories. Data
22 from other countries is reported where available and applicable.

23 **A1.2 Ambient Black Carbon Measurements**

24 Black Carbon mass concentration estimates are routinely measured at ground-level in the
25 ambient air or in deposited materials, but can also be taken in aircraft and on remote sampling
26 platforms. Globally, a significant amount of ambient data has been compiled from the following types
27 of measurements:

- 28 • *Ground-based ambient air measurements* are taken in near real time using field analyzers or
29 obtained in a laboratory following collection of PM onto a filter. This is by far the largest source
30 of observational data on BCa and ECa. Details on some of the key ambient air monitoring
31 networks producing these data are described in Table A1-2 (Global Monitoring Activities) at the
32 end of this appendix.

- 1 • *Ice core measurements* of BCa and ECa have been conducted in glaciers around the world,
2 providing a historical record of BC concentrations.
- 3 • *Surface snow measurements* have been conducted to quantify recent BC in snow based on BCa
4 and ECa concentrations in locations around the world. Snow data is much more limited in
5 spatial and temporal coverage in comparison to ambient monitoring.

6 The concentration of carbon in PM is regularly measured using methods based on the chemical,
7 physical, and light absorption properties of the particles. The chemical and physical properties of
8 carbonaceous PM vary in terms of both refractivity (the inertness of the carbon at high temperatures)
9 and light absorption. Each carbon measurement technique provides unique information about these
10 properties. All current analysis methods are operationally-defined, meaning that there is no universally
11 accepted standard measurement. When developing these methods and operational criteria, some
12 scientists use its optical properties or light-absorbing characteristics (optical or light absorption
13 methods), some use its thermal and chemical stability (thermal-optical methods), while others use its
14 morphology or microstructure or nanostructure (microscopy methods). One major class of methods,
15 thermal or thermal-optical techniques, distinguishes refractory and non-refractory carbon as ECa and
16 OCa, respectively (Figure A1.1). The second major class of methods, optical methods, quantifies the light
17 absorbing component of particles as BCa, which can be used to estimate BC concentrations and can also
18 indicate the existence of components that absorb in the near-UV (i.e., brown carbon, BrC). There is a
19 well known lack of consensus and standardization regarding the operation criteria used, calibration
20 materials used, and defining characteristics or properties of the BC measured. The methods used to
21 measure ECa and BCa require standardization and re-evaluation for climate and regulatory uses.

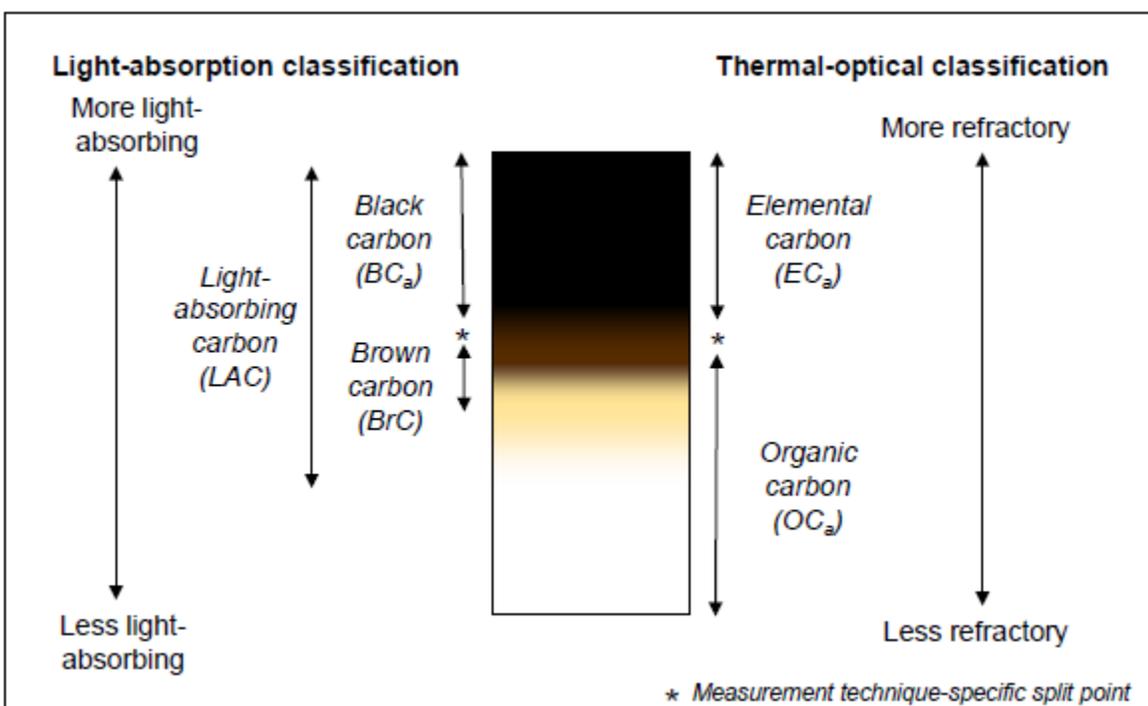


Figure A1-1. Measurement of the Carbonaceous Components of Particles. Black carbon and other types of light-absorbing materials can be characterized by measuring their specific light-absorbing properties, as seen on the left side of the figure (BCa/BrC/LAC). This contrasts with other approaches to characterizing particles based on measurements of the refractory nature of the material (inertness at high temperatures), as seen on the right-hand side of the figure (ECa and OCa).

A1.2.1 Thermal-Optical Methods, EC_a

As noted in Chapter 1, thermal-optical methods are by far the most commonly used. Since 1982, thermal-optical analysis methods have been applied to measure the EC_a and OC_a component of ambient and source aerosols (Huntzicker et. al. 1982; NIOSH 1999; Birch and Cary, 1996; Chow et. al., 1993; Chow et. al., 2007; Peterson and Richards 2002). PM collected on a filter is heated to isolate the refractory and non-refractory carbon. Laser correction measurements help prevent charred organic materials from being misinterpreted as EC_a. Thermal optical-reflectance (TOR) methods use reflectance for char correction and separation of EC_a from OC_a, while thermal-optical transmittance (TOT) uses transmittance. Long-standing reliance on these methods—which measure EC_a, rather than BC—has resulted in an extensive observational record based on EC_a and OC_a splits, and the frequent substitution of EC_a data for BC_a data, since availability of the latter is limited. In addition to laboratory-based thermal-optical methods for EC_a, semi-continuous or near real-term thermal-optical methods for EC_a and OC_a are commercially available. The semi-continuous analyzer provides hourly in-field measurements of EC_a and OC_a. This semi-continuous analyzer also provides a measure of light absorbing or optical BC_a.

1 **A1.2.2 Light-Absorption Measurements, BCa**

2 Currently, light-absorption or “optical” measurements of BCa are not consistently deployed in
3 routine monitoring programs in the United States. The one program area in which light-absorption
4 methods are used is in assessing visibility impairment in national parks and wilderness areas via the
5 Interagency Monitoring of Protected Visual Environments (IMPROVE) program. To date optical methods
6 have not been widely used in urban monitoring networks. However, such approaches are commercially
7 available and could be more widely deployed. These approaches fall into two general categories –
8 optically-absorbing and incandescent (thermal emission of light) measurement. Relative to the
9 incandescence techniques, optical techniques for BCa are in wider use. A listing of a variety of
10 commercially-available instruments used for monitoring ambient or source concentrations of BCa and
11 the wavelength selected for measurement is provided in the Table A1-2 below.

12 Modern light-absorbing techniques rely on passing a laser beam at a specific wavelength
13 through a particle sample, either in an air volume or deposited onto a filter, and observing how much
14 light is absorbed by the particles. BCa is typically measured over the green to infrared wavelengths,
15 where it absorbs more strongly than other LAC. BrC may also absorb light at shorter wavelengths (near-
16 UV and UV). Many BCa instruments can measure at multiple wavelengths, sometimes simultaneously
17 depending on the exact instrument configuration. This provides information about components that
18 absorb light over different parts of the UV/Visible spectrum. Thus, these instruments may be used to
19 distinguish between BCa and BrC; however, in many cases researchers have not been careful to
20 distinguish how much of the measured light-absorbing carbon falls into each category. In order to
21 convert light absorption to a BC mass concentration, a mass absorption coefficient or similar conversion
22 factor is used. The conversion factor is based on experiments that simultaneously measure light
23 absorption at a specific wavelength and BC mass (either as ECa from ambient measurements or particle
24 mass from soot generation experiments).

25 Incandescence is the second approach used to quantify BCa. Laser induced incandescence (LII)
26 subjects particles in an air stream to a high-intensity laser in the infrared. Some LII techniques can
27 measure individual particles, providing data on particle size, BCa mass concentration (based on an
28 assumed BC density), and an indication of the mixing state of the particles. LII is currently used in limited
29 research applications in the United States.

1

Table A1-2. Examples of Commercially-Available Optical BC_a Measurement Techniques*

Instrument (Manufacturer)	Real-time (R) or Off-line (O)	Filter (F) or air stream (A)	Wavelength to tie into above spectrum and Ch 1 properties (e.g., UV, blue, etc)
Aethalometer (Magee Scientific)	R	F	370 nm, 880 nm standard 370, 470, 520, 590, 660, 880 and 950 optional
Particle Soot Absorption Photometer (Radiance Research)	R	F	467, 530 and 660 nm
Multi-Angle Absorption Photometer (Thermo Scientific)	R	F	670 nm
Transmissometer (Magee Scientific)	O	F	370 nm and 880 nm
Densitometer (Tobias Associates Inc.)	O	F	400 – 650 nm; peak at 575 nm
Smoke Stain Reflectometer (Diffusion Systems, Ltd.)	O	F	Monochromatic light; wavelength not specified
Hybrid Integrating Plate/Sphere	O	F	633 nm
Photoacoustic soot spectrometer (Droplet Measurement Technologies, Desert Research Institute)	R	A	405, 532, 781 nm
Single particle soot photometer (Artium Technologies, Droplet Measurement Technologies)	R	A	1064 nm
Semi-continuous Field Analyzer (Sunset Labs)	R	F	632 nm
Photoacoustic Micro Soot Sensor (AVL)	R	A	808 nm

2 * The use of commercial trade names of vendor names does not constitute an endorsement by the U.S. EPA.

3

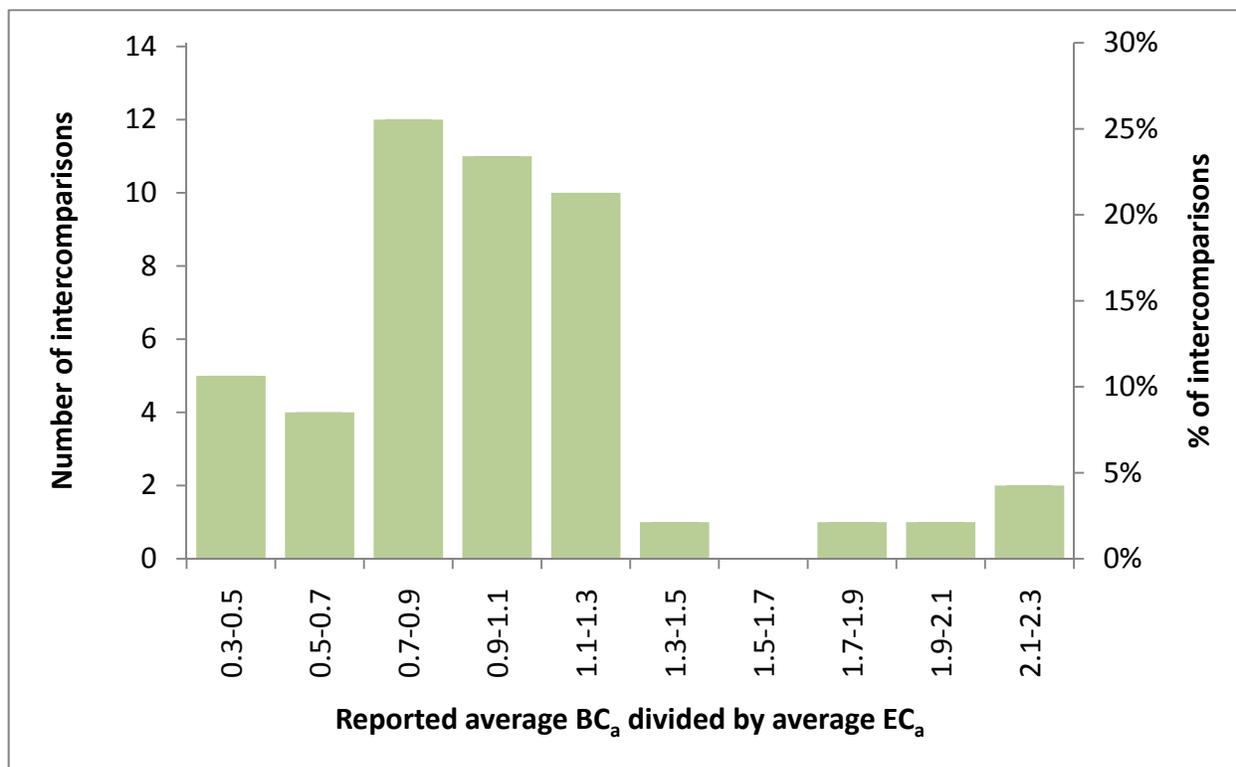
4

1 **A1.2.3 Inter-comparisons among Optical BCa and Thermal Optical ECa Measurements**

2 Given that ECa concentrations are commonly used to represent BCa, and vice versa, the
3 relationship between BCa and ECa is important to characterize. It should be noted that the two
4 measurements are not always entirely independent, as the selected conversion factor to estimate BCa is
5 sometimes based on experiments establishing a relationship between light absorption and ECa. A
6 number of inter-comparison studies have examined several different BCa or ECa measurement
7 approaches simultaneously to evaluate how well they agreed. Recent studies, published in the year
8 2000 or later, that compare ambient BCa and ECa measurements were reviewed (Chow et. al., 2009; Bae
9 et. al., 2007; Jeong et. al. 2004; Hagler et. al., 2007; Hitznerberger et. al., 2006; Snyder et. al., 2007;
10 Sharma et. al., 2002; Venkatachari et. al., 2006; Sahu et. al., 2009; Yang et. al., 2006; Miyazaki et. al.,
11 2008; Babich et. al. 2000; Ram et. al. 2010; Husain et. al., 2007; Lim et. al., 2004). In a wide variety of
12 environments, ranging from the remote Arctic to urban cities, BCa and ECa measurements were
13 reported to have consistently high correlation (average $R = 0.86 \pm 0.11$). In addition, Figure A1–2
14 shows that ratios of BCa/ECa are typically near 1 ($BCa/ECa = 0.7-1.3$ for 70% of studies), however there
15 do exist studies reporting very low BCa/ECa ratios (~ 0.5) and very high BCa/ECa ratios (~ 2).

16 The ratio of BCa to ECa and the consistency of the relationship may depend on the aerosol
17 mixture and/or the specific method used. The difference in BCa and ECa concentration may also be
18 largely influenced by the conversion factors used to change light absorption into mass concentrations
19 for optical methods as well as corrections for measurement artifacts. The differences between BCa and
20 ECa may also be due to a lack of consistency in the post-processing of the raw measurements among
21 studies (Virkkula et. al., 2007; Chow et. al., 2009). It should be noted that these inter-comparison data
22 are based on ambient measurements and similar data are needed for source measurements.

23 Table A1-3 also provides information on comparisons of BCa/BCa and ECa/ECa. The median
24 ratio for concurrent measurements of ECa is 1.4 and the median ratio of the agreement among
25 measurements of BCa was 2.7.



1
 2 **Figure A1-2. Reported BC_a/EC_a Ratios for a Pair of Measurement Techniques Reported in Ambient Field**
 3 **Studies.**

4 Instruments measuring light absorption are often capable of measuring light absorption at
 5 additional wavelengths in the near-UV or UV, which may help indicate the presence of BrC. Such
 6 approaches are currently used to attempt separation of the influence of wood smoke aerosols, which
 7 tend to be rich in BrC, from those that are dominated by diesel emissions and other fossil fuel
 8 combustion, which are rich in BC.

9 The disagreement among BC_a measurements may be due in part to differing instrument
 10 sensitivities and responses to other PM components (Slowik et. al., 2007), filter-loading artifacts, or the
 11 use of an incorrect light-absorption-to-BC-mass concentration conversion factor for studies reporting
 12 BC_a in terms of their mass concentrations. Chow et al. (2009) found that applying post-processing
 13 algorithms greatly improved the agreement among different filter-based BC_a techniques. EPA and other
 14 researchers are similarly assessing whether post-processing algorithms and site-specific conversion
 15 factors may also be beneficial to better understand the differences among BC_a and EC_a measurements.
 16 In addition to issues with the various BC_a and EC_a measurement techniques mentioned above, BC_a -to-
 17 EC_a ratios are likely to be affected by the presence of other light-absorbing species (e.g., BrC and dust).
 18 The specific inter-comparison circumstances (e.g., location, season, sample collection and analysis

1 procedures, optical wavelength, data corrections, and aerosol mixture¹) may be important to
2 understand and reconcile reported differences. A summary of the data presented in Figure A1-1 and
3 comparisons of BCa/BCa and ECa/ECa along with the circumstances for the inter-comparison
4 measurements is found in Table A1.3 (Inter-comparison of Ambient BCa and ECa Measurements).

5 **A1.2.4 Inter-comparison of Two EPA ECa Measurement Protocols**

6 The IMPROVE TOR and NIOSH-like TOT methods have been widely used in the U.S.EPA's national
7 urban Chemical Speciation (CSN) and rural IMPROVE ambient monitoring networks. EPA has transitioned
8 the urban CSN from the NIOSH-type TOT method to the IMPROVE_A TOR method. The transition began
9 in May 2007 and was completed in October 2009 and includes a change to the sampling system as well
10 as the analytical method. The major difference in the sampling method is the sampling flow rate
11 (increased to ~ 22 LPM from ~6.7 LPM) and sample filter diameter (reduced from 46.2 mm to 25mm),
12 which results in an overall increase in pressure drop across the filter during sampling. The combination
13 of these changes results in a reduction in the OC measured, which is most likely related to a change in
14 sampling artifacts. The rationale for the transition of the urban CSN to IMPROVE-like sampling and
15 analysis method was to institute consistency in the carbon measurements across the EPA's national
16 particulate monitoring networks.

17 A comparison between the previous CSN TOT data and the current CSN IMPROVE TOR data
18 indicates that measured EC is, in fact, reasonably consistent between the methods at these 11 locations
19 (Figure A1-3 below). Although this is a limited and preliminary number of comparison locations, this is
20 good news for the EPA transition from the NIOSH-type to IMPROVE protocol, in that it appears to permit
21 both data sets to be interchangeably used to evaluate BC aerosols predicted by climate and air quality
22 models, and to evaluate trends. The seasonal differences in these EC differences are modest, and may
23 be related to the combined effect of sampling rates and analytical protocol and the resulting differences
24 in measured OC as described above. EPA will continue to evaluate the differences between the two
25 measurement protocols.

26

¹ The mixture can be important because of the relative amount of non-graphitic absorbing materials, e.g., BrC and dust, as well as internal mixtures with water, organics, and sulfates.

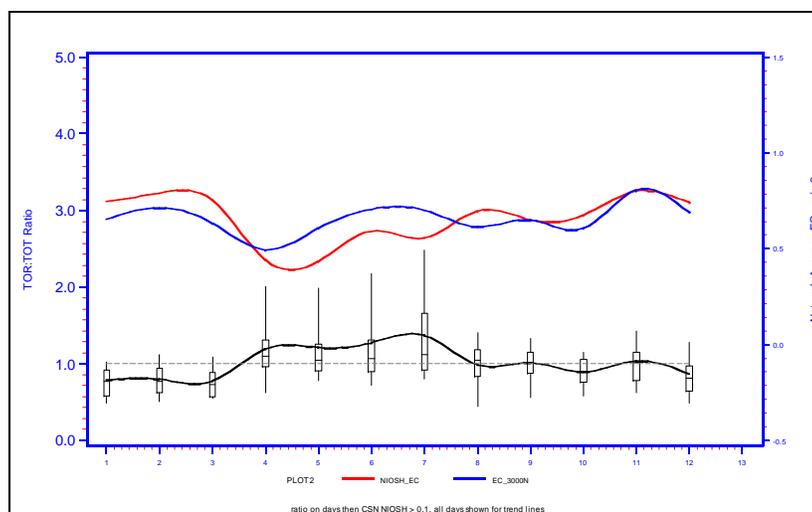


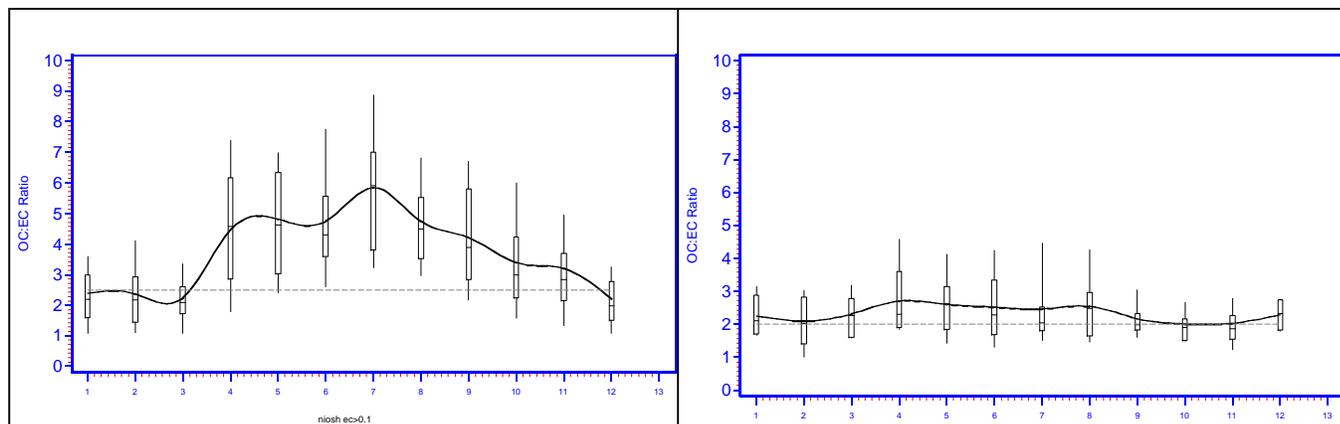
Figure A1-3. Monthly Distribution of ECa/ECa Ratio between Two EPA methods (TOR/TOT), from 897 Collocated Measurements Among 11 Urban CSN Locations. Average CSN NIOSH ECa concentration is red; IMPROVE TOR ECa is blue; the distribution of daily ratios are presented as box plots (black).

A second implication of the change from the NIOSH-like TOT to the IMPROVE TOR monitoring method along with the change in samplers, relates to measured OCa and its sampling artifacts. In some cases, sample collection procedures can lead to the inclusion of *positive artifacts*—mistakenly measuring non-PM components such as vapors as if they were in fact carbonaceous PM. Other procedures can lead to the exclusion of relevant material, producing *negative artifacts*. These artifacts are a problem particularly for measuring concentrations of OCa; sampling artifacts for EC are thought to be negligible, simply because the EC collected on the filter is more stable (non-reactive or volatile).

Because sampling artifacts are most likely to affect measurements of OCa, they may be most important for understanding OCa/ECa ratios (i.e., representing OC/BC). Figure A1-4 shows the monthly distribution of OC/BC among 897 measurements at 11 urban monitoring sites² that concurrently sampled with two alternative measurement protocols (NIOSH TOT and IMPROVE TOR) during 2009-2010. Though ECa can vary somewhat according to the monitoring protocol (see further discussion of NIOSH TOT and IMPROVE TOR below), OCa can vary even more widely as a result of the correction used for OCa sampling artifacts. As the figure shows, the OCa/ECa ratios with the CSN NIOSH TOT method have large seasonal variation and for the 11-site group, the median value is as high as 5. On the other hand, the CSN TOR OC/EC ratios do not display strong seasonality and have monthly median values of ~2-2.5. The latter are more consistent with average estimated direct emission OCa/ECa levels described in Chapter 5, as well as with the artifact corrected ratios described elsewhere (Novakov, 2005). However, they do not display the seasonal change in OCa/ECa ratios due to secondary organic aerosol (SOA) reported elsewhere. As discussed in Chapter 2 and Chapter 5, the correct characterization of OCa

² 11 site inter-comparison group includes Bronx and Queens, NY; Atlanta GA, Birmingham AL, Detroit MI, Cleveland OH; Chicago IL, Denver CO, LA (Rubidoux), CA; Sacramento CA and Seattle, WA.

1 aerosol is critical for differentiating among reflecting vs. absorbing particles for assessment of radiative
 2 forcing, where OC is assumed to be mainly light scattering. While the IMPROVE TOR OCa is adjusted for
 3 sampling artifact with backup filters, the CSN NIOSH TOT protocol is only adjusted with nominal network
 4 value of 1 ug/m³ which may be too low [Chow et. al., 2010]. On the other hand the much more
 5 suppressed seasonal behavior in the TOR carbon ratios could be related to the higher flow rate
 6 IMPROVE protocol samplers which may not fully retain semi-volatile OC particles. The latter will require
 7 further study to understand its implications for using these measurements to develop emission
 8 inventories and to evaluate climate modeling data.



9
 10 **Figure A1-4. Monthly distribution of OC/BC ratios for 11 CSN sites produced with the NIOSH-**
 11 **like TOT (a) and IMPROVE_A TOR monitoring protocols, 2009-10 (b).** Nominal OC sampling
 12 corrections of 1 µg/m³ for CSN NIOSH type TOT have been used. [Ref] The IMPROVE protocol
 13 data are adjusted with backup filters. Due in part to inability to adequately correct the CSN
 14 NIOSH OC sampling artifacts[Ref], these data may in fact overstate ambient OC/BC and imply a
 15 seasonal pattern which may be an artifact of the monitoring method.

16 **A1.2.5 Other Measurements**

17 Microscopy (the use of microscopes to view the structure of particles) and spectroscopy
 18 (measurement of a chemical as a function of wavelength) provide additional information about the
 19 physical and chemical structure of carbonaceous PM. An advantage of these methods is that they
 20 provide detailed information about how particles age and transform from the point of emission to the
 21 atmosphere. A variety of microscopy techniques have been applied to investigate carbon particles.
 22 Scanning electron microscopy (SEM) with energy dispersive X-ray spectroscopy (EDX), transmission
 23 electron microscopy (TEM), and Raman Microspectroscopy (RM) are the most widely used and have
 24 provided the most significant information about carbon aerosol composition to date. Like the thermal-
 25 optical and light absorption measurement methods, microscopy has limitations and is subject to
 26 artifacts and interpretation issues, but these techniques do provide additional information not gathered
 27 by the thermal and optical measurement techniques.

1 **A1.2.6 Limitations of Ambient Measurement Methods**

2 Specific operating conditions, such as the heating temperature, time of heating, and char
3 correction procedures, can influence thermal-optical measurement results. Chemical composition and
4 emission sources of the measured aerosol, filter loading, and uniformity of the filter particle deposit can
5 also influence OCa and ECa values obtained. Studies suggest that EC measurements for some types of
6 emissions (biomass smoke, dust) may be more strongly affected than traffic-related (e.g., diesel)
7 samples, in part because of higher levels of inorganic components and BrC (Novakov and Corrigan 1995).
8 A summary of the comparison of optical BCa to thermal ECa measurements is provided above.

9 Currently, there are no reference standards for assessing the accuracy of OCa or ECa
10 measurements by thermal methods, nor is there a standardized method protocol for distinguishing
11 between OCa and ECa. Development of standard reference materials and the consensus on
12 standardized method protocols (including data reporting procedures) will be important in the future for
13 the consistent measurement of OCa and ECa for climate purposes.

14 All optical BC measurements share a fundamental limitation in that they do not directly measure
15 BC mass concentration. Instead, conversion factors (e.g., mass absorption efficiency or mass absorption
16 cross-section) are necessary to generate BCa mass concentrations from the different optical
17 measurements. In addition, the most commonly used filter-based methods are prone to artifacts during
18 sampling. The extent of filter loading can influence particle scattering and shadowing effects which bias
19 results (Bond et. al. 1999, Weingartner et. al. 2003, Park et. al. 2010). While several filter-loading based
20 correction algorithms have been introduced (Virkkula et. al. 2007), it is uncertain as to whether a
21 correction algorithm should be universally applied as the artifacts may depend upon the particle
22 composition and concentration. Because the aerosol absorption and derived BCa depend on
23 wavelength, it should be noted that some reported BC that is based on wavelengths in the visible
24 spectrum may include other LAC.

25 **A1.2.7 Critical Gaps and Research Needs in Ambient Measurement Methods**

26 In light of the limitations discussed above, the following research can help improve the ambient
27 measurement of BC and LAC in the future and reduce the uncertainty:

- 28 • Further comparisons of the predominant thermal and optical methods in use today are needed
29 to better understand and characterize the differences and uncertainties. As comparisons are
30 made, it is important to clearly document the operational conditions of the methods used.
31
- 32 • Having a consistent, well defined calibration material would help to better understand method
33 differences and define the uncertainties in the various measurement methods.

- 1 • It is important to agree on a standardized method of operation and calibration for those
2 methods identified as most important for measuring BC in support of climate and health.
- 3 • Develop methods capable of quantifying particulate components, referred to as BrC or
4 collectively with BC as LAC, that provide additional light absorption in the near-UV and UV
5 wavelengths.
- 6 • To ensure proper use of measurements, consistent data reporting, including metadata, of the
7 sampling and analysis protocols, and data adjustments must be provided.
- 8 • Continued research and further development is needed for continuous or real-time single
9 particle measurements (e.g., aerosol time-of-flight mass spectrometry and single particle soot
10 photometers) to enhance our knowledge of particle composition and mixing state.

11 **A1.3 Emission Source Black Carbon Measurements**

12 Source measurements are used for a variety of purposes, including regulatory compliance.
13 However, in the United States and elsewhere, such measurements generally focus on total PM_{2.5} mass:
14 measurements of specific components are not required in the United States as part of regulatory
15 testing, and EPA does not have an official source measurement method. Instead, PM composition is
16 measured largely for research purposes, including development of EPA's emissions models. Available
17 source measurements are also used to develop and verify emissions inventories, refine standard
18 measurement approaches, and to assess control technologies and mitigation approaches. Due to the
19 limited amount of source emissions data for the carbonaceous content of PM, EPA often must rely on
20 data and methodologies for total PM mass, or substitute emissions models.

21 **A1.3.1. Stationary Source Emissions Measurement Methods**

22 Most current federal stationary source emission standards are focused on the regulation of
23 filterable total PM mass. For most stationary sources in the current inventory, PM_{2.5} emissions are
24 derived from use of a scaling factor applied to collection of filterable total PM and the PM₁₀ size
25 fractions. Some local/state and site specific standards also require testing for PM₁₀ and PM_{2.5} mass,
26 which sometimes includes both size fraction of filterable and condensable PM. The latter allows for
27 inclusion of certain semi-volatile particles. EPA has recently promulgated a stationary method for PM_{2.5}
28 mass and refined the condensable stationary source measurement protocol (U.S.EPA 2010); over time
29 this will help ensure greater consistency in stationary source emissions measurements. However,
30 stationary source data currently available for PM_{2.5} inventory purposes are based on non-standardized
31 methods and procedures for PM₁₀ and total filterable PM.

32 Due to the complex nature and variety of sources, regulatory and other standardized source PM
33 methods are mainly designed to provide consistent results across a certain category of sources and not
34 necessarily the entire universe of sources (Myers 2004). Thus, compilations of source emissions
35 measurements for total PM mass exist such as EPA's AP-42 [Compilation of EPA's emission factors] and
36 the U.S. National Emissions Inventory (NEI). However, none of these compilations reflects routine

1 sampling required by regulation for all sources in the inventory. Measurement of carbonaceous PM
2 components including BC or EC are not required as part of compliance testing. Such results are
3 generally available only in the academic literature.

4 There are a large variety of methods for the measurement of PM mass from stationary sources,
5 many of which measure both the filterable and condensable fractions of PM_{2.5}. These methods vary due
6 to operational differences such as filtration temperature and conditioning and treatment of the
7 different components of PM. Table A1-4 provides a list of commonly used stationary source methods
8 and some examples of operational differences for determining PM mass from a variety of sources.

9 **A1.3.2. Mobile Source Emissions Measurement Methods**

10 Mobile sources consist of a diverse group of vehicles and engines, including light-duty gasoline
11 vehicles, heavy-duty diesel trucks, gasoline-powered nonroad engines (e.g., lawnmowers, snowmobiles,
12 recreational boats), nonroad diesel engines (e.g., excavators, locomotives, and marine vessels), and
13 turbine and propeller-driven aircraft. Due to their diverse technologies and applications for highway
14 and nonroad uses, there is considerable variability in BC emissions from mobile sources.

15 In the United States, particles in mobile source exhaust emissions are measured for compliance
16 with PM emission standards and are expressed on a mass per unit work (g/bhp-hr) or mass per distance
17 traveled (g/mi) basis. For regulatory certification, diesel exhaust particle emissions are measured using
18 procedures described in 40 CFR Part 1065, which employs an engine dynamometer paired with a
19 dilution sampling system collecting sample on Teflon filters at temperatures of about 125 degrees F
20 (which reduces water condensation yet allows for condensation of organic compounds). The filters are
21 then conditioned at a specific temperature and humidity³ and weighed. This procedure is commonly
22 used to measure PM from non-diesel mobile sources for research purposes.

23 Mobile source emissions of BC are almost always measured as ECa. The latter ECa
24 measurements are, unlike PM measurements, not routinely taken and EPA, presently, does not have an
25 official (or even recommended) EC measurement method for mobile sources for regulatory purposes.
26 However, EPA does measure BC in its mobile source emissions characterization programs. There, BC is
27 measured as a particulate matter (PM) component for both gasoline vehicles such as light-duty
28 cars/trucks and diesel vehicles such as heavy-duty diesel trucks (up to 8-0,000 lbs. gross vehicle weight).
29 It is also measured to a more limited extent from nonroad diesel and even gasoline engines (both 2-
30 stroke cycle engines which have lubricating oil mixed with the fuel) and 4-stroke cycle engines. It is also
31 measured in PM from locomotives, commercial marine, and aircraft.

32
33 Sampling temperature has a major effect on the quantity and even the composition of PM. PM
34 emissions are collected on a filter from diluted exhaust. The general methodology for measuring mobile
35 source PM involves diluting the vehicle exhaust with ambient air roughly at a 10/1 dilution ratio
36 (although the dilution ratio varies greatly depending on engine operating mode) using a stainless steel

³ Mobile source measurements are made at 45%RH, while ambient measurements and many other source tests use 35%RH.

1 dilution tunnel. The filter temperature is about 125 degrees F which is high enough to prevent water
2 condensation on the filter from the copious amounts of water vapor present in vehicle exhaust formed
3 from fuel combustion. This temperature also allows some condensation of the organic hydrocarbon
4 compounds present in vehicle exhaust. This general method was developed and has been in use since
5 about 1970 for both diesel and gasoline exhaust. This methodology is also used for EPA emission
6 standards for exhaust from diesel engines including on-road trucks and more recently nonroad diesel
7 engines. This measurement system, known as constant volume sampling of an exhaust stream that is
8 diluted with varying amounts of dilution air, allows for accurate mass weighting of emissions over
9 transient driving conditions (accelerations, decelerations, steady-state cruise, and idle) where exhaust
10 volume varies. In the ambient air though, vehicle exhaust is rapidly diluted to about 1,000:1 which
11 results in somewhat different condensation of the hydrocarbon compounds into particulate.

12 The PM measurement method is more developed for diesel PM than for gasoline PM.
13 Numerous studies have been done measuring diesel PM starting with the first EPA emission standard for
14 the 1970 model year for visible smoke from diesel engines.

15 **A1.3.3. Use of Emissions Source Test Data**

16 Though carbonaceous components of PM are not systematically measured across all categories,
17 both EPA and external researchers have measured these components from some source categories.
18 EPA has compiled all available source emissions data into a database called SPECIATE, which currently
19 contains 3,326 raw PM profiles. Because many of these measurements are drawn from research on
20 emission measurements, the data comes from a variety of sampling and analytical technologies (Chang,
21 2004). Despite the uncertainties and limited size of the testing dataset compared to the total number of
22 sources, the SPECIATE database represents the best compiled source of data available. A subset of
23 these data was selected to characterize the source profiles for 15 source categories reported in Chapter
24 4, Figure 4-1.⁴ The number of individual profiles by source category can quite limited and sometimes
25 only a single value was used. Similar summaries are available elsewhere (Chow et. al.). Note that for
26 some sources, the sum of BC and OC is less than 100% of PM_{2.5} mass. The raw data used to compile
27 Figure 4-1 is available in Table A1-5, along with the percent of estimated non-carbon PM and the OC/BC
28 ratios.

29 As discussed in Chapter 4, however, EPA does not use any of these profiles for on road vehicles
30 since the mobile MOVES model directly calculates EC emissions (EPA, 2010). Mobile sources have more
31 variability in emissions than stationary sources, because mobile-source EC varies with driving mode,
32 specific model mix, and other conditions. MOVES is designed to capture this variability. Currently, EPA
33 still uses speciation profiles for nonroad diesel.

⁴ Following the procedures of Reff at al., the raw profiles in SPECIATE were modified so that all EC was adjusted to be representative of the TOR analytical method and so that the sum of the species equals the PM_{2.5} mass, if the raw profile was not provided in that format. To provide a more representative median among available test data, subsets of multiple source tests were first combined into a composite profile. Some uncertainty in expressing EC as a fraction of PM_{2.5} may be related to the water content of PM_{2.5} mass.

1 **A1.3.4. Limitations of Source Emissions Measurement Methods**

2 To estimate EC emissions for a specific source category, EC is typically assumed to be a specific
3 fraction of PM_{2.5} and then total PM_{2.5} mass is used as the starting point. Thus, the measurement
4 and/or estimate of PM_{2.5} mass is one very important source of potential uncertainty. There are
5 inconsistencies in the way PM_{2.5} is measured among source categories, including in the approach for
6 determining filterable and condensable mass, filter equilibration conditions (including laboratory RH),
7 temperature of testing, and dilution and related procedures for semi-volatile PM. Some of these
8 variables can also affect the measurement of carbon components. Because of the way estimates of PM
9 components are generated, both the PM and carbon-specific measurements can affect estimates of BC
10 and OC emissions for a given source category.

11 Current PM_{2.5} estimation methods based on PM₁₀ and total filterable PM can produce variable
12 results, particularly the methods that include condensable PM. For certain stationary source categories,
13 this can produce measurement artifacts that can overestimate the condensable PM emissions by an
14 order of magnitude.⁵

15 The use of scaling factors applied to filterable total PM and/or PM₁₀ to generate estimates of
16 PM_{2.5} introduces additional uncertainty to the estimated emission rate (NEI Quality Assurance and Data
17 Augmentation for Point Sources, Feb 2006 – Scaling/Augmentation, EPA AP-42 – PM Emission Factors,
18 FR Notice defining PM 2.5 source method – due before end of year, Research Priorities for Airborne
19 Particulate Matter: IV. Continuing Research Progress, 2004).

20 Finally, the representativeness of a particular source profile based on a limited number of
21 source tests is questionable, and derived composite profiles applied to a large number of sources is
22 another source of uncertainty. For both PM and speciation test data, there are the related
23 representativeness issues of tests conducted with actual vs. allowable emissions from the stacks and
24 effluents; tests conducted at facilities of varying age and with different degree and type of controls; and
25 tests affected by other operating conditions. These factors are often not taken into account when BC
26 profiles are applied to PM_{2.5} emissions. There are also potential issues regarding PM_{2.5} mass closure
27 (including treatment of volatile components, particle bound water) and comparison of BC data based on
28 different measurement methods.

29 **A1.3.5 Critical Gaps and Research Needs in BC Emission Sampling and Measurement** 30 **Methods**

31 In light of the limitations discussed above, the following research can help shed light on
32 amounts of BC and LAC emitted by various sources and lessens the uncertainty in developing an
33 inventory of emissions:
34

⁵ Example artifacts include the potential conversion of sulfur dioxide gas into sulfate particles, affecting the reported PM mass.

1 For all source measurements

- 2 • Understand how the source EC values relate to source BC values made by
3 currently available techniques.
- 4 • Develop high-quality source profiles for sources that need improved characterization for BC,
5 including research into how to quantify the additional light-absorbing components in the near-UV or
6 UV spectrum that are referred to as BrC or, collectively with BC, as LAC.
- 7 • Develop a standard reference material and establish a standard measurement method to report
8 source data as BC.

9

10 Stationary source measurements

- 11 • Understand the effect of varying source test methods and conditions on measured PM_{2.5} and
12 BC; and standardization of PM source testing procedures for filterable and condensable PM
- 13 • Perform uncertainty analysis of all source profiles that exist in SPECIATE and how the total mass
14 from the SPECIATE collection methods relates to the total mass from the methods used in the
15 emissions inventory.
- 16 • Increase the quantity and quality of meta-data available in the databases that better explain
17 how PM_{2.5} and EC fractions were derived for the various sources in EPA's inventories.

18

19 Mobile Source Measurements

- 20 • Develop standard measurement methods for BC for both on-road and nonroad engines,
21 especially diesels but also gasoline vehicles/engines.
- 22 • Establish more routine measurement procedures for BC, including ones that can measure these
23 quantities over short time periods (even instantaneously) as well as over an entire driving cycle.

Table A1-1. Global Monitoring Activities.

Worldwide Air Monitoring Networks^{1,2} for Black Carbon						
Network	Country/ Agency	Years of Data	BC - Indicator	Number of Sites	Measurement Method	Location of Information and/or Data
ESRL/GMD Aerosol Network Baseline Stations Regional Stations Mobile/Cooperative Platforms	United States / NOAA	1957 - Present	BCa and/or Aerosol optical properties	4 Rural 3 Rural 15 Rural	Aerosol Monitoring System – Aethalometers, Particle Soot/Absorption Photometers, Nephelometers, etc.	http://www.esrl.noaa.gov/gmd/aero/index.html
World Data Centre for Aerosols	Global Atmospheric Watch	1974 - Present	BCa and/or Aerosol optical properties	~16 Rural	Aerosols – Light Absorption/EBC, AOD, Light scattering & back scattering, Size distribution	http://wdca.jrc.it/ http://gaw.tropos.de/gaw_program.html
Nepal Climate Observatory-Pyramid (NCO-P)	Nepal	2006-2008	BCa	1 Rural	MAAP	http://www.atmos-chem-phys-discuss.net/10/8379/2010/acpd-10-8379-2010.pdf
CSN/STN—PM2.5 Speciation Trends Network ³	United States / EPA	1999 - Present	ECa	~200 urban	Thermal Optical Transmittance	http://www.epa.gov/ttnamti1/specgen.html
IMPROVE—Interagency Monitoring of Protected Visual Environments	United States / NPS	1988 - Present	ECa	110 rural (plus ~67 protocol sites)	Thermal Optical Reflectance	http://vista.cira.colostate.edu/IMPROVE/
ARIES / SEARCH—Aerosol Research Inhalation Epidemiology Study / SouthEastern Aerosol Research and Characterization Study experiment	United States / EPRI / SC	1992 - Present	ECa	5 Urban 3 Rural	Thermal Optical Reflectance	http://www.atmospheric-research.com/studies/SEARCH/index.html

NAPS—National Air Pollution Surveillance Network	Canada	2003 - Present	ECa	4 rural 13 urban	R&P Partisol-Plus 2025 R&P Partisol Model 2300	http://www.gc.ca/rnspace/naps/Default.asp?lang=En&n=5C0D33CF-1
CAPMoN—Canadian Air and Precipitation Monitoring Network	Canada	2002 - Present	ECa	29 Rural	R&P Partisol Model 2300 PM _{2.5} Speciation Sampler	http://www.msc.ec.gc.ca/capmon/particulate_general_e.cfm
European Monitoring and Evaluation Program (EMEP)	Norwegian Institute for Air Research	2002 -2003	ECa	2 Urban 12 Rural	Thermal Optical Transmittance – Sunset Lab	http://www.atmos-chem-phys.net/7/5711/2007/acp-7-5711-2007.pdf
EUropean Supersites for Atmospheric Aerosol Research (EUSAAR)	European Union	2006 - Present	BCa / ECa	20 Rural	Aerosol properties including – absorption, scattering, AOD Aethalometer / Sunset Lab	http://www.eusaar.net/files/overview/infrastructures.cfm
China Atmosphere Watch Network (CAWNET)	Chinese Meteorological Administration	1999 - Present	ECa	6 Urban 12 Rural	Thermal Optical Reflectance	http://www.agu.org/journals/jd/jd0814/2007JD009525/2007JD009525.pdf
Multiple Independent Sites – two groups by pollutant (BC & ECa) by Vignati et. al. (2010)	Multiple Agencies	Various periods 1976 -2002	BCa ECa	11 Rural 7 Rural	Various	http://www.atmos-chem-phys.net/10/2595/2010/acp-10-2595-2010.pdf
<p>Footnotes:</p> <ol style="list-style-type: none"> 1. The emphasis is on surface-based continuous air monitoring networks. Some networks listed separately may also serve as subcomponents of other larger listed networks; as a result, some double counting of the number of individual monitors is likely. 2. The information on some networks is sketchy. It is frequently unclear (1) when the network actually started up and whether all monitors were operating at that time (or were added over time), (2) whether the pollutant measured is measured as BCa, ECa or some other surrogate for BC (3) what the definition of urban/rural is for a given network and the exact numbers of urban/rural monitors, and (4) what the exact nature of the measurement method is and whether it applies to all or just some sites. 3. Collocated at CSN sites for the period 2009 to present, there are ~40 Aethalometers for measuring BC and 5 Sunset Laboratory Carbon Aerosol monitors for ECa. 						

Table A1-3. Inter-comparison of Ambient BCa and ECa Measurements

BCa-BCa comparison

Citation	Instrument A	Instrument B	r	r ²	Ratio (high/low)	Notes
Chow et. al. (2009)	7-AE (660 nm)	PSAP (660 nm)	0.98		1.28	Fresno Supersite, CA,
Chow et. al. (2009)	7-AE (660 nm)	MAAP (670 nm)	0.99		3.52	Fresno Supersite, CA,
Chow et. al. (2009)	PSAP (660 nm)	MAAP (670 nm)	0.99		2.68	Fresno Supersite, CA,
Chow et. al. (2009)	7-AE (520 nm)	PA (532 nm)	0.96		4.68	Fresno Supersite, CA,
Chow et. al. (2009)	PSAP (530 nm)	PA (532 nm)	0.95		3.69	Fresno Supersite, CA,
Chow et. al. (2009)	MAAP (670 nm)	PA (670 nm)	0.98		1.51	Fresno Supersite, CA,
Snyder (2007)	Aethalometer	PSAP	0.93	0.86	1.41	Slope of line (intercept small)

BCa-BCa comparison for study data with correction algorithms applied

Citation	Instrument A	Instrument B	r	r ²	Ratio (high/low)	Notes
Chow et. al. (2009)	7-AE adj (660 nm)	PSAP adj (660 nm)	0.95		1.02	Fresno Supersite, CA,
Chow et. al. (2009)	7-AE adj (660 nm)	MAAP (670 nm)	0.97		0.9	Fresno Supersite, CA,
Chow et. al. (2009)	PSAP adj (660 nm)	MAAP (670 nm)	0.97		0.81	Fresno Supersite, CA,
Chow et. al. (2009)	7-AE adj (660 nm)	PA (532 nm)	0.95		1.24	Fresno Supersite, CA,
Chow et. al. (2009)	PSAP adj (530 nm)	PA (532 nm)	0.95		1.17	Fresno Supersite, CA,

ECa-ECa comparisons

Citation	Instrument A	Instrument B	r	r2	Ratio (high/low)	Notes
Chow et. al. (2009)	IMPROVE_A_TOT_EC	IMPROVE_A_TOR EC	0.95		1.3	Fresno Supersite, CA
Chow et. al. (2009)	STN_TOT EC	STN_TOR EC	0.9		1.41	Fresno Supersite, CA
Chow et. al. (2009)	STN_TOR EC	IMPROVE_A_TOR EC	0.94		1.1	Fresno Supersite, CA
Chow et. al. (2009)	French EC	IMPROVE_A_TOR EC	0.9		1.03	Fresno Supersite, CA
Chow et. al. (2009)	Sunset Thermal EC	IMPROVE_A_TOR EC	0.87		1.82	Fresno Supersite, CA
Bae et. al. (2007)	NIER-EC	UT-EC	0.99	0.98	1.05	Semicontinuous Sunset instruments, with different temperature protocols: NIER - shortened protocols, UT: nine-step
Klouda (2005)	IMPROVE TOR	STN-NIOSH TOT			1.66	RM 8785 suspended PM
Chow et. al. (2001)	IMPROVE ECR	NIOSH ECT	0.91		4.3	r, mixed ambient samples
Chow et. al. (2001)	IMPROVE ECR	IMPROVE ECT	0.98		1.5	r, mixed ambient samples
Chow et. al. (2001)	NIOSH ECR	NIOSH ECT	0.98		1.56	r, mixed ambient samples
Schmid (2001)	IMPROVE TOR	Sunset TOT 900/900C			1.06	Berlin Nov 7
Schmid (2001)	IMPROVE TOR	Sunset TOT 900/900C			1.03	Berlin Nov 8
Schmid (2001)	IMPROVE TOR	Sunset TOT 900/900C			1.44	Berlin Nov 10 heavily loaded
Schmid (2001)	IMPROVE TOR	Sunset TOT 820/850C			1.35	Berlin Nov 7
Schmid (2001)	IMPROVE TOR	Sunset TOT 820/850C			1.28	Berlin Nov 8
Schmid (2001)	IMPROVE TOR	Sunset TOT 820/850C			1.78	Berlin Nov 10 heavily loaded
Sharma (2002)	TOT	DRI TOR	0.96	0.93	1.09	ambient

BCa-ECa comparisons

Citation	BCa Method	MAC (m ² g ⁻¹)	λ nm	EC Method	r	Avg BCa	Avg EC	Ratio BCa/EC	Location
Chow et. al. (2009)	Aethalometer AE-31 PM _{2.5}	16.6	660	IMPROVE_A_TOR PM _{2.5}	0.89	0.94	1.01	0.93	Fresno, CA
Chow et. al. (2009)	MAAP PM _{2.5}	6.6	670	IMPROVE_A_TOR PM _{2.5}	0.96	0.95	0.95	1.00	Fresno, CA
Chow et. al. (2009)	Sunset Optical BC PM _{2.5}		660	IMPROVE_A_TOR PM _{2.5}	0.87	0.52	1.01	0.51	Fresno, CA
Bae et. al. (2007)	Aethalometer AE-16 PM _{2.5}	16.6	880	Sunset PM _{2.5} hourly	0.93	0.59	0.68	0.87	Gosan, Korea
Bae et. al. (2007)	Aethalometer AE-16 PM _{2.5}	16.6	880	Sunset PM _{2.5} hourly	0.92	0.59	0.74	0.80	Gosan, Korea
Bae et. al. (2007)	Aethalometer AE-16 PM _{2.5}	16.6	880	Sunset PM _{2.5} hourly	0.80	1.89	2.18	0.87	Gosan, Korea
Bae et. al. (2007)	Aethalometer AE-16 PM _{2.5}	16.6	880	Sunset PM _{2.5} hourly	0.70	1.89	2.3	0.82	Gosan, Korea
Jeong et. al. (2004)	Aethalometer AE-20	16.6	880	Sunset PM _{2.5} every two hrs.	0.92	0.9 *	0.4 *	2.25	Rochester, NY
Jeong et. al. (2004)	Aethalometer AE-20	16.6	880	Sunset PM _{2.5} every two hrs.	0.77	0.9	0.4 *	2.25	Philadelphia, PA
Jeong et. al. (2004)	Sunset Optical BC PM _{2.5}	16.6	660	Sunset PM _{2.5} every two hrs.	0.97	0.3 *	0.4 *	0.75	Rochester, NY
Jeong et. al. (2004)	Sunset Optical BC PM _{2.5}	16.6	660	Sunset PM _{2.5} every two hrs.	0.85	0.4 *	0.4 *	1.00	Philadelphia, PA
Hagler et. al. (2007)	PSAP	**	565	NIOSH TOT	0.95		7ngm ⁻³	**	Greenland - no BC mass
Hitzenberger (2006)	AE-9	19	830	Cachier two step 1000C in O ₂	0.72			1.14	Vienna, Austria
Hitzenberger (2006)	MAAP	6.5	670	Cachier two step 1000C in O ₂	0.91			1.20	Vienna, Austria
Hitzenberger (2006)	Integrating sphere	Calc***	550	Cachier two step 1000C in O ₂	0.86			0.98	Vienna, Austria
Hitzenberger (2006)	Light transmission-white light		white	Cachier two step 1000C in O ₂	0.89			1.20	Vienna, Austria
Hitzenberger (2006)	AE-9	19	830	VDI 650 C in O ₂	0.66			0.95	Vienna, Austria
Hitzenberger (2006)	MAAP	6.5	670	VDI 650 C in O ₂	0.88			1.05	Vienna, Austria
Hitzenberger (2006)	Integrating sphere	Calc***	550	VDI 650 C in O ₂	0.78			0.84	Vienna, Austria
Hitzenberger (2006)	Light transmission-white light		white	VDI 650 C in O ₂	0.79			1.05	Vienna, Austria

Hitzenberger (2006)	Aethalometer AE-9	19	830	TOT 800C in O ₂	0.61			1.11	Vienna, Austria
Hitzenberger (2006)	MAAP	6.5	670	TOT 800C in O ₂	0.88			1.11	Vienna, Austria
Hitzenberger (2006)	Integrating sphere	Calc***	550	TOT 800C in O ₂	0.67			0.93	Vienna, Austria
Hitzenberger (2006)	Light transmission-white light		white	TOT 800C in O ₂	0.83			1.13	Vienna, Austria
Snyder (2007)	PSAP	**	565	Sunset PM _{2.5} hourly	0.91			**	Riverside, CA
Snyder (2007)	Aethalometer AE-31	**	880	Sunset PM _{2.5} hourly	0.93			**	Riverside, CA
Sharma (2002)	Aethalometer AE-11 PM _{2.5}	19	880	IMPROVE TOR PM _{2.5}	0.89		0.58 *		Egbert, Canada
Sharma (2002)	PSAP PM _{2.5}	10	565	IMPROVE TOR PM _{2.5}	0.99		0.58 *		Egbert, Canada
Sharma (2002)	Aethalometer AE-11 PM _{2.5}	19	880	IMPROVE TOR PM _{2.5}	0.98		1.42 *		Downsview, Canada
Sharma (2002)	PSAP PM _{2.5}	10	565	IMPROVE TOR PM _{2.5}	0.69		1.42 *		Downsview, Canada
Sharma (2002)	Aethalometer AE-11	19	880	Cachier two step EC 1100C in O ₂	0.91		0.087, 0.012*		Alert, Canada (Arctic)
Sharma (2002)	PSAP	10	565	Cachier two step EC 1100C in O ₂	0.93		0.087, 0.012*		Alert, Canada (Arctic)
Sharma (2002)	Aethalometer AE-11 PM _{2.5}	19	880	NIOSH TOT PM _{2.5}	0.92		1.95 *		Evans Ave, Canada
Sharma (2002)	PSAP PM _{2.5}	10	565	NIOSH TOT PM _{2.5}	0.96		1.95 *		Evans Ave, Canada
Sharma (2002)	Aethalometer AE-11 PM _{2.5}	19	880	NIOSH TOT PM _{2.5}	0.89		1.82 *		Palmerston, Canada
Sharma (2002)	PSAP PM _{2.5}	10	565	NIOSH TOT PM _{2.5}	0.89		1.82 *		Palmerston, Canada
Sharma (2002)	Aethalometer AE-11 PM _{2.5}	19	880	NIOSH TOT PM _{2.5}	0.92		1.48 *		Winchester, Canada
Sharma (2002)	PSAP PM _{2.5}	10	565	NIOSH TOT PM _{2.5}	0.54		1.48 *		Winchester, Canada
Venkatachari (2006)	Aethalometer AE-20 PM _{2.5}	16.6	880	Sunset PM _{2.5} hourly	n/a	1.01	0.85	1.2	New York City, NY
Venkatachari (2006)	Aethalometer AE-20 PM _{2.5}	16.6	880	R&P 5400 PM _{2.5} hourly	n/a	1.01	0.55	1.8	New York City, NY
Venkatachari (2006)	Aethalometer AE-20 PM _{2.5}	16.6	880	CSN TOT PM _{2.5}	n/a	1.01	0.53	1.9	New York City, NY
Sahu (2009)	PSAP PM _{2.5}	8.9	565	Sunset PM _{2.5} hourly	0.98	1.18	n/a	1.0	Jeju Island, South Korea

Yang (2006)	Aethalometer AE-16 PM _{2.5}	16.6	880	IMPROVE TOR PM _{2.5}	0.72	16.5	12	1.4	Xi'an, China
Miyazaki (2008)	COSMOS	9.8	565	Sunset PM _{1.0} hourly	0.96	n/a	n/a	n/a	Thailand
Babich (2000)	Aethalometer AE-20	19	880	IMPROVE TOR PM _{2.5}	0.87	1.1	1.4	0.79	Bakersfield, CA
Babich (2000)	Aethalometer AE-20	19	880	IMPROVE TOR PM _{2.5}	0.98	1.2	1.5	0.80	Chicago, IL
Babich (2000)	Aethalometer AE-20	19	880	IMPROVE TOR PM _{2.5}	0.95	0.8	1.3	0.62	Dallas, TX
Babich (2000)	Aethalometer AE-20	19	880	IMPROVE TOR PM _{2.5}	0.95	1.1	1.5	0.73	Philadelphia, PA
Babich (2000)	Aethalometer AE-20	19	880	IMPROVE TOR PM _{2.5}	0.96	3.1	3.9	0.79	Phoenix, AZ
Babich (2000)	Aethalometer AE-20	19	880	IMPROVE TOR PM _{2.5}	0.92	1.6	1.9	0.84	Riverside, CA
Ram (2010)	Aethalometer	16.6	880	Sunset TOT NIOSH PM ₁₀	0.79	4.45	3.84	1.2	Kanpur, India
Husain (2007)	Aethalometer AE-21 PM _{3.2}	16.6	880	Sunset TOT NIOSH PM _{2.5}	0.84	n/a	n/a	1.3	Lohore, Pakistan
Lim (2004)	PSAP PM _{2.5}	10	565	R&P 5400 PM _{2.5} hourly	n/a	1.26	2.8	0.5	Atlanta, GA
Lim (2004)	PSAP PM _{2.5}	10	565	RU/OGI TOT PM _{2.5} hourly Sunset predecessor	n/a	1.26	2.33	0.5	Atlanta, GA
Lim (2004)	Aethalometer AE-16 PM _{2.5}	12.6	880	R&P 5400 PM _{2.5} hourly	n/a	2.61	2.8	0.9	Atlanta, GA
Lim (2004)	Aethalometer AE-16 PM _{2.5}	12.6	880	RU/OGI TOT PM _{2.5} hourly Sunset predecessor	n/a	2.61	2.33	1.1	Atlanta, GA

* Median concentration

** BC data presented as absorption coefficients (Mm⁻¹) - ratio of BCa/Eca and linear regression equations not extracted for these papers, although it could be calculated

*** Calibration curve based on dissolved carbon black

Table A1-4. Stationary Source Emission Measurement Methods.

Method	PM Type	Filtration Temperature ¹ (°F)	Purpose	
EPA Method 5	Filterable	248 ± 25	General	40 CFR 60 Appendix A-3
EPA Method 5A	Filterable	108 ± 18	Asphalt Roofing	40 CFR 60 Appendix A-3
EPA Method 5B	Filterable	320 ± 25	Utility Plants	40 CFR 60 Appendix A-3
EPA Method 5D	Filterable	248 ± 25	Positive Pressure baghouses	40 CFR 60 Appendix A-3
EPA Method 5E	Filterable and Total Organic Material	248 ± 25	Wool Fiberglass	40 CFR 60 Appendix A-3
EPA Method 5F	Filterable	320 ± 25	Non sulfate Filterable PM	40 CFR 60 Appendix A-3
EPA Method 5G	Filterable and Condensable	<90	Wood Heaters - Dilution	40 CFR 60 Appendix A-3
EPA Method 5H	Filterable and Condensable	<248 and <68	Wood Heaters	40 CFR 60 Appendix A-3
EPA Method 5I	Filterable	248 ± 25	Low level general	40 CFR 60 Appendix A-3
EPA Method 17	Filterable	Stack Temperature	General	40 CFR 60 Appendix A-6
EPA Method 201	Filterable 10µm	Stack Temperature	General – Particle Sizing	40 CFR 51 Appendix M
EPA Method 201A	Filterable 10µm/2.5 µm	Stack Temperature	General – Particle Sizing	40 CFR 51 Appendix M
EPA Method 202	Condensable	85	General – Condensable PM	40 CFR 51 Appendix M
EPA Conditional Test Method -039	Total 10µm/2.5 µm PM (Filterable and Condensable)	85???	General – Dilution based PM	
Example State, VCS and International Methods				
CARB 5	Filterable	248 ± 25		
CARB 501	Filterable, multiple aerodynamic sizes	Stack Temperature	General – Particle Size	http://www.arb.ca.gov/test/meth/vol1/Meth_501.pdf
ASTM D6831 - 05a	Filterable	Stack Temperature	Continuous PM	
ISO 9096 and EN 13284	Filterable			
VDI 2066 Part. 10 method and in the Norm EN 13284-1	Filterable 10/2.5			

Table A1-5. Data Used to Prepare BC and OC source profile box plots (Chapter 4, Figure 4-1). OC/BC ratios and OC+BC as percent of PM are also included.

Stats	Agricultural Burning	Bituminous Combustion	Charbroiling	DistillateOil Combustion	HDDV Exhaust	LDDV Exhaust	NaturalGas Combustion	Noncatalyst Gasoline Exhaust	Onroad Gasoline Exhaust	Prescribed Burning	ProcessGas Combustion	Residential Wood Combustion: HardSoft	SubBituminous Combustion	Wildfires	Wood Fired Boiler
OC Fraction of PM2.5															
10th	0.30	0.02	0.34	0.25	0.18	0.20	0.25	0.84	0.30	0.65	0.05	0.39	0.02	0.47	0.33
25th %ile	0.34	0.02	0.46	0.25	0.18	0.25	0.25	0.84	0.44	0.70	0.20	0.47	0.02	0.47	0.33
50th %ile	0.40	0.03	0.70	0.25	0.18	0.32	0.25	0.84	0.55	0.71	0.35	0.53	0.03	0.56	0.33
75th %ile	0.44	0.07	0.84	0.25	0.18	0.39	0.25	0.84	0.67	0.79	0.46	0.58	0.04	0.64	0.33
90th	0.56	0.12	0.87	0.25	0.18	0.44	0.25	0.84	0.75	0.83	0.57	0.68	0.04	0.64	0.33
N	9	3	4	1	1	4	1	1	10	7	3	12	2	2	1
BC Fraction of PM2.5															
10th	0.05	0.01	0.00	0.10	0.77	0.31	0.38	0.01	0.09	0.01	0.10	0.01	0.02	0.03	0.14
25th %ile	0.08	0.01	0.01	0.10	0.77	0.38	0.38	0.01	0.14	0.01	0.13	0.04	0.02	0.03	0.14
50th %ile	0.10	0.02	0.02	0.10	0.77	0.53	0.38	0.01	0.19	0.02	0.17	0.06	0.04	0.09	0.14
75th %ile	0.12	0.05	0.06	0.10	0.77	0.63	0.38	0.01	0.23	0.04	0.22	0.10	0.07	0.16	0.14
90th	0.13	0.08	0.10	0.10	0.77	0.64	0.38	0.01	0.34	0.07	0.27	0.12	0.07	0.16	0.14
N	9	3	3	1	1	4	1	1	10	7	3	12	2	2	1

	Agricultural Burning	Bituminous Combustion	Charbroiling	DistillateOil Combustion	HDDV Exhaust	LDDV Exhaust	NaturalGas Combustion	Noncatalyst Gasoline Exhaust	Onroad Gasoline Exhaust	Prescribed Burning	Process Gas Combustion	Residential Wood Combustion: HardSoft	SubBituminous Combustion	Wildfires	Wood Fired Boiler
OC:BC ratios															
10th	5.9	1.9		2.5	0.2	0.6	0.6	59.9	3.3	54.3	0.5	27.6	1.0	14.5	2.4
25th	4.2	1.7	41.2	2.5	0.2	0.7	0.6	59.9	3.2	49.4	1.5	12.4	1.0	14.5	2.4
50th	4.1	1.6	31.1	2.5	0.2	0.6	0.6	59.9	2.9	38.6	2.1	9.4	0.7	5.9	2.4
75th	3.6	1.5	13.5	2.5	0.2	0.6	0.6	59.9	3.0	19.3	2.1	5.9	0.7	4.1	2.4
90th	4.3	1.4	8.5	2.5	0.2	0.7	0.6	59.9	2.2	12.0	2.1	5.5	0.7	4.1	2.4
BC+OC, as % PM															
10th	36%	3%	34%	35%	95%	51%	63%	85%	39%	66%	15%	40%	4%	50%	46%
25th	43%	4%	47%	35%	95%	63%	63%	85%	57%	72%	34%	51%	4%	50%	46%
50th	49%	4%	72%	35%	95%	86%	63%	85%	74%	73%	53%	58%	7%	65%	46%
75th	57%	12%	90%	35%	95%	102%	63%	85%	90%	83%	68%	68%	11%	80%	46%
90th	69%	20%	97%	35%	95%	108%	63%	85%	109%	90%	83%	81%	11%	80%	46%

1 Appendix 2. Black Carbon Emissions Inventory 2 Methods and Comparisons

3 A2.1 Introduction

4 This appendix provides specific details on the approach used to generate domestic inventories
5 for stationary, area, and mobile sources, and compares that approach to methods used in compiling
6 international inventories. It explores key methodological similarities and differences between
7 inventories, and also outlines the specific methodologies and data inputs used to construct key global
8 and regional inventories currently available.

9 In general, existing inventories for black carbon are based on calculations rather than actual
10 emissions measurements. Direct emissions measurements of BC and other PM constituents are rare,
11 and no known inventory is based on direct BC emissions data. Instead, “BC” inventories are calculated
12 mathematically from PM_{2.5} inventories. These calculations divide the direct carbonaceous particle
13 emissions from the PM_{2.5} inventory into two categories: EC and OC. Thus most “BC” inventories are
14 actually EC inventories. Though sometimes the terms EC and BC are used interchangeably, EC is actually
15 more narrowly defined (see Chapter 1). By tracking only EC, current inventories fail to account for the
16 portion of primary OC emissions that is light absorbing (including some BC and also BrC). As discussed in
17 Chapter 2, this means current domestic and international inventories systematically underestimate total
18 LAC; however, the magnitude of this gap has not been adequately quantified to date.

19 Most inventories use a **bottom-up approach**, first pairing *PM_{2.5} emission factors* with *activity*
20 *level data* for the source category to generate PM_{2.5} emissions estimates, and then applying a *speciation*
21 *factor* to estimate the amount of BC (or other constituents) contained in the total mass of PM_{2.5}
22 emissions. The BC emissions for individual source categories are then aggregated to form total BC
23 emissions estimates. The speciation factors for an individual source category relate emissions of specific
24 constituents to total PM_{2.5} mass—for example, the amount of BC to total PM_{2.5}. PM_{2.5} emission factors
25 and the speciation factors for particular constituents can be based on either fuel consumption data (i.e.,
26 estimated emissions of total PM_{2.5} or specific constituents per unit of fuel) or actual measured source
27 emissions from emissions testing (see Appendix 1). Some inventories use a combination of these
28 different approaches, depending on the information available for each source category.

29 In a few cases, emissions may be back-calculated from remote sensing of primary PM_{2.5}
30 emissions or measured ambient data of the amount of carbonaceous aerosols in the atmosphere. This
31 kind of **top-down approach** is far less common; currently only a few regional inventories in Asia rely on
32 such methods.

33 Additional information on approaches used to calculate the U.S. emissions inventory and other
34 global/regional inventories is provided below.

A2.2 Development of U.S. National Emissions Inventory for Black and Organic Carbon

EPA's National Emissions Inventory (NEI) is a bottom-up compilation of estimates of air pollutants discharged on an annual basis by source category (EPA, 2008). The Consolidated Emissions Reporting Rule (EPA, 2002) provides a regulatory basis for the collection of emissions inventory information. Currently, emissions of BC and other PM constituents (OC, nitrates, sulfates, crustal material) are not directly reported as part of the NEI. Rather, BC emissions for most sources are estimated by matching PM_{2.5} emissions from the NEI for those sources to source-specific BC speciation profiles from the "SPECIATE" database (see Appendix 1 for details on this database). The one exception is onroad mobile sources, for which BC emissions are estimated directly through models. The following sections provide more information on the specific methods used to compile the inventory for both stationary and mobile sources. More detail is provided for mobile source inventories due to the dominant contribution of these sources to U.S. BC emissions.

Stationary Sources

Stationary sources in the NEI include both point (fossil fuel combustion, industrial sources) and nonpoint (biomass burning) source categories. The basic method for estimating PM_{2.5} emissions for all of these sources follows the simple conceptual formula described in Equation 1:

$$E = A \times EF (1-ER/100) \quad (\text{Equation 1})$$

Where:

- E = PM_{2.5} emissions (for example, in Tons);
- A = activity rate;
- EF = the emission factor, and
- ER = overall emissions reduction efficiency, %

Direct PM_{2.5} emissions are composed of both filterable (solid) and condensable (gaseous) fractions. The condensable fraction condenses rapidly in the ambient air to form tiny liquid droplets. The sum of the filterable and condensable fractions is what is reported in the NEI for all source categories, and these estimates are used in Chapter 2 of this report.

Emission Factors

AP-42, Compilation of Air Pollutant Emission Factors (EFs), has been published since 1972 as the primary compilation of EPA's EF information. It contains EFs and process information for more than 200 air pollution source categories. More recently, AP-42 has been transitioned into the FIRE 6.25 Data System, which currently represents the most comprehensive collection of emission factors (EPA, 2009). It currently contains thousands of records of (mostly) filterable PM_{2.5} EFs updated through calendar year 2004 (Online at <http://cfpub.epa.gov/webfire/>).

A source category is a specific industry sector or a group of similar emitting sources. These EFs have been developed and compiled from source test data, material balance studies, and engineering

1 estimates. EFs can be as simple as an average rate per unit process input. In most cases, EFs depend on
2 many variables such as process parameters, effluent temperatures, ambient temperatures, wind speed,
3 and soil moisture. In these cases the formula is applied to estimate emissions for a particular set of
4 conditions. Under some circumstances in the inventorying of PM_{2.5}, EFs and estimation techniques are
5 applied for analyses other than those for which they were developed. The accuracy and
6 representativeness of the EFs are determined by the reliability of the testing methodology, how
7 uniformly it is applied across sources, or the engineering process information used to derive the EFs.

8 EFs for some emission categories are more reliable than others. In some cases an EF may not be
9 available for a source category because of insufficient or unacceptable data for generalization across
10 source type. Often it is difficult to determine precisely what the certainty in the EF is. Thus, the
11 application of EFs requires subjectivity and judgment from knowledgeable technical staff for the
12 application of concern. As discussed in a previous chapter, users of EFs in national-, regional-, and
13 urban- scale studies should be cognizant of their potential limitations, and other techniques should be
14 considered to improve the confidence in PM emission inventories. Several such approaches have been
15 developed and some continue to be explored: continuous emission-monitoring sensors (not available
16 for PM currently?), material balances, specialized source profiling for composition and compositional
17 material balances, source sampling to obtain improved particle-size distributions and location-specific
18 emission rates, near-source ambient characterization, and source apportionment techniques
19 (references needed). It is important to note that the reliability of EF estimation decreases when only a
20 few source tests are used as the basis for the factor, or when judgmental decisions are made from
21 analogy between technologies. Differences in EF estimates also can develop if the current operations or
22 processes are significantly different from those upon which the original EFs were derived.

23 When most people traditionally think of particulate matter, they envision solid particles that
24 exist in the exhaust stream. However, unlike traditional particulate matter, PM_{2.5} is comprised of both a
25 filterable fraction and a condensable fraction (see earlier discussion). The filterable fraction already
26 exists in solid particle form in the exhaust. The condensable fraction exists in gaseous form in the
27 exhaust stream but condenses rapidly in the ambient air to form tiny liquid droplets. Together, the
28 filterable and condensable fractions are referred to as direct emissions of PM_{2.5}, and the summed
29 number is what is reported in the NEI for all source categories. Most AP-42 EFs do not quantify the
30 condensable fraction of total PM_{2.5} emissions. “Gap filling” techniques are used to estimate
31 condensable PM_{2.5} for many stationary and area source categories. This introduces some uncertainties
32 in the emission estimates.

33

1 Emissions Reduction Factors

2 The Emission-reduction Factor (ER) in Equation 1 accounts for emission controls employed on a
3 source. For example, these include various effluent exit devices such as bag house filters and
4 electrostatic precipitators for removal of PM. Like other process equipment, emission controls have
5 variable operating performance depending on their design, maintenance, and nature of the process
6 controlled. Thus, like EFs, values of ERs are overall averages for specific processes and emission-control
7 designs based on limited testing. Actual values of ERs vary in time and by process in an undocumented
8 manner, adding significant uncertainty to emission estimates. Note that if no emission controls are
9 applied, the abatement efficiency equals zero ($ER=0$) and the emission calculation is reduced to the
10 product of activity and the emission factor, EF

11 Activity Levels:

12 The last piece of information needed in equation 1 to estimate PM_{2.5} emissions for sources is
13 activity patterns. Activity patterns (AP) describe average temporal operating characteristics of a
14 process, including estimates of the down time for maintenance or process failure. Values of AP for point
15 and non-point sources are each obtained in different ways owing to the differing nature of the sources.

16 Most point sources or industrial sources operate with local permits, and these require
17 information about process emissions, including temporal characteristics. For sources with CEMs for
18 monitoring opacity (roughly proportional to fine PM loading), such as large utility boilers, real-time data
19 are available to derive activity patterns, and deduce emission variability over extended time periods.
20 Further, point sources keep and report records of output during operating periods, and maintenance or
21 other down times.

22 There is a great deal of complexity in acquiring activity data for nonpoint sources, which are
23 diverse in character, individually small, and often intermittent, but collectively significant. Though such
24 sources are difficult to characterize, they are generally important to PM emission estimation because
25 their aggregated mass emissions can be large and their chemical composition (e.g. black carbon) may be
26 important for estimating source attribution. One good example of such a category is forest fires,
27 burning of land-clearing debris, and agricultural burning.

28 Temporal resolution depends on the allocation of emissions aggregated seasonally, weekly, daily
29 or by diurnal variation, depending on use and industry activity patterns. The temporal allocations allow
30 for improved approximation of the actual temporal patterns that can be important not only for precise
31 annual averaging using seasonal or daily allocations, but also for short-term impairment taken over
32 periods of 24 hours or less. "Typical" temporal variations for different sources have been developed
33 from surveys, activity analyses, and expert consensus. These temporal models are approximations that
34 may deviate substantially from actual emissions in a given location. Depending on the requirements for
35 precision in estimations, local testing through observations and activity data may be required, not only
36 for large point or nonpoint sources, but for smaller ones that may be of special interest.

1 For nonpoint sources, emissions can be estimated coarsely from “top-down” measures of
2 activity at the state- or national- level demographics, land use, and economic activity. The construction
3 industry, for example, is based on the total annual expenditures at the regional level. These estimates
4 are then allocated by county, using a procedure linked with construction costs and estimated area under
5 construction. Because of their potential importance as PM sources, considerable effort has been
6 devoted recently to the characterization of emissions and activity patterns for non-point sources.
7 Another example is estimation of emissions from fires, which depends upon knowledge of the time,
8 location, and areal extent of the burn, fuel loading, types of combustible material and moisture content.
9 Recent efforts by EPA include the use of process modeling and remote sensing data to better estimate
10 fire activity patterns and emissions from fires (BlueSky Framework, 2009). Finally, residential wood
11 burning is also an important local sources of PM and black carbon. Quantification of emissions from this
12 source category has been approached through acquisition of data on how much fuel is burned in
13 fireplaces and woodstoves using national consumption estimates. Where this source is a large
14 contributor to PM, local surveys of firewood use are used to supplement and improve activity level
15 estimates. For all burning categories, the PM emissions reported via AP-42 contain both condensable
16 and filterable emissions, so that the uncertainties involved with arriving at total PM_{2.5} is less compared
17 to other point and non-point sources.

18 Estimating BC and OC Emissions:

19 Next, these PM_{2.5} emissions can be converted to BC and OC by proper application of speciation
20 factors from the SPECIATE database. (See Appendix 1 for details on SPECIATE.) The equation used is
21 quite simple: PM_{2.5} Emissions (in tons) * fraction of PM_{2.5} that is BC = BC emissions. This can be
22 difficult given that there are thousands of PM_{2.5} source categories but only a limited number of
23 speciation profiles. Therefore, special attention must be given in mapping specific profiles to source
24 categories. These details are explained in Reff et al. (2009). Application of these methods to the
25 inventories results in the 90 source categories for which BC and OC emissions are reported in Chapter 2.

26 While the process for compiling BC emissions inventories is reasonably straightforward, there are
27 important limitations in this process that introduce uncertainties in final BC emissions estimates. These
28 include:

- 29 • The reliability of the PM_{2.5} emission factors used in equation (1). Some emission factors for
30 point and non point sources are more reliable than others (NARSTO, 2002).
- 31 • The reliability of condensable PM estimates by source category. Some sources include PM
32 condensables as part of their testing protocol (fires, residential wood combustion). Others do
33 not, and a generic emission factor (via AP-42) is applied to estimate the amount of condensable
34 PM the source emits, this introduces a level of uncertainty in determining final BC emissions
35 that is not currently accounted for. The source measurements section of this report gives a
36 clearer indication of what the issues are and how they can be improved.

- 1 • The reliability of activity levels used in equation (1). Some activity levels are generated using
2 process models (EPA, 2006), while some are generated using surrogate information (EPA,
3 2006B).
- 4 • Finally, many “augmentations” are done in the emissions inventory processing steps. These
5 augmentations include scaling measured PM to PM_{2.5} as well as assigning condensable emission
6 estimates to point and nonpoint sources that are not available via source testing. Some of the
7 impacts of the uncertainties in doing this have been explored (NARSTO, 2005), but the issue has
8 not been dealt holistically.

9 Mobile Sources

10 In the U.S. inventory, mobile sources consist of the following general categories of vehicles and engines:

- 11 • *On-road gasoline*, such as passenger cars and light-duty trucks
- 12 • *On-road diesel*, including light-duty passenger cars, light-duty trucks, and heavy-duty trucks.
13 Unlike in Europe, very few diesel passenger cars are sold in the United States, making heavy-
14 duty diesel trucks the dominant vehicle type in this category.
- 15 • *Nonroad diesel*, including construction, agricultural, and other equipment
- 16 • *Nonroad gasoline*, including both 2-stroke and 4-stroke cycle engines such as those used in
17 lawn/garden equipment and recreational marine
- 18 • *Commercial marine*, classified by engine displacement as categories C1, C2, and C3 (ocean-
19 going)
- 20 • *Locomotives*
- 21 • *Aircraft*, which are generally turbine aircraft rather than the smaller piston gasoline-powered
22 aircraft

23 BC emissions from on-road vehicles, both gasoline and diesel, are now calculated directly using EPA’s
24 new MOVES 2010 model. For other mobile source categories, BC emissions are calculated using
25 methods similar to those described above for stationary sources.

26 *On-road Gasoline and Diesel*

27 For onroad gasoline and diesel vehicles, EPA’s emissions models directly calculate both total
28 PM_{2.5} emissions and BC emissions. Recent improvements in EPA’s new MOVES 2010 model (EPA, 2010)
29 as compared to the earlier MOBILE6.2 model (EPA, 2003) include accounting for high emitters,
30 deterioration of PM emissions (i.e., increase in PM mass) with higher mileage, and increased PM

1 emissions at lower temperatures.¹ This model directly calculates BC emissions (as well as other exhaust
2 PM components such as sulfates and OC), and accounts for the significantly reduced BC fraction emitted
3 from onroad diesels due to application of diesel particulate filters (DPFs) (required for heavy-duty diesel
4 trucks up to 80,000 pounds GVW beginning with the 2007 model year). An important input for the
5 gasoline vehicle PM_{2.5} portion of the MOVES model is a recent study examining PM emissions from
6 about 500 in-use vehicles (Coordinating Research Council, 2008).

7 Gasoline OC and BC emissions increase dramatically at lower ambient temperatures. To
8 calculate this increase for gasoline vehicles, we used calculations done for EPA rulemaking packages for
9 gasoline PM, for which an hourly grid-cell temperature adjustment was done as part of emissions
10 processing at the county level for each of the over 3,200 counties. As a general rule, diesel PM
11 emissions are less sensitive to temperature for a variety of reasons (lower importance of cold start since
12 many diesels trucks do not operate on short trips; easier engine warm up since older diesels do not have
13 catalysts which take a finite time to warm up during which emissions are higher). This means that BC
14 emissions from diesel vehicles are not projected to increase as much at lower temperatures as would be
15 the case with gasoline vehicles.

16 MOVES can also be used to calculate tire and brake wear PM_{2.5}, with speciation factors applied
17 to calculate BC. Only a small fraction of the PM from tire and break wear is in the PM_{2.5} range, so
18 estimated BC emissions from these categories are fairly small. However, a large fraction of tire wear PM
19 (about 22%) is BC. In the U.S. inventories reported in Chapter 2, these detailed calculations at the
20 county level were done for 2005 and projection years (2020, 2030) along with some less detailed
21 calculations (at the national level) for 1990. One important thing to note is that the PM, BC, and OC are
22 relatively high from on-road gasoline vehicles for 1990 due to the presence of a large number of non-
23 catalyst vehicles still remaining in the fleet.

24 *Nonroad Gasoline and Diesel*

25 For nonroad engines (both gasoline and diesel powered), EPA calculates BC emissions based on
26 PM emissions estimates from the NONROAD model (EPA, 2008a). Also, the National Mobile Inventory
27 Model (NMIM) uses the current version of the NONROAD model (NONROAD2008) to calculate emissions
28 inventories. The model incorporates emission factors (in g/BHP-hr – that is, grams per brake
29 horsepower-hour), engine output (BHP-hr), and usage data for a wide number of NONROAD sources.
30 For gasoline engines, 2-stroke cycle engines are a separate category from 4-stroke cycle engines. These
31 engines have lubricating oil mixed with the fuel so the exhaust VOC (and PM) will be markedly different
32 from that for the more standard 4-stroke cycle engines. For these engines, the profile used to derive the
33 emission estimates in Chapter 4 is that used for non-catalyst equipped gasoline-powered motor vehicles
34 since these non-road gasoline do not have catalysts. This profile (92049) comes from the EPA Speciate

¹ MOVES also accounts for emissions changes with use of gasoline/ethanol blends, although the effect on PM exhaust emissions from use of gasoline/ethanol blends is extremely small if not zero (EPA, 2010).

1 data base and shows 10% of the PM being BC. Admittedly, data specific for nonroad gasoline engines,
2 especially 2-stroke engines with their oil combustion, are needed.

3 BC emissions are then calculated by using speciation factors denoting the percent of PM
4 emissions represented by BC. A speciation factor for nonroad diesel engines not equipped with diesel
5 particulate filters comes from EPA's SPECIATE data base (EPA, 2008b). The profile used to derive the
6 emissions estimates in Chapter 4 (Profile 92035) is actually derived from heavy-duty on-road diesels and
7 has 77% of the PM being BC. Beginning in calendar year 2012, many if not most newly manufactured
8 nonroad diesels for that "model year" will be equipped with diesel particulate filters (DPFs). This
9 technology reduces exhaust PM mass by over 90%, and the small amount of PM remaining has relatively
10 little BC. In effect, DPFs preferentially reduce BC. Roughly 10% of the PM from a diesel with a DPF
11 consists of BC based on a large emissions characterization program on four 2007 model year on-road
12 diesel truck engines equipped with DPFs. The testing was done by Southwest Research Institute for this
13 program conducted by the Coordinating Research Council (Khalek et al., 2009).

14 A critical factor in compiling BC inventories for nonroad diesels is to correctly apportion the BC
15 emissions between pre-trap equipped diesel engines and trap equipped diesels in any given calendar
16 year. The NONROAD model correctly calculates the combined PM mass in a given calendar year
17 accounting for pre-trap and trap-equipped diesels. Though it does not presently calculate BC emissions
18 separately, a later version of the model under development will do so. Meanwhile, when NONROAD is
19 run, one can get a model year emissions output for specific calendar years. One can then probably
20 manually take that model year input and apply the higher BC speciation percent (77%) to the pre-trap
21 equipped engines and the lower percentage (roughly 10%) to the new diesel engines equipped with
22 DPFs. The inventory numbers presented account for this difference. For nonroad gasoline, a speciation
23 profile of 10% of the PM being BC is used based on tests on older non-catalyst light-duty vehicles. Most
24 nonroad engines do not have catalysts. Since almost no or limited PM speciation has been obtained for
25 the exhaust of these engines, the most appropriate factor to apply is based on older non-catalyst
26 vehicles (of the types produced before introduction of catalysts with the 1975 model year). It is also
27 important to note, however, that 2-stroke cycle engine production will be changed with the advent of
28 new EPA emission standards.

29 *Commercial Marine, Locomotive, and Aircraft*

30 Commercial marine, locomotive, and aircraft emissions are calculated separately in spreadsheet
31 models, with separate BC speciation factors for C1/C2 commercial marine and C3 commercial marine
32 (the larger ocean-going vessels). For the smaller vessels, the profile for non-road diesel engines is
33 applied even though the higher sulfur content of the fuel will lead to the PM containing higher sulfate
34 emissions than for nonroad diesels. DPFs will be required for these vessels starting in 2014, reducing
35 the BC fraction to about 10% of the PM. However, DPFs will only be used on some engine classes, and
36 implementation dates will vary (depending on factors such as engine size). Thus, there is a need for a
37 model to correctly account for the implementation of these standards. For now, a model year break-out
38 of PM emissions was done for both 2020 and 2030. Separate BC/PM speciation factors were applied to
39 the PM emissions from the diesels with and without DPFs. Currently, the diesel BC speciation factor of

1 77% BC/PM is used for C1/C2 commercial marine for all years of analysis: 2005 as well as the non-DPF
 2 equipped engines in 2020 and 2030. Evidence from recent studies (Lack et al. 2009) suggests that a
 3 lower BC speciation factor may be more appropriate for C1/C2 marine.

4 PM emissions from C3 Marine have substantially different PM speciation profiles than smaller
 5 diesel engines used in C1/C2 Marine and on-road and nonroad diesel. C3 marine diesels burn a high
 6 molecular weight residual oil that contains very high sulfur levels (up to 45,000 ppm versus the 15 ppm
 7 in on-road and non road diesel fuel). Past EPA evaluations of C3 marine have used the EPA PM SPECIATE
 8 profile of Residual Oil Combustion (U.S. EPA, 2008c), which estimates a 1% BC speciation factor.

9 For this report, an updated BC speciation profile was estimated from studies available in the
 10 literature. Results from relevant studies that measured BC and PM emission rates from marine sources
 11 are summarized in Table A2-1.

12 **Table A2-1. Summary of Recent Studies that Measured BC and PM Emission Rates from C3 Marine.**

Study	Vessel	Fuel	Fuel Sulfur Content	BC/PM ¹
Murphy et al. 2008	Post-Panamax Container	Heavy Fuel Oil	30,000 ppm	0.31%
Agrawal et al. 2008	Suezmax Marine Tanker	Heavy Fuel Oil	28,500 ppm	0.50%
Petzold et al. 2010 ²	Medium Speed Diesel Engine	Heavy Fuel Oil	22,100 ppm	2.63%
Lack et al. 2009	Slow-speed diesel vessels ³	Variety	Variety ⁴	7.33%
Lack et al. 2009	Medium-speed diesel vessels ⁵	Variety	Variety ⁶	28.00%

13

14 NOTES

- 15 ¹ Lack et al. 2009 measured BC not EC
- 16 ² Engine test, with engine load 85-110%
- 17 ³ PM measurements come from 29 SSD ships, BC Emissions come from 52 SSD ships
- 18 ⁴ Mostly high sulfur fuel (>5,000 ppm)
- 19 ⁵ PM measurements come from 12 vessels, BC emissions come from 51 vessels
- 20 ⁶ Mostly low sulfur fuel (<5,000 ppm)

21 As noted in Table 1, there is substantial variation in the reported BC/PM emission profiles from
 22 these studies. The discrepancies among different BC emission factors for marine sources are additionally
 23 noted in the literature (Petzold et al. 2010). Considering the uncertainty of the values, the EPA selected
 24 a BC/PM speciation factor of **3%** which falls in the middle of the range of reported values. EPA
 25 recognizes that this is an area of active research, and recommends further work be conducted.

26 Table A2-2 displays the BC, OC, and hydrated sulfate speciation rates from the relevant marine
 27 studies. The results from the Lack et al. 2009 study, is subdivided according to vessel type: slow-speed
 28 diesel (SSD) and medium speed diesel (MSD). Lack et al. 2009 also grouped the BC emission observations
 29 according to fuel sulfur content. 51 ship observations had low sulfur fuel content (<5,000 ppm) and 42

1 ship observations of vessels had fuel sulfur content greater than 5,000 ppm. From the available data, BC,
 2 OC, and hydrated sulfate speciation factors were calculated for each of the subcategories. The ships in
 3 the Lack et al., 2009 study with low sulfur content had much lower sulfate speciation factors and higher
 4 BC speciation factors than the other studies.

5 **Table A2-2. Summary of Speciation Ratios of PM from Relevant Marine Studies.**

	BC	OC	Hydrated Sulfate
Agrawal et al. 2008	0.5%	11%	84%
Murphy et al. 2008	0.3%	7.4%	NA
Petzhold et al. 2010	2.6%	21%	93%
(SSD) Lack et al. 2009	7.3%	23%	65%
(MSD) Lack et al. 2009	28%	16%	53%
(>5,000 ppm) Lack et al. 2009	7.1%	19%	70%
(<5,000 ppm) Lack et al. 2009	51%	35%	6.5%

6

7 To estimate the BC/PM factor for future years (2020 and 2030), the International fuel sulfur
 8 limits were considered (Table A4-2) as well as the speciation rates from the studies evaluating. Lowering
 9 the fuel sulfur content is an effective method to reduce the particulate sulfate, which comprises the
 10 majority of the PM from marine vessels using heavy fuel oil. Due to the substantial drop in fuel sulfur
 11 levels, the BC speciation factor should rise in 2020 and 2030. Due to limited data, the EPA chose a C3
 12 marine BC speciation factor of **6%** for 2020 and 2030. For now, EPA is choosing 11% as an OC/PM
 13 speciation fraction for 1990 and 2005 with a higher fraction (58.6%) for 2020 and 2030 when fuel sulfur
 14 reductions occur, especially in ECA areas.

15 For locomotives, as for C1/C2 marine, the HDDV on-road profile (77% BC) is presently being used
 16 for pre-2014 engines although available data suggest this number might be too high. DPFs will be used
 17 in subsequent years, reducing BC to about 10%. For 2020 and 2030, the PM model outputs are obtained
 18 by calendar year and for the years when the standards take effect, the 10% number is used.

19 For purposes of emissions inventory estimates, aircraft operations are often broken into two
 20 basic portions. The first portion, landing and take-off (LTO) cycle is normally defined to include aircraft
 21 ground operations (taxi/idle) as well as aircraft operations below 3000 feet elevation in the local airport
 22 terminal area. The second portion is referred to as non-LTO that includes climb (above 3000 feet) to
 23 cruise altitude and descent from cruise to 3000 feet. Together these portions comprise what is called
 24 “full-flight” emissions.

25 Emissions for the LTO portion are fairly well characterized. Engine emission rates are measured
 26 in jet engine test cells during FAA certification testing; it is believed that these measurements
 27 reasonably predict engine emissions rates for aircraft in actual LTO operations. Programs for evaluating
 28 and controlling LTO emissions have been in place in the United States for about thirty years. Today
 29 these are LTO engine emissions standards for hydrocarbons, carbon monoxide, oxides of nitrogen and
 30 smoke number. While work is now underway to develop a sampling and measurement procedure and

1 certification requirement for aircraft jet turbine engine particulate matter (PM) emissions, there are not
2 yet specific engine emission standards for PM. To address this shortfall on at least an interim basis, FAA,
3 working with EPA, industry and academic experts developed a methodology to estimate LTO PM
4 emissions. This methodology, known as the “First Order Approximation” (FOA) uses information on
5 smoke number and other engine and fuel parameters to estimate LTO PM emission rates for each
6 engine model.² This information is then matched with airframe information on number of engines to
7 get a per LTO emission rate for each aircraft type. Using the airport specific information and the aircraft
8 activity for each airframe/ engine combination, the LTO PM Inventory estimates are made. The total PM
9 emission rate includes all types of compounds contributing to the PM mass. It is estimated that only
10 about 13 percent of the PM mass is black carbon (BC); the remainder is comprised of sulfates and
11 organics. The average BC PM emission index (EI) is in the range of 0.04-0.05 g/kg fuel burned for the
12 LTO portion.

13 The estimation of non-LTO BC emissions depends on a very limited set of measurements.
14 Emissions testing in jet engine test cells does not fully characterize PM emission rates at altitude
15 because they are conducted at sea level static conditions and have to be carefully extrapolated to
16 altitude conditions, due to the differences in the atmospheric environment and engine operating
17 conditions outside of the LTO -- including cruise altitudes. Although there are research models available
18 to estimate non-LTO BC, there is not yet a consensus approach for estimating non-LTO PM emissions as
19 exists for LTO PM emissions. This is an area of ongoing research within the scientific and technical
20 aviation communities.

21 However, two important points should be recognized with regard to non-LTO PM BC emissions.
22 First, results from FAA’s model entitled “System for Assessing Aviation’s Global Emissions” (SAGE)
23 indicate that total fuel burn during non-LTO operations is about ten times that during the LTO.³ Since
24 the PM emission inventory is linked to fuel burn, overall PM emissions during the non-LTO portion of the
25 “full flight” would be expected to be larger than those during the LTO portion. Second, this is an area of
26 ongoing research and to-date there are no less than six researchers who have used various methods to
27 estimate the EI for PM BC emissions during the non-LTO portion of “full flight”(Lilenfield 1995, Pueschel
28 1997, Anderson 1998, Petzold 1999a, Petzold 1999b, Doppelheuer 2001, Kinsey 2009). Some researchers
29 have used equivalent non-LTO thrust levels on the ground while the estimates of others were based on
30 in-situ plume measurements of aircraft in flight. It is difficult to make direct comparisons among these
31 values or to use this data to derive a point estimate for the non-LTO BC EI since they were developed on
32 different airframe/engine models of different technology vintages using different measurement
33 approaches. While data from the published researchers ranges from about 0.01-0.35 g/kg fuel burned,
34 the majority of the data lies in the range of about 0.02-0.11 g/kg fuel burned. Each study has its relative
35 strengths and weaknesses and most of the older engines with higher EIs are no longer in service. Table
36 A2.3, below summarizes the publicly available literature on this issue.

37 _____

² Wayson R.L., J.S. Kinsey, (2007), *PM Methodology Discussion Paper* (FOA3a).

³ The FAA SAGE website is as follows: http://www.faa.gov/about/office_org/headquarters_offices/apl/research/models/sage/.

1 **Table A2-3. Estimates of Aircraft Black Carbon Emissions.**

	Aircraft	Engine(s)	Measurement Condition	EIBC g/kg fuel
Kinsey (2009)	DC8 Various air frames APEX 1 to 3	CFM56-2C1 CFM56-7B24 CFM56-3B1/3B2 CFM56-3B1 RB211-535E4B	Non-LTO thrust levels at Sea Level Static (SLS)	0.021, 0.026, 0.032 0.028, 0.025 0.092 0.098 0.275
Petzold (1999a)	ATTAS	Rolls-Royce/SNECMA M45HMk501	Non-LTO thrust levels at SLS In-flight	0.118-0.149 0.11-0.15
Pueschel(1997)	Concorde	Olympus 593	In-flight 16300 m altitude	0.07-0.11
Petzold (1999b)	B737-300	CFM56-3B1	In-flight 7925 m altitude	0.01
"	A310-300	CF6-80C2A2	In-flight 10670 m altitude	0.021
"	VFW 614	M45H	In-flight 7925 m altitude	0.07-0.11
Lilenfield (1995)	DC8	GE 404	All thrust levels at SLS	0.03-0.4
Anderson (1998)	Multiple	Multiple	In-flight; mass EIs estimated from number EIs based on average particle volume and mass	0.01-0.35
Doppelheuer (2001)	n/a	CF6-50C2	Modeled Cruise Simulation - DLR Method using empirical calculation	0.015

2

3

1 **A2.3 Development of International Emissions Inventories for Black and Organic** 2 **Carbon**

3 There are a number of methodological differences between the approaches used to compile
4 domestic and international inventories. Specifically, in contrast to EPA's method of using emission
5 factors paired with activity levels to estimate BC and OC emissions, the most widely used global
6 emissions inventory described in Chapter 2 (Bond et al., 2004) incorporates other factors to derive
7 estimated BC emissions, including fuel type, combustion source technology type, and emissions
8 controls. There is extensive usage data on emissions from specific vehicles and engines in the United
9 States which is used for input for EPA models. These data exists to a lesser extent outside the United
10 States. Fuel consumption data, which are often used as a substitute in global inventory calculations, are
11 also useful but do not have the detail that vehicle/engine usage data have.

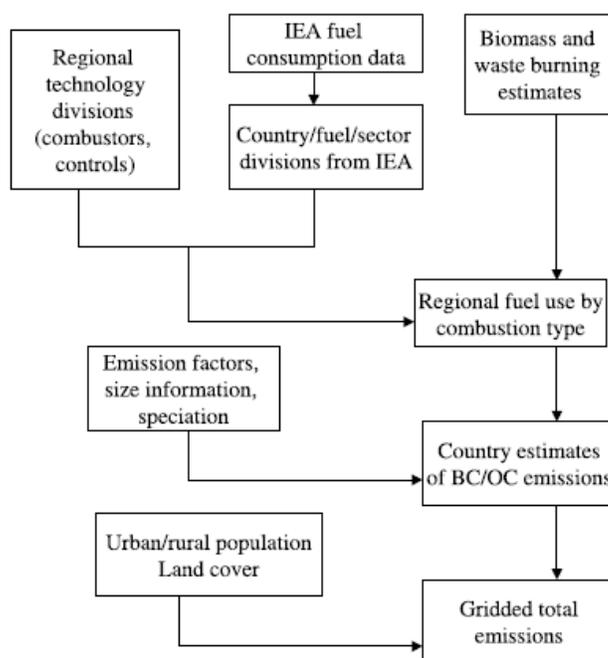
12 Like the U.S. inventory, global inventories typically rely on Equation 1 (outlined above), for
13 estimating BC and OC emissions. That is, country-specific emission factors are combined with
14 appropriate activity level information to yield an estimate of emissions. For example, onroad cars,
15 trucks, buses, and all onroad mobile sources are generally assessed through travel-based emission
16 factors and vehicle miles traveled (VMT). This approach associates mobile source emissions with traffic
17 patterns, providing spatial and temporal information about the distribution of emission that can be used
18 in a variety of applications (for example, air quality modeling). However, motor vehicle emission factors
19 are highly variable and uncertain because of different vehicle types, ages, maintenance, and operating
20 conditions (Cadle et al., 2005). Fuel composition data often can be obtained more easily and accurately
21 than activity measures such as VMT. Fuel-based emission factors for fossil fuel and bio-fuel combustion,
22 for example, can be derived easily from diluted in-plume measurements, using simultaneous CO₂
23 measurements to determine dilution ratios and to relate other pollutants to the weight of carbon in the
24 consumed fuel (Reference needed). Fuel-based emission factors are very common in global and large-
25 scale inventories where detailed information on source activity is very limited.

26 Global BC and OC inventories are complicated by a lack of specific, detailed information on
27 source types, emission factors, activities, and controls, especially in the developing world. In such cases
28 a simple equation for calculating emissions based solely on emission factors and activity levels cannot be
29 applied rigorously. Therefore, certain proxies have been used to estimate BC and OC emissions.

30 Specific Approaches used in Global Inventories

31 Most global inventories attempt to estimate BC emissions, even though some of the emission
32 factors used seemingly represent testing on EC. In Chapter 2, Table 2.X compared Alternative Global
33 Estimates of BC/EC and OC Emissions from Combustion (in Million Tons). There are important
34 differences in the way these various global estimates were generated, resulting in some variation in the
35 total emissions estimates generated in different studies. It is useful to compare the approaches in more
36 detail.

1 The Bond inventories, which are the most extensively used global inventories, were
 2 characterized in Chapter 2. Bond et al. (Bond, 2004) identified about 50 different combinations of fuel
 3 type and usage and subdivided these into processes with different emission characteristics. This
 4 approach is based on combining fuel composition data and assumptions of combustion technologies and
 5 emissions controls, and is very similar to earlier work done in the literature (Klimont et al., 2002).
 6 Emissions for a fuel/sector combination are calculated as an aggregate of the contributions of all
 7 technologies within that sector. The total emissions for each country, in turn, are the sum across all
 8 fuel/sector combinations. The reader is referred to published literature for more details on the methods
 9 used and the uncertainties inherent in their methodology (Bond, 2004; Bond, 2007). An overview of the
 10 Bond estimation procedure is given in Figure 2A.1 below. Using this method, global BC emissions were
 11 estimated at about 8.9 million tons per year, with an uncertainty range of 4.8 -24 million tons/year. The
 12 United States accounted for about 6% of global BC emissions in this inventory.



13

14 **Figure A2-1. Bond et al. Methodology for Developing Emission Estimates (from Bond 2004).**

15 Other authors have compiled inventories based on alternative methods. Penner et al. (1993)
 16 looked at developing BC emission in two ways: first, based on fuel consumption estimates, and second
 17 based on BC/SO₂ ratios. In examining the relationship between ambient BC and SO₂ concentrations in
 18 urban areas around the world, the authors found strong correlations in source areas and also that
 19 various sources had characteristic BC/SO₂ ratios. Site-specific BC/SO₂ ratios were transformed to BC
 20 emissions using available SO₂ emission estimates for each country/world region. The result was a global
 21 BC emission estimate of about 26 million tons per year from urban fuel use. Penner (1993) also
 22 calculated global BC emissions on the basis of fuel consumption, assuming constant emission factors for
 23 commercial and domestic coal, diesel fuel, wood, and bagasse combustion, yielding a total of 14 million
 24 tons BC/yr, with 7.3 million tons/year from fossil fuel combustion and 6.7 million tons/year from wood

1 and bagasse fuel burning. Even though global estimates from the two methods differed by a factor of
2 about 2, larger differences were found for individual countries and regions.

3 Cooke and Wilson (1996) compiled published estimates for biomass areal density, amounts
4 above ground, the fractions burned, and emission factors for different fuel types (E.g., forests, savanna).
5 Agricultural burning and biomass combustion for heating and cooking were not included. Country-
6 specific fuel consumption rates were compiled for industrial, domestic, and combined sectors for solid,
7 liquid, and gaseous fuels. Country-wide emissions were distributed to grids according to population
8 density. The Cooke and Wilson global BC inventory of about 15 million tons was comprised of 9 million
9 tons and 6 million tons from fossil fuel and biomass combustion, respectively. The fossil fuel component
10 of 9 million tons was approximately one-third that estimated by Penner using the BC/SO₂ ratio approach
11 (i.e. 26 million tons), but similar to emissions based on Penner's fuel consumption approach (i.e., 7.3
12 million tons from fossil fuel combustion).

13 Liousse et al. (1996) reported global BC and OM (organic mass) emissions. OM is divided by 1.3
14 and converted to measured elements in organic molecules. Their inventory includes categories for
15 biomass (i.e., savanna and forest fires), agricultural waste, wood fuel, and dung combustion as well as
16 domestic coal and diesel fuel combustion. Global fossil fuel combustion (7.3 million tons BC /year) and
17 biomass burning (6.2 million tons EC /year) total 13.5 million tons EC/year, lower than the estimate of
18 15 million tons BC/year from Cooke and Wilson that excluded agricultural burning and biomass
19 combustion for fuel and energy production. Liousse et al. also estimated global OC emissions of 69
20 million tons/year, with 24 million tons OC/year from fossil fuel and 45 million tons/year from biomass
21 burning.

22 Then, Cooke et al. (1999) refined Cooke and Wilson (1996) by considering the relative ages of
23 vehicles in developed and developing countries and particle size differences for controlled and
24 uncontrolled combustion processes. The estimated global BC inventory from fossil fuel combustion of 7
25 million tons in Cooke et al. was consistent with estimates of 9 million tons BC/year by Cooke and Wilson
26 and 7.3 million tons BC/year by Liousse et al.. The Cooke et al. estimate of global OC emissions was
27 about 11 million tons, about half that of Liousse et al. (24 million tons OC/year).

28 Cofala et al (Cofala, 2007) have used a global version of the Regional Air Pollution Information
29 and Simulation (RAINS) model to estimate anthropogenic emissions of BC and OC (along with numerous
30 other pollutants). The authors rely on the RAINS methodology (Klimont, 2002) for particle emissions,
31 which they modify to capture regional and country-specific characteristics of BC and OC emissions as
32 laid out in other references (Kupiainen and Klimont, 2007) and extend it to developing regions with data
33 from Bond (Bond, 2004). Their methods result in an estimate of 6 million tons of BC emitted globally in
34 1995 and about 5.9 million tons of BC emitted in 2000.

35

1 Specific Approaches Used in Regional Inventories

2 Chapter 2 also discusses the information available from alternative inventories available for
 3 specific countries or world regions. Table 2A.2 provides a comparison of key differences in data and
 4 methods between some of these inventories.

5 **Table A2-4. Regional Inventories of BC and OC Emissions.**

Carbon Emission Inventory Reference	Region and Resolution (Base Year)	Emission Source Categories	Source of Emission Factors	Source of Activity Information
Reddy and Venkataraman (2002a; Fossil Fuel) Reddy and Venkataraman (2002b; Biomass Combustion) Black Carbon and Organic Carbon	India .25 x .25 (1996-1997)	4 utilities 5 coal combustion 8 industrial 2 residential/commercial 8 transportation 4 biomass/biofuels burning	Literature review U.S. EPA AP-42 Customized emission factors to fit Indian technology	Fossil fuel consumption from Central Board of Irrigation and Power (1997), Cement Manufacturers' Association (1999), Centre for Monitoring of India Economy (1999), The Fertilizer Association of India (1998), Ministry of Coal, Ministry of Industry (1998), and Ministry of Petroleum and Natural Gas (1998), Statistics for Iron and Steel Industry in India (Steel Authority of India, 1998). Biofuel consumption in rural and urban from Tata Research Institute (Joshi et al., 1992) and National Sample Survey (1996). Forest coverage from Forest Survey of India (1998).
Streets et al. (2003) Black Carbon Only	Asia 1 x 1 to .08 x .08 (2000)	Each of the 22 Asia countries (plus international shipping) has power generation, industry, and domestic sectors divided into 3 categories (i.e. coal, oil, or biofuel, and other), 10 transportation categories, and 3 biomass burning categories.	Literature review U.S. EPA AP-42 MOBILE5 model	RAINS-Asia simulation (2000 forecast from the 1995 base year), except for China which was based on Sinton and Fridley (2000) on a provincial basis. For the transportation sector, used World Road Statistics (International Road Federation, 2000) and World Motor Vehicle Data (American Automobile Manufacturers Association 1998).
Cao et al. (2006) Black Carbon and Organic Carbon	China .2 x .2 (2000)	Includes 5 sectors (i.e. power generation, industry, residential, transportation, and	Literature review Laboratory tests of biofuel emissions from cooking stoves	Point source activity from State Power Corporation of China (2001) and Editorial Board of China Rural Energy Yearbook (2001). Area sources activity from National

		biomass burning) separated by 363 large point (including 285 power plants) and area sources (e.g. population, gross domestic product) with 18 different sector-fuel type combination.		Bureau of Statistics and various government agencies, mainly at the county level
Streets et al. (2001) Black Carbon	China (will supply later)	Covers 37 different source types over five sectors (power, industry, residential, transport, field combustion) and 13 different fuel types, including biofuels	Literature Review	Fuel consumption data by sector and fuel type were developed within the framework of the RAINS-Asia model (Foell et al., 1995; Downing et al., 1997). Generated by China's Energy Research Institute.
Parashar et al. (2005) Black Carbon and Organic Carbon	India (will supply later)	Fossil fuel, biofuel, and biomass burning	Literature Review Laboratory tests of biofuels and soft coke	Fossil fuel consumption from TEDDY 2001/2002, biofuel use and biomass burning taken from Reddy and Venkataraman (2002).
Sahu et al. (2008) Black Carbon	India 1 x 1 (2001)	Categorized into area sources and LPS and then by fuel type	Literature review	Activity data collected from Central Electricity Authority (CEA), Census of India, Ministry of Coal, Ministry of Road Transport and Highways, Ministry of Agriculture.
Dickerson et al. (2002) Black Carbon	Asia (will supply later)	Same as 37 different source types identified in Streets et al. (2001)	Literature Review MOBILE 5, also used CO/BC ratio to estimate emissions	RAINS-Asia model, Tata Energy Research Institute
Lamarque et al. (2010) Black Carbon and Organic Carbon	Global .5 x .5 (2000)	12 different sectors over 40 different regions	Literature Review (Bond and Liousse combination emission factors)	Biomass from RETRO, GICC, and GFEDv2 inventories, ship data from International Maritime Organization (IMO) aircraft data from AERO2K database, EUROCONTROL, Bond et al. (2007) and Junker and Liousse (2008)
Mitra et al. (2002) Black Carbon and Organic Carbon	India	7 different fuel types	Literature Review (from Cooke 1999)	
Mayol-Bracero et al. (2002) Black Carbon	India	8 different fuel types across four (five?)	Literature Review (Streets et al. 2001)	RAINS-Asia model

		sectors		
Ohara et al. (2007) Black Carbon and Organic Carbon	Asia .5 x .5 (2000)	Four sectors broken down into fuel type (coal, oil, biofuel, others)	Literature Review U.S. EPA AP-42	LPS activity data from China State Grid Company, RAINS-Asia, Fuel consumption from International Energy Agency (IEA) or UN Energy Statistics Yearbook. Biofuel consumption from Yan et al. (2006), RAINS-Asia
Derwent et al (2001)	Western Europe (1995-1998)	All sectors	Back calculated using dispersion modeling and ambient data	NA

1

2 A comparison among the regional inventories listed in Table 2A.2 yields some interesting
3 information. It is also helpful to compare the emissions estimates from these inventories with estimates
4 in the appropriate portion(s) of the Bond/Streets global inventory. Specifically:

- 5 • Cao et al., 2006 used emission factors from Cooke’s 1999 global inventory and Streets’ inventory
6 of China BC emissions for the year 2000, along with Andreae and Merlet 2001 emission factors
7 for biofuels. They developed a local inventory based on specific emission factors for crop straw
8 used in cooking stoves by testing five different types of straw in a combustion tower designed to
9 simulate Chinese cooking stoves. Their national BC emissions estimate for 2000 in China is
10 about 1.7 tons, which is somewhat higher than the Bond estimate of 1.6 million tons and the
11 Streets estimate of 1.2 million tons. The authors attributed this to their inclusion of coal
12 combustion in rural industry and rural residential sources, which they noted are often
13 underestimated in more global estimates. They outlined the residential and industrial sectors as
14 being the most important in contributing to Chinese BC emissions.
- 15 • Streets et al., 2001 measured Chinese BC emissions for 1995, using mostly emission factors from
16 other literature sources. Their inventory focuses on submicron black carbon emissions rather
17 than bulk emissions, because submicron emissions are more relevant to radiative transfer
18 calculations. Their study noted also that most inventories assume that the fraction of BC that
19 makes up PM_{2.5} remains constant throughout the combustion process. Streets et al 2001 states
20 that smoldering combustion, while releasing a higher amount of particulate matter, does not
21 exhibit temperatures high enough to produce the same proportion of BC. They propose a
22 negative correlation between particulate emissions and the fraction of BC emitted. Other
23 observations by this study were that removal efficiency of particles for the industrial sector is
24 lower and less documented than that of the power sector, making emissions from the Chinese
25 industrial sector more uncertain and variable and that domestic emissions in China are
26 responsible for over 80% of Chinese BC emissions. The final estimate of BC emissions was about
27 1.5 million tons, which is lower than the Bond estimate of 1.6 tons but higher than emissions

- 1 estimated via other regional Asian studies REAS and TRACE-P (1.3 million tons and 1.0 million
2 tons, respectively).
- 3 • Reddy and Venkataraman in 2002 estimated BC emissions in India for the year 1996 by
4 developing emission factors using Indian fuel composition and indigenous pollution control
5 technology. Emission factors of coal for the power and industrial sector were derived from
6 those of the EPA. Domestic emission factors were taken from Indian literature sources (Gray,
7 1986). Transportation factors were taken from an average of PM emission factors from
8 countries with similar transportation statistics. The study noted that most of the BC emissions
9 from India were from the transportation sector (almost 60%). Overall they predicted BC
10 emissions from 1996 – 1997 to be 0.36 million tons annually. This estimate is lower than almost
11 all other estimates of Indian BC emissions. The authors claimed their lower estimate was due to
12 different emission factors – other emission factors used in more global inventories were too
13 high due to improper differentiation of fuel composition, combustion type, and PM_{2.5}
14 composition differences.
 - 15 • Parashar et al., (2005) estimated Indian BC emissions for 1995 and used Bond emission factors
16 for fossil fuel combustion. Biomass combustion emission factors were determined by actually
17 combusting different types of fuels in a U shaped chimney. They found that dung cakes released
18 a particularly high amount of particles due to a smoldering combustion, which releases more
19 particles than other types of burning. Their final emissions estimate for India was 0.92 million
20 tons BC, which is higher than Bond’s estimate of 0.64 million tons of BC. The higher emissions
21 could be due to the higher emission factor they found for dung cakes, which accounted for a
22 higher than proportional amount of domestic emissions.
 - 23 • Mitra et al estimated Indian BC emissions for the year 1996 using Cooke’s 1999 emission factors
24 for “under developed” countries. They only calculated emissions for four fossil fuel categories:
25 coal, diesel, gasoline, kerosene. Their annual emission estimate was approximately 1.1 million
26 tons of BC. This is much higher than Bond’s estimate of 0.63 million tons BC in India. The higher
27 emissions could be due to the use of emission factors for under developed countries, in that
28 Bond et al. in their estimates may have used emission factors from more developed countries to
29 represent fossil fuel combustion characteristics in India.
 - 30 • Sahu et al., estimated BC emissions from India for the years 1991 and 2001. They used Cooke’s
31 1999 emission factors for “under developed countries” for fossil fuel combustion and Reddy’s
32 regionalized 2002 emission factors for biofuel combustion. Their final estimate of BC emission in
33 India was about 1.5 million tons per year, higher than any other inventory despite the fact that
34 they did not inventory small industry. Their high estimates could be due to using under-
35 developed country emission factors for all fossil fuel combustion sectors and also from use of
36 diesel activity information which did not represent current conditions.
 - 37 • Mayol-Bracero et al calculated BC emissions in India using the Chinese emission factors
38 developed by Streets et al (Streets, 2001) and national activity data from the GAINS model.

1 Their final estimate of BC emissions in India was 0.5 million tons, slightly lower than Bond's
2 estimate of about 0.6 million tons.

- 3 • Ohara et al developed REAS (Regional Emission Inventory for many parts of Asia) for several
4 pollutants including black carbon for the period 1980 – 2003. Emissions were calculated as a
5 product of activity data, emission factors, and removal efficiency of controls. BC emission
6 factors were taken from Streets et al 2003, and characterized into developed countries and
7 countries with no known emission controls (this category included India and China). The
8 emission factors for developed countries changed several times over the time period of the
9 inventory. Chinese BC emissions in 2000 were 1.2 million tons and Indian emissions were
10 estimated to be about 0.9 million tons. These numbers compare fairly well to other estimates
11 for those countries. The inventory noted the domestic sector as the main contributor to BC
12 emissions.

- 13 • Dickerson used two different approaches to measuring BC emissions in India and other South
14 Asian countries. They first did a bottom up inventory using emission factors and activity level
15 data. They assumed that South Asia source types of BC were similar to that of China, and they
16 obtained energy use information from the RAINS-Asia model. For residential biofuel
17 combustion, they used the emission factor 1 g/kg, which was taken from measurements in the
18 literature (Muhlbaier 1982) and were similar to that used in the Reddy and Venkataraman study
19 outlined earlier. Their estimates of BC emissions from power plants were lower than Reddy and
20 Venkataraman because of smaller emission factors due to a high level of ash in the particulate
21 emissions. Indian vehicles were assumed to be similar to Chinese vehicles; the authors used
22 emission factors from Streets' 2001 work. Final BC estimate for India was 0.56 million tons.
23 Their estimates differ from Penner 1993, Cooke and Wilson 1996, and Cooke 1999, as well as
24 Bond et al, because of possible inclusion of ash in the emission factors, omission of biofuels, and
25 difference in time periods. For the second method, CO emissions were used as surrogates to
26 estimate BC emissions. They found that that total BC emissions for India using this method were
27 more on the order of 2 million tons. The team concluded that bottom up inventory estimates
28 produce much smaller values of BC emissions than do actual in field observations (or "top
29 down" estimates), which could imply errors in calculating these inventories "bottom up."

- 30 • Zhang et al., 2009 focus on the INTEX-B mission, the goals of which were to quantify transport
31 and evolution of Asian pollution to North America and assess its implications for regional air
32 quality and climate. The inventory improved China's emission estimates by balancing the spread
33 of new and old technology in China's industrial sector and improving energy statistics. For other
34 Asian countries, the mission used IEA energy statistics and emission factors documented in
35 Klimont et al 2002. The INTEX-B mission also incorporated inventories that were thought to be
36 more accurate representations of individual countries, such as the Indian inventory from Reddy
37 et al 2002, the Japan inventory from Kannari et al 2007, and the South Korean inventory from
38 Park. China emissions were 1811 Gg and India emissions were 344 Gg for 2006. INTEX-B also
39 included small industry emissions, but noted that they were uncertain of the numbers. The

1 authors also noted that for Southeast Asia, the activity level data was extrapolated and there
2 were few local emission factors, so the data may not be very accurate. This mission was seen as
3 an improvement on the previous TRACE-P due to the updated methodology and collaboration
4 with local inventory efforts.

(This page intentionally left blank)

Appendix 3. Studies Estimating Global and Regional Health Benefits of Reductions in Black Carbon

Geographic Scale	Results of Study	Mitigating Measures	Pollutants	Reference
<u>Studies of Mitigation Strategies for Ambient Reductions in BC</u>				
Global, Arctic	Fossil fuel soot (FS) and bio-fuel soot and gases (BSG) contribute to global warming, with FS being the greater contributor per unit mass. However, BSG may contribute 8 times more in premature mortalities than FS due to the population exposures of potential reductions	Elimination of global anthropogenic FS and BSG.	PM _{2.5} from fossil fuel soot, biofuels soot, & methane	Jacobson (2010)
Global	Avoid 240,000 annual premature mortalities in China, 30,000 elsewhere globally. Find reductions in sulfates, OC, and BC collectively lead to loss in net negative radiative forcing.	50% reduction in China's 2030 SO ₂ , OC, BC emissions from 2000 levels	SO ₂ , OC, BC	Saikawa et al. (2009)
Global	Halving global anthropogenic BC emissions avoids 157,000 premature deaths annually worldwide, the vast majority of which occur within the source region. Most of the avoided deaths are achieved by halving East Asia emissions, but South Asian emissions have 50% greater mortality impacts per unit BC emitted than East Asian emissions. Residential and industrial emissions contribute disproportionately to mortality due to co-location with global population. About 8 times more avoided deaths estimated when anthropogenic BC+OC emissions halved compared with halving BC alone.	50% reductions in anthropogenic BC emissions globally, from 8 major world regions, and from 3 major economic sectors (residential, industrial, transportation).	BC, OC	Anenberg et al. (in preparation)
Global	Implementing all measures would avoid 1-5 million PM _{2.5} and O ₃ -related premature deaths annually based on 2030 population, with the majority achieved by the technical and non-technical BC measures. About 80% of the avoided deaths occur in Asia. Avoided deaths occur regardless of simultaneous implementation of low-carbon CO ₂ measures.	Suite of methane mitigation measures, "technical" BC mitigation measures (ex. Improving coke ovens and brick kilns and increasing use of diesel particulate filters), and "non-technical" BC mitigation measures (ex. eliminating high-emitting vehicles, banning open burning of agricultural waste, and eliminating biomass cook stoves in developing countries).	Particles and gases	UNEP Short-lived Climate Forcers report (in preparation)

<u>Studies of Indoor and Ambient Mitigating Strategies for BC</u>				
China, India, Africa	Benefits of mitigation exceeded costs by factors of 3.6 to 13.6 to one	Improved stoves in China and India for domestic heating and cooking, coal to briquette use for domestic cooking and heating, and community forestry programs to control savannah and open burning in Africa	BC, OC, SO ₂	Baron et al. (2009)
China, India	Find BC mitigating strategies involving indoor BC stove emissions provide as well as certain strategies for diesel BC emission reductions in urban cities are win-win opportunity for climate and public health.	Indoor reduction of BC from replacement of stoves used for cooking and home heating, and strategies to reduce BC emission from diesel vehicles used in urban cities.	BC, OC, SO ₂	Kandlikar et al. (2009)
<u>Indoor Mitigating Strategies to Reduce BC</u>				
China	Climate and human health benefits to cost ratio of 6 with about 69% of these benefits associated with human health.	Household fuel intervention	BC, OC, SO	Smith et al. (2008)
India and UK	Low emission stoves in India result in 12,500 fewer DALYs ¹ annually and energy efficiency in the UK households result in 850 DALYs per year. .	Energy efficiency in UK household heating and, 150 million low-emission cookstoves in India	BC, OC, sulfates	Wilkinson et al. (2009)

¹ Disability-adjusted life years (DALYs) is a measure of overall disease-burden expressed as the number of years lost due to ill-health, disability or early death.

1

2 **Appendix 4. Efforts to Limits Diesel Fuel Sulfur Levels**

3

4

5

6 As discussed in Chapter 7, the availability of low-sulfur diesel fuel is imperative for many
7 emissions control strategies. Sulfur in fuel will poison the catalysts that are built into passive DPFs, thus
8 rendering them ineffective. DPFs work ideally with 50 ppm or less sulfur diesel fuel (“low-sulfur diesel”).
9 Thus, nations that have adopted low sulfur requirements for diesel fuel of 50 ppm or less are best
10 positioned to adopt more stringent emission standards for new motor vehicles, and have more flexibility
11 to target emissions from in-use vehicles. Nations with established standards of 500 ppm or less have
12 more limited institutional and technological potential for further reductions. Nations with nominal or no
13 limits on sulfur in diesel fuel are unable to adopt technology-based standards or controls on in-use
14 engines that would offer significant reductions in elemental carbon.

14

15

16

17

18

19

20

21

22

23

24

25

26

27

28

29

30

31

32

33

34

35

36

37

38

39

40

41

- 15 Aside from the United States, Canada, Japan, and the European Union, 50 ppm or less sulfur
16 diesel fuel is not common. Only a few metropolitan areas in developing Asia have 50 ppm sulfur diesel
17 available (USAID, 2010). However, several countries around the world have adopted schedules that
18 require the use of lower sulfur diesel fuel between 2010 and 2015:
- 19 • Africa: Morocco established limits of 50 ppm in 2009, and Tunisia will require 50 ppm fuel in
20 2014-2015.
 - 21 • Americas and Caribbean: Mexico adopted ULSD in 2009 nationwide, while Chile and Brazil have
22 mandated ULSD in urban areas between 2009 and 2013. Several other nations have established
23 requirements requirements for diesel fuel with 50 ppm sulfur, either nationwide (Columbia
24 2013, Chile 2010, Uruguay 2010) or in large urban areas (Argentina 2012, Colombia 2010).
 - 25 • Caucasus and Central Asia: Armenia and Kazakhstan both introduced requirements for 10 ppm
26 diesel fuel in 2010. Georgia adopted national standards for 50 ppm diesel fuel in 2010.
 - 27 • East Asia and Pacific Islands: Malaysia required 50 ppm diesel fuel in 2010 and is requiring 10
28 ppm diesel fuel in 2015. Singapore, Malaysia, and the Republic of Korea have established
29 national sulfur standards of 50 ppm in diesel fuel between 2007 and 2010. Thailand is limiting
30 diesel fuel to 50 ppm sulfur in 2012. Malaysia and the Republic of Korea plan to adopt 10-15
31 ppm sulfur limits between 2010 and 2015.
 - 32 • Eastern Europe: Albania and Belarus plan to require 10 ppm sulfur in diesel fuel in 2011-2012.
33 Croatia, Russia, and Turkey have adopted standards of 50 ppm between 2008 and 2010, though
34 numerous fuel grades continue to be sold.
 - 35 • South Asia: China limits diesel sulfur to 50 ppm in Beijing (2008), Hong Kong, and Macao; diesel
36 fuel in Taiwan is limited to 50 ppm sulfur after 2005 and 10 ppm starting in 2011. For selected
37 urban areas, India is requiring the use of 50 ppm sulfur diesel fuel in 2010.
 - 38

- 1 • Southwest Asia / Middle East: Israel required 10 ppm sulfur in diesel fuel in 2009, while Qatar is
2 requiring it in 2012. Saudi Arabia and Syria will require 50 ppm fuel in 2014-2015.

3
4 Numerous other countries have established diesel sulfur limits of 500 ppm prior to 2015, including
5 Azerbaijan, Brazil (outside urban areas), Ecuador, Fiji, India, Malawi, Mozambique, Oman, Pakistan, the
6 Philippines, South Africa, Sri Lanka, Thailand, Vietnam, and Zimbabwe.

7
8 Among nations with less stringent standards on fuel sulfur (e.g, 2,000-10,000 ppm) in either all
9 or part of their territory, some have lowered the limits in recent years. For example, outside urban
10 areas, Argentina and Peru are reducing allowable sulfur to 1500 ppm between 2010 and 2012, from
11 levels of 2500-3000 ppm introduced in 2006. Venezuela reduced allowable sulfur from a standard of
12 5,000 ppm established to a new standard of 2,000 ppm in 2010. Notable among nations of sub-Saharan
13 Africa, Mauritius established a diesel fuel sulfur standard of 2500 in 2001. Moving to lower sulfur levels
14 in these regions is hampered by economic and technical barriers.

15
16 Among nations without sulfur standards, some include oil producing nations, such as Egypt, Iran,
17 and Kuwait. Many sub-Saharan African nations lack national sulfur standards. In the former Soviet
18 Union, many central Asian countries base their national standards on Russia's GOST 305/82 standard for
19 diesel fuel (2,000 ppm). Nevertheless, some nations have diesel fuel with sulfur levels that meet the
20 national standards of countries from which they export. For example, diesel fuel in Lesotho, Namibia,
21 Swaziland, and Botswana meets the 500 ppm national standard established in South Africa, from which
22 they import their fuel.

23
24 Through the Partnership for Clean Fuels and Vehicles (PCFV)
25 (<http://www.unep.org/transport/pcfvl/>), UNEP continues to work with developing nations to identify
26 opportunities and build capacity to establish lower sulfur levels. For example, the PCFV holds
27 workshops in Africa, Asia, and the Americas, gathering local scientists, engineers, and officials to discuss
28 scientific evidence and economic impacts of how diesel fuel sulfur levels affect cities in developing
29 countries. These meetings follow on PCFV's successful campaign to eliminate lead in gasoline, which
30 recently celebrated the complete phase-out of lead in African gasoline.

31
32 Several regional intergovernmental agreements have also been signed by representatives at the
33 ministerial level. In February 2008, environmental ministerial officials from Latin America and the
34 Caribbean in Santo Domingo, Dominican Republic agreed to promote sulfur reduction in fuel throughout
35 the region, with a target goal of 50 ppm. In July 2009, several west and central African environmental
36 ministers signed a regional framework agreement on air pollution, including goals to adopt 3500 ppm
37 fuel sulfur limits by the end of 2011, with a goal of 50 ppm fuel by 2020. Though non-binding on
38 governments, these agreements suggest that there is significant impetus to reduce sulfur levels in fuels
39 used in the developing world.

40

1 In addition to governmental and intergovernmental efforts to reduce diesel fuel sulfur levels,
2 several private sector initiatives also exist. Vehicle industries around the world have recognized the
3 value of reduced sulfur for enabling lower-emissions vehicles and high-efficiency combustion
4 technologies. In 2002, vehicle and engine manufacturers from the U.S, Europe, and Japan published a
5 report on worldwide fuels harmonization, which promoted lower sulfur levels in gasoline and diesel fuel.
6 More recently, the African Refiners Association has developed a set of “AFRI” fuel specifications (AFRI-1
7 through AFRI-4) as a developmental pathway for African development of ≤ 50 ppm sulfur.

8
9 Table A4-1 below gives recent information on national standards for on-road diesel sulfur limits,
10 and estimates of current sulfur levels. In addition to the efforts described above, Chapter 5 also
11 mentions the limits on sulfur content of marine fuel being phased in under requirements from the IMO.
12 Table A4-2 below provides details regarding the fuel sulfur levels allowed for C3 marine fuel within ECAs
13 and globally outside of ECAs, and the schedule for phase-in of tighter limits on sulfur content of this fuel.

14
15

1 **TABLE A4-1**

INTERNATIONAL REGULATIONS AND INTERNATIONAL AGREEMENTS ON DIESEL FUEL SULFUR LEVELS (LEVELS IN PPM SHOWN)

Region	Country	Year																				
		2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Americas and Caribbean	Mexico (National)	?	?	?	?	?	?	500	500	500	15	15										
	Mexico (Northern)	?	?	?	?	?	?	300	15	15	15	15										
	Argentina (Urban)	?	?	?	?	?	?	1500	1500	1500	1500	(500)	(500)	50								
	Argentina (Non-Urban)	?	?	?	?	?	?	2500	2500	2500	2500	2500	2500	1500								
	Barbados	No existing or planned standards																				
	Bolivia	?	?	?	?	?	?	?	?	?	?	?	(2000)									
	Brazil (Non-Urban)	?	3500	3500	3500	3500	3500	3500	3500	3500	3500	1800	1800	1800	1800	500						
	Brazil (Metropolitan)	?	2000	2000	2000	2000	500	500	500	500	500	50	50	50	10							
	Chile (National)	?	?	?	?	?	?	?	350	350	350	350	50									
	Chile (Santiago)	?	?	?	?	?	?	?	50	50	50	50	10									
	Colombia (National)	?	?	?	?	?	?	?	2500	2500	2500	500			50							
	Colombia (Bogota)	?	?	?	?	?	?	?	?	?	?	?	50									
	Costa Rica	No existing or planned standards																				
	Cuba	No existing or planned standards																				
	Dominican Republic																					
	Ecuador (National)	?	?	?	?	?	?	?	(500)	500	500	50										
	Ecuador (Urban)	?	?	?	?	?	?	5000														
	Ecuador (Non-Urban)	?	?	?	?	?	?	7000														
	El Salvador	?	?	?	?	?	?	?	?	?	?	?	500									
	Guatemala	?	?	?	?	?	?	?	?	?	?	?	(500)									
Honduras	?	?	?	?	?	?	?	?	?	?	?	(500)										
Panama	?	?	?	?	?	?	?	?	?	?	?	(1000)										
Peru (Urban)	?	?	?	?	?	?	?	1500	1500	1500	1500	50										
Peru (Non-Urban)	?	?	?	?	?	?	?	3000	3000	3000	3000	1500										
Uruguay	?	?	?	?	?	?	?	8000	8000	8000	8000	50										
Venezuela	?	?	?	?	?	?	?	5000	5000	5000	5000	2000										
East Asia and Pacific Islands	Cambodia	2000	?	?	?	?	?															
	China (National)	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	350										
	China (Beijing)	?	?	?	?	?	?	?	?	?	50	50	50	50	10							
	China (Hong Kong)	500	50	50	50	50	50	50	50	50	50	50										
	Fiji	?	?	?	?	?	?	?	500													
	Indonesia	5000	5000	5000	5000	5000	5000	5000	3500	3500	3500	(500)										
	Japan	100	100	100	100	100	100	50	50	50	50	50										
	Malaysia	3000	3000	(500)	(500)	(500)	(500)	(500)	(500)	(500)	(500)	(500)	50	50	50	50	10					
	Nepal	10000	10000	10000	10000	10000	10000	10000	10000	10000	10000	10000										
	Philippines	?	?	?	?	?	?	?	?	?	?	?	500									
	Republic of Korea	500	500	500	500	500	500	500	500	500	500	500	50									
	Singapore	500	500	500	500	500	500	500	500	50	50	50	50									
	Thailand	500	500	500	500	500	500	500	500	500	500	500	500	500	50	50						
Vietnam	10000	10000	2000	2000	2000	500	500	500	500	500	500	500										
Eastern Europe, Caucasus, and Central Asia	Albania	?	?	?	?	?	?	?	?	?	?	350	150	10								
	Belarus	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Bosnia & Herzegovina	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Croatia	?	?	?	?	?	?	?	?	?	?	?	50	50	50	10						
	Russia	?	?	?	?	?	?	?	?	?	?	?	500	350	50							
	Turkey	?	?	?	?	?	?	?	?	?	?	?	50/1000	50/1000	50/1000							
	Armenia	?	?	?	?	?	?	?	?	?	?	?	350	50	10							
	Azerbaijan	?	?	?	?	?	?	?	?	?	?	?	?	?	?	2000	2000	2000	2000	2000	500	
	Georgia	?	?	?	?	?	?	?	?	?	?	?	?	350	50							
	Kazakhstan	?	?	?	?	?	?	?	?	?	?	?	?	2000	2000	10						
Kyrgyzstan	?	?	?	?	?	?	?	?	?	?	?	(350)	(350)	(350)								
Serbia	?	?	?	?	?	?	?	?	?	?	?	?	?	?	10000	(350)						
Uzbekistan	?	?	?	?	?	?	?	?	?	?	?	?	?	?	5000							
South Asia	Afghanistan	No existing or planned standards																				
	Bangladesh	?	?	?	?	?	?	5000														
	India	2500	2500	2500	2500	2500	500	500	500	500	500	350										
	India (Selected Areas)						350	350	350	350	350	50										
	Nepal	10000	10000	10000	10000	10000	10000	(500)	(500)	(500)	(500)	(500)	(350)									
Pakistan	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?		
Sri Lanka	10000	10000	10000	3000	?	?	?	?	?	?	?	?	?	?	500							
Southwest Asia and North Africa	Algeria	No existing or planned standards																				
	Bahrain	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Egypt	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Iran	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Iraq	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Israel	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Jordan	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Kuwait	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Lebanon	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
	Libya	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	
Morocco	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?	?		

2
3
4

1

TABLE A4-2 International Fuel Sulfur Limits for C3 Marine Fuel, by Target Year

Global		ECA	
2004	45,000 ppm	2005	15,000 ppm
2012	35,000 ppm	2010	10,000 ppm
2020	5,000 ppm	2015	1,000 ppm

2

(This page intentionally left blank)

Appendix 5. U.S. Emission Standards for Mobile Sources

The tables below show the various mobile source PM standards in the United States, including the more detailed standards based on engine type and size applicable to certain categories, such as nonroad diesel engines.

On-Road Diesel Regulations for PM Control

Regulation	Sector	Model Years Applicable	PM standard	units
Tier 0 rule	light duty diesel trucks 1	1981-1993	0.26	g/mile
Tier 0 rule	light duty diesel trucks 2	1981-1993	0.13	g/mile
1985 Heavy Duty Diesel Rule	heavy-duty highway CI engines	1988-90	0.60	g/bhp-hr
1991 Heavy Duty Diesel Rule	heavy-duty highway CI engines	1991-1993	0.25	g/bhp-hr
1991 Heavy Duty Diesel Rule	urban buses	1991-1993	0.10	g/bhp-hr
Tier 1 rule	light duty diesel trucks 1 and 2	1994-1999	0.10	g/mile
1994 Heavy Duty Diesel Rule	heavy-duty highway CI engines	1994-2006	0.10	g/bhp-hr
1994 Heavy Duty Diesel Rule	urban buses	1994-1995	0.07	g/bhp-hr
1994 Heavy Duty Diesel Rule	urban buses	1996-2006	0.05	g/bhp-hr
2001 Heavy Duty Diesel Rule	heavy duty onroad CI engines	2007+	0.01	g/bhp-hr
NLEV rule	light duty diesel LEV cars and trucks	1999-2003	0.08	g/mile
NLEV rule	light duty diesel ZLEV cars and trucks	1999-2003	0.04	g/mile
2000 Tier 2 Rule	LDV LLDT bins 2-6	2004	0.01	g/mile
2000 Tier 2 Rule	LDV LLDT - 50% bins 2-6	2005	0.01	g/mile
2000 Tier 2 Rule	LDV LLDT - 75% bins 2-6	2006	0.01	g/mile
2000 Tier 2 Rule	LDV LLDT - bins 2-6	2007	0.01	g/mile
2000 Tier 2 Rule	LDV LLDT - 25% bins 7-8	2004	0.02	g/mile
2000 Tier 2 Rule	LDV LLDT - 50% bins 7-8	2005	0.02	g/mile
2000 Tier 2 Rule	LDV LLDT - 75% bins 7-8	2006	0.02	g/mile
2000 Tier 2 Rule	LDV LLDT - bins 7-8	2007	0.02	g/mile
2000 Tier 2 Rule	LDV LLDT - bin 9	2004-2006	0.06	g/mile
2000 Tier 2 Rule	LDV LLDT - bin 10	2004-2006	0.08	g/mile

On-Road Gasoline-Vehicle Regulations

Regulation	Sector	Model Years Applicable	PM standard	units
2000 Tier 2 Rule	Onroad Gasoline - 25% bins 2-6	2004	0.01	g/mile
2000 Tier 2 Rule	Onroad Gasoline - 50% bins 2-6	2005	0.01	g/mile
2000 Tier 2 Rule	Onroad Gasoline - 75% bins 2-6	2006	0.01	g/mile
2000 Tier 2 Rule	Onroad Gasoline - bins 2-6	2007	0.01	g/mile
2000 Tier 2 Rule	Onroad Gasoline - 25% bins 7-8	2004	0.02	g/mile
2000 Tier 2 Rule	Onroad Gasoline - 50% bins 7-8	2005	0.02	g/mile
2000 Tier 2 Rule	Onroad Gasoline - 75% bins 7-8	2006	0.02	g/mile
2000 Tier 2 Rule	Onroad Gasoline - bins 7-8	2007	0.02	g/mile
2000 Tier 2 Rule	Onroad Gasoline - bin 9	2004-2006	0.06	g/mile
2000 Tier 2 Rule	Onroad Gasoline - bin 10	2004-2006	0.08	g/mile
2001 Heavy Duty Diesel Rule	Heavy Duty Onroad Gasoline 50%	2008	0.01	g/bhp-hr
2001 Heavy Duty Diesel Rule	Heavy Duty Onroad Gasoline 100%	2009	0.01	g/bhp-hr

Nonroad Diesel Regulations

Regulation	Sector	Model Years Applicable	PM standard	Units
2004 Nonroad Tier 4	Nonroad Diesel, hp<25	2008	0.3	g/bhp-hr
2004 Nonroad Tier 4	Nonroad Diesel, 25<=hp<75	2008	0.22	g/bhp-hr
2004 Nonroad Tier 4	Nonroad Diesel, 75<=hp<175	2012	0.01	g/bhp-hr
2004 Nonroad Tier 4	Nonroad Diesel, 175<=hp<750	2011	0.01	g/bhp-hr
2004 Nonroad Tier 4	Nonroad Diesel, hp750	2011	0.075	g/bhp-hr
2004 Nonroad Tier 4	Nonroad Diesel, hp750 generator sets	2015	0.02	g/bhp-hr
2004 Nonroad Tier 4	Nonroad Diesel, hp750, all other eqpt	2015	0.03	g/bhp-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, kW<8	2000	1	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, kW<8	2005	0.8	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 8<=kW<19	2000	0.8	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 8<=kW<19	2005	0.8	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 19<=kW<38	1999	0.8	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 19<=kW<38	2004	0.6	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 37<=kW<75	2004	0.4	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 75<=kW<130	2003	0.3	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 130<=kW<225	1996	0.54	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 130<=kW<225	2003	0.2	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 225<=kW<450	1996	0.54	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 225<=kW<450	2001	0.2	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 450<=kW<560	1996	0.54	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, 450<=kW<560	2002	0.2	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, kW>560	2000	0.54	g/kW-hr
1998 Nonroad Diesel Engines Rule	Nonroad Diesel, kW>560	2006	0.2	g/kW-hr

Locomotive Regulations

Regulation	Sector	Model Years Applicable	PM standard	Units
2008 LocoMarine Rule	Locomotive - Line Haul	Tier 0 (1973-1992)	0.22	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Switch	Tier 0 (1973-2001)	0.26	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Line Haul	Tier 1 (1993-2004)	0.22	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Switch	Tier 1 (2002-2004)	0.26	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Line Haul	Tier 2 (2005-2011)	0.10	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Switch	Tier 2 (2005-2010)	0.13	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Line Haul	Tier 3 (2012-2014)	0.10	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Switch	Tier 3 (2011-2014)	0.10	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Line Haul	Tier 4 (2015+)	0.03	g/bhp-hr
2008 LocoMarine Rule	Locomotive - Switch	Tier 4 (2015+)	0.03	g/bhp-hr
1997 Locomotive Stds	Locomotive - Line Haul	Tier 0 (1973-2001)	0.60	g/bhp-hr
1997 Locomotive Stds	Locomotive - Switch	Tier 0 (1973-2001)	0.72	g/bhp-hr
1997 Locomotive Stds	Locomotive - Line Haul	Tier 1 (2002-2004)	0.45	g/bhp-hr
1997 Locomotive Stds	Locomotive - Switch	Tier 1 (2002-2004)	0.54	g/bhp-hr
1997 Locomotive Stds	Locomotive - Line Haul	Tier 2 (2005+)	0.20	g/bhp-hr
1997 Locomotive Stds	Locomotive - Switch	Tier 2 (2005+)	0.24	g/bhp-hr

2008 LocoMarine Rule - Tier 3	C1 Commercial Stnd Power Density, <19kW Max power, <0.9L/cylinder	2009+	0.30	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Commercial Stnd Power Density, 19<75kW Max power, <0.9L/cylinder	2009+	0.22	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Commercial Stnd Power Density, 75<3700kW Max power, <0.9L/cylinder	2012	0.10	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Commercial Stnd Power Density, 75<3700kW Max power, 0.9<1.2L/cylinder	2013	0.09	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Commercial Stnd Power Density, 75<3700kW Max power, 1.2<2.5L/cylinder	2014	0.08	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Commercial Stnd Power Density, 75<3700kW Max power, 2.5<3.5L/cylinder	2013	0.08	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Commercial Stnd Power Density, 75<3700kW Max power, 3.5<7.0L/cylinder	2012	0.08	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Rec and Com High Power Density, <19kW Max power, <0.9L/cylinder	2009+	0.30	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Rec and Com High Power Density, 19<75kW Max power, <0.9L/cylinder	2009+	0.22	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Rec and Com High Power Density, 75<3700kW Max power, <0.9L/cylinder	2012	0.11	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Rec and Com High Power Density, 75<3700kW Max power, 0.9<1.2L/cylinder	2013	0.10	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Rec and Com High Power Density, 75<3700kW Max power, 1.2<2.5L/cylinder	2014	0.09	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Rec and Com High Power Density, 75<3700kW Max power, 2.5<3.5L/cylinder	2013	0.09	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C1 Rec and Com High Power Density, 75<3700kW Max power, 3.5<7.0L/cylinder	2012	0.08	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C2,<3700kW,7-<15L/cylinder	2013	0.10	g/bhp-hr
2008 LocoMarine Rule - Tier 3	C2,<3700kW,15-<30L/cylinder	2014	0.20	g/bhp-hr
2008 LocoMarine Rule - Tier 4	C1&C2, >3700 kW	2014	0.09	g/bhp-hr
2008 LocoMarine Rule - Tier 4	C1&C2, >3700 kW	2016	0.04	g/bhp-hr
2008 LocoMarine Rule - Tier 4	C1&C2, 600 to <3700	2014	0.03	g/bhp-hr

Commercial Marine Regulations

Regulation	Sector	Model Years Applicable	PM standard	Units
1999 C1&C2 Marine Engine Rule	C1, power >=37 kW disp. <0.9	2005+	0.40	g/kW-hr
1999 C1&C2 Marine Engine Rule	C1, 0.9<=disp.< 1.2	2004+	0.30	g/kW-hr
1999 C1&C2 Marine Engine Rule	C1, 1.2<=disp.<2.5	2004+	0.20	g/kW-hr
1999 C1&C2 Marine Engine Rule	C1, 2.5<=disp.<5.0	2007+	0.20	g/kW-hr
1999 C1&C2 Marine Engine Rule	C2, 5.0<=disp.<15.0	2007+	0.27	g/kW-hr
1999 C1&C2 Marine Engine Rule	C2, 15.0<=disp.<30.0	2007+	0.50	g/kW-hr

(This page intentionally left blank)

Appendix 6. International Emission Standards for Heavy-Duty Vehicles

Heavy-duty on-road diesel vehicles represent the predominant mobile source of BC in most areas although nonroad diesel (and locomotives and commercial marine can also be significant). The following discussion addresses emission standards in other parts of the world.

Outside the U.S., Europe, and Japan, other nations adopt heavy-duty engine emission standards developed by these governments using schedules determined by legislative or executive standards. As noted earlier, Canada generally adopts U.S. standards on a timeframe similar to the U.S. Australia also bases its national standards on those developed in the U.S., Europe, or Japan. Outside these nations, other countries adopt emission standards, generally based on European standards, albeit on a different time frame. As discussed in Appendix 3, countries must ensure that fuel quality is requisite to allow emissions-reduction technologies to be implemented.

A number of countries have adopted schedules for phasing in PM emission standards for heavy-duty diesel engines that are likely to require advanced aftertreatment, such as a DPF, to meet the relevant national standard. In the Americas, Brazil's PROCONVE P7 standards beginning in the 2012 model year are likely to require advanced aftertreatment. Russia has adopted standards based on EURO IV starting in the 2010 model year and standards based on EURO V in the 2014 model year. In the Beijing area, China adopted standards equivalent EURO IV in 2008, and has proposed adoption of EURO V-equivalent standards in 2012). In addition, several countries that have applied for membership in the European Union will adopt EURO standards if accepted. These countries include Croatia, Iceland, Macedonia, and Turkey. Other potential candidate countries that have not formally petitioned for EU membership include Albania, Bosnia and Herzegovina, Kosovo, Montenegro, and Serbia.

Numerous other countries have adopted or proposed heavy-duty engine emission standards equivalent to earlier U.S. or EURO emission standards. In the Americas, these countries include Argentina, Brazil, Chile, Mexico, and Peru. In the western Pacific and Asia, these countries include China, India, the Republic of Korea, Singapore, and Thailand. In Europe outside of the European Union, Russia and Turkey have adopted earlier EURO standards. These countries are making progress in reducing BC emissions from heavy-duty vehicles.

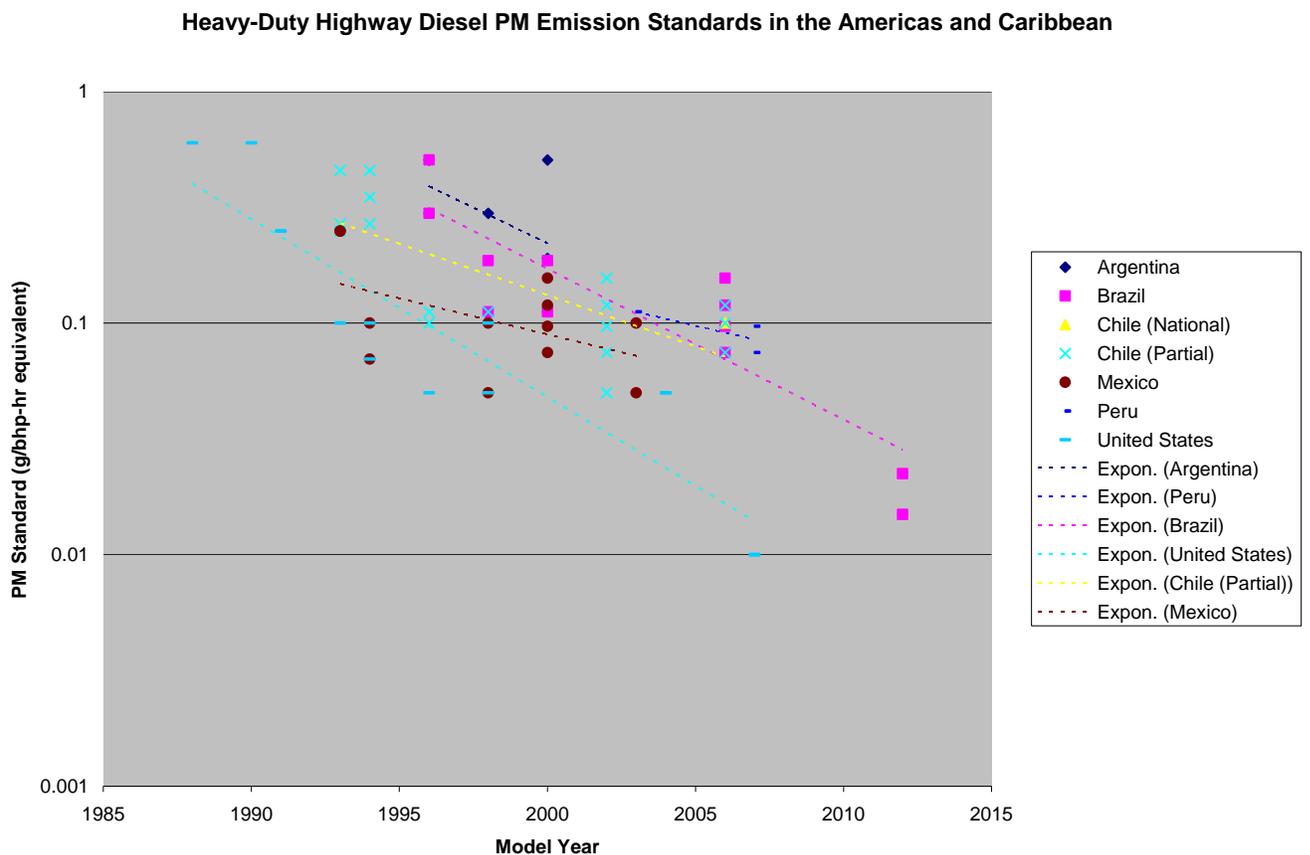
Figures A6-1, A6-2, and A6-3 show a graph of how PM emission standards are changing over time in the Americas, Asia and Australia, and Europe, respectively. The figures also include trend lines, indicating the general trend of emission standards over time. As illustrated, most countries with emission standards in place have introduced progressively more stringent standards over time. The

1 scatter around the trendline of each country reflects differences in standards based on vehicle type
 2 (truck vs. bus), test procedure (e.g., operating cycle), and/or location (e.g., urban vs. rural).

3
 4 Beyond nations that have regulations with emission standards, other nations have been
 5 addressing vehicle emissions in some manners. Some other nations are adopting emission standards for
 6 light-duty vehicles, generally based on EURO standards. Others have eliminated or are scheduled to
 7 eliminate lead from gasoline, which enables the implementation of standards to reduce tailpipe
 8 emissions using catalytic aftertreatment. An example of this progress is found in Africa, where all
 9 nations have eliminated lead in gasoline. Others have banned the import of light-duty vehicles without
 10 a catalytic converter or established opacity testing requirements for cars, trucks, or scooters. This
 11 progress suggests room for additional technology-based approaches to reducing BC emissions.

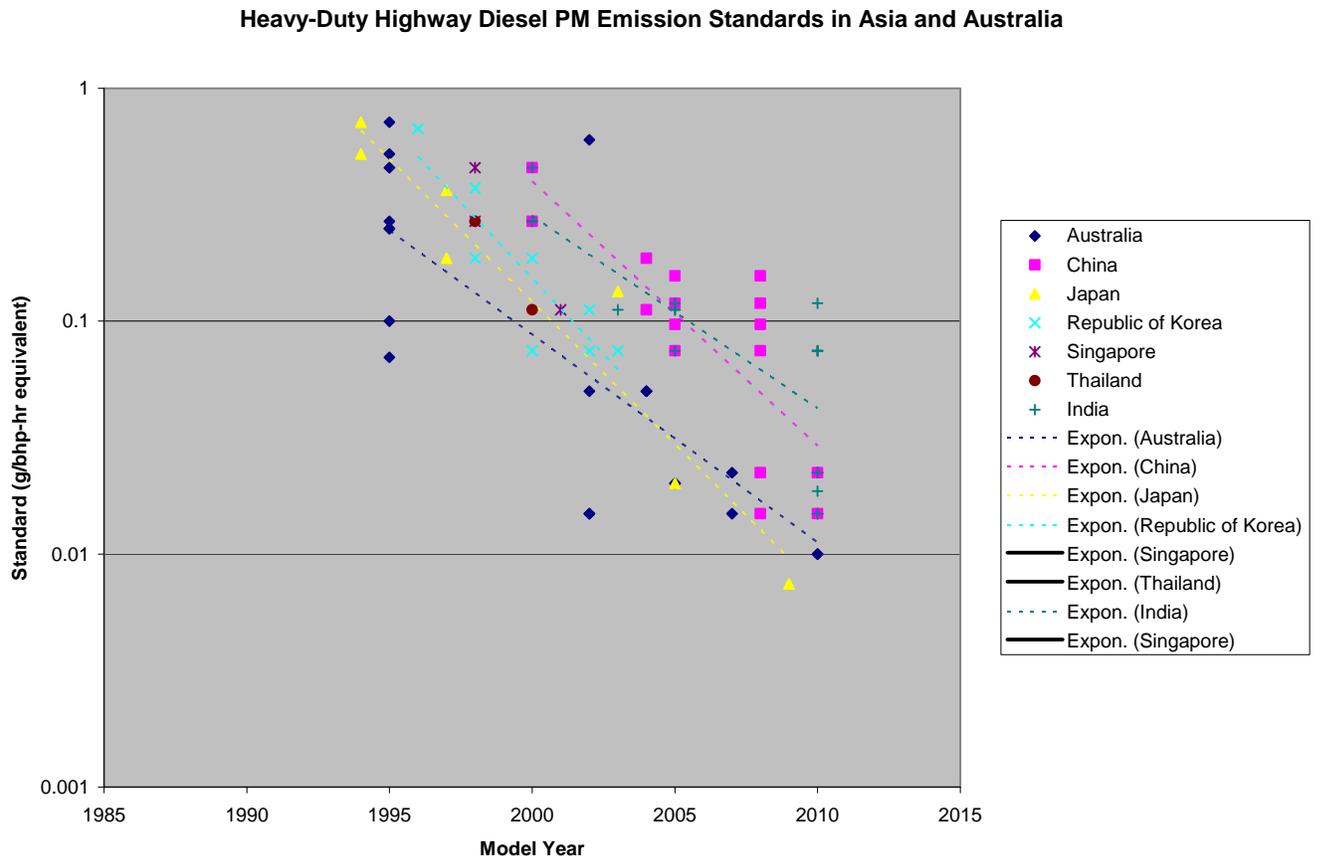
12
 13 Many other countries lack any emission standards. The reasons for their lack of emission
 14 standards may be attributable to several causes, including insufficient governmental capacity, poverty
 15 and other economic factors, and government policy. Many such countries face many other problems
 16 related to economic development, public health, violence, and authoritarian rule. Addressing BC from
 17 motor vehicle emissions in these locations may requires attention to factors other than technology.

18
 19 Figure A6-1 (Logarithmic Scale)
 20

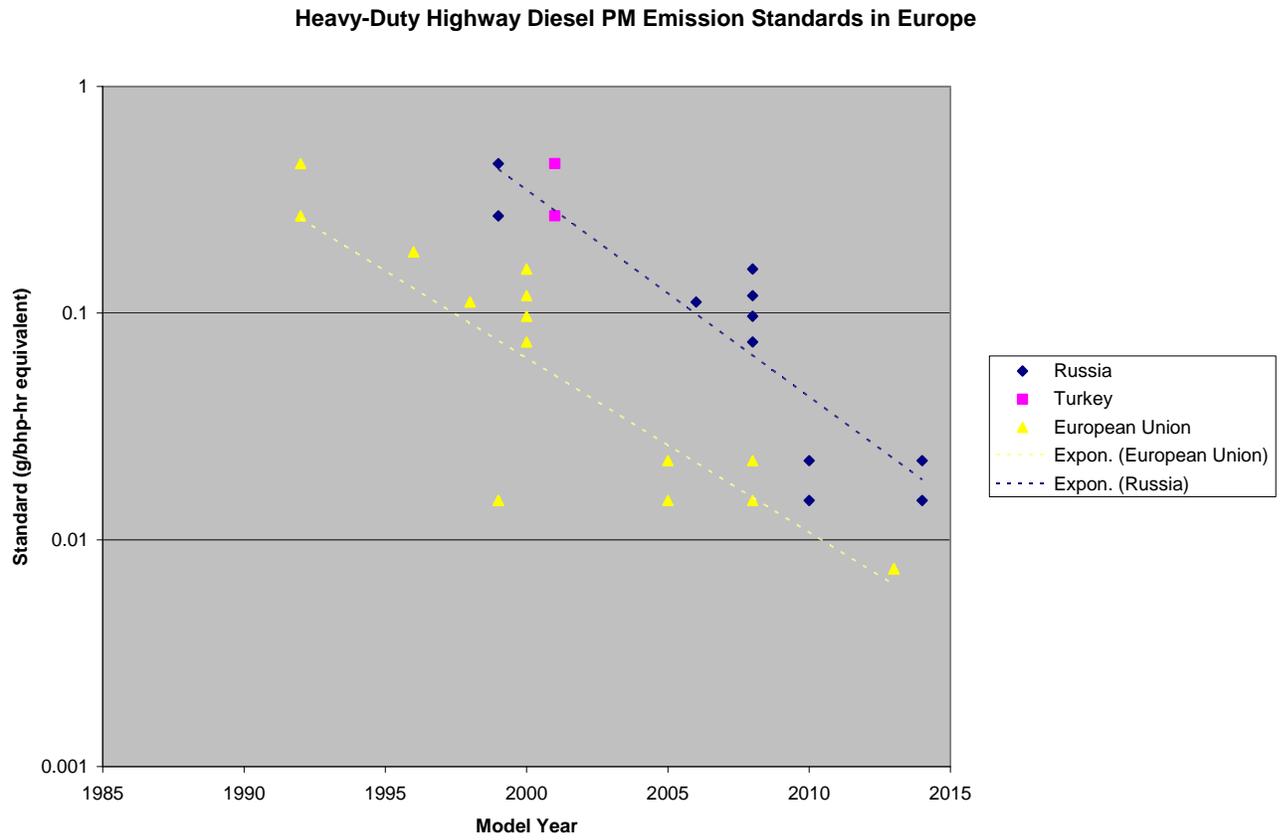


21
 22

1 Figure A6-2 (Logarithmic Scale)
2
3



1
2 Figure A6-3 (Logarithmic Scale)
3



4

Bibliography

- 1
- 2
- 3 Abel S.J., Haywood J.M., Highwood E.J., Li J., and Buseck P.R. (2003) Evolution of biomass burning
4 aerosol properties from an agricultural fire in southern Africa. *Geophys. Res. Lett.* **30** (15), page
5 1783 (doi:10.1029/2003GL017342).
- 6 Abt Associates (2005) BenMAP Technical Appendices. Prepared for the Office of Air Quality Planning and
7 Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. Available on the
8 Internet at <http://www.epa.gov/air/benmap/models/BenMAPappendicesSept08.pdf>.
- 9 ACES (Advanced Collaborative Emissions Study) (2009) Phase 1 of the Advanced Collaborative Emissions
10 Study. Available on the Internet at
11 [http://www.crcao.com/reports/recentstudies2009/ACES%20Phase%201/ACES%20Phase1%20Fi
12 nal%20Report%2015JUN2009.pdf](http://www.crcao.com/reports/recentstudies2009/ACES%20Phase%201/ACES%20Phase1%20Final%20Report%2015JUN2009.pdf).
- 13 ACIA (2004) *Impacts of a warming Arctic: Arctic climate impact assessment*, Cambridge University Press.
- 14 Ackerman A.S., Toon O.B., Stevens D.E., Heymsfield A.J., Ramanathan V., and Welton E.J. (2000)
15 Reduction of tropical cloudiness by soot. *Science* **288**.
- 16 Adachi K. and Buseck P.R. (2008) Internally mixed soot, sulfates, and organic matter in aerosol particles
17 from Mexico City. *Atmos. Chem. Phys.* **8**, 6469-6481. Available on the Internet at
18 <http://www.atmos-chem-phys.net/8/6469/2008/acp-8-6469-2008.pdf>.
- 19 Adams T.T., Walsh J., Brown M., Goodrum J., Sellers J., and Das K. (2002) A demonstration of fat and
20 grease as an industrial boiler fuel. by the University of Georgia Engineering Outreach Service,
21 Athens, GA.
- 22 Adar S.D., Gold D.R., Coull B.A., Schwartz J., Stone P.H., and Suh H. (2007) Focused exposures to airborne
23 traffic particles and heart rate variability in the elderly. *Epidemiology* **18** (1), 95-103.
- 24 Adkins, et al. (2010) Field testing and survey evaluation of household biomass cook stoves in rural sub-
25 Saharan Africa. *Energy for Sustainable Development* **14**, 172-185.
- 26 Adler G., Riziq A.A., Erlick C., and Rudich Y. (2010) Effect of intrinsic organic carbon on the optical
27 properties of fresh diesel soot. *Proceedings of the National Academy of Sciences* **107**, 6699-
28 6704.
- 29 Adler T. (2010) Better burning, better breathing: improving health with cleaner cook stoves.
30 *Environmental Health Perspectives* **118** (3).
- 31 Alaska Department of Environmental Conservation (2010) 2008 Alaska wildland fire emissions inventory.
32 prepared by Alaska Department of Environmental Conservation, Division of Air Quality, October
33 6. Available on the Internet at
34 [http://fire.ak.blm.gov/content/admin/awfcg_committees/Air%20Quality%20and%20Smoke%20
35 Management/5_2008_AK_WF_EI_rpt_oct10.pdf](http://fire.ak.blm.gov/content/admin/awfcg_committees/Air%20Quality%20and%20Smoke%20Management/5_2008_AK_WF_EI_rpt_oct10.pdf).
- 36 Albrecht B.A. (1989) Aerosols, cloud microphysics, and fractional cloudiness. *Science*.
- 37 Alexander D.T.L., Crozier P.A., and Anderson J.R. (2008) Brown carbon spheres in East Asian outflow and
38 their optical properties. *Science* **321**, 833-836.
- 39 Alfaro S.C., Lafon S., Rajot L., Formenti P., Gaudichet A., and Maille M. (2004) Iron oxides and light
40 absorption by pure desert dust: An experimental study. *J. Geophys. Res.* **109** (D08208)
41 (doi:10.1029/2003JD004374).
- 42 Allen C.D., et al. (2002) Ecological restoration of southwestern Ponderosa Pine ecosystems: a broad
43 perspective. *Ecological Applications* **12** (5).

- 1 Allen J.O., Dookeran N., Smith K.A., and Sarofim A. (1996) Measurement of polycyclic aromatic
2 hydrocarbons associated with size-segregated Atmospheric Aerosols in Massachusetts. *Environ.*
3 *Sci. Technol.* **30**.
- 4 Amador-Muñoz O., Villalobos-Pietrini R., Agapito-Nadales M.C., Munive-Colín Z., Hernández-Mena L.,
5 Sánchez-Sandoval M., Gómez-Arroyo S., Bravo-Cabrera J.L., and Guzmán-Rincón J. (2010)
6 Solvent extracted organic matter and polycyclic aromatic hydrocarbons distributed in size-
7 segregated airborne particles in a zone of México City: Seasonal behavior and human exposure.
8 *Atmospheric Environment* **44** (1), 122-130.
- 9 AMAP (2009) *AMAP 2009 update on selected climate issues of concern* (AMAP Technical Report No. 2).
10 Available on the Internet at <http://www.amap.no/assessment/generalpublic.htm>.
- 11 Anderson B.E., Cofer W.R., Bagwell D.R., Barrick J.W., Hudgins C.H., and Brunke K.E. (1998) Airborne
12 observations of aircraft aerosol emissions I: total nonvolatile particle emission indices.
13 *Geophysical Research Letters* **25** (10), 1689-1692.
- 14 Andreae M.O. and Gelencsér A. (2006) Black carbon or brown carbon? The nature of light-absorbing
15 carbonaceous aerosols. *Atmospheric Chemistry and Physics* **6**, 3131-3148.
- 16 Andreae M.O. (2009) A new look at aging aerosols. *Science* **326** (5959), 1493-1494
17 (doi:10.1126/science.1183158). Available on the Internet at
18 <http://www.sciencemag.org/content/326/5959/1493.short>
- 19 Anenberg S.C., Talgo K., Arunachalam S., Dolwick P., Jang C., and West J.J. Impacts of global, regional,
20 and sectoral black carbon emission reductions on surface air quality and human mortality. (In
21 preparation).
- 22 Anenberg S.C., Horowitz, L. W., Tong D.Q., and West J.J. (2010) An estimate of the global burden of
23 anthropogenic ozone and fine particulate matter on premature human mortality using
24 atmospheric modeling. *Environmental Health Perspectives* **118**.
- 25 Ang K.P., Gunasingham, H., and B. T. Tay (1985) Polycyclic aromatic hydrocarbons in diesel exhaust
26 emissions. *International Journal of Environmental Studies* **26** (1&2), 91-96. Available on the
27 Internet at <http://www.informaworld.com/smpp/content~db=all~content=a770528942>.
- 28 Arctic Council Task Force on Short-Lived Climate Forcers (2011) Progress report and recommendations
29 for ministers, Nuuk Greenland.
- 30 Audet P. and Charest C. (2007) Heavy metal phytoremediation from a meta-analytical perspective.
31 *Environ Pollut* **147**, 231-237.
- 32 Ault A.P., Gaston C.J., Wang Y., Dominguez G., Thiemens M.H., and Prather K.A. (2010) Characterization
33 of the single particle mixing state of individual ship plume events measured at the Port of Los
34 Angeles. *Environ. Sci. Technol.* **44** (6), 1954-1961.
- 35 Aunan K., Berntsen T.K., Myhre G., Rypdal K., Streets D.G., Woo J.-H., and Smith K.R. (2009) Radiative
36 forcing from household fuel burning in Asia. *Atmospheric Environment* (doi:
37 10.1016/j.atmosenv.2009.07.053).
- 38 Babich P., Davey M., Allen G., and Koutrakis P. (2000) Method comparisons for particulate nitrate,
39 elemental carbon, and PM_{2.5} mass in seven U.S. cities. *JAWMA* **50** (7), 1095-1105.
- 40 Baccarelli A., Wright R.O., Bollati V., Tarantini L., Litonjua A.A., Suh H.H., Zanobetti A., Sparrow D.,
41 Vokonas P.S., and Schwartz J. (2009) Rapid DNA methylation changes after exposure to traffic
42 particle. *Am J Respir Crit Care Med.* **179**, 572-578.
- 43 Bachmann J. (2009) *Black carbon: A science/policy primer*, Pew Center on Global Climate Change,
44 Arlington, VA.
- 45 Baduel C., Voisin D., and Jaffrezo J.L. (2009) Comparison of analytical methods for humic-like substances
46 (HULIS) measurements in atmospheric particles. *Atmos. Chem. Phys.* **9**.
- 47 Bae M.-S., et al. (2004) Validation of a semi-continuous instrument for elemental carbon and organic
48 carbon using a thermal-optical method. *Atmospheric Environment* **38**.

- 1 Bae M.-S., Demerjian K.L., and Schwab J.J. (2006) Seasonal estimation of organic mass to organic carbon
2 in PM_{2.5} at rural and urban locations in New York State. *Atmospheric Environment* **40** (39), 7467-
3 7479.
- 4 Bae M.-S., Schauer J.J., Turner J.R., and Hopke P.K. (2009) Seasonal variations of elemental carbon in
5 urban aerosols as measured by two common thermal-optical carbon methods. *Science of the*
6 *Total Environment* **407** (18), 5176-5183 (DOI: 10.1016/j.scitotenv.2009.05.035).
- 7 Bahadur R., Feng Y., Russell L.M., and Ramanathan V. (2011) Impact of California's air pollution laws on
8 black carbon and their implications for direct radiative forcing. *Atmos Environ* **45**, 1162-1167.
- 9 Bailey D.L.R. and Clayton P. (1982) The measurement of suspended particle and total carbon
10 concentrations in the atmosphere using standard smoke shade methods. *Atmos Environ* **16**.
- 11 Bailis R., et al. (2005) Mortality and greenhouse gas impacts of biomass and petroleum energy futures in
12 Africa. *Science* **308**, 98-103.
- 13 Barnes and Kumar (2002) Success factors in improved stoves programmes: lessons from six states in
14 India. *Journal of Environmental Studies and Policy* **5** (2), 99-112.
- 15 Barnett T.P., et al. (2005) Potential impacts of a warming climate on water availability in snow-
16 dominated regions. *Nature* **438** (doi:10.1038/nature04141).
- 17 Baron R.E., Montgomery W.D., and Tuladhar S.D. (2010) An analysis of black carbon mitigation as a
18 response to climate change Prepared by the Copenhagen Consensus on Climate. Available on
19 the Internet at <http://fixtheclimate.com/component-1/the-solutions-new-research/>
- 20 Barth M.J. and Tadi R.R. (1998) Emissions comparison between truck and rail: case study of California I-
21 40. *Transportation Res Record* **1520**, 44-52.
- 22 Bauer S.E., Menon S., Koch D., Bond T.C., and Tsigaridis K. (2010) A global modeling study on
23 carbonaceous aerosol microphysical characteristics and radiative effects. *Atmospheric Chemistry*
24 *and Physics* **10**, 7439-7456.
- 25 Bell M., Ebisu K., Peng R., Samet J., and Dominici F. (2009) Hospital admissions and chemical
26 composition of fine particle air pollution. *Am J Respir Crit Care Med.* **179**, 1115-1120.
- 27 Bell T.L., Rosenfeld D., Kim K.-M., Yoo J.-M., Lee M.-I., and Hahnenberger M. (2008) Midweek increase in
28 U.S. summer rain and storm heights suggests air pollution invigorates rainstorms. *Journal of*
29 *Geophysical Research* **113** (D02209) (doi: 10.1029/2007JD008623).
- 30 Bergstrom R.W., Pilewskie P., Russell P.B., Redemann J., Bond T.C., Quinn P.K., and Sierau B. (2007)
31 Spectral absorption properties of atmospheric aerosols. *Atmos. Chem. Phys.* **7** (5937-5943).
- 32 Berntsen T., Fuglestvedt J., Myhre G., Stordal F., and Berglen T.F. (2006) Abatement of greenhouse
33 gases: does location matter? *Climatic Change* **74** (4).
- 34 Bevan S.L., North P.R.J., Grey W.M.F., Los S.O., and Plummer S.E. (2009) Impact of atmospheric aerosol
35 from biomass burning on Amazon dry-season drought. *J. Geophys. Res.* **114** (D09204) (doi:
36 10.1029/2008JD011112).
- 37 Bice K., Eil A., Habib B., Heijmans P., Kopp R., Nogue J., Norcross F., Sweitzer-Hamilton M., and
38 Whitworth A. (2007) Black carbon: a review and policy recommendations. Technical report
39 prepared by Princeton University, Woodrow Wilson School of Public and International Affairs.
40 Available on the Internet at
41 <http://www.wws.princeton.edu/research/PWReports/F08/wws591e.pdf>.
- 42 Bilger B. (2009) Hearth surgery. *The New Yorker*, 89.
- 43 Birch M.E. and Cary R.A. (1996) Elemental carbon-based method for monitoring occupational exposures
44 to particulate diesel exhaust. *Aerosol Science and Technology* **25**.
- 45 Birch M.E. (1998) Analysis of carbonaceous aerosols: inter-laboratory comparison. *Analyst* **123**.
- 46 Bluestein and et al (2008) Sources and mitigation opportunities to reduce emissions of short-term arctic
47 climate forcers. Technical Report No. 2 prepared by Arctic Monitoring and Assessment Program
48 (AMAP), Oslo, Norway.

- 1 Bonazza A. and et al (2007) Carbon black crusts from the Tower of London. *Environmental Science and*
 2 *Technology* **41** (12).
- 3 Bond T., Streets D., Yarber K., Nelson S., Woo J., and Klimont Z. (2004) A technology based global
 4 inventory of black and organic carbon emissions from combustion. *J. Geophys. Res.* **109**
 5 (doi:10.1029/2003JD003697).
- 6 Bond T., Habib G., and Bergstrom R.W. (2006a) Limitations in the enhancement of visible light
 7 absorption due to mixing state *Journal of Geophysical Research* **111** (D20211).
- 8 Bond T.C. (2001) Spectral dependence of visible light absorption by carbonaceous particles emitted from
 9 coal combustion. *Geophysical Research Letters* **28** (21).
- 10 Bond T.C. and Sun H.L. (2005) Can reducing BC emissions counteract global warming? *Environmental*
 11 *Science & Technology* **39** (16).
- 12 Bond T.C. and Bergstrom R.W. (2006) Light absorption by carbonaceous particles: an investigative
 13 review. *Aerosol Science and Technology* **40** (1), 27-67.
- 14 Bond T.C., Wehner B., Plewka A., Wiedensohler A., Heintzenberg J., and Charlson R.J. (2006b) Climate-
 15 relevant properties of primary particulate emissions from oil and natural gas combustion.
 16 *Atmospheric Environment* **40**, 3574-3587.
- 17 Bond T.C. (2007) Can warming particles enter global climate discussions? *Environmental Research*
 18 *Letters* **2** (4).
- 19 Bond T.C., Bhardwaj E., Dong R., Jogani R., Jung S., Roden C., Streets D.G., and Trautmann. N.M. (2007)
 20 Historical emissions of black and organic carbon aerosol from energy-related combustion.
 21 *Global Biogeochemical Cycles* **21** (GB2018), 1850-2000.
- 22 Bond T.C. (2008) Targeting black carbon for climate reasons: what do we know about emissions, and is it
 23 enough to get started? Presentation to *Princeton University Woodrow Wilson School 59e Policy*
 24 *Workshop, Princeton, NJ.*
- 25 Bond T.C., Zarzycki M.G., Flanner, and Koch D.M. (2009) A science-policy handshake metric for climate
 26 forcing by short-lived pollutants, with applications to black carbon and organic matter. *Environ.*
 27 *Res. Lett.*
- 28 Bond T.C., Zarzycki C., Flanner M.G., and Koch D.M. (2010) Quantifying immediate radiative forcing by
 29 black carbon and organic matter with the Specific Forcing Pulse. *Atmos. Chem. Phys. Discuss.*
 30 **10**, 15713-15753 (doi:10.5194/acpd-10-15713-2010).
- 31 Bond T.C., Zarzycki C., Flanner M.G., and Koch D.M. (2011) Quantifying immediate radiative forcing by
 32 black carbon and organic matter with the Specific Forcing Pulse. *Atmos. Chem. Phys.* **11**, 1505-
 33 1525 (doi:10.5194/acp-11-1505-2011).
- 34 Boucher O. and Reddy M.S. (2008) Climate trade-off between BC and carbon dioxide emissions. *Energy*
 35 *Policy* **36**.
- 36 Boy E., Bruce N., and Delgado H. (2002) Birth weight and exposure to kitchen wood smoke during
 37 pregnancy in rural Guatemala. *Environ Health Perspect* **110**.
- 38 Braun A. and et al (2004) A study of diesel PM with X-ray microspectroscopy. *Fuel* **83**.
- 39 Braun A. (2005) Carbon speciation in airborne particulate matter with C (1s) NEXAFS spectroscopy.
 40 *Journal of Environmental Monitoring* **7**.
- 41 Broadway R.M. and Cass R.W. (1975) Fractional efficiency of a utility boiler baghouse: Nucla Generating
 42 Plant. EPA-600/2-75-013-a (NITS PB 246 641), August.
- 43 Bruce N., Perez-Padilla R., and Albalak R. (2000) Indoor air pollution in developing countries: a major
 44 environmental and public health challenge for the new millennium. *Bull. World Health Organ.*
 45 **78** (9).
- 46 Brunekreef B., Beelen R., Hoek G., Schouten L., Bausch-Goldbohm S., Fischer P., Armstrong B., Hughes E.,
 47 Jerrett M., and van den Brandt P. (2009) Effects of long-term exposure to traffic-related air

- 1 pollution on respiratory and cardiovascular mortality in the Netherlands: the NLCS-AIR Study.
 2 Prepared by the Health Effects Institute, Boston, MA, 139.
- 3 Buonicore A.J. and W.T. Davis (eds.) (1992) *Air pollution engineering manual*. Air & Waste Management
 4 Association, Van Nostrand Reinhold, New York, NY.
- 5 Burt R., Wilson M.A., Mays M.D., and Lee C.W. (2003) Major and trace elements of selected pedons in
 6 the USA. *J Environ Qual* **32**, 2109-2121.
- 7 Buseck P.R. and Pósfai M. (1999) Airborne minerals and related aerosol particles: effects on climate and
 8 the environment. *PNAS* **96** (7).
- 9 Butterfield D., Beccaceci S., Sweeney B., Green D., Alexander J., and Grieve A. (2010) 2009 annual report
 10 for the UK Black Carbon Network. *NPL Report AS-52* (May).
- 11 Buzcu-Guven B., Harriss R., and Hertzmark D. (2010) Gas flaring and venting: extent, impacts, and
 12 remedies. Prepared for the Energy Forum of the James A. Baker III Institute of Public Policy, Rice
 13 University. Available on the Internet at
 14 <http://www.rice.edu/energy/publications/docs/Birnur%20Carbon.pdf>.
- 15 Cabada J.C., Pandis S.N., Subramanian R., Robinson A.L., and Polidori A. (2004) Estimating the secondary
 16 organic aerosol contribution to PM_{2.5} using the EC Tracer Method. *Aerosol Science and*
 17 *Technology* **38** (S1 December).
- 18 Cachier H. and Pertuisot M.H. (1994) Particulate carbon in Arctic ice. *Analusis Magazine* **22**, 34-37.
- 19 Cachier H. (1997) Particulate and dissolved carbon in air and snow at the Summit site. *Transfer of*
 20 *aerosols and gases to Greenland snow and ice*, J.-L. Jaffrezo ed., Cedex, France, 21-27
- 21 Cadle S.H., Groblicki P.J., and Stroup D.P. (1980) Automated carbon analyzer for particulate samples.
 22 *Analytical Chemistry* **52** (13), 2201-2206.
- 23 Cakmak S., Dales R.E., and Vida C.B. (2009) Components of particulate air pollution and mortality in
 24 Chile. *Int J Occup Environ Health* **15**.
- 25 California Air Resources Board (2010) Preliminary discussion paper: proposed amendments to
 26 California's low-emission vehicle regulations – particulate matter mass, ultrafine solid particle
 27 number, and black carbon emissions. *May*.
- 28 Cao G., Zhang X., and et al (2006) Inventory of black carbon and organic carbon emissions from China.
 29 *Atmos Environ* **40** (34), 6516-6527 (CARB, 2010).
- 30 Carlton A.G., Wiedinmyer C., and Kross J.H. (2009) A review of Secondary Organic Aerosol (SOA)
 31 formation from isoprene. *Atmospheric Chemistry and Physics* **9**, 4987-5005. Available on the
 32 Internet at www.atmos-chem-phys.net/9/4987/2009/.
- 33 Carmichael G.R., Adhikary B., Kulkarni S., D'Allura A., Tang Y., Streets D., Zhang Q., Bond T., Ramanathan
 34 V., Jamroensan A., and Marrapu P. (2009) Asian aerosols: current and year 2030 distributions
 35 and implications to human health and regional climate change. *Environ. Sci. Technol.* **43**.
- 36 Casper S. (2008) Using EPA's environmental Benefits Mapping and Analysis Program (BenMAP) for global
 37 health impact analysis. Available on the Internet at
 38 http://www.epa.gov/air/benmap/docs/BenMAP_Global.pdf
- 39 Cass R.W. and Broadway R.M. (1976) Fractional efficiency of a utility boiler baghouse: Sunbury Steam
 40 Electric Station. EPA-600/2-76-077a [NTIS PB253 943], March.
- 41 Cerverny R.S. and Balling R.C. (1998) Weekly cycles of air pollutants, precipitation, and tropical cyclones
 42 in the coastal NW Atlantic region. *Nature* **394**, 561-563.
- 43 Chakrabarty R.K., Moosmuller H., Chen L.-W.A., Lewis K., Arnott W.P., Mazzoleni C., Dubey C.K., Wold
 44 C.E., Hao W.M., and Kreidenweis S.M. (2010) Brown carbon in tar balls from smoldering biomass
 45 combustion. *Atmospheric Chemistry and Physics* **10**, 6363-6370 (doi:10.5194/acp-10-6363-
 46 2010).
- 47 Chameides W.L., Yu H., Liu S.C., Bergin M., Zhou X., Mearns L., Wang G., Kiang C.S., Saylor R.D., Luo C.,
 48 Huang Y., Steiner A., and Giorgi F. (1999) Case study of the effects of atmospheric aerosols and

- 1 regional haze on agriculture: an opportunity to enhance crop yields in China through emission
 2 controls. *PNAS* **96**, 13626-13633.
- 3 Chan T.W., Huang L., Leaitch W.R., Sharma S., Brook J.R., Slowik J.G., Abbatt J.P.D., Brickell P.C., Liggio J.,
 4 Li S.-M., and Moosmuller H. (2010) Observations of OM/OC and specific attenuation coefficients
 5 (SAC) in ambient fine PM at a rural site in central Ontario. *Atmospheric Chemistry and Physics*
 6 **10**. Available on the Internet at www.atmos-chem-phys.net/10/2393/2010/.
- 7 Chandra, et al. (2004) Can the state of mixing of black carbon explain the mystery of ‘excess’
 8 atmospheric absorption? *Geophys. Res. Lett.* **31**.
- 9 Chang O.M.C. and England G.C. (2004) Development of fine particulate emission factors and speciation
 10 profiles for oil and gas-fired combustion systems, update: critical review of source sampling and
 11 analysis methodologies for characterizing organic aerosol and fine particulate source emission
 12 profiles.
- 13 Charlson R.J. (1992) Climate forcing by anthropogenic aerosols. *Science* **255**, 423.
- 14 Chen W.-T., Lee Y.H., Adams P.J., Nenes A., and Seinfeld J.H. (2010) Will BC mitigation dampen aerosol
 15 indirect forcing? *Geophys. Res. Lett.* **37** (L09801) (doi:10.1029/2010GL042886).
- 16 Chen Y. and Bond T.C. (2010) Light absorption by organic carbon from wood combustion. *Atmospheric*
 17 *Chemistry and Physics* **10**. Available on the Internet at [http://www.atmos-chem-](http://www.atmos-chem-phys.net/10/1773/2010/acp-10-1773-2010.pdf)
 18 [phys.net/10/1773/2010/acp-10-1773-2010.pdf](http://www.atmos-chem-phys.net/10/1773/2010/acp-10-1773-2010.pdf).
- 19 Chestnut L.G. and Rowe R.D. (1990) A new national park visibility value estimates. In visibility and fine
 20 particles, transactions of an AWMA/EPA International Speciality Conference. Prepared for the
 21 Air and Waste Management Association, Pittsburg, PA.
- 22 Chi K.H., Chauan Y.L., Chang F.O.Y., Ji L., Neng H., Guey R.S., and Chung T. (2010) PCDD/F measurement
 23 at a high-altitude station in central Taiwan: evaluation of long-range transport of PCDD/Fs
 24 during the Southeast Asia Biomass Burning Event. *Environmental Science & Technology* **44**.
- 25 Chin, et al. (2009) Light absorption by pollution, dust, and biomass burning aerosols: a global model
 26 study and evaluation with AERONET measurements. *Ann. Geophys.* **27**.
- 27 Chin M., et al. (2007) Intercontinental transport of pollution and dust aerosols: implications for regional
 28 air quality. *Atmospheric Chemistry and Physics* **7** (21), 5501-5517.
- 29 Chow J.C., et al. (1993) The DRI thermal optical reflectance carbon analysis system description,
 30 evaluation and applications in United States air quality studies. *Atmos Environ* **27** (Part A-
 31 General Topics).
- 32 Chow J.C., et al. (2001) Comparison of IMPROVE and NIOSH carbon measurements. *Aerosol Science and*
 33 *Technology* **34**.
- 34 Chow J.C., et al. (2007) The IMPROVE-A temperature protocol for thermal/optical carbon analysis:
 35 maintaining consistency with a long-term database. *JAWMA* **57**.
- 36 Chow J.C., Yu J.Z., Watson J.G., Ho S.S.H., Bohannon T.L., Hays M.D., and Fung K.K. (2007) The application
 37 of thermal methods for determining chemical composition of carbonaceous aerosols: a review;
 38 Part A. *Journal of Environmental Science and Health* **42**, 1521-1541.
- 39 Chow J.C., et al. (2008) Advances in integrated and continuous measurements for particle mass and
 40 chemical, composition. *JAWMA* **58**.
- 41 Chow J.C., Watson J.G., Doraiswamy P., Chen L.-W.A., Sodeman D.A., Lowenthal D.H., Arnott W.P., and
 42 Motallebi N. (2009) 2009: Aerosol light absorption, black carbon, and elemental carbon
 43 measurements at the Fresno Supersite, California. *Atmospheric Research* **93** (4), 874-887.
- 44 Chow J.C., Watson J.G., Chen L.-W.A., Rice J., and Frank N.H. (2010a) Quantification of PM_{2.5} organic
 45 carbon sampling artifacts in US networks. *Atmos. Chem. Phys.* **10**, 5223-5239 (doi:10.5194/acp-
 46 10-5223-2010). Available on the Internet at www.atmos-chem-phys.net/10/5223/2010/ and at
 47 <http://www.atmos-chem-phys.org/10/5223/2010/acp-10-5223-2010.pdf>.

- 1 Chow J.C., Watson J.G., Green M.C., and Frank N.H. (2010b) Filter light attenuation as a surrogate for
 2 elemental carbon. *JAWMA* **60** (November).
- 3 Chow J.C., Watson J.G., Lowenthal D.H., Chen L.-W.A., and Motallebi N. (2010c) Black and organic
 4 carbon emission inventories: Review and application to California. *JAWMA* **60** (4), 497-507.
- 5 Chu S.-H. (2005) Stable estimate of primary OC/EC ratios in the ECTracer method. *Atmos Environ* **39**,
 6 1383-1392.
- 7 Chuang K.J., Coull B.A., Zanobetti A., Suh H., Schwartz J., Stone P.H., Litonjua A., Speizer F.E., and Gold
 8 D.R. (2008) Particulate air pollution as a risk factor for ST-segment depression in patients with
 9 coronary artery disease. *Circulation* **118**.
- 10 Chung C., Ramanathan V., Kim D., and Podgorny I. (2005) Global anthropogenic aerosol direct forcing
 11 derived from satellite and ground-based observations. *Journal of Geophysical Research* **110**
 12 (doi: 10.1029/2005JD006356).
- 13 Chung C.E. and Zhang G.J. (2004) Impact of absorbing aerosol on precipitation: Dynamic aspects in
 14 association with convective available potential energy and convective parameterization closure
 15 and dependence on aerosol heating profile. *Journal of Geophysical Research* **109** (D22103)
 16 (doi:10.1029/2004JD004726).
- 17 Chung S.H. and Seinfeld J.H. (2002) Global distribution and climate forcing of carbonaceous aerosols.
 18 *Journal of Geophysical Research* **107** (D19).
- 19 Chung S.H. and Seinfeld J.H. (2005) Climate response of direct radiative forcing of anthropogenic black
 20 carbon. *Journal of Geophysical Research* **110** (D11102) (doi: 10.1029/2004JD005441).
- 21 Chylek P., Srivastava V., Cahenzli L., Pinnick R.G., Dod R.L., Novakov T., Cook T.L., and Hinds B.D. (1987)
 22 Aerosol and graphitic carbon content of snow. *Journal of Geophysical Research* **92**, 9801-9809.
- 23 Chylek P., Johnson B., and Wu H. (1992) Black carbon concentration in a Greenland Dye-3 ice core.
 24 *Geophys. Res. Lett.* **19**, 1951-1953.
- 25 Chylek P., Kou L., Johnson B., Boudala F., and Lesins G. (1999) Black carbon concentrations in
 26 precipitation and near surface air in and near Halifax, Nova Scotia. *Atmos Environ* **33**, 2269-
 27 2277.
- 28 Clarke A.D. and Noone K.J. (1985) Soot in the Arctic snowpack: A cause for perturbations in radiative
 29 transfer. *Atmos Environ* **19**, 2045-2053.
- 30 Clarke L., Edmonds J., Jacoby H., Pitcher H., Reilly J., and Richels R. (2007) Scenarios of greenhouse gas
 31 emissions and atmospheric concentrations, sub-report 2.1A. In *Synthesis and Assessment*
 32 *Product 2.1*, U.S. Climate Change Science Program and the Subcommittee on Global Change
 33 Research. Department of Energy, Office of Biological & Environmental Research, Washington,
 34 D.C., 154
- 35 Clean Air Task Force (2009) The carbon dioxide-equivalent benefits of reducing black carbon emissions
 36 from U.S. Class 8 trucks using diesel particulate filters: a preliminary analysis. Available on the
 37 Internet at <http://www.catf.us/resources/publications/view/100>.
- 38 Cofala J., Amann M., Klimont Z., Kupiainen K., and Hoglund-Isaksson L. (2007) Scenarios of global
 39 anthropogenic emissions of air pollutants and methane until 2030. *Atmos Environ* **41**, 8486-
 40 8499 (doi:10.1016/j.atmosenv.2007.07.010).
- 41 Conny J.M. and Slater J.F. (2002) Black carbon and organic carbon in aerosol particles from crown fires in
 42 the Canadian boreal forest. *Journal of Geophysical Research* **107** (D11), 4116
 43 (doi:10.1029/2001JD001528).
- 44 Conway A., Gades A., and Raymond C.F. (1996) Albedo of dirty snow during conditions of melt. *Water*
 45 *Resources Research* **32** (6), 1713-1718.
- 46 Cooke W.F. and Wilson J.J.N. (1996) A global black carbon aerosol model. *Journal of Geophysical*
 47 *Research* **101** (D14), 19395-19409.

- 1 Coordinating Research Council (2008) Kansas City characterization study. Final Report prepared for the
2 Environmental Protection Agency, Project E69. Based on EPA Contract Report (ERG No.
3 0133.18.007.001).
- 4 Corbett J.J., Lack D.A., Winebrake J.J., Harder S., Silberman J.A., and Gold M. (2010) Arctic shipping
5 emissions inventories and future scenarios. *Atmospheric Chemistry and Physics* **10** (19), 9689
6 (DOI: 10.5194/acp-10-9689-2010).
- 7 Croft B., Lohmann U., and K. von Salzen (2005) Black carbon ageing in the Canadian Centre for Climate
8 modeling and analysis atmospheric general circulation model. *Atmospheric Chemistry and*
9 *Physics* **5**.
- 10 Curry J.A. (1995) Interactions among aerosols, clouds, and climate of the Arctic Ocean. *Sci. Tot. Environ.*
11 **160/161**, 777-791.
- 12 Davis S.C., Diegel S.W., and Boundy R.G. (2009) *Transportation energy data book: edition 28*, Table 2.12,
13 ORNL-6984 (Edition 28 of ORNL-5198), U.S. Department of Energy. Available on the Internet at
14 <http://cta.ornl.gov/data/Index.shtml>.
- 15 Debell L.J., Gebhart K.A., Malm W.C., Marc L. P., Schichtel B.A., and White W.H. (2006) Spatial and
16 seasonal patterns and temporal variability of haze and its constituents in the United States:
17 Appendix A - annual average reconstructed fine mass and aerosol extinction budgets for each
18 site. Report IV, November.
- 19 Delfino R.J., Staimer N., Gillen D., Tjoa T., Sioutas C., Fung K., George S.C., and Kleinman M.T. (2006)
20 Personal and ambient air pollution is associated with increased exhaled nitric oxide in children
21 with asthma. *Environ Health Perspect* **114**.
- 22 Delfino R.J., Staimer N., Tjoa T., Polidori A., Arhami M., Gillen D.L., Kleinman M.T., Vaziri N.D., Longhurst
23 J., Zaldivar F., and Sioutas C. (2008) Circulating biomarkers of inflammation, antioxidant activity,
24 and platelet activation are associated with primary combustion aerosols in subjects with
25 coronary artery disease. *Environ Health Perspect* **116** (7), 898-906.
- 26 Delfino R.J., Staimer N., Tjoa T., Gillen D.L., Polidori A., Arhami M., Kleinman M.T., Vaziri N.D., Longhurst
27 J., and Sioutas C. (2009) Air pollution exposures and circulating biomarkers of effect in a
28 susceptible population: Clues to potential causal component mixtures and mechanisms. *Environ*
29 *Health Perspect* **117**.
- 30 Delfino R.J., Gillen D.L., Tjoa T., Staimer N., Polidori A., Arhami M., Sioutas C., and Longhurst J. (2010a)
31 Electrocardiographic ST segment depression and exposure to traffic-related aerosols in elderly
32 subjects with coronary artery disease. *Environ Health Perspect* (In Press).
- 33 Delfino R.J., Rjoa T., Gillen D.L., Staimer N., Polidor A., Arhami M., Jamner L., Sioutas C., and Longhurst J.
34 (2010b) Traffic-related air pollution and blood pressure in elderly subjects with coronary artery
35 disease. *Epidemiology* **21**, 396-404.
- 36 Denissenko M.F., Pao A., Tang M.S., and Pfeiffer G.P. (1996).: Preferential formation of benzo[a]pyrene
37 adducts at lung cancer mutational hotspots in P53. *Science* **274**.
- 38 Denman K.L., Brasseur G., Chidthaisong A., Ciais P., Cox P.M., Dickinson R.E., Hauglustaine D., Heinze C.,
39 Holland E., Jacob D., Lohmann U., Ramachandran S., Dias P.L.d.S., Wofsy S.C., and Zhang X.
40 (2007) Couplings between changes in the climate system and biogeochemistry. In *Climate*
41 *Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth*
42 *Assessment Report of the Intergovernmental Panel on Climate Change*, S. Solomon, D. Qin, M.
43 Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor, and H.L. Miller eds., Cambridge University
44 Press, Cambridge, United Kingdom and New York, NY, USA
- 45 Dickerson R.R., Andreae M.O., Campos T., Mayol-Bracero O.L., Neusuess C., and Streets D.G. (2002)
46 Analysis of black carbon and carbon monoxide observed over the Indian Ocean: implications for
47 emissions and photochemistry. *Journal of Geophysical Research* **107** (D19), 8017
48 (doi:10.1029/2001JD000501).

- 1 Diesel Technology Forum (2006) Retrofitting America's diesel engines. *November*. Available on the
 2 Internet at [http://www.dieselforum.org/news-center/pdfs/Retrofitting-America-s-Diesel-](http://www.dieselforum.org/news-center/pdfs/Retrofitting-America-s-Diesel-Engines-11-2006.pdf/view)
 3 [Engines-11-2006.pdf/view](http://www.dieselforum.org/news-center/pdfs/Retrofitting-America-s-Diesel-Engines-11-2006.pdf/view).
- 4 Diesel Technology Forum (2007) CMAQ funded diesel retrofit projects. *April*. Available on the Internet at
 5 <http://www.dieselforum.org/news-center/pdfs/CMAQ-web.pdf/view>.
- 6 Dockery D.W., Luttmann-Gibson H., Rich D.Q., Link M.S., Mittleman M.A., Gold D.R., Koutrakis P.,
 7 Schwartz J.D., and Verrier R.L. (2005a) Association of air pollution with increased incidence of
 8 ventricular tachyarrhythmias recorded by implanted cardioverter defibrillators. *Environ Health*
 9 *Perspect* **113**.
- 10 Dockery D.W., Luttmann-Gibson H., Rich D.Q., Link M.S., Schwartz J.D., Gold D.R., Koutrakis P., Verrier
 11 R.L., and Mittleman M.A. (2005b) Particulate air pollution and nonfatal cardiac events Part II
 12 Association of air pollution with confirmed arrhythmias recorded by implanted defibrillators.
 13 Prepared by Health Effects Institute, Cambridge, MA.
- 14 Doherty S.J., Warren S.G., Grenfell T.C., Clarke A.D., and Brandt R.E. (2010) Light-absorbing impurities in
 15 Arctic snow. *Atmospheric Chemistry and Physics* **10**, 11647-11680.
- 16 Doppelheuer A. (2001) Quantities, characteristics and reduction potentials of aircraft engine emissions.
 17 By the Society of Automotive Engineers, Technical Paper 2001-01-3008.
- 18 Doran J.C., Barnard J.C., Arnott W.P., Cary R., Coulter R., Fast J.D., Kassianov E.I., Kleinman L., Laulainen
 19 N.S., Martin T., Paredes-Miranda G., Pekour M.S., Shaw W.J., Smith D.F., Springston S.R., and Yu
 20 X.Y. (2007) The T1-T2 study: evolution of aerosol properties downwind of Mexico City.
 21 *Atmospheric Chemistry and Physics* **7**.
- 22 Doran J.C., Fast J.D., Barnard J.C., Laskin A., Desyaterik Y., and Gilles M.K. (2008) Applications of
 23 Lagrangian dispersion modeling to the analysis of changes in the specific absorption of
 24 elemental carbon. *Atmospheric Chemistry and Physics* **8**. Available on the Internet at
 25 <http://www.atmos-chem-phys.net/8/1377/2008/>.
- 26 Dubovik O. and King M.D. (2000) A flexible inversion algorithm for retrieval of aerosol optical properties
 27 from Sun and sky radiance measurements. *Journal of Geophysical Research* **105**, 20673-20696.
- 28 Dukan M. (2010) Mitigating industrial black carbon through energy recycling. *Climate Alert* **20** (2).
- 29 Dusek U., Frank G.P., Hildebrandt L., Curtius J., Schneider J., Walter S., Chand D., Drewnick F., Hings S.,
 30 Jung D., Borrmann S., and Andreae M.O. (2006) Size matters more than chemistry for cloud-
 31 nucleating ability of aerosol particles. *Science* **312** (5778), 1375-1378.
- 32 Dwyer, et al. (2010) Do climate models reproduce observed solar dimming and brightening over China
 33 and Japan? *Journal of Geophysical Research* **115** (D00k08) (doi:10.1029/2009JD012945).
- 34 Einfeld W., Ward D.E., and Hardy C. (1991) Effects of fire behavior on prescribed fire smoke
 35 characteristics: a case study. *Global biomass burning: Atmospheric, climatic, and biospheric*
 36 *implications*, MIT Press, Cambridge, MA
- 37 Engelhart G.J., Asa-Awuku A., Nenes A., and Pandis S.N. (2008) CCN activity and droplet growth kinetics
 38 of fresh and aged monoterpene secondary organic aerosol. *Atmospheric Chemistry and Physics*
 39 **8**.
- 40 Ezzati M., Rodgers A., Lopez A.D., Hoorn S.V., and Murray C.J.L. (2006) Mortality and burden of disease
 41 attributable to individual risk factors. In *Comparative quantification of health risks: global and*
 42 *regional burden of disease due to selected major risk factors*, M. Ezzati, A.D. Lopez, A. Rodgers,
 43 and C.J.L. Murray eds., World Health Organization, Geneva, Switzerland
- 44 Falke S.R., Husar R.B., and Schichtel B.A. (2001) Fusion of SeaWiFS and TOMS satellite data with surface
 45 observations and topographic data during extreme aerosol events. *JAWMA* **51** (11), 1579-1585.
- 46 Fann N., Fulcher C., and Hubbell B. (2009) The influence of location, source, and emission type in
 47 estimates of the human health benefits of reducing a ton of air pollution. *Air Quality,*
 48 *Atmosphere & Health* **2** (3) (DOI: 10.1007/s11869-009-0044-0).

- 1 Feingold G., Jiang H., and Harrington J.Y. (2005) On smoke suppression of clouds in Amazonia. *Geophys. Res. Lett.* **32** (L02804) (doi:10.1029/2004GL021369).
- 2
3 Fiebig M., Stohl A., Wendisch M., Eckhardt S., and Petzold A. (2003) Dependence of solar radiative
4 forcing of forest fire aerosol on ageing and state of mixture. *Atmospheric Chemistry and Physics*
5 **3**.
- 6 Field C.B. (2007) Contribution of working group II to the fourth assessment report of the
7 intergovernmental panel on climate change. In *Climate change 2007: impacts, adaptation and*
8 *vulnerability*, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden, and C.E. Hanson eds.,
9 Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA
- 10 Flanner, et al. (2011) Radiative forcing and albedo feedback from the Northern Hemisphere cryosphere
11 between 1979 and 2008. *Nature Geoscience*.
- 12 Flanner M.G., Zender C.S., Randerson J.T., and Rasch P.J. (2007) Present-day climate forcing and
13 response from BC in snow. *Journal of Geophysical Research-Atmospheres* **112** (D11)
14 (doi:10.1029/2006JD008003).
- 15 Flanner M.G., Zender C.S., Hess P.G., Mahowald N.M., Painter T.H., Ramanathan V., and Rasch P.J.
16 (2009) Springtime warming and reduced snow cover from carbonaceous particles. *Atmospheric*
17 *Chemistry and Physics* **9**, 2481-2497.
- 18 Food and Agriculture Organization of the United Nations (2007) Fire management – global assessment
19 2006. FAO Forestry Paper 151.
- 20 Forster P., Ramaswamy V., Artaxo P., Bernsten T., Betts R., Fahey D.W., Haywood J., Lean J., Lowe D.C.,
21 Myhre G., Nganga J., Prinn R., Raga G., Schulz M., and R. Van Dorland (2007a) Changes in
22 atmospheric constituents and in radiative forcing. In *Climate change 2007: the physical science*
23 *basis. Contribution of working group I to the fourth assessment report of the Intergovernmental*
24 *Panel on Climate Change*, S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt,
25 M.Tignor, and H.L. Miller eds., Cambridge University Press, Cambridge, United Kingdom and
26 New York, NY, USA
- 27 Forster P., Shine K., and Stuber N. (2007b) It is premature to include non-CO₂ effects of aviation in
28 climate trading schemes. *Atmos Environ* **40** (6).
- 29 Foryd K.D., Holloway J.S., Middlebrook. A., Miller L., Montzka S., Murphy D.M., Peischl J., Ryerson T.B.,
30 Schwarz J.P., Spackman J.R., and Veres P. (2009) Biomass burning in Siberia and Kazakhstan as
31 an important source for haze of the Alaskan Arctic in April 2008. *Geophys. Res. Lett.* **36**
32 (L02813).
- 33 Frank N. (2006) Retained nitrate, hydrated sulfates, and carbonaceous mass in Federal reference
34 method PM_{2.5} for six eastern US cities. *Journal of the Air and Waste Management Association*
35 **56**, 500-511.
- 36 Franklin M., Koutrakis P., and Schwartz J. (2008) The role of particle composition on the association
37 between PM_{2.5} and mortality. *Epidemiology* **19**.
- 38 French H.W. (2007) Reports of forced labor at brick kilns unsettle China. *New York Times* **June 16**.
- 39 Fuglestedt J. (2009) Impacts of metric choice on analyzing climate effects of emissions. *IPCC expert*
40 *meeting on the science of alternative metrics, The Grand Hotel, Oslo, Norway, 18-20 March*
41 *2009*, 75 pages, G.-K. Plattner, T.F. Stocker, P. Midgley, and M. Tignor, eds., IPCC Working Group
42 I Technical Support Unit, University of Bern, Bern, Switzerland (ISBN 978-92-9169-126-5).
43 Available on the Internet at [http://www.ipcc.ch/pdf/supporting-material/expert-meeting-](http://www.ipcc.ch/pdf/supporting-material/expert-meeting-metrics-oslo.pdf)
44 [metrics-oslo.pdf](http://www.ipcc.ch/pdf/supporting-material/expert-meeting-metrics-oslo.pdf)
- 45 Fuglestedt J.S., Bernsten T.K., Godal O., Sausen R., Shine K.P., and Skodvin T. (2003) Metrics of climate
46 change: assessing radiative forcing and emission indices. *Climatic Change* **58**.
- 47 Fuglestedt J.S., Shine K.P., Bernsten T., Cook J., Lee D.S., Stenke A., Skeie R.B., Velders G.J.M., and Waitz
48 I.A. (2010) Transport impacts on atmosphere and climate: metrics. *Atmos Environ* **44** (37), 4648-

- 1 4677. Available on the Internet at [http://www.pbl.nl/en/publications/2009/Transport-impacts-](http://www.pbl.nl/en/publications/2009/Transport-impacts-on-atmosphere-and-climate_-_Metrics.html)
2 on-atmosphere-and-climate_-_Metrics.html.
- 3 Fujino J., Nair R., Kainuma M., Masui T., and Matsuoka Y. (2006) Multi-gas mitigation analysis on
4 stabilization scenarios using AIM global model. *The Energy Journal* (Special Issue Number 3:
5 Multigas Mitigation and Climate Policy), 343-354.
- 6 Fujita E.M., Campbell D.E., and Zielinska B. (2006) Chemical analysis of lubrication oil samples from a
7 study to characterize exhaust emissions from light-duty gasoline vehicles in the Kansas City
8 metropolitan area,. CRC Report No. E-69a, December.
- 9 Fung K. (1990) Particulate carbon speciation by MnO₂ oxidation. *Aerosol Science and Technology* **12**,
10 122-127.
- 11 Fuzzi S., Andreae M.O., Huebert B.J., Kulmala M., Bond T.C., Boy M., Doherty S.J., Guenther A.,
12 Kanakidou M., Kawamura K., Kerminen V.M., Lohmann U., Russell L.M., and Pöschl U. (2006)
13 Critical assessment of the current state of scientific knowledge, terminology, and research needs
14 concerning the role of organic aerosols in the atmosphere, climate, and global change.
15 *Atmospheric Chemistry and Physics* **6**.
- 16 Galbavy E.S., Anastasio C., Lefer B.L., and Hall S.R. (2007) Light penetration in the snowpack at Summit,
17 Greenland: part 1, nitrite and hydrogen peroxide photolysis. *Atmos Environ* **41**, 5077-5090.
- 18 Gao R.S., Hall S.R., Swartz W.H., Schwarz J.P., Spackman J.R., Watts L.A., Fahey D.W., Aikin K.C., Shetter
19 R.E., and Bui T.P. (2008) Calculations of solar shortwave heating rates due to black carbon and
20 ozone absorption using in situ measurements. *Journal of Geophysical Research* **113** (doi:
21 10.1029/2007JD009358). Available on the Internet at
22 <http://oaqpswww.epa.gov/wiki/BlackCarbon/files/2008Gao.pdf>.
- 23 Garrett T.J., Zhao C., Dong X., Mace G.G., and Hobbs P.V. (2004) Effects of varying aerosol regimes on
24 low-level Arctic stratus. *Geophys. Res. Lett.* **31**, L17105, AGU (17). Available on the Internet at
25 <http://dx.doi.org/10.1029/2004GL019928>
- 26 Gautam R., Hsu N.C., Lau K.-M., and Kafatos M. (2009) Aerosol and rainfall variability over the Indian
27 monsoon region: distributions, trends and coupling. *Ann. Geophys.* **27**. Available on the Internet
28 at <http://www.ann-geophys.net/27/3691/2009/>.
- 29 Generoso, et al. (2007) A satellite- and model-based assessment of the 2003 Russian fires: Impact on the
30 Arctic region. *Journal of Geophysical Research* **112** (D15302) (doi:10.1029/2006JD008344).
- 31 Gent J.F., Koutrakis P., Belanger K., Triche E., Holford T.R., Bracken M.B., and Leaderer B.P. (2009)
32 Symptoms and medication use in children with asthma and traffic-related sources of fine
33 particle pollution. *Environ Health Perspect* **117**.
- 34 Godleski J.J., Clarke R.W., Coull B.A., Saldiva P.H.N., Jiang N.F., Lawrence J., and Koutrakis P. (2002)
35 Composition of inhaled urban air particles determines acute pulmonary responses. *Ann. Occup.*
36 *Hyg* **46**.
- 37 Gold D.R., Litonjua A.A., Zanobetti A., Coull B.A., Schwartz J., MacCallum G., Verrier R.L., Nearing B.D.,
38 Canner M.J., Suh H., and Stone P.H. (2005) Air pollution and ST-segment depression in elderly
39 subjects. *Environ Health Perspect* **113**.
- 40 Gomez-Arroyo S., Jose Luis Bravo-Cabrera, and Guzman-Rincon J. (2010) Solvent extracted organic
41 matter and polycyclic aromatic hydrocarbons distributed in size-segregated airborne particles in
42 a zone of Mexico City: seasonal behavior and human exposure. *Atmospheric Environment* **44**.
- 43 Gomot-de Vaufleury A., , and Kerhoas I. (2000) Effects of cadmium on the reproductive system of the
44 land snail *Helix aspersa*. *Bull. Environ. Contam. Toxicol.* **64**, 434-442.
- 45 Graber E.R. and Rudich Y. (2006) Atmospheric HULIS: how humic-like are they? A comprehensive and
46 critical review. *Atmospheric Chemistry and Physics* **6**, 729-753.
- 47 Graham, et al. (2005) Partnership for clean indoor air, household energy, indoor air pollution and health:
48 overview of experiences and lessons in China.

- 1 Grahame T.J. (2009) Does improved exposure information for PM_{2.5} constituents explain differing results
 2 among epidemiological studies? . *Inhal. Toxicol.* (Epub ahead of print) **21**.
- 3 Grannas A.M., Shepson P.B., and Filley T.R. (2004) Photochemistry and nature of organic matter in Arctic
 4 and Antarctic snow. *Global Biogeochemical Cycles* **18** (GB1006) (doi:10.1029/2003GB002133).
- 5 Green D., Alexander J., Fuller G., Quincey P., and Butterfield D. (2007) Marylebone Road Aethalometer
 6 trial report. Prepared by the Environmental Research Group, King's College, London, England,
 7 May.
- 8 Grenfell T.C., Perovich D.K., and Ogren J.A. (1981) Spectral albedos of an alpine snow pack. *Cold Reg. Sci.*
 9 *Technol.* **4**, 121-127.
- 10 Grenfell T.C., Warren S.G., and Mullen P.C. (1994) Reflection of solar radiation by the Antarctic snow
 11 surface at ultraviolet, visible, and near-infrared wavelengths. *J. Geophys. Res.* **99**, 18669-18684.
- 12 Grenfell T.C., Light B., and Sturm M. (2002) Spatial distribution and radiative effects of BC in the snow
 13 and sea ice during the SHEBA experiment. *Journal of Geophysical Research* **107** (C10), 8032.
- 14 Grieshop A.P., Reynolds C.C.O., Kandlikar M., and Dowlatabadi H. (2009) black-carbon mitigation wedge.
 15 *Nature Geoscience* **2** (doi: 10.1038/ngeo595). Available on the Internet at
 16 <http://oaqpswww.epa.gov/wiki/BlackCarbon/files/a-black-carbon-mitigation-wedge.pdf>.
- 17 Guan H., Esswein R., Lopez J., Bergstrom R., A.Warnock, Follette-Cook M., Fromm M., and Iraci L.T.
 18 (2010) A multi-decadal history of biomass burning plume heights identified using aerosol index
 19 measurements. *Atmos. Chem. Phys.* **10**.
- 20 Gundel L.A., R.L. D., Rosen H., and Novakov T. (1984) The relationship between optical attenuation and
 21 black carbon concentration for ambient and source particles. *The Science of the Total*
 22 *Environment* **36**, 197-202.
- 23 Gupta J. (2003) Informal labour in brick kilns: need for regulation. *Economic and Political Weekly* **38**
 24 (31), 3282-3292.
- 25 Hadley O., et al. (2007) Trans-Pacific transport of black carbon and fine aerosol (d < 2.5 μm) into North
 26 America. *Journal of Geophysical Research* **112** (D05309).
- 27 Hadley O.L., Corrigan C.E., Kirchstetter T.W., Cliff S.S., and Ramanathan V. (2010) Measured black carbon
 28 deposition on the Sierra Nevada snow pack and implication for snow pack retreat. *Atmos. Chem.*
 29 *Phys. Discuss.* **10**, 10463-10485 (doi:10.5194/acpd-10-10463-2010).
- 30 Hagler G.S.W., Bergin M.H., Smith E.A., and Dibb J.E. (2007a) A summer time series of particulate carbon
 31 in the air and snow at Summit, Greenland. *J. Geophys. Res.* **112**, D21309
 32 (doi:10.1029/2007JD008993).
- 33 Hagler G.S.W., Bergin M.H., Smith E.A., Dibb J.E., Anderson C., and Steig E.J. (2007b) Particulate and
 34 water-soluble carbon measured in recent snow at Summit, Greenland. *Geophys. Res. Lett.* **34**,
 35 L16505 (doi:10.1029/2007GL030110).
- 36 Hammer C.U., Clausen, H.B., Dansgaard, W., Gundestrup, N., Johnsen, S.J., and Reeh, N. (1978) Dating of
 37 Greenland ice cores by flow models, isotopes, volcanic debris, and continental dust. *J. Glaciology*
 38 **20** (82), 3-26.
- 39 Hammitt J.K., Jain A.K., Adams J.L., and Wuebbles D.J. (1996) A welfare-based index for assessing
 40 environmental effects of greenhouse-gas emissions. *Nature* **381**.
- 41 Hansen A.D.A., Rosen H., and Novakov T. (1982) Real-time measurement of the absorption coefficient of
 42 aerosol particles. *Appl. Opt.* **21**, 3060–3062.
- 43 Hansen J. and Nazarenko L. (2004) Soot climate forcing via snow and ice albedos. *Proceedings of the*
 44 *National Academy of Sciences* **101**.
- 45 Hansen J., Sato M., Ruedy R., and others (2005) Efficacy of climate forcings. *J. Geophys. Res.* **110**,
 46 D18104 (doi: 10.1029/2005JD005776).
- 47 Hansen J., Sato M., Kharecha P., Russell G., Lea D.W., and Siddall M. (2007a) Climate change and trace
 48 gases. *Phil. Trans. Royal. Soc. A* (365).

- 1 Hansen J., Sato M., Ruedy R., Kharecha P., A. Lacis, Miller R., Nazarenko L., Lo K., Schmidt G.A., Russell
2 G., Aleinov I., Bauer S., Baum E., Cairns B., Canuto V., Chandler M., Cheng Y., Cohen A., Genio
3 A.D., Faluvegi G., Fleming E., Friend A., Hall T., Jackman C., Jonas J., Kelley M., Kiang N.Y., Koch
4 D., Labow G., Lerner J., Menon S., Novakov T., Oinas V., Perlwitz J., Perlwitz J., Rind D., Romanou
5 A., Schmunk R., Shindell D., Stone P., Sun S., Streets D., Tausnev N., Thresher D., Unger N., Yao
6 M., and Zhang S. (2007b) Dangerous human-made interference with climate: A GISS model E
7 study. *Atmospheric Chemistry and Physics* **7**.
- 8 Hansen J., Sato M., Ruedy R., Kharecha P., Lacis A., Miller R., Nazarenko L., Lo K., Schmidt G.A., Russell
9 G., Aleinov I., Bauer S., Baum E., Cairns B., Canuto V., Chandler M., Cheng Y., Cohen A., Del Genio
10 A., Faluvegi G., Fleming E., Friend A., Hall T., Jackman C., Jonas J., Kelley M., Kiang N.Y., Koch D.,
11 Labow G., Lerner J., Menon S., Novakov T., Oinas V., Perlwitz J., Perlwitz J., Rind D., Romanou A.,
12 Schmunk R., Shindell D., Stone P., Sun S., Streets D., Tausnev N., Thresher D., Unger N., Yao M.,
13 and Zhang S. (2007c) Climate simulations for 1880-2003 with GISS modelE. *Clim Dyn* **29**, 661–
14 696.
- 15 Haynie F.H. (1986) Environmental factors affecting corrosion of weathering steel. In *Materials*
16 *degradation caused by acid rain: developed from the 20th state-of-the-art symposium of the*
17 *American Chemical Society, June 1985, Arlington, VA., ACS symposium series 318, 163-171, R.*
18 *Baboian, ed., American Chemical Society, Washington, D.C. (HERO ID 157198).*
- 19 Hays M.D., Geron C.D., Linna K.J., Smith N.D., and Schauer J.J. (2002) Speciation of gas-phase and fine
20 particle emissions from burning of foliar fuels. *Environmental Science & Technology* **36** (11),
21 2281-2295.
- 22 Hays M.D., Smith N.D., Kinsey J.S., Dong Y., and Kariher P. (2003) A polycyclic aromatic hydrocarbon
23 (PAH) size distributions in aerosols from appliances of residential wood combustion as
24 determined by direct thermal desorption. *Journal of Aerosol Science* **34**.
- 25 Hays M.D., Fine P.M., Geron C.D., Kleeman M.J., and Gullett B.K. (2005) Open burning of agricultural
26 biomass: Physical and chemical properties of particle-phase emissions,. *Atmospheric*
27 *Environment* **39** (36), 6747-6764 (doi: 10.1016/j.atmosenv.2005.07.072).
- 28 Hays M.D. and VanderWal R.L. (2007) Heterogeneous soot nanostructure in atmospheric and
29 combustion source aerosols. *Energy Fuels* **21**.
- 30 Hays M.D., Beck L., Barfield P., Lavrich R.J., Dong Y., and Vander Wal R.L. (2008) Physical and chemical
31 characterization of residential oil boiler emissions. *Environmental Science & Technology* **42** (7),
32 2496-2502.
- 33 Haywood J.M., and Shine K.P. (1995) The effect of anthropogenic sulfate and soot aerosol on the clear
34 sky planetary radiation budget. *Geophysical Research Letters* **22** (5).
- 35 Hecobian A., Zhang X., Zheng M., Frank N., Edgerton E.S., and Weber R.J. (2010) Water-soluble organic
36 aerosol material and the light-absorption characteristics of aqueous extracts measured over the
37 Southeastern United States. *Atmos. Chem. Phys.* **10**, 5965–5977. Available on the Internet at
38 www.atmos-chem-phys.net/10/5965/2010/.
- 39 Hedman B., Naslund M., and Marklund S. (2006) Emission of PCDD/F, PCB, and HCB from combustion of
40 firewood and pellets in residential stoves and boilers. *Environ. Sci. Technol.* **40**.
- 41 Hegerl G.C., Zwiers F.W., Braconnot P., Gillett N.P., Luo Y., Orsini J.A.M., Nicholls N., Penner J.E., and
42 Stott P.A. (2007) Understanding and attributing climate change. In *Climate Change 2007: The*
43 *Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the*
44 *Intergovernmental Panel on Climate Change* S. Solomon, D. Qin, M. Manning, Z. Chen, M.
45 Marquis, K.B. Averyt, M. Tignor, and H.L. Miller eds., Cambridge University Press, Cambridge,
46 United Kingdom and New York, New York, USA
- 47 Hegg D.A., Warren S.G., Grenfell T.C., Doherty S.J., Larson T.V., and Clarke A.D. (2009) Source attribution
48 of black carbon in arctic snow. *Environ. Sci. Technol.* (43), 4016–4021.

- 1 Hegg D.A., Warren S.G., Grenfell T.C., Doherty S.J., and Clarke A.D. (2010) Sources of light-absorbing
2 aerosol in arctic snow and their seasonal variation. *Atmos. Chem. Phys.* **10**, 10923–10938
3 (doi:10.5194/acp-10-10923-2010).
- 4 Herman J.R., Bhartia P.K., Torres O., Hsu C., Seftor C., and Celarier E. (1997) Global distribution of UV-
5 absorbing aerosols from Nimbus 7/TOMS data. *J. Geophys. Res.* **102**, D14.
- 6 Hijioka Y., Matsuoka Y., Nishimoto H., Masui M., and Kainuma M. (2008) Global GHG emissions scenarios
7 under GHG concentration stabilization targets. *Journal of Global Environmental Engineering* **13**,
8 97-108.
- 9 Hinckley J. and Doshi K. (2010) Emission controls for wood-fired boilers. Prepared by the U.S. Forest
10 Service Biomass Energy Resource Center, Montpelier, VT, May.
- 11 Hirdman D., Burkhardt J.F., Sodemann H., Eckhardt S., Jefferson A., Quinn P.K., Sharma S., Strom J., and
12 Stohl A. (2010) Long-term trends of black carbon and sulphate aerosol in the Arctic: changes in
13 atmospheric transport and source region emissions. *Atmos. Chem. Phys. Discuss.* **10**, 12133–
14 12184. Available on the Internet at www.atmos-chem-phys-discuss.net/10/12133/2010/A.
- 15 Hirdman D., et al. (2010) Source identification of short-lived pollutants in the Arctic using statistical
16 analysis of measurement data and particle dispersion model output. *Atmospheric Chemistry &*
17 *Physics* **10**, 669-693.
- 18 Hitzenberger R., Petzold A., Bauer H., Ctyroky P., Pouresmaeil P., Laskus L., and Puxbaum H. (2006)
19 Intercomparison of thermal and optical measurement methods for elemental carbon and black
20 carbon at an urban location. *Environ Sci Technol.* **40** (20), 6377-6383.
- 21 Hoffer A., Gelencsér A., Guyon P., Kiss G., Schmid O., Frank G.P., Artaxo P., and Andreae M.O. (2006)
22 Optical properties of humic-like substances (HULIS) in biomass-burning aerosols. *Atmos. Chem.*
23 *Phys.* **6**.
- 24 Holben B.N., Eck T.F., Slutsker I., Tanre D., Buis J.P., Setzer A., Vermote E., Reagan J.A., Kaufman Y.J.,
25 Nakjima T., Lavenu F., Jankowiak I., and Smirnov A. (1998) AERONET – A federated instrument
26 network and data archive for aerosol characterization. *Remote Sens. Environ.* **66**, 1-16.
- 27 Hopkins R.J., Lewis K., Desyaterik Y., Wang Z., Tivanski A.V., Arnott W.P., Laskin A., and Gilles M. (2007)
28 Correlations between optical, chemical and physical properties of biomass burn aerosols.
29 *Geophys. Res. Lett.* **34**, L18806.
- 30 Huang Y.-C.T., Ghio A.J., Stonehuerner J., McGee J., Carter J.D., Grambow S.C., and Devlin R.B. (2003)
31 The role of soluble components in ambient fine particles-induced changes in human lungs and
32 blood. *Inhal Toxicol* (15).
- 33 Huntzicker J.J., Johnson R.L., Shah J.J., and Cary R.A. (1982) Analysis of organic and elemental carbon in
34 ambient aerosols by a thermal-optical method. In. *Particulate Carbon: Atmospheric Life Cycle*,
35 G.T. Wolff and R.L. Klimisch eds., Plenum Press, New York, New York, 79-88
- 36 Husain L., Dutkiewicz V.A., Khan A.J., and Ghauri B.M. (2007) Characterization of carbonaceous aerosols
37 in urban air. *Atmospheric Environment* **41**, 6872–6883.
- 38 Husain L., Khan A.J., Ahmed T., Swami K., Bari A., Webber J.S., and Li J. (2008) Trends in atmospheric
39 elemental carbon concentrations from 1835 to 2005. *J. Geophys. Res.* **113**, D13102
40 (doi:10.1029/2007JD009398).
- 41 Husar R.B. (2011) Satellite measurements of atmospheric aerosols. Draft chapter for *Aerosol*
42 *measurement: principles, techniques, and applications* Available on the Internet at
43 [http://capita.wustl.edu/capita/capitareports/090120_AerosolMeasHandbook/Manuscript/Satell](http://capita.wustl.edu/capita/capitareports/090120_AerosolMeasHandbook/Manuscript/SatelliteMeasurementsOfAtmosphericAerosols_100105.pdf)
44 [iteMeasurementsOfAtmosphericAerosols_100105.pdf](http://capita.wustl.edu/capita/capitareports/090120_AerosolMeasHandbook/Manuscript/SatelliteMeasurementsOfAtmosphericAerosols_100105.pdf).
- 45 Im J.-S., Saxena V.K., and Wenny B.N. (2001) Temporal trends of black carbon concentrations and
46 regional climate forcing in the southeastern United States. *Atmos Environ* (35).
- 47 Indigo Agglomerator. An overview of Indigo Agglomerator technology. Available on the Internet at
48 http://www.indigotechnologies.com.au/agg_overview.php.

- 1 Intergovernmental Panel on Climate Change (2007) *Climate change 2007: the physical science basis,*
2 *contribution of working group I to the fourth assessment report of the intergovernmental panel*
3 *on climate change*, S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor,
4 and H.L. Miller, eds., Cambridge University Press, Cambridge United Kingdom and New York, NY,
5 USA, 996.
- 6 Intergovernmental Panel on Climate Change (2009a) Meeting Report. *IPCC expert meeting on the*
7 *science of alternative metrics, The Grand Hotel, Oslo, Norway, 18-20 March 2009*, 75 pages, G.-
8 K. Plattner, T.F. Stocker, P. Midgley, and M. Tignor, eds., IPCC Working Group I Technical Support
9 Unit, University of Bern, Bern, Switzerland (ISBN 978-92-9169-126-5). Available on the Internet
10 at <http://www.ipcc.ch/pdf/supporting-material/expert-meeting-metrics-oslo.pdf>
- 11 Intergovernmental Panel on Climate Change (2009b) IPCC fourth assessment report: climate change
12 2007. *IPCC expert meeting on the science of alternative metrics, The Grand Hotel, Oslo, Norway,*
13 *18-20 March 2009*, 75 pages, G.-K. Plattner, T.F. Stocker, P. Midgley, and M. Tignor, eds., IPCC
14 Working Group I Technical Support Unit, University of Bern, Bern, Switzerland (ISBN 978-92-
15 9169-126-5). Available on the Internet at [http://www.ipcc.ch/pdf/supporting-material/expert-](http://www.ipcc.ch/pdf/supporting-material/expert-meeting-metrics-oslo.pdf)
16 [meeting-metrics-oslo.pdf](http://www.ipcc.ch/pdf/supporting-material/expert-meeting-metrics-oslo.pdf)
- 17 International Bank for Reconstruction and Development/The World Bank (2010) Improved cook stoves
18 and better health in Bangladesh: lessons from household energy and sanitation programs.
- 19 International Energy Agency (2006) *World energy outlook 2006*, International Energy Agency, Paris
20 Cedex 15, France (ISBN: 92-64-10989-7).
- 21 International Energy Agency (2009) *World energy outlook 2009*, International Energy Agency (ISBN: 978-
22 92-64-06130-9).
- 23 International Energy Agency (2010) Energy poverty: how to make modern energy access universal.
24 Special early excerpt of the *World energy outlook 2010*, International Energy Agency (ISBN 978-
25 92-64-08624-1). Available on the Internet at
26 http://www.worldenergyoutlook.org/docs/weo2010/weo2010_poverty.pdf.
- 27 Ito A. and Penner J.E. (2005) Historical emissions of carbonaceous aerosols from biomass and fossil fuel
28 burning for the period 1870-2000. *Global Biogeochem. Cycles* **19**, GB2028, AGU (2). Available on
29 the Internet at <http://dx.doi.org/10.1029/2004GB002374>
- 30 Ivleva N.P., et al. (2007a) Raman microspectroscopic analysis of size-resolved atmospheric aerosol
31 particle samples collected with an ELPI: soot, humic-like substances, and inorganic compounds.
32 *Aerosol Science and Technology* **41**, 655-671.
- 33 Ivleva N.P., et al. (2007b) Raman microspectroscopic analysis of changes in the chemical structure and
34 reactivity of soot in a diesel exhaust aftertreatment model system. *Environmental Science &*
35 *Technology* **41**.
- 36 Jacobson M.Z. (2001) Strong radiative heating due to the mixing state of BC in atmospheric aerosols.
37 *Nature* **409**.
- 38 Jacobson M.Z. (2002) Control of fossil-fuel particulate black carbon and organic matter, possibly the
39 most effective method of slowing global warming. *Journal of Geophysical Research* **107**, 4410
40 (doi:10.1029/2001jd001376).
- 41 Jacobson M.Z. (2004) Climate response of fossil fuel and biofuel soot, accounting for soot's feedback to
42 snow and sea ice albedo and emissivity. *Journal of Geophysical Research* **109** (D21201)
43 (doi:10.1029/2004JD004945).
- 44 Jacobson M.Z. (2005) Correction to "control of fossil-fuel particulate BC and organic matter, possibly the
45 most effective method of slowing global warming". *Journal of Geophysical Research-*
46 *Atmospheres* **110** (D14105) (doi:10.1029/2005JD005888).

- 1 Jacobson M.Z. (2007) Testimony for the hearing on Black Carbon and Climate Change. U.S. House
 2 Committee on Oversight and Government Reform [Cited October 18]. . Available on the Internet
 3 at <http://oversight.house.gov/images/stories/documents/20071018110606.pdf>.
- 4 Jacobson M.Z. (2010) Short-term effects of controlling fossil-fuel soot, biofuel soot and gases, and
 5 methane on climate, Arctic ice, and air pollution health. *Journal of Geophysical Research* **115**
 6 (D14209).
- 7 Jansen K.L., Larson T.V., Koenig J.Q., Mar T.F., Fields C., Stewart J., and Lippmann M. (2005) Associations
 8 between health effects and particulate matter and black carbon in subjects with respiratory
 9 disease. *Environ Health Perspect* **113**.
- 10 Japar S.M., Szkarlat A.C., and Pierson W.R. (1984) The determination of the optical properties of
 11 airborne particle emissions from diesel vehicles. *Science of the Total Environment* **36**, 121-130.
- 12 Jeong G.-R. and Wang C. (2010) Climate effects of seasonally varying biomass burning emitted
 13 carbonaceous aerosols. *Atmospheric Chemistry and Physics* **10**.
- 14 Jeong M.-J. and Hsu N.C. (2008) Retrievals of aerosol single-scattering albedo and effective aerosol layer
 15 height for biomass-burning smoke: synergy derived from “A-Train” sensors. *Geophys. Res. Lett.*
 16 **35** (L24801) (doi: 10.1029/2008GL036279).
- 17 Jetter and Kariher P. (2008) Solid-fuel household cook stoves: Characterization of performance and
 18 emissions. *Biomass and Bioenergy* **33** (2), 294-305.
- 19 Jiang M., Marr L.C., Dunlea E.J., Herndon S.C., Jayne J.T., Kolb C.E., Knighton W.B., Rogers T.M., Zavala
 20 M., Molina L.T., and Molina M.J. (2005) Vehicle fleet emissions of black carbon, polycyclic
 21 aromatic hydrocarbons, and other pollutants measured by a mobile laboratory in Mexico City.
 22 *Atmospheric Chemistry and Physics* **5**.
- 23 Jimenez, et al. (2009) Evolution of organic aerosols in the atmosphere. *Science* **326**, 1525-1529.
- 24 Joeng C.-H., Hopke P.K., Kim E., and Lee D.-W. (2004) The comparison between thermal-optical
 25 transmittance elemental carbon and Aethalometer black carbon measured at multiple
 26 monitoring sites *Atmos Environ* **38** (31), 5193-5204.
- 27 Johansson D. (2009) Economics vs. physical-based metrics for relative greenhouse gas valuations,
 28 working papers in economics. Prepared by School of Business, Economics and Law, Göteborgs
 29 Universitet, Sweden.
- 30 Johansson D.J.A., Persson U.M., and Azar C. (2008) Uncertainty and learning: implications for the trade-
 31 off between short-lived and long-lived greenhouse gases. *Climatic Change* **88**.
- 32 Johnson, et al. (2007) In-field greenhouse gas emissions from cook stoves in rural Mexican households.
 33 *Atmos Environ* **42**.
- 34 Junker C. and Liousse C. (2008) A global emission inventory of carbonaceous aerosol from historic
 35 records of fossil fuel and biofuel consumption for the period 1860-1997. *Atmospheric Chemistry*
 36 *and Physics* **8**, 1195-1207.
- 37 Kahn R.A., Garay M.J., Nelson D.L., Yau K.K., Bull M.A., Gaitley B.J., Martonchik J.V., and Levy R.C. (2007)
 38 Satellite-derived aerosol optical depth over dark water from MISR and MODIS: comparisons with
 39 AERONET and implications for climatological studies. *Journal of Geophysical Research* **112**
 40 (D18205) (doi:10.1029/2006JD008175).
- 41 Kahn R.A., Nelson D.L., Garay M., Levy R.C., Bull M.A., Diner D.J., Martonchik J.V., Paradise S.R., Hansen
 42 E.G., and Remer L.A. (2009) MISR aerosol product attributes, and statistical comparisons with
 43 MODIS. *IEEE T. Geosci. Remote*, 4095-4114.
- 44 Kahn R.A., Gaitley B.J., Garay M.J., Diner D.J., Eck T., Smirnov A., and Holben B.N. (2010) MISR aerosol
 45 product assessment by comparison with AERONET. *Journal of Geophysical Research* (submitted).
- 46 Kahnert M. (2010) Modeling the optical and radiative properties of freshly emitted light absorbing
 47 carbon with an atmospheric chemical transport model. *Atmospheric Chemistry and Physics* **10**.

- 1 Kandlikar M. (1996) Indices for comparing greenhouse gas emissions: Integrating science and
 2 economics. *Energy Economics* **18**.
- 3 Kandlikar M., Reynolds C.C.O., and Grieshop A.P. (2009): A perspective paper on black carbon mitigation
 4 as a response to climate change: *Copenhagen Consensus on Climate*.
- 5 Kannari A., Streets D.G., Tonooka Y., Murano K., and Baba T. (2008) MICS-Asia II: an inter-comparison
 6 study of emission inventories for the Japan Region. *Atmos Environ* **42** (15), 3584-3591.
- 7 Kaspari S.D., Schwikowski M., Gysel M., Flanner M.G., Kang S., Hou S., and Mayewski P.A. (2011) Recent
 8 increase in black carbon concentrations from a Mt. Everest ice core spanning 1860–2000 AD.
 9 *Geophys. Res. Lett.* **38** (L04703) (doi:10.1029/2010GL046096).
- 10 Katrinak K.A., Rez P., and Buseck P.R. (1992) Structural variations in individual carbonaceous particles
 11 from an urban aerosol. *Environmental Science & Technology* **26**.
- 12 Kazadzis S., Bais A., Arola A., Krotkov N., Kouremeti N., and Meleti C. (2009) Ozone monitoring
 13 instrument spectral UV irradiance products: comparison with ground based measurements at an
 14 urban environment. *Atmospheric Chemistry and Physics* **9**, 585-594. Available on the Internet at
 15 www.atmos-chem-phys.net/9/585/2009/.
- 16 Khalek I.A., Bougher T.L., and Merritt P.M. (2009) Coordinating Research Council, Advanced
 17 Collaborative Emissions Study, Phase 1.
- 18 Kim J.J., Smorodinsky S., Lipsett M., Singer B.C., Hodgson A.T., and Ostro B. (2004) Traffic-related air
 19 pollution near busy roads: the East Bay children's Respiratory Health Study. *Am J Respir Crit Care*
 20 *Med.* **170**.
- 21 Kim Y., Hatsushika H., Muskett R.R., and Yamazaki K. (2005) Possible effect of boreal wildfire soot on
 22 Arctic sea ice and Alaska glaciers. *Atmos Environ* **39**, 3513-3520.
- 23 Kinsey J.S., Hays M.D., Dong Y., Williams D.C., and Logan R. (2009) Chemical characterization of the fine
 24 particle emissions from commercial aircraft engines during the Aircraft Particle Emissions
 25 eXperiment (APEX) 1 to 3. *Environmental Science and Technology* (Under review).
- 26 Kiraz K., Kart L., Demir R., Oymak S., Gulmez I., Unalacak M., and Ozesmi M. (2003) Chronic pulmonary
 27 disease in rural women exposed to biomass fumes. *Clinical and Investigative Medicine* **26** (5),
 28 243-248 (PubMed PMID: 14596485).
- 29 Kirchstetter T.W., Novakov T., and Hobbs P.V. (2004) Evidence that the spectral dependence of light
 30 absorption by aerosols is affected by organic carbon. *Journal of Geophysical Research* **109**
 31 (D21208) (doi:10.1029/2004JD004999).
- 32 Kirchstetter T.W., Aguiar J., Tonse S., and Novakov T. (2008) Black carbon concentrations and diesel
 33 vehicle emission factors derived from coefficient of haze measurements in California: 1967-
 34 2003. Prepared by Lawrence Berkeley National Laboratory, Berkeley, CA, LBNL Paper LBNL-
 35 63493.
- 36 Kleinman L.I., Springston S.R., Daum P.H., Lee Y.N., Nunnermacker L.J., Senum G.I., Wang J., Weinstein-
 37 Lloyd J., Alexander M.L., Hubbe J., Ortega J., Canagaratna M.R., and Jayne J. (2008) The time
 38 evolution of aerosol composition over the Mexico City plateau. *Atmospheric Chemistry and*
 39 *Physics* **8**, 1559-1575 (doi:10.5194/acp-8-1559-2008). Available on the Internet at
 40 <http://www.atmos-chem-phys.net/8/1559/2008/acp-8-1559-2008.html>.
- 41 Kleinman L.I., Springston S.R., Wang J., Daum P.H., Lee Y.N., Nunnermacker L.J., Senum G.I., Weinstein-
 42 Lloyd J., Alexander M.L., Hubbe J., Ortega J., Zaveri R.A., Canagaratna M.R., and Jayne J. (2009)
 43 The time evolution of aerosol size distribution over the Mexico City plateau. *Atmos. Chem. Phys.*
 44 **9** (13), 4261-4278, Copernicus Publications (doi:10.5194/acp-9-4261-2009). Available on the
 45 Internet at <http://www.atmos-chem-phys.net/9/4261/2009/> and at <http://www.atmos-chem-phys.net/9/4261/2009/acp-9-4261-2009.pdf>
- 46
 47 Klimont C.J., et al. (2009) Projections of SO₂, NO_x and carbonaceous aerosols emissions in Asia. *Tellus B*
 48 **61** (4), 602-617.

- 1 Klouda G.A., Filliben J.J., Parish H.J., Chow J.C., Watson J.G., and Cary R.A. (2005) Reference Material
2 8785: Air particulate matter on filter media. *Aerosol Science and Technology* **39**, 173-183.
- 3 Knauer M. and et. al. (2009) Soot structure and reactivity analysis by raman microspectroscopy,
4 temperature-programmed oxidation, and high-resolution transmission electron microscopy. *The*
5 *Journal of Physical Chemistry A* **113**.
- 6 Koch D., Bond T.C., Streets D., and Unger N. (2007a) Linking future aerosol radiative forcing to shifts in
7 source activities. *Geophys. Res. Lett.* **34** (L05821), AGU (doi:10.1029/2006GL028360, 2007).
8 Available on the Internet at <http://dx.doi.org/10.1029/2006GL028360>
- 9 Koch D., Bond T.C., Streets D., Unger N., and van der Werf G.R. (2007b) Global impacts of aerosols from
10 particular source regions and sectors. *Journal of Geophysical Research-Atmospheres* **112**
11 (D02205).
- 12 Koch D., Schulz M., Kinne S., McNaughton C., Spackman J.R., Balkanski Y., Bauer S., Berntsen T., Bond
13 T.C., Boucher O., Chin M., Clarke A., Luca N.D., Dentener F., Diehl T., Dubovik O., Easter R., Fahey
14 D.W., Feichter J., Fillmore D., Freitag S., Ghan S., Ginoux P., Gong S., Horowitz L., Iversen T.,
15 Kirkevåg A., Klimont Z., Kondo Y., Krol M., Liu X., Miller R., Montanaro V., Moteki N., Myhre G.,
16 Penner J.E., Perlwitz J., Pitari G., Reddy S., Sahu L., Sakamoto H., Schuster G., Schwarz J.P.,
17 Seland Ø., Stier P., Takegawa N., Takemura T., Textor C., van Aardenne J.A., and Zhao Y. (2009)
18 Evaluation of black carbon estimations in global aerosol models. *Atmospheric Chemistry and*
19 *Physics* **9**, 9001-9026. Available on the Internet at www.atmos-chem-phys.net/9/9001/2009/.
- 20 Koch D. and Del Genio A. (2010) Black carbon absorption effects on cloud cover: review and synthesis.
21 *Atmospheric Chemistry and Physics Discussions* **10**.
- 22 Kopacz M., et al. (2010) Origin and radiative forcing of black carbon transported to the Himalayas and
23 Tibetan Plateau. *Atmospheric Chemistry and Physics Discussions* **10**, 21615-21651.
- 24 Kopacz M., Mauzerall D.L., Leibensperger E.M., Wang J., Henze D.K., and Singh K. (2010) Black carbon
25 receptor to source relationship in the Himalayas and the Tibetan Plateau: preliminary results
26 using the adjoint of GEOS-Chem. Presented at the *EPA Workshop on Short-Lived Climate Forcers*,
27 *Chapel Hill, NC, March 3-4*.
- 28 Kopp R.E. and Mauzerall D.L. (2010) Assessing the climatic benefits of BC mitigation. *Proceedings of the*
29 *National Academy of Sciences* **107** (26), 11703-11708.
- 30 Kralovec A.C., Christensen E.R., and Van Camp R.P. (2002) Fossil and wood combustion as recorded by
31 carbon particles in Lake Erie sediments 1850-1998. *Environ. Sci. Technol.* **36**, 1405-1413.
- 32 Krinner G., Boucher O., and Balkanski Y. (2006) Ice-free glacial northern Asia due to dust deposition on
33 snow. *Climate Dynamics* **27**, 613-625.
- 34 Kristjánsson J.E. (2002) Studies of the aerosol indirect effect from sulfate and black carbon aerosols. *J.*
35 *Geophys. Res.* **107** (D15), 4246 (doi: 10.1029/2001JD000887).
- 36 Kucera T., Horakova H., and Sonska A. (2008) Toxic metal ions in photoautotrophic organisms.
37 *Photosynthetica* **46**, 481-489.
- 38 Kuki K.N., Oliva M.A., and Costa A.C. (2009) The simulated effects of iron dust and acidity during the
39 early stages of establishment of two coastal plant species. *Water Air Soil Pollut* **196**, 287-295.
- 40 Kukla G.J. and Kukla H.J. (1974) Increased surface albedo in the northern hemisphere. *Science* **183**, 709-
41 714.
- 42 Lack D.A. and Cappa C.D. (2010) Impact of brown and clear carbon on light absorption enhancement,
43 single scattering albedo and absorption wavelength dependence of black carbon. *Atmospheric*
44 *Chemistry and Physics* **10**, 4207-4220 (doi:10.5194/acp-10-4207-2010).
- 45 Lamarque J.-F., Bond T.C., Eyring V., Granier C., Heil A., Klimont Z., Lee D., Liousse C., Mieville A., Owen
46 B., Schultz M.G., Shindell D., Smith S.J., Stehfest E., Aardenne J.V., Cooper O.R., Kainuma M.,
47 Mahowald N., McConnell J.R., Naik V., Rishi K., and van Vuuren D.P. (2010) Historical (1850-

- 1 2000) gridded anthropogenic and biomass burning emissions of reactive gases and aerosols:
2 methodology and application. *Atmospheric Chemistry and Physics* **10**, 7017-7039.
- 3 Lanki T., Pekkanen J., Aalto P., Elosua R., Berglind N., D'Ippoliti D., Kulmala M., Nyberg F., Peters A.,
4 Picciotto S., Salomaa V., Sunyer J., Tiittanen P., Von Klot S., and Forastiere F. (2006) Associations
5 of traffic-related air pollutants with hospitalization for first acute myocardial infarction: the
6 HEAPSS study. *Occup Environ Med* **63**.
- 7 Lashof D.A. and D.R. A. (1990) The relative contributions of greenhouse gas emissions to global warming.
8 *Nature* 344.
- 9 Lau K.M. and Kim K.M. (2006) Observational relationships between aerosol and Asian monsoon rainfall,
10 and circulation. *Geophysical Research Letters* **33** L21810 (doi:10.1029/2006GL027546).
- 11 Lau W.K.M., et al. (2010) Enhanced surface warming and accelerated snow melt in the Himalayas and
12 Tibetan Plateau induced by absorbing aerosols. *Environmental Research Letters* **5** (2) (doi:
13 10.1088/1748-9326/5/2/025204).
- 14 Lauer A., and Hendricks, J. (2006) Simulating aerosol microphysics with the ECHAM4/MADE GCM - Part
15 II: results from a first multiannual simulation of the submicrometer aerosol. *Atmos. Chem. Phys.*
16 **6**.
- 17 Lavanchy V.M.H., Gaggeler H.W., Schotterer U., Schwikowski M., and Baltensperger U. (1999) Historical
18 record of carbonaceous particle concentrations from a European high-alpine glacier (Colle
19 Gnifetti, Switzerland). *J. Geophys. Res.* **104**, 21227-21236.
- 20 Legrand M., Preunkert S., Schock M., Cerqueira M., Kasper-Giebl A., Afonso J., Pio C., Gelencser A., and
21 Dombrowski-Etchevers I. (2007) Major 20th century changes of carbonaceous aerosol
22 components (EC, WinOC, DOC, HULIS, carboxylic acids, and cellulose) derived from Alpine ice
23 cores. *J. Geophys. Res.* **112**, D23S11 (doi:10.1029/2006JD008080).
- 24 Lesins G., Chylek P., and Lohmann U. (2002) A study of internal and external mixing scenarios and its
25 effect on aerosol optic properties and direct radiative forcing. *J. Geophys. Res.* **104**.
- 26 Liepert B.G., Feichter J., Lohmann U., and Roeckner E. (2004) Can aerosols spin down the water cycle in a
27 warmer and moister world? *Geophysical Research Letters* **31** (L06207).
- 28 Lilenfeld H., Whitefield P.D., and Hagen D. (1995) Ground testing of emissions from aircraft exhaust.
29 Paper by the American Institute of Aeronautics and Astronautics, Paper AIAA-95-0110.
- 30 Lim B. and Cachier H. (1996) Determination of black carbon by chemical oxidation and thermal
31 treatment in recent marine and lake sediments and cretaceous-tertiary clays. *Chem. Geol.* **131**,
32 143-154.
- 33 Lim H.-J., Turpin B.J., Edgerton E., Hering S.V., Allen G., Maring H., and Solomon P. (2003)
34 Semicontinuous aerosol carbon measurements: comparison of Atlanta Supersite measurements.
35 *Journal of Geophysical Research* **108**, 8419 (doi:10.1029/2001JD001214).
- 36 Lin C.-I., Baker M., and Charlson R.J. (1973) Absorption coefficient of atmospheric aerosol: a method for
37 measurement. *Applied Optics* **12** (6), 1356-1363.
- 38 Lioussé C., Penner J.E., Chuang C., Walton J.J., and Eddleman H. (1996) A global three-dimensional study
39 of carbonaceous aerosols. *J. Geophys. Res.* **101** (D14), 19411- 19432.
- 40 Lipfert F.W., Baty J.D., Wyzga R.E., and Miller J.P. (2006) PM_{2.5} constituents and related air quality
41 variables as predictors of survival in a cohort of U.S. military veterans. *Inhal Toxicol.* **18**, 645-
42 657.
- 43 Lipfert F.W., Wyzga R.E., Baty J.D., and Miller J.P. (2009) Air pollution and survival within the Washington
44 University-EPRI veterans cohort: risks based on modeled estimates of ambient levels of
45 hazardous and criteria air pollutants. *J Air Waste Manage Assoc.* **59**, 473-489.
- 46 Lippmann M., Ito K., Hwang J.S., Maciejczyk P., and Chen L.C. (2006) Cardiovascular effects of nickel in
47 ambient air. *Environ Health Perspect* **114**.

- 1 Liu D., Wang Z., Liu Z., Winker D., and Trepte C. (2008a) A height resolved global view of dust aerosols
 2 from the first year CALIPSO lidar measurements. *J. Geophys. Res.* **113** (D16214)
 3 (doi:10.1029/2007JD009776).
- 4 Liu X., Xu B., Yao T., Wang N., and Wu G. (2008b) Carbonaceous particles in Muztagh Ata ice core, West
 5 Kunlun Mountains, China. *Chinese Science Bulletin* **53** (21), 3379-3386.
- 6 Liv, et al. (2007).
- 7 Lohmann U. and Hoose C. (2009) Sensitivity studies of different aerosol indirect effects in mixed-phase
 8 clouds. *Atmos. Chem. Phys.* **9**, 8917-8934.
- 9 Lohmann U., et al. (2010) Total aerosol effect: Radiative forcing or radiative flux perturbation? .
 10 *Atmospheric Chemistry and Physics* **10**.
- 11 Ma Y., L. and Hays M.D. (2008) Thermal extraction-two-dimensional gas chromatography-mass
 12 spectrometry with heart-cutting for nitrogen heterocyclics in biomass burning aerosols. *Journal*
 13 *of Chromatography A* **1200** (2), 228-234.
- 14 MacCarty, et al. (2008) A laboratory comparison of the global warming impact of five major types of
 15 biomass cooking stoves. *Energy for Sustainable Development* **XII**.
- 16 MacCarty, et al. (2010) Fuel use and emissions performance of fifty cooking stoves in the laboratory and
 17 related benchmarks of performance. *Energy for Sustainable Development* **14**, 161-171.
- 18 MACTEC (2005) Boiler best available retrofit technology (BART) engineering analysis. *Lake Michigan Air*
 19 *Directors Consortium (LADCO), March 30*.
- 20 Madrigano J., Baccarelli A., Wright R., Suh H., Sparrow D., Vokonas P., S., and Schwartz J. (2010) Air
 21 pollution, obesity, genes, and cellular adhesion molecules. *Occup Environ Med.* **67**, 312-317.
- 22 Manne A.S. and Richels R.G. (2001) An alternative approach to establishing trade-offs among
 23 greenhouse gases. *Nature* **410** (6829).
- 24 Mar T.F., Norris G.A., Koenig J.Q., and Larson T.V. (2000) Associations between air pollution and
 25 mortality in Phoenix, 1995-1997.
- 26 Maria S.F., et al. (2004) Organic aerosol growth mechanisms and their climate-forcing implications.
 27 *Science* **306**.
- 28 Marinoni, et al. (2010) Aerosol mass and black carbon concentrations, two year-round observations at
 29 NCO-P (5079m, Southern Himalayas). *Atmospheric Chemistry and Physics Discussions* **10**, 8379-
 30 8413.
- 31 Marr L.C., Dzepina K., Jimenez J.L., Reisen F., Bethel H.L., Arey J., Gaffney J.S., Marley N.A., Molina L.T.,
 32 and Molina M.J. (2006) Sources and transformations of particle-bound polycyclic aromatic
 33 hydrocarbons in Mexico City. *Atmospheric Chemistry and Physics* **6**.
- 34 Martins J.A., Silva Dias M.A.F., and Gonalves F.L.T. (2009) Impact of biomass burning aerosols on
 35 precipitation in the Amazon: I. A modeling study. *Journal of Geophysical Research* **114** (D02207)
 36 (doi: 10.1029/2007JD 009587).
- 37 Masclet P., Hoyau V., Jaffrezo J.L., and Cachier H. (2000) Polycyclic aromatic hydrocarbon deposition on
 38 the ice sheet of Greenland. Part I: superficial snow. *Atmos Environ* **34**, 3195-3207.
- 39 Masera, et al. (2007) Impact of Patsari improved cook stoves on indoor air quality in Michoacán, Mexico.
 40 *Energy for Sustainable Development* **11** (2).
- 41 Mauritsen T., Sedlar J., Tjernström M., Leck C., Martin M., Shupe M., Sjogren S., Sierau B., Persson
 42 P.O.G., Brooks I.M., and Swietlicki E. (2010) Aerosols indirectly warm the Arctic. *Atmospheric*
 43 *Chemistry and Physics* **10**, 16775-16796 (doi:10.5194/acpd-10-16775-2010).
- 44 Mayol-Bracero O.L., Gabriel R., Andreae M.O., Kirchstetter T.W., Novakov T., Ogren J., Sheridan P., and
 45 Streets D.G. (2002) Carbonaceous aerosols over the Indian Ocean during the Indian Ocean
 46 Experiment (INDOEX): chemical characterization, optical properties, and probable sources.
 47 *Journal of Geophysical Research* **107** (D19), 8030 (doi:10.1029/2000JD000039).

- 1 McCarty J.L. (2011) Remote sensing-based estimates of annual and seasonal emissions from crop
 2 residue burning in the contiguous United States. *J. Air & Waste Manage. Assoc.* **61**, 22-34.
- 3 McConnell J.R., Edwards R., Kok G.L., Flanner M.G., Zender C.S., Saltzman E.S., Banta J.R., Pasteris D.R.,
 4 Carter M.M., and Kahl J.D.W. (2007) 20th-century industrial black carbon emissions altered
 5 Arctic climate forcing. *Science* **317** (5843), 1381-1384. Available on the Internet at
 6 <http://www.sciencemag.org/cgi/content/abstract/1144856>.
- 7 McConnell J.R. and Edwards R. (2008) Coal burning leaves toxic heavy metal legacy in the Arctic.
 8 *Proceedings of the National Academy of Sciences* **105** (34), 12140-12144.
- 9 McConnell J.R. (2010) Historical black carbon and other ice core aerosol records in the Arctic for GCM
 10 evaluation. *Atmos Environ* **44**, 2665-2666.
- 11 McConnell R., Berhane K., Gilliland F., Molitor J., Thomas D., Lurmann F., Avol E., Gauderman W.J., and
 12 Peters J.M. (2003) Prospective study of air pollution and bronchitic symptoms in children with
 13 asthma. *Am J Respir Crit Care Med.* **168**.
- 14 McCracken, et al. (2007) Chimney stove intervention to reduce long-term wood smoke exposure lowers
 15 blood pressure among Guatemalan women. *Environ Health Perspect* **115** (7).
- 16 McKelvey K.S. and Busse K.K. (1996) Twentieth-century fire patterns on Forest Service lands. In *Sierra*
 17 *Nevada ecosystem project: final report to Congress, vol. II, assessments and scientific basis for*
 18 *management options*, University of California Davis Centers for Water and Wildland Resources,
 19 Davis, CA
- 20 MECA Diesel Retrofit Technology for Clean Air. Manufacturers of Emission Controls Association (MECA)
 21 (2010): What is retrofit?
- 22 Meehl and et. al. (2008) Effects of black carbon aerosols on the Indian monsoon. *Journal of Climate*
 23 (doi: 10.1175/2007JCLI1777.1).
- 24 Menon S., Hansen J., Nazarenko L., and Luo Y. (2002) Climate effects of black carbon aerosols in China
 25 and India. *Science* **297**, 2250-2253.
- 26 Menon S., Koch D., Beig G., Sahu S., Fasullo J., and Orlikowski D. (2010) Black carbon aerosols and the
 27 third polar ice cap. *Atmospheric Chemistry and Physics* **10**, 4559-4571 (doi: 10.5194/acp-10-
 28 4559-2010).
- 29 Mid-Atlantic Regional Air Management Association (2006) Control Analysis and Documentation for
 30 Residential Wood Combustion in the MANE-VU Region. Available on the Internet at
 31 http://www.marama.org/visibility/ResWoodCombustion/RWC_FinalReport_121906.pdf.
- 32 Miller J.D., Safford H.D., Crimmins M., and Thode A.E. (2009) Quantitative evidence for increasing forest
 33 fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA.
 34 *Ecosystems* **12**.
- 35 Ming J., Cachier H., Xiao C., Qin D., Kang S., Hou S., and Xu J. (2008) Black carbon record based on a
 36 shallow Himalayan ice core and its climatic implications. *Atmospheric Chemistry and Physics* **8**,
 37 1343-1352.
- 38 Ming J., Xiao C., Cachier H., Qin D., Qin X., Li Z., and Pu J. (2009) Black carbon in the snow of glaciers in
 39 west China and its potential effects on albedos. *Atmospheric Research* **92**, 114-123.
- 40 Mishchenko M.I., Cairns B., Kopp G., Schueler C.F., Fafaul B.A., Hansen J.E., Hooker R.J., Itchkawich T.,
 41 Maring H.B., and Travis L.D. (2007) Accurate monitoring of terrestrial aerosols and total solar
 42 irradiance: introducing the Glory mission. *Bulletin of the American Meteorological Society* **88**,
 43 677-691 (doi:10.1175/BAMS-88-5-677).
- 44 Mitra A.P. and Sharma C. (2002) Indian aerosols: present status. *Chemosphere* **49** (9).
- 45 Miyazaki Y., Kondo Y., Sahu L.K., Imaru J., Fukushima N., and Kano M. (2008) Performance of a newly
 46 designed continuous soot monitoring system (COSMOS). *Journal of Environmental Monitoring*
 47 **10**, 1195-1201 (doi: 10.1039/b806957c).
- 48 Moffet and Prather (2009a) Climate effects of black carbon aerosols in China and India. *Science* **297**.

- 1 Moffet R.C. and Prather K.A. (2009b) *In-situ* measurements of the mixing state and optical properties of
2 soot with implications for radiative forcing estimates. *Proceedings of the National Academy of*
3 *Sciences* **106** (29), 11872-11877.
- 4 Moffet R.C. and Prather K.A. (2009c) Climate effects of black carbon aerosols in China and India. *Science*
5 **297**.
- 6 Moffet R.C., Henn T.R., Tivanski A.V., Hopkins R.J., Desyaterik Y., Kilcoyne A.L.D., Tyliczszak T., Fast J.,
7 Barnard J., Shutthanandan V., Cliff S.S., Perry K.D., Laskin A., and Gilles M.K. (2010) Microscopic
8 characterization of carbonaceous aerosol particle aging in the outflow from Mexico City.
9 *Atmospheric Chemistry and Physics* **10**.
- 10 Molina M., Zaelke D., Sarma K.M., Andersen S.O., Ramanathan V., and Kanariu D. (2009) Reducing
11 abrupt climate change risk using the Montreal protocol and other regulatory actions to
12 complement cuts in CO₂ emissions. *PNAS* **106** (49).
- 13 Mooney J.J. (2006) Toxic solid nanoparticles: the importance of retrofitting diesel engine particle
14 emission control systems to older in-use diesel engines and available methods. November.
15 Available on the Internet at http://www.vert-certification.eu/attachments/061_CARB-EPA-
16 [VERT.pdf](http://www.vert-certification.eu/attachments/061_CARB-EPA-).
- 17 Mordukhovich I., Wilker E., Suh H., Wright R., Sparrow D., Vokonas P.S., and Schwartz J. (2009) Black
18 carbon exposure, oxidative stress genes, and blood pressure in a repeated-measures study.
19 *Environ Health Perspect.* **117** (11), 1767–1772 (doi: 10.1289/ehp.0900591).
- 20 Morgan G. (2009) Asia and Australasia wildfire management: a regional perspective. In *Proceedings of*
21 *the Third International Symposium on Fire Economics, Planning, and Policy: Common Problems*
22 *and Approaches*, A. González-Cabán ed., USDA Forest Service Pacific Southwest Research
23 Station, Albany, CA (PSW-GTR-227).
- 24 Muhlbaier-Dasch J. (1982) Particulate and gaseous emissions from wood-burning fireplaces.
25 *Environmental Science & Technology* **16**, 639-645.
- 26 Munive-Colin Z., Hernandez-Mena L., Sanchez-Sandoval M., Zhang,, et al. (2008) Carbonaceous aerosol
27 composition over various regions of China during 2006. *Journal of Geophysical Research* **113**,
28 D14111.
- 29 Muri G., Cermelj B., Faganeli J., and Brancelj A. (2002) Black carbon in Slovenian alpine lacustrine
30 sediments. *Chemosphere* **46**, 1225-1234.
- 31 Muri G., Wakeham S.G., and Faganeli J. (2003) Polycyclic aromatic hydrocarbons and black carbon in
32 sediments of a remote alpine lake (Lake Planina, Northwest Slovenia). *Environ. Toxic. Chem.* **22**,
33 1009-1016.
- 34 Murphy D.M., Chow J.C., Leibensperger E.M., Malm W.C., Pitchford M., Schichtel B.A., Watson J.G., and
35 White W.H. (2011) Decreases in elemental carbon and fine particle mass in the United States.
36 *Atmospheric Chemistry and Physics* **11**, 2057-2076. Available on the Internet at [www.atmos-](http://www.atmos-chem-phys-discuss.net/11/2057/2011/)
37 [chem-phys-discuss.net/11/2057/2011/](http://www.atmos-chem-phys-discuss.net/11/2057/2011/).
- 38 Mussati D. and Hemmer P. (2002) Section 6. Particulate matter controls. Chapter 2: wet scrubbers for
39 particulate matter. In *EPA air pollution control cost manual (sixth edition)*, U.S. Environmental
40 Protection Agency, (EPA/452/B-02-001).
- 41 Myers R. (2006) Back to the back to the future: stationary source testing for fine PM. *AWMA EM* **April**,
42 25-30.
- 43 Myhre and et. al. (2009) Modelled radiative forcing of the direct aerosol effect with multi-observation
44 evaluation. *Atmospheric Chemistry and Physics* **9**. Available on the Internet at [www.atmos-](http://www.atmos-chem-phys.net/9/1365/2009/)
45 [chem-phys.net/9/1365/2009/](http://www.atmos-chem-phys.net/9/1365/2009/).
- 46 Naeher, et al. (2000) Particulate matter and carbon monoxide in highland Guatemala: indoor and
47 outdoor levels from traditional and improved wood stoves and gas stoves. *Indoor Air* **10**.

- 1 Naidoo G. and Chirkoot D. (2004) The effects of coal dust on photosynthetic performance of the
2 mangrove, *Avicennia marina* in Richards Bay, South Africa. *Environmental Pollution* **127**, 359-
3 366.
- 4 Nam E., Kishan S., Baldauf R.W., Fulper C.R., Sabisch M., and Warila J. (2010) Temperature effects on
5 particulate matter emissions from light-duty, gasoline-powered motor vehicles. *Environ. Sci.*
6 *Technology* **44**, 4672-4677.
- 7 NARSTO (2005) A NARSTO assessment: improving emission inventories for effective air quality
8 management across North America. Prepared by the NARSTO Emission Inventory Assessment
9 Team. Available on the Internet at <http://www.narsto.org/section.src?SID=8>.
- 10 National Academies Press (2001) *Evaluating vehicle inspections and maintenance programs*,
11 Washington, DC.
- 12 National Institute for Occupational Safety and Health (1999) National Institute for Occupational Safety
13 and Health (NIOSH) method 5040, elemental carbon (diesel particulate). In *NIOSH manual of*
14 *analytical methods (NMAM), 4th Ed., Issue 3*, National Institute for Occupational Safety and
15 Health
- 16 National Interagency Fire Center (2011) Prevention and education. Available on the Internet at
17 http://www.nifc.gov/prevention_education.html
- 18 National Research Council Board on Atmospheric Sciences and Climate-National Academy of Sciences
19 (2010) *Climate stabilization targets: emissions, concentrations, and impacts over decades to*
20 *millennia*, National Academies Press, Washington, DC.
- 21 National Research Council of the National Academies (2005) *Radiative forcing of climate change:*
22 *expanding the concept and addressing uncertainties*, The National Academies Press,
23 Washington, DC.
- 24 National Wildfire Coordinating Group (2007) Chapter 8. Fire prevention. In *Communicator's guide for*
25 *wildland fire management: fire education, prevention, and mitigation practices*. Available on the
26 Internet at
27 [http://www.nifc.gov/preved/comm_guide/wildfire/files/pdf%20%20files/linked%20pdfs/8%20fi](http://www.nifc.gov/preved/comm_guide/wildfire/files/pdf%20%20files/linked%20pdfs/8%20fire%20prevention.pdf)
28 [re%20prevention.pdf](http://www.nifc.gov/preved/comm_guide/wildfire/files/pdf%20%20files/linked%20pdfs/8%20fire%20prevention.pdf).
- 29 Nazaroff W.W. and Cass G.R. (1991) Protecting museum collections from soiling due to the deposition of
30 airborne particles. *Atmos Environ* **25**, 841-852.
- 31 Nenes A., Conant W.C., and Seinfeld J.H. (2002) Black carbon radiative heating effects on cloud
32 microphysics and implications for the aerosol indirect effect – 2. Cloud microphysics. *Journal of*
33 *Geophysical Research* **107**, 4605.
- 34 NESCAUM (2005) Assessment of control technology options for BART-eligible sources steam electric
35 boilers, industrial boilers, cement plants and paper and pulp facilities. In *Air Pollution*
36 *Engineering Manual, Air & Waste Management Association*, A.J. Buonicore and W.T. Davis eds.,
37 Van Nostrand Reinhold, New York, NY
- 38 NESCAUM (2009) Applicability and feasibility of NO_x, SO₂, and PM emissions control technologies for
39 industrial, commercial, and institutional (ICI) boilers. January.
- 40 NESCAUM (2010) Technologies for control and measurement of mercury emissions from coal-fired
41 power plants in the United States: a status report. July.
- 42 Niwa Y., Hiura Y., Murayama T., Yokode M., and Iwai N. (2007) Nano-sized carbon black exposure
43 exacerbates atherosclerosis in LDL-receptor knockout mice. *Circulation J* **71**.
- 44 Noone and Clarke (1998) Soot scavenging measurements in arctic snowfall. *Atmos Environ* **22** (12).
- 45 Noss R.F., Franklin J.F., Baker W.L., Schoennagel T., and Moyle P.B. (2006) Managing fire-prone forests in
46 the western United States. *Front Ecol Environ* **4** (9).
- 47 Notten M., Oosthoek A.J.P., Rozema J., and Aerts R. (2005) Heavy metal concentrations in a soil-plant-
48 snail food chain along a terrestrial soil pollution gradient. *Environ Pollut* **138**, 178-190.

- 1 Novakov T. and Corrigan C.E. (1995a) Thermal characterization of biomass smoke particles. *Mikrochim*
 2 *Acta* **119**, 157-166.
- 3 Novakov T. and Corrigan C.E. (1995b) Influence of sample composition on aerosol organic and black
 4 carbon determinations. Prepared by Lawrence Berkeley National Laboratory, Berkeley, CA, LBL-
 5 37513 UC-407.
- 6 Novakov T., Ramanathan V., Hansen J., Kirchstetter T., Sato M., Sinton J., and Sathaye J. (2003) Large
 7 historical changes of fossil-fuel black carbon aerosols. *Geophys. Res. Lett.* **30**
 8 (doi:10.1029/2002gl016345).
- 9 Novakov T., Menon S., Kirchstetter T.W., Koch D., and Hansen J.E. (2005) Aerosol organic carbon to black
 10 carbon ratios: analysis of published data and implications for climate forcing. *Journal of*
 11 *Geophysical Research* **110** (D21205) (doi: 10.1029/2005JD005977).
- 12 O'Neill B.C. (2000) The jury is still out on global warming potentials. *Climatic Change* **44**, 427-443.
- 13 O'Neill B.C. (2003) Economics, natural science, and the costs of global warming potentials. *Climatic*
 14 *Change* **58** (3).
- 15 O'Neill M.S., Veves A., Sarnat J.A., Zanobetti A., Gold D.R., Economides P.A., Horton E.S., and Schwartz J.
 16 (2007) Air pollution and inflammation in type 2 diabetes: a mechanism for susceptibility. *Occup*
 17 *Environ Med* **64**.
- 18 Oakes M., Rastogi N., Majestic B.J., Shafer M., Schauer J.J., Edgerton E.S., and Weber R.J. (2010)
 19 Characterization of soluble iron in urban aerosols using near-real time data. *Journal of*
 20 *Geophysical Research* **115** (D15302).
- 21 Office of the Press Secretary-White House (2010) Presidential memorandum regarding fuel efficiency
 22 standards. May. Available on the Internet at [http://www.whitehouse.gov/the-press-](http://www.whitehouse.gov/the-press-office/presidential-memorandum-regarding-fuel-efficiency-standards)
 23 [office/presidential-memorandum-regarding-fuel-efficiency-standards](http://www.whitehouse.gov/the-press-office/presidential-memorandum-regarding-fuel-efficiency-standards).
- 24 Ogren J.A., Charlson R.J., and Groblicki P.J. (1983) Determination of elemental carbon in rainwater. *Anal.*
 25 *Chem.* **55**, 1569-1572.
- 26 Ogren J.A., Groblicki P.J., and Charlson R.J. (1984) Measurement of the removal rate of elemental carbon
 27 from the atmosphere. *The Science of the Total Environment* **36**, 329-338.
- 28 Ohara T., Akimoto H., Kurokawa K., Horii N., Yamaji K., Yan X., and Hayasaka T. (2007) An Asian emission
 29 inventory of anthropogenic emission sources for the period 1980-2020. *Atmospheric Chemistry*
 30 *and Physics* **7** (16), 4419-4444.
- 31 Orozco-Levi M., Garcia-Aymerich J., Villar J., Ramirez-Sarmiento A., Anto J.M., and Gea J. (2006) Wood
 32 smoke exposure and risk of chronic obstructive pulmonary disease. *European Respiratory*
 33 *Journal* **27** (3).
- 34 Ostro B., Feng W.-Y., Broadwin R., Green S., and Lipsett M. (2007) The effects of components of fine
 35 particulate air pollution on mortality in California: results from CALFINE. *Environ Health Perspect*
 36 **115**.
- 37 Ostro B.D., Feng W.Y., Broadwin R., Malig B.J., Green R.S., and Lipsett M.J. (2008) The impact of
 38 components of fine particulate matter on cardiovascular mortality in susceptible
 39 subpopulations. *Occup Environ Med* **65**.
- 40 Ottmar R.D., Peterson J.L., Leenhouts B., and Core J.E. (2001) Smoke management: techniques to reduce
 41 or redistribute emissions. In *Smoke management guide for prescribed and wildland fire: 2001*
 42 *edition*, C.C. Hardy, R.D. Ottmar, J.L. Peterson, J.E. Core, and P. Seamon eds., National Wildfire
 43 Coordination Group.
- 44 Pacala S. and Socolow R. (2004) Stabilization wedges: solving the climate problem for the next 50 years
 45 with current technologies *Science* **305** (5686), 968-972.
- 46 Painter, et al. (2007) Impact of disturbed desert soils on duration of mountain snow cover. *Geophys. Res.*
 47 *Lett.* **34**.

- 1 Parashar D.C., Gadi R., et al. (2005) Carbonaceous aerosol emissions from India. *Atmos Environ* **39** (40),
 2 7861-7871.
- 3 Park S.K., O'Neill M.S., Vokonas P.S., Sparrow D., and Schwartz J. (2005) Effects of air pollution on heart
 4 rate variability: The VA normative aging study. *Environ Health Perspect* **113**.
- 5 Park S.K., O'Neill M.S., Vokonas P.S., Sparrow D., Spiro A., Tucker K.L., Suh H., Hu H., and Schwartz J.
 6 (2008) Traffic-related Particles are Associated with Elevated Homocysteine. *Am J Respir Crit Care*
 7 *Med.* **178**, 283-289.
- 8 Park S.S., Hansen A.D.A., and Cho S.Y. (2010) Measurement of real time black carbon for investigating
 9 spot loading effects of Aethalometer data. *Atmospheric Environment* **44**, 1449-1455.
- 10 Parrish D.D. (2006) Critical evaluation of US on-road vehicle emission inventories. *Atmos Environ* **40**,
 11 2288-2300.
- 12 Partanen W.E. and Allen C. (2005) Green energy from fats, oil & grease (FOG). Prepared by Power-Gen
 13 International, Las Vegas, NV.
- 14 Partnership for Clean Fuels and Vehicles (2009) Cleaning up urban bus fleets.
- 15 Partnership for Clean Indoor Air (2005) *Household energy, indoor air pollution and health: overview of*
 16 *experiences and lessons in India*, Winrock International, Little Rock, AR.
- 17 Patterson E.M. and McMahon C.K. (1984) Absorption characteristics of forest fire particulate matter.
 18 *Atmos Environ* **18**, 184.
- 19 Peck, et al. (2008) Burns and fires from non-electric domestic appliances in low and middle income
 20 countries. *Burns* **34**, 303-311.
- 21 Peng R., Bell M., Geyh A., McDermott A., Zeger S., Samet J., and Dominici F. (2009) Emergency
 22 admissions for cardiovascular and respiratory diseases and the chemical composition of fine
 23 particle air pollution. *Environ Health Perspect* **117**.
- 24 Penner J.E., Eddleman H., et al. (1993) Towards the development of a global inventory for black carbon
 25 emissions. *Atmos Environ* **27** (8), 1277-1295 (Part A. General Topics).
- 26 Perovich D.K., Grenfell T.C., Light B., Elder B.C., Harbeck J., Polashenski C., Tucker III W.B., and Stelmach
 27 C. (2009) Transpolar observations of the morphological properties of Arctic sea ice. *Journal of*
 28 *geophysical Research* **114** (C00A04) (doi:10.1029/2008JC004892).
- 29 Peters A., Dockery D.W., Muller J.E., and Mittleman M.A. (2001) Increased particulate air pollution and
 30 the triggering of myocardial infarction. *Circulation* **103**.
- 31 Peterson M.R. and Richards M.H. (2002) Thermal-optical-transmittance analysis for organic, elemental,
 32 carbonate, total carbon, and OCX2 in PM2.5 by the EPA/NIOSH method. In *Proceedings,*
 33 *symposium on air quality measurement methods and technology-2002*, E.D. Winegar and R.J.
 34 Tropp eds., Air & Waste Management Association, Pittsburgh, PA (83-1-83-19).
- 35 Petzold A., Kopp C., et al. (1997) The dependence of the specific attenuation cross-section on black
 36 carbon mass fraction and particle size. *Atmos Environ* **31** (5), 661-672.
- 37 Petzold A., Doppelheuer A., Brock C.A., and Schroder F. (1999a) *In situ* observations and model
 38 calculations of black carbon emission by aircraft at cruise altitude. *Journal of Geophysical*
 39 *Research-Atmospheres* **104** (D18), 22171-22181.
- 40 Petzold A., Strom F.P., Schroder F.P., and Karcher B. (1999b) Carbonaceous aerosol in jet engine exhaust:
 41 emission characteristics and implications for heterogeneous chemical reactions. *Atmospheric*
 42 *Environment* **33**, 2689-2698.
- 43 Pew Center (2010) Workshop summary report. Assessing the benefits of avoided climate change: cost-
 44 benefit analysis and beyond. Washington, DC.
- 45 Pitchford M. and Malm W. (1993) Development and applications of a standard visual index. *Atmos*
 46 *Environ* **28** (5), 1049-1054.
- 47 Polenske K. and McMichael F. (2002) A Chinese cokemaking process-flow model for energy and
 48 environmental analyses. *Energy Policy* **30**.

- 1 Pope D.P., Mishra V., Thompson L., Siddiqui A.R., Rehfuess E.A., Weber M., and Bruce N.G. (2010) Risk of
2 low birth weight and stillbirth associated with indoor air pollution from solid fuel use in
3 developing countries. *Epidemiology Reviews* (doi: 10.1093/epirev/mxq005).
- 4 Pósfai M., Gelencsér A., Simonics R., Arató K., Li J., Hobbs P.V., and Buseck P.R. (2004) Atmospheric tar
5 balls: Particles from biomass and biofuel burning. *Journal of Geophysical Research* **109** (D06213)
6 (doi:10.1029/2003JD004169).
- 7 Pósfai M. and Buseck P.R. (2010) Nature and climate effects of individual tropospheric aerosol particles.
8 *Annu. Rev. Earth Planet. Sci.* **38**, 17-43.
- 9 Pratt K.A. and Prather K. (2009) Real-time, single-particle volatility, size, and chemical composition
10 measurements of aged urban aerosol. *Environ Sci Technol.* **43** (21), 8276-8282.
- 11 Pratt K.A. and Prather K.A. (2010) Aircraft measurements of vertical profiles of aerosol mixing states.
12 *Journal of Geophysical Research* **115** (D11305) (doi:10.1029/2009JD013150).
- 13 Prospero J.M., Landing W.M., and Schulz M. (2010) African dust deposition to Florida: Temporal and
14 spatial variability and comparisons to models. *Journal of Geophysical Research* **115** (D13304)
15 (doi:10.1029/2009JD012773).
- 16 Pueschel R.F., Boering K.A., Verma S., Howard S.D., Ferry G.V., Goodman J., Allen D.A., and Hamill P.
17 (1997) Soot aerosol in the lower stratosphere: pole-to-pole variability and contributions by
18 aircraft. *Journal of Geophysical Research* **102** (D11), 13113-13118.
- 19 Qian Y., Gustafson W.I., Leung L.R., and Ghan S.J. (2009) Effects of soot-induced snow albedo change on
20 snowpack and hydrological cycle in western United States based on Weather Research and
21 Forecasting chemistry and regional climate simulations. *Journal of Geophysical Research* **114**
22 (D03108) (doi:10.1029/2008JD011039).
- 23 Quincey P. (2007) A relationship between black smoke index and black carbon concentration. *Atmos*
24 *Environ* **41**, 7964-7968.
- 25 Quinn, et al. (2008) Short-lived pollutants in the Arctic: their climate impact and possible mitigation
26 strategies. *Atmospheric Chemistry and Physics* **8**.
- 27 Quinn P.K., Shaw G., Andrews E., Dutton E.G., Ruoho-Airola T., and Gong S.L. (2007) Arctic haze: current
28 trends and knowledge gaps. *Tellus* **59B**, 99-114 (doi: 10.1111/j.1600-0889.2006.00238.x).
- 29 Quiroz R., Grimalt J.O., and Fernandez P. (2010) Toxicity assessment of polycyclic aromatic hydrocarbons
30 in sediments from European high mountain lakes. *Ecotoxicology and Environmental Safety* **73**.
- 31 Ram K., Sarin M.M., and Tripathi S.N. (2010) Inter-comparison of thermal and optical methods for
32 determination of atmospheric black carbon and attenuation coefficient from an urban location
33 in northern India. *Atmospheric Research* **97**, 335-342.
- 34 Ramana M.V., Ramanathan V., Feng Y., Yoon S.-C., Kim S.-W., Carmichael G.R., and Schauer J.J. (2010)
35 Warming influenced by the ratio of black carbon to sulphate and the black-carbon source.
36 *Nature Geoscience* **3**, 542-545 (doi:10.1038/NGE0918).
- 37 Ramanathan V., Crutzen P.J., Kiehl J.T., and Rosenfeld D. (2001) Aerosols, climate, and the hydrological
38 cycle. *Science* **294**, 2119-2124.
- 39 Ramanathan V., Chung C., Kim D., Bettge T., Buja L., Kiehl J.T., Washington W.M., Fu Q., Sikka D.R., and
40 Wild M. (2005) Atmospheric brown clouds: impacts on South Asian climate and hydrological
41 cycle. *PNAS* **102** (15), 5326-5333.
- 42 Ramanathan V. (2007) Atmospheric brown clouds: hemispherical and regional variations in long-range
43 transport, absorption, and radiative forcing. *Journal of Geophysical Research* **112** (D22821) (doi:
44 10.1029/2006JD008124).
- 45 Ramanathan V. and Carmichael G. (2008) Global and regional climate changes due to black carbon.
46 *Nature Geoscience* **1**, 221-227.
- 47 Ramanathan V. and Feng W.Y. (2009) Air pollution, greenhouse gases and climate change: Global and
48 regional perspectives. *Atmos Environ* **43**.

- 1 Ramanathan V. (2010a) Clearing the smoke: understanding the impacts of black carbon pollution.
2 Testimony before the Select Committee on Energy Independence and Global Warming, United
3 States House of Representatives, 111th Congress, March 16.
- 4 Ramanathan V. (2010b) Interactive comment on "Quantifying immediate radiative forcing by black
5 carbon and organic matter with the Specific Forcing Pulse" by T.C. Bond, et al. *Atmospheric*
6 *Chemistry and Physics Discussions* **10**, C6227-C6241. Available on the Internet at
7 <http://www.atmos-chem-phys-discuss.net/10/C6227/2010/acpd-10-C6227-2010.pdf>.
- 8 Ramirez-Venegas A., Sansores R.H., Perez-Padilla R., Regalado J., Velazquez A., Sanchez C., and Mayar
9 M.E. (2006) Survival of patients with chronic obstructive pulmonary disease due to biomass
10 smoke and tobacco. *Am. J. Respir. Crit. Care Med.* **173**, 393-397.
- 11 Ramos L., Hernandez L.M., and Gonzalez M.J. (1994) Sequential fractionation of copper, lead, cadmium
12 and zinc in soils from or near Donana National Park. *Journal of Environmental Quality* **23**, 50-57.
- 13 Randerson J.T., Liu H., Flanner M.G., Chambers S.D., Jin Y., Hess P.G., Pfister G., Mack M.C., Treseder
14 K.K., Welp L.R., Chapin F.S., Harden J.W., Goulden M.L., Lyons E., Neff J.C., Schuur E.A.G., and S.
15 Z.C. (2006) The impact of Boreal forest fire on climate warming. *Science* **314**, 1130-1132 (doi:
16 10.1126/science.1132075).
- 17 Reddy M.S. and Venkataraman C. (2002a) Inventory of aerosol and sulphur dioxide emissions from India:
18 part I--fossil fuel combustion. *Atmos Environ* **36** (4), 677-697.
- 19 Reddy M.S. and Venkataraman C. (2002b) Inventory of aerosol and sulphur dioxide emissions from India.
20 Part II--biomass combustion. *Atmos Environ* **36** (4).
- 21 Reddy M.S. and Boucher O. (2007) Climate impact of black carbon emitted from energy consumption in
22 the world's regions. *Geophys. Res. Lett.* **34** (L11802).
- 23 Reff A., Bhawe P.V., Simon H., Pace T.G., Pouliot G.A., Mobley J.D., and Houyoux M. (2009) Emissions
24 inventory of PM2.5 trace elements across the United States. *Environmental Science &*
25 *Technology* **43**, 5790-5796.
- 26 Regalado J., Perez-Padilla R., Sansores R.H., et al. (2006) The effect of biomass burning on respiratory
27 symptoms and lung function in rural Mexican women. *Am J Respiratory and Critical Care*
28 *Medicine* **174**, 901-905.
- 29 Regional Planning Organization (2004a) Research and development of planned burning emission
30 inventories for the Central States Regional Air Planning Association. Final report prepared for
31 the Central States Air Resource Agencies and the Central Regional Air Planning Association,
32 Oklahoma City, OK by Sonoma Technology, Inc., Petaluma, CA, STI-902514-2516-FR, July.
33 Available on the Internet at [http://www.dnr.mo.gov/env/apcp/docs/d2-cenrap-](http://www.dnr.mo.gov/env/apcp/docs/d2-cenrap-plannedburn.pdf)
34 [plannedburn.pdf](http://www.dnr.mo.gov/env/apcp/docs/d2-cenrap-plannedburn.pdf).
- 35 Regional Planning Organization (2004b) Fire emissions inventory development for the Midwest Regional
36 Planning Organization. Prepared for the Lake Michigan Air Director's Consortium by EC/R
37 Incorporated, September.
- 38 Regional Planning Organization (2005) 2002 fire emission inventory for the WRAP Region – Phase II.
39 Prepared for Western Governors Association/Western Regional Air Partnership by Air Sciences,
40 Inc., Project No. 178-6, July. Available on the Internet at
41 http://www.wrapair.org/forums/fejf/documents/WRAP_2002_PhII_EI_Report_20050722.pdf.
- 42 Regional Planning Organization (2006) Technical support document on agricultural and forestry smoke
43 management in the MANE-VU Region. Draft. Available upon request, March.
- 44 Regional Planning Organization (2008) Documentation of the base G2 and best & final 2002 base year,
45 2009 and 2018 emission inventories for VISTAS. Prepared by Mactec, Inc., Revision 1, April.
46 Available on the Internet at
47 http://www.dep.state.fl.us/air/rules/regulatory/regional_haze_imp/app_d.pdf.

- 1 Regoli F., Gorbi S., Fattorini D., Tedesco S., Notti A., Machella N., Bocchetti R., Benedetti M., and Piva F.
2 (2006) Use of the land snail *Helix aspersa* sentinel organism for monitoring ecotoxicologic
3 effects of urban pollution: an integrated approach. *Environ Health Perspect* **114**, 63-69.
- 4 Rehfuess E., Mehta S., and Pruss-Ustun A. (2006) Assessing household solid fuel use: multiple
5 implications for the Millennium Development Goals. *Environ Health Perspect* **114** (3), A178.
- 6 Reid J.S., Koppmann R., Eck T.F., and Eleuterio D.P. (2005) A review of biomass burning emissions part II:
7 intensive physical properties of biomass burning particles. *Atmospheric Chemistry and Physics* **5**.
- 8 Ren C., Park S.K., Vokonas P.S., Sparrow D., Wilker E., Baccarelli A., Suh H.H., Tucker K.L., Wright R.O.,
9 and Schwartz J. (2010) Air Pollution and Homocysteine. *Epidemiology* **21**, 198-206.
- 10 Riahi K., Gruebler A., and Nakicenovic N. (2007) Scenarios of long-term socio-economic and
11 environmental development under climate stabilization. *Technological Forecasting and Social*
12 *Change* **74** (7), 887-935.
- 13 Rich D.Q., Schwartz J., Mittleman M.A., Link M., Luttmann-Gibson H., Catalano P.J., Speizer F.E., and
14 Dockery D.W. (2005) Association of short-term ambient air pollution concentrations and
15 ventricular arrhythmias. *American Journal of Epidemiology* **161**.
- 16 Rich K.E., Petkau J., Vedal S., and Brauer M. (2004) A case-crossover analysis of particulate air pollution
17 and cardiac arrhythmia in patients with implantable cardioverter defibrillators. *Inhal Toxicol* **16**.
- 18 Riemer N., Vogel H., and Vogel B. (2004) Soot aging time scales in polluted regions during day and night.
19 *Atmos. Chem. Phys.* **4**.
- 20 Riemer N., West M., Zaveri R., and Easter R. (2010) Estimating black carbon aging time-scales with a
21 particle-resolved aerosol model. *Aerosol Science and Technology* **41**.
- 22 Rinne S.T., Rodas E.J., Bender B.S., Rinne M.L., Simpson J.M., R. G.-U., and Glickman L.T. (2006)
23 Relationship of pulmonary function among women and children to indoor air pollution from
24 biomass use in rural Ecuador. *Respiratory Medicine* **100** (7), 1208-1215.
- 25 Roberts D.L. and Jones A. (2004) Climate sensitivity to black carbon aerosol from fossil fuel combustion.
26 *J. Geophys. Res.* **109**, D16202 (doi:10.1029/2004JD004676).
- 27 Robinson, et al. (2007) Rethinking organic aerosols: semivolatile emissions and photochemical aging.
28 *Science* **315**, 1259-1262.
- 29 Rodhe H. (1990) A comparison of the contribution of various gases to the greenhouse effect. *Science*
30 **248**, 4960.
- 31 Rosen H. and Novakov T. (1977) Raman scattering and the characterization of atmospheric aerosol
32 particles. *Nature* **266**.
- 33 Rosen H. and Novakov T. (1983) Optical transmission through aerosol deposits on diffusely reflective
34 filters: a method for measuring the absorbing component of aerosol particles. *Applied Optics* **22**
35 (9), 1265-1267.
- 36 Ruckstuhl C., Philipona R., Behrens K., Coen M.C., Duerr B., Heimo A., Mätzler C., Nyeki S., Ohmura A.,
37 Vuilleumier L., Weller M., Behrens K., Wehrli C., and Zelenka A. (2008) Aerosol and cloud effects
38 on solar brightening and the recent rapid warming. *Geophys. Res. Lett.* **35**, L12708
39 (doi:10.1029/2008GL034228).
- 40 Russell P.B., Bergstrom R.W., Shinozuka Y., Clarke A.D., DeCarlo P.F., Jimenez J.L., M.Livingston J.,
41 Redemann J., Dubovik O., and Strawa A. (2010) Absorption angstrom exponent in AERONET and
42 related data as an indicator of aerosol composition. *Atmos. Chem. Phys.* **10**, 1155-1169.
43 Available on the Internet at www.atmos-chem-phys.net/10/1155/2010/.
- 44 Ryan M., Archer S., Birdsey R., Dahm C., Heath L., Hicke J., Hollinger D., Huxman T., Okin G., Oren R.,
45 Randerson J., and Schlesinger W. (2008) Land resources. In *The effects of climate change on*
46 *agriculture, land resources, water resources, and biodiversity*, U.S. Climate Change Science
47 Program and the Subcommittee on Global Change Research, Washington, DC.

- 1 Rypdal K., Rive N., Berntsen T., Klimont Z., Mideksa T.K., Myhre G., and Skeie R.B. (2009) Costs and
2 global impacts of BC abatement strategies. *Tellus B* **61** (4).
- 3 Sacramento Metropolitan Air Quality Management District (2009) *Rule 421, mandatory episodic*
4 *curtailment of wood and other solid fuel burning*. Staff Report. Available on the Internet at
5 <http://www.airquality.org/bod/2009/SepRule421HearingAttD-StaffReport.pdf>.
- 6 Sahu L.K., Kondo Y., Miyazaki Y., Kuwata M., Koike M., Takegawa N., Tanimoto H., Matsueda H., Yoon
7 S.C., and Kim Y.J. (2009) Anthropogenic aerosols observed in Asian continental outflow at Jeju
8 Island, Korea, in spring 2005. *Journal of Geophysical Research* **114**, D03301
9 (doi:10.1029/2008JD010306).
- 10 Sahu S.K., Beig G., et al. (2008) Decadal growth of black carbon emissions in India. *Geophys. Res. Lett.*
11 **35** (2), L02807.
- 12 Saikawa E., Naik V., Horowitz L.W., Liu J., and Mauzerall D.L. (2009) Present and potential future
13 contributions of sulfate, black and organic carbon aerosols from China to global air quality
14 premature mortality and radiative forcing. *Atmospheric Environment* **43**.
- 15 Saldiva P.H.N., Clarke R.W., Coull B.A., Stearns R.C., Lawrence J., Krishna-Murthy G.G., Diaz E., Koutrakis
16 P., Suh H., Tsuda A., and Godleski J.J. (2002) Lung inflammation induced by concentrated
17 ambient air particles is related to particle composition. *Am J Respir Crit Care Med.* **165**.
- 18 Salma I., Ocskay R., Chi X., and Maenhaut W. (2007) Sampling artifacts, concentrations and chemical
19 composition of fine water soluble organic carbon and humic-like substances in a continental
20 urban atmospheric environment. *Atmos. Environ.* **41**, 4106–4118.
- 21 Salma I., T., M'esz'aros W., Maenhaut E., Vass, and Majer Z. (2010) Chirality and the origin of
22 atmospheric humic-like substances. *Atmos. Chem. Phys.* **10**.
- 23 Sarnat J.A., Marmur A., Klein M., Kim E., Russell A.G., Sarnat S.E., Mulholland J.A., Hopke P.K., and
24 Tolbert P.E. (2008) Fine particle sources and cardiorespiratory morbidity: An application of
25 chemical mass balance and factor analytical source-apportionment methods. *Environ Health*
26 *Perspect* **116**.
- 27 Sarofim M. (2010) Using BC metrics in climate policy. *Journal of Integrative Environmental Sciences.* (*in*
28 *press*).
- 29 Sato M., et al. (2003) Global atmospheric black carbon inferred from AERONET. *PNAS* **100** (11).
- 30 Saylor R.D., Edgerton E.S., and Hartsell B.E. (2006a) Linear regression techniques for use in the EC tracer
31 method of secondary organic aerosol estimation. *Atmospheric Environment* **40** (39).
- 32 Saylor R.D., Edgerton E.S., and Hartsell B.E. (2006b) Linear regression techniques for use in the EC tracer
33 method of secondary organic aerosol estimation. *Atmospheric Environment* **40** (39), 7546-7556.
- 34 Schauer J.J., et al. (2003) ACE-Asia intercomparison of a thermal-optical method for the determination
35 of particle-phase organic and elemental carbon. *Environmental Science & Technology* **37**.
- 36 Schimel D., Alves D., Enting I., Heimann M., Joos F., Raynaud D., Wigley T., Prather M., Derwent R.,
37 Ehhalt D., Fraser P., Sanhueza E., Zhou X., Jonas P., Charlson R., Rodhe H., Sadasivan S., Shine
38 K.P., Fouquart Y., Ramaswamy V., Solomon S., Srinivasan J., Albritton D., Isaksen I., Lal M., and
39 Wuebbles D. (1996) Radiative forcing of climate change. In *Climate change 1995: the science of*
40 *climate change. Contribution of Working Group I to the Second Assessment Report of the*
41 *Intergovernmental Panel on Climate Change*, J.T. Houghton, L.G.M. Filho, B.A. Callander, N.
42 Harris, A. Kattenberg, and K. Maskell eds., Cambridge University Press, Cambridge, United
43 Kingdom and New York, NY, 65-131
- 44 Schmid H., et al. (2001) Results of the "carbon conference" international aerosol carbon round robin test
45 stage I. *Atmospheric Environment* **35**.
- 46 Schmidt G., Ruedy R., Hansen J., Aleinov I., Bell N., Bauer M., Bauer S., Cairns B., Canuto V., Cheng Y., A.
47 Del Genio, Faluvegi G., Friend A., Hall T., Hu Y., Kelley M., Kiang N., Koch D., Lacis A., Lerner J., Lo
48 K., Miller R., Nazarenko L., Oinas V., Perlwitz J., Perlwitz J., Rind D., Romanou A., Russel G., Sato

- 1 M., Shindell D., Stone P., Sun S., Tausnev N., Thresher D., and Yao M. (2006) Present-day
2 atmospheric simulation using GISS Model: Comparison to in situ, satellite, and reanalysis data. *J.*
3 *Climate* **19**.
- 4 Schnaiter M., Horvath H., Möhler O., Naumann K.H., Saathoff H., and Schock O.W. (2003) UV-VIS-NIR
5 spectral optical properties of soot and soot-containing aerosols. *J. Aerosol Sci.* **34**
6 (doi:10.1016/S0021-8502(03)00361-6).
- 7 Schnaiter M., Linke C., Möhler O., Naumann K.H., Saathoff H., Wagner R., Schurath U., and Wehner B.
8 (2005) Absorption amplification of black carbon internally mixed with secondary organic
9 aerosol. *J. Geophys. Res.* **110**, D19204 (doi:10.1029/2005JD006046).
- 10 Schneider S.H., et al. (2007) Assessing key vulnerabilities and the risk from climate change. In *Climate*
11 *change 2007: impacts, adaptation and vulnerability. Contribution of Working Group II to the*
12 *Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, M.L. Parry, O.F.
13 Canziani, J.P. Palutikof, P.J. van der Linden, and C.E. Hanson eds., Cambridge University Press,
14 Cambridge, UK, 779-810
- 15 Schulz M., et al. (2006) Radiative forcing by aerosols as derived from the aerocom present-day and pre-
16 industrial simulations. *Atmos Chem Phys* **6**.
- 17 Schwartz J., Litonjua A., Suh H., Verrier M., Zanobetti A., Syring M., Nearing B.D., Verrier R.L., Stone P.,
18 MacCallum G., Speizer F.E., and Gold D.R. (2005) Traffic related pollution and heart rate
19 variability in a panel of elderly subjects. *Thorax* **60**.
- 20 Seinfeld J. and Pandis S. (2006) *Atmospheric chemistry and physics: From air pollution to climate change*
21 *(second edition)*, John Wiley & Sons, Inc., Hoboken, New Jersey, 1203 (ISBN 0-471-72018-6).
- 22 Sharma S., Brook J.R., Cachier H., Chow J., Gaudenzi A., and Lu G. (2002) Light absorption and thermal
23 measurements of black carbon in different regions of Canada. *Journal of Geophysical Research*
24 **107** (D24), 4771 (doi:10.1029/2002JD002496).
- 25 Shaw G.E. and Stamnes K. (1980) Arctic haze: perturbation of the polar radiation budget. *Ann. N.Y. Acad.*
26 *Sci.* **338**, 533-539.
- 27 Shih S.-I., Lee W.-J., Lin L.-F., Huang J.-Y., Su J.-W., and Chang-Chien G.-P. (2008) Significance of biomass
28 open burning on the levels of polychlorinated dibenzo-p-dioxins and dibenzofurans in the
29 ambient air. *Journal of Hazardous Materials* **153**.
- 30 Shindell, et al. (2009) Improved attribution of climate forcing to emissions. *Science*. Available on the
31 Internet at <http://www.sciencemag.org/cgi/content/full/326/5953/716>.
- 32 Shindell D., Lamarque J.-F., Unger N., Koch D., Faluvegi G., Bauer S., Ammann M., Cofala J., and Teich H.
33 (2008a) Climate forcing and air quality change due to regional emissions reductions by economic
34 sector. *Atmospheric Chemistry and Physics* **8**.
- 35 Shindell D. and Faluvegi G. (2009) Climate response to regional radiative forcing during the twentieth
36 century. *Nature Geoscience* **2** (4), 294 (doi: 10.1038/ngeo473).
- 37 Shindell D. and Faluvegi G. (2010) The net climate impact of coal-fired power plant emissions. *Atmos.*
38 *Chem. Phys.* **10**, 3247-3260 (doi:10.5194/acp-10-3247-2010).
- 39 Shindell D.T., et al. (2008) A multi-model assessment of pollution transport to the Arctic. *Atmospheric*
40 *Chemistry and Physics* **8**, 5353-5372.
- 41 Shindell D.T., H. Levy II, Gilliland A., Schwarzkopf M.D., and Horowitz L.W. (2008b) Climate change from
42 short-lived emissions due to human activities in climate projections based on emissions
43 scenarios for long-lived and short-lived radiatively active gases and aerosols. In *A report by the*
44 *U.S. Climate Change Science Program and the Subcommittee on Global Change Research*, D.T.
45 Shindell, H. Levy II, A. Gilliland, M.D. Schwarzkopf, and L.W. Horowitz eds., Washington, D.C.
- 46 Shine K.P., Fuglestedt J.S., Hailemariam K., and Stuber N. (2005) Alternatives to the global warming
47 potential for comparing climate impacts of emissions of greenhouse gases. *Climatic Change*
48 **96**.

- 1 Shine K.P., Berntsen T.K., Fuglestedt J.S., Skeie R.B., and Stuber N. (2007) Comparing the climate effect
2 of emissions of short- and long-lived climate agents. *Phil. Trans. Royal. Soc., a-Mathematical*
3 *Physical and Engineering Sciences* **365** (1856).
- 4 Shine K.P. (2009) The global warming potential--the need for an interdisciplinary retrieval. An editorial
5 comment. *Climatic Change* **96**.
- 6 Shiraiwa M., Kondo Y., Iwamoto T., and Kita K. (2009) Amplification of light absorption of black carbon
7 by organic coating. *Aerosol Science and Technology* **44**.
- 8 Simon H., L. B., Bhave P.V., Divita F., Hsu Y., Luecken D., Mobley J.D., Pouliot G.A., Reff A., Sarwar G., and
9 Strum M. (2010) The development and uses of EPA's SPECIATE database. *Atmospheric Pollution*
10 *Research* **1**, 196-296.
- 11 Sinha B. (2002) The Indian stove programme: an insider's view – the role of society, politics, economics
12 and education. *Boiling Point* **48** (23-26).
- 13 Sinton, et al. (2004) An assessment of programs to promote improved household stoves in China. *Energy*
14 *for Sustainable Development* **VIII** (3).
- 15 Slater J.F., Currie L.A., Dibb J.E., and Benner B.A., Jr. (2002) Distinguishing the relative contribution of
16 fossil fuel and biomass combustion aerosols deposited at Summit, Greenland through isotopic
17 and molecular characterization of insoluble carbon. *Atmos. Environ.* **36**, 4463-4477.
- 18 Slowik J.G., E. C., Han J.-H., Davidovits P., Onasch T.B., Jayne J.T., Williams L.R., Canagaratna M.R.,
19 Worsnop D.R., Chakrabarty R.K., Moosmuller H., Arnott W.P., Schwarz J.P., Gao R.-S., Fahey
20 D.W., Kok G.L., and Petzold A. (2007) An inter-comparison of instruments measuring black
21 carbon content of soot particles. *Aerosol Sci. Technol.* **41** (3), 295-314.
- 22 Smallwood G.J., Snelling D.R., Gülder Ö.L., Clavel D., Gareau D., Sawchuck R.A., and Graham L. (2001)
23 Transient particulate matter measurements from the exhaust of a direct injection spark ignition
24 automobile. SAE paper 2001-01-3581.
- 25 Smith J.D., Kroll J.H., Cappa C.D., Che D.L., Liu C.L., Ahmed M., Leone S.R., Worsnop D.R., and Wilson K.R.
26 (2009a) The heterogeneous reaction of hydroxyl radicals with sub-micron squalane particles: a
27 model system for understanding the oxidative aging of ambient aerosols. *Atmos. Chem. Phys.* **9**.
- 28 Smith K.R., Samet J.M., Romieu I., and Bruce N. (2000a) Indoor air pollution in developing countries and
29 acute lower respiratory infections in children. *Thorax* **55** (6).
- 30 Smith K.R., Uma R., Zhang J., Rasmussen R.A., and Khalil M.A.K. (2000b) Household stoves in India,
31 greenhouse gases from small-scale combustion devices in developing countries, phase IIa.
32 EPA/600/R-00-052, June. Available on the Internet at
33 <http://www.epa.gov/nrmrl/pubs/600r00052/600R00052.pdf>.
- 34 Smith K.R. (2002) In praise of petroleum. *Science* **298**, 1847.
- 35 Smith K.R., Mehta S., and Maeusezahl-Feuz M. (2004) Indoor smoke from household solid fuels. In
36 *Comparative quantification of health risks: global and regional burden of disease due to selected*
37 *major risk factors*, M. Ezzati, A.D. Rodgers, A.D. Lopez, and C.L.J. Murray eds., World Health
38 Organization, Geneva, Switzerland
- 39 Smith K.R. and Haigler E. (2008) Co-benefits of climate mitigation and health protection in energy
40 systems: scoping methods. *Annual Review Public Health* **29**.
- 41 Smith K.R., Jerrett M., Anderson H.R., Burnett R.T., Stone V., Derwent R., Atkinson R.W., Cohen A.,
42 Shonkoff S.B., Krewski D., Pope C.A., Thun M.J., and Thurston G. (2009b) Public health benefits
43 of strategies to reduce greenhouse-gas emissions: health implications of short-lived greenhouse
44 pollutants. *The Lancet* **374**.
- 45 Smith S.J. and Wigley T.M.L. (2006) Multi-gas forcing stabilization with the MiniCAM. *The Energy Journal*
46 (Special Issue Number 3: Multigas Mitigation and Climate Policy), 373-391.

- 1 Snyder D.C. and Schauer J.J. (2007) An inter-comparison of two black carbon aerosol instruments and a
 2 semi-continuous elemental carbon instrument in the urban environment. *Aerosol Science and*
 3 *Technology* **41**, 463-474.
- 4 Solomon S., Plattner G.K., Knutti R., and Friedlingstein P. (2009) Irreversible climate change due to
 5 carbon dioxide emissions. *Proc Natl Acad Sci USA* **106** (6).
- 6 Springmann M., Knopf D.A., and Riemer N. (2009) Detailed heterogeneous chemistry in an urban plume
 7 box model: reversible co-adsorption of O₃, NO₂, and H₂O on soot coated with benzo[a]pyrene.
 8 *Atmos. Chem. Phys.* **9**.
- 9 Sram R.J., Binkova B., Dejmek J., and Bobak M. (2005) Ambient air pollution and pregnancy outcomes: a
 10 review of the literature. *Environ Health Perspect* **113**.
- 11 Stanhill G. and Cohen S. (2001) Global dimming: a review of the evidence for a widespread and
 12 significant reduction in global radiation with discussion of its probable causes and possible
 13 agricultural consequences. *Agric. Forest Meteorol.* **107**.
- 14 State and Territorial Air Pollution Program Administrators/Association of Local Air Pollution Control
 15 Officers (2006) Controlling fine particulate matter under the Clean Air Act: a menu of options.
 16 Report by the State and Territorial Air Pollution Program Administrators/Association of Local Air
 17 Pollution Control Officers (now National Association of Clean Air Agencies, NACAA), March.
- 18 Stier P., Seinfeld J.H., Kinne S., Feichter J., and Boucher O. (2006) Impact of nonabsorbing anthropogenic
 19 aerosols on clear-sky atmospheric absorption. *J. Geophys. Res.* **111**, D18201
 20 (doi:10.1029/2006JD007147).
- 21 Stoffyn-Egli P., et al. (1997) The identification of black carbon particles with the analytical scanning
 22 electron microscope: methods and initial results. *Science of the Total Environment* **198**.
- 23 Stohl A., Andrews E., Burkhart J.F., Forster C., Herber A., Hoch S.W., Kowal D., Lunder C., Mefford T.,
 24 Ogren J.A., Sharma S., Spichtinger N., Stebel K., Stone R., Strom J., Torseth K., Wehrl C., and Yttri
 25 K.E. (2006) Pan-arctic enhancements of light absorbing aerosol concentrations due to North
 26 American boreal forest fires during summer 2004. *Journal Geophysical Research* **111**, D22214
 27 (doi:10.1029/2006JD007216).
- 28 Stohl A., Berg T., Burkhart J.F., Fjærraa A.M., Forster C., Herber A., Hov Ø., Lunder C., McMillan W.W.,
 29 Oltmans S., Shiobara M., Simpson D., Solberg S., Stebel K., Ström J., Tørseth K., Treffeisen R.,
 30 Virkkunen K., and Yttri K.E. (2007) Arctic smoke – record high air pollution levels in the European
 31 Arctic due to agricultural fires in Eastern Europe in spring 2006. *Atmos. Chem. Phys.* **7**, 511-534,
 32 Copernicus Publications (doi:10.5194/acp-7-511-2007). Available on the Internet at
 33 <http://www.atmos-chem-phys.net/7/511/2007/> and at <http://www.atmos-chem-phys.net/7/511/2007/acp-7-511-2007.pdf>
- 34 Strack, et al. (2007) Arctic tundra shrub invasion and soot deposition: Consequences for spring snowmelt
 35 and near-surface air temperatures. *Journal of Geophysical Research* **112**, G04S44.
- 36 Strawa A.W., Kirchstetter T.W., Hallar A.G., Ban-Weiss G.A., McLaughlin J.P., Harley R.A., and Lunden
 37 M.M. (2010) Optical and physical properties of primary on-road vehicle particle emissions and
 38 their implications for climate change. *Journal of Aerosol Science* **41**, 36-50.
- 39 Streets D., Bond T., Lee T., and Jang C. (2004) On the future of carbonaceous aerosol emissions. *J.*
 40 *Geophys. Res.* **109** (doi: 10.1029/2004JD004902).
- 41 Streets D.G., Gupta S., Waldhoff S.T., Wang M.Q., Bond T.C., and Yiyun B. (2001) Black carbon emissions
 42 in China. *Atmospheric Environment* **35**, 4281-4296.
- 43 Streets D.G., Bond T.C., Carmichael G.R., Fernandes S.D., Fu Q., He D., Klimont K., Nelson S.M., Tsai N.Y.,
 44 and Wang M.Q. (2003a) An inventory of gaseous and primary aerosol emissions in Asia in the
 45 year 2000. *J. Geo.Res.* **108** (D21), Art. No. 8809.
- 46 Streets D.G., Yarber K.F., Woo J.H., and Carmichael G.R. (2003b) Biomass burning in Asia: Annual and
 47 seasonal estimates and atmospheric emissions. *Global Biogeochemical Cycles* **17** (4), 1099.
- 48

- 1 Streets D.G., Wu Y., and Chin M. (2006) Two-decadal aerosol trends as a likely explanation of the global
 2 dimming/brightening transition. *Geophysical Research Letters* **33**, L15806
 3 (doi:10.1029/2006GL026471).
- 4 Strydom C., Robinson C., Pretorius E., Whitcutt J.M., Marx J., and Bornman M.S. (2006) The effect of
 5 selected metals on the central metabolic pathways in biology: a review. *WaterSA* **32**, 543-554.
- 6 Suglia S.F., Gryparis A., Schwartz J., and Wright R.J. (2008) Association between traffic-related black
 7 carbon exposure and lung function among urban women. *Environ Health Perspect*, **116**.
- 8 Sümer H. (2004) The association of biomass fuel combustion on pulmonary function tests in the adult
 9 population of Mid-Anatolia. *SPM-Sozial- und Präventivmedizin-International Journal of Public*
 10 *Health*.
- 11 Takahama S., Liu S., and Russell L.M. (2010) Coatings and clusters of carboxylic acids in carbon-
 12 containing atmospheric particles from spectromicroscopy and their implications for cloud-
 13 nucleating and optical properties *Geophys. Res.* **115** (D1), D01202 (doi:10.1029/2009JD012622).
- 14 Tankersley C.G., Campen M., Bierman A., Flanders S.E., Broman K.W., and Rabold R. (2004) Particle
 15 effects on heart-rate regulation in senescent mice. *Inhal Toxicol* **16**, 381-390.
- 16 Tankersley C.G., Bierman A., and Rabold R. (2007) Variation in heart rate regulation and the effects of
 17 particle exposure in inbred mice. *Inhal Toxicol*, **19**, 621-629.
- 18 Tankersley C.G., Champion H.C., Takimoto E., Gabrielson K., Bedja D., Misra V., El-Haddad H., Rabold R.,
 19 and Mitzner W. (2008) Exposure to inhaled particulate matter impairs cardiac function in
 20 senescent mice. *Am J Physiol Regul Integr Comp Physiol* **295**, R252-R263.
- 21 Task Force on Hemispheric Transport of Air Pollution (2010a) *Hemispheric transport of air pollution*
 22 *2010*, F. Dentener, T. Keating, and H. Akimoto, eds., United Nations Economic Commission for
 23 Europe, Geneva, Switzerland (ECE/EB.AIR/100).
- 24 Task Force on Hemispheric Transport of Air Pollution (2010b) Part A: ozone and particulate matter. In
 25 *Hemispheric transport of air pollution 2010*, F. Dentener, T. Keating, and H. Akimoto eds., United
 26 Nations Economic Commission for Europe, Geneva, Switzerland (ECE/EB.AIR/100).
- 27 Texas Commission on Environmental Quality (2010) Texas Emissions Reduction Plan (TERP). Available on
 28 the Internet at <http://www.tceq.state.tx.us/implementation/air/terp>. November 30, 2010.
- 29 The American Trucking Associations, Inc., (2007) The impact of rising gas prices on America's small
 30 businesses. Statement before the Senate Committee on Small Business and Entrepreneurship,
 31 June 14, 2007. Available on the Internet at
 32 <http://www.truckline.com/Newsroom/Testimony1/Testimony%20of%20Tim%20Lynch%20before%20the%20Senate%20Committee%20on%20Small%20Business%20on%20Impact%20of%20Rising%20Gas%20Prices.pdf>.
- 33
 34
- 35 Thevenon F., Anselmetti F.S., Bernasconi S.M., and Schwikowski M. (2009) Mineral dust and elemental
 36 black carbon records from an Alpine ice core (Colle Gnifetti glacier) over the last millennium. *J.*
 37 *Geophys. Res.* **114**, D17102 (doi:10.1029/2008JD011490).
- 38 Tol R.S., Berntsen T., O'Neill B.C., Fuglestvedt J.S., Shine K.P., Balkanski Y., and Makra L. (2008) Metrics
 39 for aggregating the climate effect of different emissions: a unifying framework. Paper by ESRI
 40 (formerly Economic and Social Research Institute), ESRI Working Paper No. 257.
- 41 Torres O., Tanskanen A., Veiheilmann B., Ahn C., Braak R., Bhartia P.K., Veeffkind P., and Levelt P. (2007)
 42 Aerosols and surface UV products from Ozone Monitoring Instrument observations: An
 43 overview. *J. Geophys. Res.* **112**, D24S47 (doi:10.1029/2007JD008809).
- 44 Trenberth K.E., et al. (2007) Observations: surface and atmospheric climate change. In *Climate change*
 45 *2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment*
 46 *Report of the Intergovernmental Panel on Climate Change*, S. Solomon, D. Qin, M. Manning, Z.
 47 Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller eds., Cambridge University Press,
 48 Cambridge, United Kingdom, and New York, NY, USA

- 1 Truce R. and Wilkison L. (2008) Enhanced Fine particle and mercury emission control using the Indigo
2 Agglomerator. In the proceedings from the *11th International Conference on Electrostatic*
3 *Precipitation, Hangzhou, China*, K. Yan, ed., Zhejiang University Press.
- 4 Tuinstra F. and Koenig J.L. (1970) Raman spectrum of graphite. *The Journal of Chemical Physics* **53**.
- 5 Turco R.P., Toon O.B., Whitten R.C., Pollack J.B., and Hamill P. (1983) The global cycle of particulate
6 elemental carbon. In *Precipitation scavenging, dry deposition, and re-suspension*, H.R.
7 Pruppacher, R.G. Semonin, and W.G.N. Slinn eds., Elsevier, New York, N.Y., 1337-1351
- 8 Turner J.H., Lawless P.A., Yamamoto T., Coy D.W., McKenna J.D., Mycock J.C., Nunn A.B., Greiner G.P.,
9 and Vataavuk W.M. (2002a) Section 6. Particulate matter controls. Chapter 3: electrostatic
10 precipitators. In *EPA air pollution control cost manual (sixth edition)*, U.S. Environmental
11 Protection Agency, (EPA/452/B-02-001).
- 12 Turner J.H., McKenna J.D., Mycock J.C., Nunn A.B., and Vataavuk W.M. (2002b) Section 6. Particulate
13 matter controls. Chapter 1: baghouses and filters. In *EPA air pollution control cost manual (sixth*
14 *edition)*, U.S. Environmental Protection Agency, (EPA/452/B-02-001).
- 15 Turpin B.J. and Lim H.-J. (2010) Species contributions to PM_{2.5} mass concentrations: revisiting common
16 assumptions for estimating organic mass. *Aerosol Science and Technology* **35** (1), 602-610 (DOI:
17 10.1080/02786820119445).
- 18 Twomey (1977) The influence of pollution on the shortwave albedo of clouds. *J. of Atm. Sci.* **34**.
- 19 Twomey S. (1974) Pollution and planetary albedo. *Atmospheric Environment* (8).
- 20 U.S. Agency for International Development (2004) Vehicle inspection and maintenance programs:
21 international experience and best practices.
- 22 U.S. Agency for International Development (2007) Fuel-efficient stove programs in IDP settings -
23 summary evaluation report, Uganda. Available on the Internet at
24 [http://www.usaid.gov/our_work/humanitarian_assistance/disaster_assistance/sectors/files/uga](http://www.usaid.gov/our_work/humanitarian_assistance/disaster_assistance/sectors/files/uganda_final_summary.pdf)
25 [nda_final_summary.pdf](http://www.usaid.gov/our_work/humanitarian_assistance/disaster_assistance/sectors/files/uganda_final_summary.pdf).
- 26 U.S. Agency for International Development (2010a) Evaluation of manufactured wood-burning stoves in
27 Dadaab refugee camps, Kenya.
- 28 U.S. Agency for International Development (2010b) Black carbon emissions in Asia: sources, impacts,
29 and abatement options. April.
- 30 U.S. Census Bureau (2004) 2002 Economic census: vehicle inventory and use survey. Census Bureau
31 report EC02TV-US.
- 32 U.S. Climate Change Science Program and Subcommittee on Global Change Research (2009) CCSP 2009:
33 atmospheric aerosol properties and climate impacts, a report by the U.S. Climate Change
34 Science Program and the Subcommittee on Global Change Research. 128, M. Chin, R.A. Kahn,
35 and S.E. Schwartz, eds.
- 36 U.S. Department of Agriculture (2010) USDA Arctic Black Carbon Initiative. Fact sheet by the US
37 Department of Agriculture (USDA), Washington, D.C., September.
- 38 U.S. Energy Information Administration (1999) Energy efficiency - transportation sector. Report by the
39 U.S. Department of Energy, Energy Information Administration, October 17. Available on the
40 Internet at http://www.eia.doe.gov/emeu/efficiency/ee_ch5.htm and at
41 http://www.eia.gov/emeu/efficiency/ee_ch5.htm.
- 42 U.S. Energy Information Administration (2010) Electric Power Annual (with data for 2008). January 21.
- 43 U.S. Environmental Protection Agency Study of particulate matter emissions. Report.
- 44 U.S. Environmental Protection Agency (1992) Prescribed burning background document and technical
45 information document for best available control measures. Report.
- 46 U.S. Environmental Protection Agency (1998) Interim air quality policy on wildland and prescribed fires.
47 Report.

- 1 U.S. Environmental Protection Agency (1999) Draft regulatory impact analysis. Report, EPA 420D-99-
2 001, April.
- 3 U.S. Environmental Protection Agency (2001a) On-highway heavy-duty engine and vehicle standards and
4 highway diesel fuel sulfur control requirements. *Federal Register* **66** (12), 5002-5193.
- 5 U.S. Environmental Protection Agency (2001b) Control of air pollution from new motor vehicles: heavy-
6 duty engine and vehicle standards and highway diesel fuel sulfur control requirements, final
7 rule. Regulatory impact analysis: heavy-duty engine and vehicle standards and highway diesel
8 fuel sulfur control requirements; Chapter VI, Table VI F-4. *Federal Register* **66** (12), VI-19
9 (EPA420-R-00-026). Available on the Internet at [http://www.epa.gov/otaq/highway-](http://www.epa.gov/otaq/highway-diesel/regs/ria-vi.pdf)
10 [diesel/regs/ria-vi.pdf](http://www.epa.gov/otaq/highway-diesel/regs/ria-vi.pdf).
- 11 U.S. Environmental Protection Agency (2001c) Final regulatory impact analysis, Chapter 6: control of air
12 pollution from new motor vehicles: heavy-duty engine and vehicle standards and highway diesel
13 fuel sulfur control requirements; final rule. Report.
- 14 U.S. Environmental Protection Agency (2002a) Health assessment document for diesel engine exhaust,
15 Chapter 2. Report by the National Center for Environmental Assessment Office of Research and
16 Development, U.S. Environmental Protection Agency, Washington, D.C., EPA/600/8-90/057F.
17 Available on the Internet at
18 http://oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=36319.
- 19 U.S. Environmental Protection Agency (2002b) Health assessment document for diesel engine exhaust.
20 Report by the National Center for Environmental Assessment Office of Research and
21 Development, U.S. Environmental Protection Agency, Washington, D.C., EPA/600/8-90/057F.
22 Available on the Internet at
23 http://oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=36319.
- 24 U.S. Environmental Protection Agency (2003a) EPA MOBILE6.2 model. Available on the Internet at
25 <http://www.epa.gov/otaq/m6.htm>.
- 26 U.S. Environmental Protection Agency (2003b) National air quality and emissions trends report, 2003
27 special studies edition. September. Available on the Internet at
28 <http://www.epa.gov/airtrends/aqtrnd03/>.
- 29 U.S. Environmental Protection Agency (2004a) Final regulatory impact analysis: control of emissions
30 from nonroad diesel engines. Report, EPA420-%-04-007. Available on the Internet at
31 <http://www.epa.gov/ttn/ecas/ria.html>.
- 32 U.S. Environmental Protection Agency (2004b) Particle pollution report. Report.
- 33 U.S. Environmental Protection Agency (2004c) EPA's designations for PM_{2.5} nonattainment areas in New
34 England questions and answers. Available on the Internet at
35 http://www.epa.gov/ne/airquality/pdfs/pm25_qa.pdf.
- 36 U.S. Environmental Protection Agency (2005) AirControlNET version 4.1 documentation report. Report
37 Prepared by E.H. Pechan & Associates, Inc.
- 38 U.S. Environmental Protection Agency (2005) National Emissions Inventory data and documentation,
39 CHIEF website. Available on the Internet at
40 <http://www.epa.gov/ttn/chief/net/2005inventory.html>.
- 41 U.S. Environmental Protection Agency (2006a) 2006 regulatory impact analysis for the National Ambient
42 Air Quality Standards for Particle Pollution. Report. Available on the Internet at
43 <http://www.epa.gov/ttn/ecas/ria.html>.
- 44 U.S. Environmental Protection Agency (2006b) Diesel retrofit technology: an analysis of the cost-
45 effectiveness of reducing particulate matter emissions from heavy-duty diesel engines through
46 retrofits. Report, EPA420-S-06-002.
- 47 U.S. Environmental Protection Agency (2007) 2002 NEI booklet. Available on the Internet at
48 <http://www.epa.gov/ttn/chief/net/2002neibooklet.pdf>.

- 1 U.S. Environmental Protection Agency (2008a) Regulatory impact analysis: Control of emissions of air
2 pollution from locomotive engines and marine compression-ignition engines less than 30 liters
3 per cylinder; republication. *Federal Register* **73** (126) (EPA420-R-08-001a). Available on the
4 Internet at <http://www.epa.gov/oms/regs/nonroad/420r08001a.pdf>.
- 5 U.S. Environmental Protection Agency (2008b) Kansas City PM characterization study - final report
6 (based on EPA contract report ERG No. 0133.18.007.001). Final Report.
- 7 U.S. Environmental Protection Agency (2008c) NONROAD model (nonroad engines, equipment, and
8 vehicles). Available on the Internet at <http://www.epa.gov/otaq/nonrdmdl.htm>.
- 9 U.S. Environmental Protection Agency (2008d) EPA National Mobile Inventory Model (NMIM). Available
10 on the Internet at <http://www.epa.gov/otaq/nmim.htm>.
- 11 U.S. Environmental Protection Agency (2008e) Technology transfer network clearinghouse for
12 inventories and emission factors, SPECIATE Version 4.2. Available on the Internet at
13 http://cfpub.epa.gov/si/speciate/ehpa_speciate_browse_details.cfm?ptype=P&number=4737.
- 14 U.S. Environmental Protection Agency (2008f) Final regulatory impact analysis, Chapter 5: control of
15 emissions of air pollution from locomotive engines and marine compression ignition engines less
16 than 30 liters per cylinder; final rule.
- 17 U.S. Environmental Protection Agency (2009a) Integrated science assessment for particulate matter.
18 Report by the U.S. Environmental Protection Agency. Washington, D.C., EPA/600/R-08/139.
19 Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recorddisplay.cfm?deid=216546>.
- 20 U.S. Environmental Protection Agency (2009b) Report to Congress: highlights of the Diesel Emissions
21 Reduction Program. Report, EPA-420-R-09-006, August.
- 22 U.S. Environmental Protection Agency (2009c) *Strategies for reducing residential wood smoke*. Available
23 on the Internet at <http://www.epa.gov/ttn/oarpg/t1/memoranda/strategies-doc-8-11-09.pdf>.
- 24 U.S. Environmental Protection Agency (2010a) Regulatory impact analysis: control of emissions of air
25 pollution from category 3 marine diesel engines. Report.
- 26 U.S. Environmental Protection Agency (2010b) Regulatory impact analysis: NESHAP for industrial,
27 commercial, and institutional boilers and process heaters. Draft Report, April.
- 28 U.S. Environmental Protection Agency (2010c) MOVES2010 model. Available on the Internet at
29 <http://www.epa.gov/otaq/models/moves>.
- 30 U.S. Environmental Protection Agency (2010d) "Emerging Technology List" National Clean Diesel
31 Campaign.
- 32 U.S. Environmental Protection Agency (2010e) EPA and NHTSA to propose greenhouse gas and fuel
33 efficiency standards for heavy-duty trucks; begin process for further light-duty standards. EPA-
34 420-F-10-038.
- 35 U.S. Environmental Protection Agency (2010f) Regulatory impact analysis: NESHAP for industrial,
36 commercial, and institutional boilers and process heaters. Draft Report.
- 37 U.S. Environmental Protection Agency (2010g) Renewable Fuel Standard Program (RFS2), regulatory
38 impact analysis. Available on the Internet at
39 <http://www.epa.gov/otaq/renewablefuels/420r10006.pdf>.
- 40 U.S. Environmental Protection Agency (2010h) Methods for measurement of filterable PM₁₀ and PM_{2.5}
41 and measurement of condensable PM emissions from stationary sources. *Federal Register* **75**
42 (244). Available on the Internet at [http://origin.www.gpo.gov/fdsys/pkg/FR-2010-12-
43 21/pdf/2010-30847.pdf](http://origin.www.gpo.gov/fdsys/pkg/FR-2010-12-21/pdf/2010-30847.pdf)
- 44 U.S. Environmental Protection Agency (2011 (forthcoming)) Regulatory impact analysis for proposed
45 particulate matter National Ambient Air Quality standard. Available on the Internet at
46 www.epa.gov/ttn/ecas.

- 1 U.S. Forest Service AirFire Team and Sonoma Technology I. (2010) Blue Sky Framework: website for a
 2 modular modeling system that enables fire information, consumption, and smoke modeling.
 3 April 1. Available on the Internet at <http://blueskyframework.org/>.
- 4 Unger N., Shindell D.T., and Wang J.S. (2009) Climate forcing by the on-road transportation and power
 5 generation sectors. *Atmospheric Environment* **43**, 3077-3085.
- 6 Unger N., Bond T.C., Wang J.S., Koch D.M., Menon S., Shindell D.T., and Bauer S. (2010) Attribution of
 7 climate forcing to economic sectors. *Proceedings of the National Academy of Sciences* **107** (8).
- 8 United Nations Development Programme Bangladesh (2010) Technical and financial fact sheet. Available
 9 on the Internet at
 10 <http://www.undp.org.bd/projects/prodocs/IKEBMI/Technical%20Fact%20Sheet.pdf>.
- 11 United Nations Development Programme Global Environment Facility (2007) Improving kiln efficiency
 12 for the brick making industry in Bangladesh - PDF B Phase. BDG/04/014.
- 13 United Nations Environmental Programme (2008) Atmospheric brown clouds: regional assessment
 14 report with focus on Asia. Report by the United Nations Environment Programme, Nairobi,
 15 Kenya. Available on the Internet at <http://www.unep.org/pdf/ABCsummaryFinal.pdf>.
- 16 United Nations Environmental Programme (2011) The integrated assessment of black carbon and
 17 tropospheric ozone. In progress.
- 18 Uno I., Eguchi K., Yumimoto K., Takemura T., Shimizu A., Uematsu M., Liu Z., Y., Wang Z., F., Hara Y., and
 19 Sugimoto N. (2009) Asian dust transported one full circuit around the globe. *Nature Geoscience*
 20 **2** (8), 557-560.
- 21 Val Martin M., Logan J.A., Kahn R.A., Leung F.-Y., Nelson D.L., and Diner D.J. (2010) Smoke injection
 22 heights from fires in North America: analysis of 5 years of satellite observations. *Atmos. Chem.*
 23 *Phys. Discuss.* **10**.
- 24 Van der Werf G.R., Randerson J.T., Giglio L., Collatz G.J., Kasibhatla P.S., and Arellano A.F., Jr. (2006)
 25 Interannual variability of global biomass burning emissions from 1997 to 2004. *Atmos. Chem.*
 26 *Phys. Discuss.* **6**.
- 27 van Poppel L.H., Friedrich H., Spinsby J., Chung S.H., Seinfeld J.H., and Buseck P.R. (2005) Electron
 28 tomography of nanoparticle clusters: Implications for atmospheric lifetimes and radiative forcing
 29 of soot. *Geophys. Res. Lett.* **32**, L24811, AGU (24). Available on the Internet at
 30 <http://dx.doi.org/10.1029/2005GL024461>
- 31 van Setten B.A.A.L., Makkee M., and Moulijn J.A. (2001) Science and technology of catalytic diesel
 32 particulate filters. *Catalysis Reviews* **43** (4).
- 33 van Vuuren D., den Elzen M., Lucas P., Eickhout B., Strengers B., van Ruijven B., Wonink S., and van
 34 Houdt R. (2007) Stabilizing greenhouse gas concentrations at low levels: an assessment of
 35 reduction strategies and costs. *Climatic Change* **81**, 119-159, Springer Netherlands (2). Available
 36 on the Internet at <http://dx.doi.org/10.1007/s10584-006-9172-9>
- 37 Vander Wal R.L., et al. (2004) Carbon nanostructure examined by lattice fringe analysis of high-
 38 resolution transmission electron microscopy images. *Applied Spectroscopy* **58**.
- 39 Venkatachari P., Zhou L., Hopke P.K., Schwab J.J., Demerjian K.L., Weimer S., Hogrefe O., Felton D., and
 40 Rattigan O. (2006) An intercomparison of measurement methods for carbonaceous aerosol in
 41 the ambient air in New York City. *Aerosol Science and Technology* **40**, 788-795.
- 42 Venkataraman C., Habib G., Eiguren-Fernandez A., Miguel A.H., and Friedlander S.K. (2005) Residential
 43 biofuels in south Asia: carbonaceous aerosol emissions and climate impacts. *Science* **307** 5714.
- 44 Venkataraman C., et al. (2010) The Indian National Initiative for Advanced Biomass Cookstoves: the
 45 benefits of clean combustion. *Energy for Sustainable Development* **14**, 63-72.
- 46 Vignati, et al. (2010) Sources of uncertainties in modeling black carbon at the global scale. *Atmos. Chem.*
 47 *Phys. Discuss.* **10**.

- 1 Virkkula A., Mäkelä T., Hillamo R., Yli-Tuomi T., Hirsikko A., Hämeri K., and Koponen I.K. (2007) A simple
2 procedure for correcting loading effects of Aethalometer data. *J Air Waste Manag Assoc* **57**
3 (10), 1214-1222.
- 4 Walgraeve C., Demeestere K., Dewulf J., Zimmermann R., and Van Langenhove H. (2010) Oxygenated
5 polycyclic aromatic hydrocarbons in atmospheric particulate matter: Molecular characterization
6 and occurrence. *Atmospheric Environment* **44**.
- 7 Wang C. (2004) A modeling study on the climate impacts of black carbon aerosols. *J. Geophys. Res.* **109**,
8 D03106, AGU (D3). Available on the Internet at <http://dx.doi.org/10.1029/2003JD004084>
- 9 Wang C. (2007) Impact of direct radiative forcing of black carbon aerosols on tropical convective
10 precipitation. *Geophys. Res. Lett.* **34**, L05709, AGU (5). Available on the Internet at
11 <http://dx.doi.org/10.1029/2006GL028416>
- 12 Wang F., Roldin P., Massling A., Kristensson A., Swietlicki E., Fang D., and Ketzel M. (2010) Aerosol
13 dynamics in the Copenhagen urban plume during regional transport. *Atmos. Chem. Phys.*
14 *Discuss.* **10**, 8553-8594 (doi:10.5194/acpd-10-8553-2010).
- 15 Warnatz J., Maas U., and Dibble R.W. (2006) *Physical and chemical fundamentals, modeling and*
16 *simulation, experiments, pollutant formation, 4th ed.*, Springer-Verlag, Berlin, Germany, 378.
- 17 Warneke C., Bahreini R., Brioude J., Brock C.A., de Gouw J.A., Fahey D.W., Froyd K.D., Holloway J.S.,
18 Middlebrook A., Miller L., Montzka S., Murphy D.M., Peischl J., Ryerson T.B., Schwarz J.P.,
19 Spackman J.R., and Veres P. (2009) Biomass burning in Siberia and Kazakhstan as an important
20 source for haze over the Alaskan Arctic in April 2008. *Geophys. Res. Lett.* **36**, L02813, AGU (2).
21 Available on the Internet at <http://dx.doi.org/10.1029/2008GL036194>
- 22 Warneke C., Froyd K.D., Brioude J., Bahreini R., Brock C.A., Cozic J., de Gouw J.A., Fahey D.W., Ferrare R.,
23 Holloway J.S., Middlebrook A.M., Miller L., Montzka S., Schwarz J.P., Sodemann H., Spackman
24 J.R., and Stohl A. (2010) An important contribution to springtime Arctic aerosol from biomass
25 burning in Russia. *Geophys. Res. Lett.* **37**, L01801, AGU (1). Available on the Internet at
26 <http://dx.doi.org/10.1029/2009GL041816>
- 27 Warren and Clarke (1990) Soot in the atmosphere and snow surface of Antarctica. *Journal Geophysical*
28 *Research* **95**.
- 29 Warren S.G. and Wiscombe W.J. (1980) A model for the spectral albedo of snow, II: snow containing
30 atmospheric aerosols. *J. Atmos. Sci.* **37**, 2734-2745.
- 31 Watmough S.A., Hutchinson T.C., and Dillon P.J. (2004) Lead dynamics in the forest floor and mineral soil
32 in south-central Ontario. *Biogeochemistry* **71**, 43-68.
- 33 Weingartner E., Burtscher H., and Baltensperger H. (1997) Hygroscopic properties of carbon and diesel
34 soot particles. *Atmospheric Environment* **31**.
- 35 Weingartner E., Saatho H., Schnaiter M., Streit N., Bitnar B., and Baltensperger U. (2003) Absorption of
36 light by soot particles: determination of the absorption coefficient by means of Aethalometers.
37 *Aerosol Science and Technology* **34** 1445-1463.
- 38 Wellenius G.A., Yeh G.Y., Coull B.A., Suh H.H., Phillips R.S., and Mittleman M.A. (2007) Effects of ambient
39 air pollution on functional status in patients with chronic congestive heart failure: a repeated-
40 measures study. *Environ Health* **6**.
- 41 Western Air Regional Partnership (2002) Non-burning management alternatives on agricultural lands in
42 the western United States, Volume II: non-burning management alternatives and
43 implementation plan strategies. Prepared by Eastern Research Group, Inc.
- 44 Wiedinmyer C. and Hurteau M.D. (2010) Prescribed fire as a means of reducing forest carbon emissions
45 in the western United States. *Environ. Sci. Technol.* **44**, 1926-1932.
- 46 Wientjes I.G.M. and Oerlemans J. (2010) An explanation for the dark region in the western melt zone of
47 the Greenland ice sheet. *The Cryosphere* **4**, 261-268 (doi:10.5194/tc-4-261-2010).

- 1 Wild M., Gilgen H., Roesch A., Ohmura A., Long C., Dutton E., Forgan B., Kallis A., Russak V., and Tsvetkov
 2 A. (2005) From dimming to brightening: decadal changes in solar radiation at the Earth's surface.
 3 *Science* **308**.
- 4 Wild M. (2009) Global dimming and brightening: a review. *J. Geophys. Res.* **114**, D00D16, AGU. Available
 5 on the Internet at <http://dx.doi.org/10.1029/2008JD011470>
- 6 Wilker E.H., Baccarelli A., Suh H., Vokonas P., Wright R.O., et al. (2010) Black carbon exposures, blood
 7 pressure, and interactions with single nucleotide polymorphisms in microRNA processing genes.
 8 *Environ Health Perspect.* **118** (7) (doi:10.1289/ehp.0901440).
- 9 Wilkinson P., Smith K.R., Davies M., Adair H., Armstrong B.G., Barrett M., et al. (2009) Public health
 10 benefits of strategies to reduce greenhouse-gas emissions: household energy. *The Lancet* **374**.
- 11 Winebrake J.J., Corbett J.J., Falzarano A., Hawker J.S., Korfmacher K., Ketha S., and Zilora S. (2008)
 12 Assessing energy, environmental, and economic tradeoffs in intermodal freight transportation. *J*
 13 *Air & Waste Manage Assoc* **58**, 1004-1013.
- 14 Winker D.M., Hunt W.H., and McGill M.J. (2007) Initial performance assessment of CALIOP. *Geophys.*
 15 *Res. Lett.* **34**, L19803, AGU (19). Available on the Internet at
 16 <http://dx.doi.org/10.1029/2007GL030135>
- 17 Wiscombe W. and Warren S. (1980) A model for the spectral albedo of snow II. Snow containing
 18 atmospheric aerosols. *J. Atmos. Sci.* **37**, 2734-2274.
- 19 Wise M.A., Calvin K.V., Thomson A.M., Clarke L.E., Bond-Lamberty B., Sands R.D., Smith S.J., Janetos A.C.,
 20 and Edmonds J.A. (2009) Implications of limiting CO₂ concentrations for land use and energy.
 21 *Science* **324**, 1183-1186.
- 22 Wolff G.T., Groblicki P.J., Cadle S.H., and Countiss (1982) Particulate carbon at various locations in the
 23 United States. In *Particulate Carbon: Atmospheric Life Cycle*, G. T. Wolff and R.L. Klimisch eds.,
 24 Plenum Press, New York, NY, 79-88
- 25 Wonaschutz A., et al. (2009) Application of the integrating sphere method to separate the contributions
 26 of brown and black carbon in atmospheric aerosol. *Environmental Science & Technology* **43**.
- 27 Wong J. and Li Z. (2002) Retrieval of optical depth for heavy smoke aerosol plumes: Uncertainties and
 28 sensitivities to the optical properties. *J. Atmos. Sci.* **59**.
- 29 Wooster M.J. and Zhang Y.H. (2004) Boreal forest fires burn less intensely in Russia than in North
 30 America. *Geophys. Res. Lett.* **31**, L20505.
- 31 World Health Organization (2006) *Fuel for life: household energy and health*, WHO Press, Geneva,
 32 Switzerland Available on the Internet at
 33 <http://www.who.int/indoorair/publications/fuelforlife/en/index.html>.
- 34 Wu J., Fu C., Xu Y., Tang J.P., Wang W., and Wang Z. (2008) Simulation of direct effects of black carbon
 35 aerosol on temperature and hydrological cycle in Asia by a Regional Climate Model.
 36 *Meteorology and Atmospheric Physics* **100**, 179-193, Springer Wien (1). Available on the
 37 Internet at <http://dx.doi.org/10.1007/s00703-008-0302-y>
- 38 Xu B.-Q., Wang M., Joswiak D.R., Cao J.-J., Yao T.-D., Wu G.-J., Yang W., and Zhao H.-B. (2009a)
 39 Deposition of anthropogenic aerosols in a southeastern Tibetan glacier. *J. Geophys. Res.* **114**,
 40 D17209, AGU (D17). Available on the Internet at <http://dx.doi.org/10.1029/2008JD011510>
- 41 Xu B., Yao T., Liu X., and Wang N. (2006) Elemental and organic carbon measurements with a two-step
 42 heating gas chromatography system in snow samples from the Tibetan Plateau. *Annals of*
 43 *Glaciology* **43** (1), 257-262. Available on the Internet at
 44 <http://www.ingentaconnect.com/content/igsoc/agl/2006/00000043/00000001/art00038> and at
 45 <http://dx.doi.org/10.3189/172756406781812122>
- 46 Xu B., Cao J., Hansen J., Yao T., Joswia D.R., Wang N., Wu G., Wang M., Zhao H., Yang W., Liu X., and He J.
 47 (2009b) Black soot and the survival of Tibetan glaciers. *Proc. Nat. Acad. Sci. U.S.A.* **106**, 22114-
 48 22118.

- 1 Yadav V.K., Prasad S., Patel D.K., Khan A.H., Tripathi M., and Shukla Y. (2010) Identification of polycyclic
2 aromatic hydrocarbons in unleaded petrol and diesel exhaust emission. *Environ Monit Assess*
3 **168**.
- 4 Yang L., Xiaoye Z., Sunling G., Huizheng C., Dan W., Wenjun Q., and Junying U. (2006) Comparison of EC
5 and BC and evaluation of dust aerosol contribution to light absorption in Xi'an, China.
6 *Environmental Monitoring and Assessment* **120**, 301-312, Springer Netherlands (1). Available on
7 the Internet at <http://dx.doi.org/10.1007/s10661-005-9062-z>
- 8 Yttri, et al. (2007) Elemental and organic carbon in PM₁₀: a one year measurement campaign within the
9 European Monitoring and Evaluation Programme EMEP. *Atmospheric Chemistry and Physics* **22**,
10 5711-5725.
- 11 Yu H., Liu S.C., and Dickinson R.E. (2002a) Radiative effects of aerosols on the evolution of the
12 atmospheric boundary layer. *J. Geophys. Res.* **107**, 4142, AGU (D12). Available on the Internet at
13 <http://dx.doi.org/10.1029/2001JD000754>
- 14 Yu J.Z., Xu J., and Yang H. (2002b) Charring characteristics of atmospheric organic particulate matter in
15 thermal analysis. *Environ. Sci. Technol.* **36** (4), 754-761.
- 16 Zanobetti A. and Schwartz J. (2006) Air pollution and emergency admissions in Boston, MA. *J Epidemiol*
17 *Community Health* **60**.
- 18 Zanobetti A., Stone P.H., Speizer F.E., Schwartz J.D., Coull B.A., Suh H.H., Nearing B.D., Mittleman M.A.,
19 Verrier R.L., and D.R. G. (2009) T-wave alternans, air pollution and traffic in high-risk subjects.
20 *Am J Cardiol* **104**, 665-670.
- 21 Zeka A., Sullivan J.R., Vokonas P.S., Sparrow D., and Schwartz J. (2006) Inflammatory markers and
22 particulate air pollution: characterizing the pathway to disease. *Int J Epidemiol* **35**.
- 23 Zhang G., Streets, and et. al. (2009) Asian emissions in 2006 for the NASA INTEX-B mission. *Atmos. Chem.*
24 *Phys. Discuss.*
- 25 Zhang Q., et al. (2007) Major components of China's anthropogenic primary particulate emissions.
26 *Environ. Res. Lett.* **2**, 045027.
- 27 Zhang R., Khalizov A.F., Pagels J., Zhang D., Xue H., and McMurry P.H. (2008a) Variability in morphology,
28 hygroscopicity, and optical properties of soot aerosols during atmospheric processing.
29 *Proceedings of the National Academy of Sciences* **105**, 10291-10296 (30). Available on the
30 Internet at <http://www.pnas.org/content/105/30/10291.abstract>
- 31 Zhang Y.X., Schauer J.J., Zhang Y.H., Zeng L.M., Wei Y.J., Liu Y., and Shao M. (2008b) Characteristics of
32 particulate carbon emissions from real-world Chinese coal combustion. *Environmental Science &*
33 *Technology* **42** (14).
- 34 Zhi G., Peng C., Chen Y., Liu D., Sheng G., and Fu J. (2009) Deployment of coal briquettes and improved
35 stoves: possibly an option for both environment and climate. *Environmental Science &*
36 *Technology* **43** (15), 5586-5591.
- 37 Zhu Q. (2003) *Developments in particulate control*, IEA Clean Coal Centre (ISBN 92-9029-388-8).
38

(This page intentionally left blank)

United States
Environmental Protection
Agency

Office of Air and Radiation
Office of Air Quality Planning and Standards
Research Triangle Park, NC

Publication No. EPA-450/D-11-001
March 2011
