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Highlights



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

- BC is emitted directly into the atmosphere in the form of fine particles ($PM_{2.5}$). The United States contributes about 8% of the global emissions of BC. Within the United States, BC is estimated to account for approximately 12% of all direct $PM_{2.5}$ emissions in 2005.

- BC contributes to the adverse impacts on human health, ecosystems, and visibility associated with $PM_{2.5}$.

- BC influences climate by: 1) directly absorbing light, 2) reducing the reflectivity ("albedo") of snow and ice through deposition, and 3) interacting with clouds.

- The direct and snow/ice albedo effects of BC are widely understood to lead to climate warming. However, the globally averaged net climate effect of BC also includes the effects associated with cloud interactions, which are not well quantified and may cause either warming or cooling. Therefore, though most estimates indicate that BC has a net warming influence, a net cooling effect cannot be ruled out.

- Sensitive regions such as the Arctic and the Himalayas are particularly vulnerable to the warming and melting effects of BC.

- BC is emitted with other particles and gases, many of which exert a cooling influence on climate. Therefore, estimates of the net effect of BC emissions sources on climate should include the offsetting effects of these co-emitted pollutants. This is particularly important for evaluating mitigation options.

- BC's short atmospheric lifetime (days to weeks), combined with its strong warming potential, means that targeted strategies to reduce BC emissions can be expected to provide climate benefits within the next several decades.

- The different climate attributes of BC and long-lived greenhouse gases make it difficult to interpret comparisons of their relative climate impacts based on common metrics.

- Based on recent emissions inventories, the majority of global BC emissions come from Asia, Latin America, and Africa. Emissions patterns and trends across regions, countries and sources vary significantly.

- Control technologies are available to reduce BC emissions from a number of source categories.

- BC mitigation strategies, which lead to reductions in $PM_{2.5}$, can provide substantial public health and environmental benefits.

- Considering the location and timing of emissions and accounting for co-emissions will improve the likelihood that mitigation strategies will be properly guided by the balance of climate and public health objectives.

- Achieving further BC reductions, both domestically and globally, will require adding a specific focus on reducing direct $PM_{2.5}$ emissions to overarching fine particle control programs.

- The most promising mitigation options identified in this report for reducing BC (and related "soot") emissions are consistent with control opportunities emphasized in other recent assessments.
    - **United States:** The United States will achieve substantial BC emissions reductions by 2030, largely due to controls on new mobile diesel engines. Other source categories in the United States, including stationary sources, residential wood combustion, and open biomass burning also offer potential opportunities.
    - **Global:** The most important BC emissions reduction opportunities globally include residential cookstoves in all regions; brick kilns and coke ovens in Asia; and mobile diesels in all regions.
    - **Sensitive Regions:** To address impacts in the Arctic, other assessments have identified the transportation sector; residential heating; and forest, grassland and agricultural burning as primary mitigation opportunities. In the Himalayas, studies have focused on residential cooking; industrial sources; and transportation, primarily on-road and off-road diesel engines.

- A variety of other options may also be suitable and cost-effective for reducing BC emissions, but these can only be identified with a tailored assessment that accounts for individual countries' resources and needs.

- Despite some remaining uncertainties about BC that require further research, currently available scientific and technical information provides a strong foundation for making mitigation decisions to achieve lasting benefits for public health, the environment, and climate.

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AAE	Absorption Angstrom Exponent
AAOD	Aerosol Absorption Optical Depth
ABC	Atmospheric Brown Cloud
ACIA	Arctic Climate Impact Assessment
ACS	American Cancer Society
AERONET	Aerosol Robotic (Measurement) Network
AIE	Aerosol Indirect Effect
AMAP	Arctic Monitoring and Assessment Program
AOD	Aerosol Optical Depth
AP-42	Compilation of Air Pollutant Emission Factors (EPA)
APU	Auxiliary Power Unit
AQS	Air Quality System (EPA)
AR5	Fifth Assessment Report of the Intergovernmental Panel on Climate Change
ARIES	Aerosol Research Inhalation Epidemiology Study
BB	Biomass Burning
BC	Black Carbon
BCa	Apparent Black Carbon
BF	Biofuel
BrC	Brown Carbon
BS	Black Smoke
BSG	Biofuel Soot and Gases
C1	Commercial Marine Engines less than 5 liters/cylinder
C2	Commercial Marine Engines between 5-30 liters/cylinder
C3	Commercial Marine Engines greater than 30 liters/cylinder
CAA	Clean Air Act
CALIPSO	Cloud-Aerosol Lidar and Infrared Pathfinder Satellite Observation
CAPMoN	Canadian Air and Precipitation Monitoring Network
CARB	California Air Resources Board
CAWNET	China Atmosphere Watch Network
CCN	Cloud Condensation Nuclei
CCVS	Closed Crankcase Ventilation System
CF_4	Carbon Tetrafluoride
CH_4	Methane
CI	Compression Ignition
CLRTAP	Convention on Long-Range Transboundary Air Pollution
CMAQ	Congestion Mitigation and Air Quality
CNG	Compressed Natural Gas
CO	Carbon Monoxide
CO_2	Carbon Dioxide
COH	Coefficient of Haze

CRF	Concentration-Response Function
CSN	Chemical Speciation Network
DALYs	Disability Adjusted Life Years
DERA	Diesel Emissions Reduction Act
DOC	Diesel Oxidation Catalyst
DPF	Diesel Particulate Filter
EC	Elemental Carbon
ECa	Apparent Elemental Carbon
ECA	Emissions Control Area
EDX	Energy Dispersive X-ray Spectroscopy
EF	Emission Factor
EI	Emission Index
EMEP	European Monitoring and Evaluation Programme
EPA	Environmental Protection Agency
ER	Emission Reduction Factor
ERT	Emission Reduction Technique
ESP	Electrostatic Precipitator
ESRL/GMD	Earth System Research Laboratory/Global Monitoring Division (NOAA)
EU	European Union
EURO	European Emissions Standards
EUSAAR	European Supersites for Atmospheric Aerosol Research
FAO	Food and Agriculture Organization
FF	Fossil Fuel
FHWA	Federal Highway Administration
FRM	Federal Reference Method
g/bhp-hr	Grams per Break Horsepower-Hour
GACC	Global Alliance for Clean Cookstoves
GAW Aerosol	Global Atmospheric Watch Aerosol Program
GCP	Global Cost Potential
GDI	Gasoline Direct Injection
GDP	Global Damage Potential
GSFC	Goddard Space Flight Center
Gg	Gigagram (10^9 g = 1 kilotonne)
GHG	Greenhouse Gas
GIZ	German Agency for International Development (Deutsche Gesellschaft für Internationale Zusammenarbeit)
GLAS	Geoscience Laser Altimeter System (NASA)
GTP	Global Temperature Potential
GWP	Global Warming Potential
H_2O	Water
HAP	Hazardous Air Pollutant
HC	Hydrocarbon
HFC	Hydrofluorocarbon
HHK	Hybrid Hoffman Kiln
HKHT	Hindu-Kush Himalayan Tibetan Region
HULIS	Humic-Like Substances
hPa	Hectopascal (a unit of barometric pressure)
IAM	Integrated Assessment Model
ICI	Industrial, Commercial and Institutional (Boilers)
IEA	International Energy Agency

IGAC	International Global Atmospheric Chemistry Project
IIASA	International Institute for Applied Systems Analysis
IMPROVE	Interagency Monitoring of Protected Visual Environments
IPCC	Intergovernmental Panel on Climate Change
IR	Infrared
ISA	Integrated Science Assessment
LAC	Light-Absorbing Carbon
LDGV	Light Duty Gasoline Vehicle
LEV	Low Emissions Vehicle
LII	Laser Induced Incandescence
LNG	Liquefied Natural Gas
LPG	Liquid Petroleum Gas
LPM	Liters per Minute
LTO	Landing and Take-Off
MAC	Mass Absorption Coefficients
MACT	Maximum Achievable Control Technology
MANE-VU	Mid-Atlantic/Northeast Visibility Union
MARAMA	Mid-Atlantic Regional Air Management Association
MARPOL	International Convention on the Prevention of Pollution from Ships
MI	Myocardial Infarction
MISR	Multi-angle Imaging Spectroradiometer
Mm	Millimeter
MODIS	Moderate Resolution Imaging Spectroradiometer
MOVES	Mobile Vehicle Emission Simulator Model (EPA)
N_2O	Nitrous Oxide
NAAQS	National Ambient Air Quality Standard
NAPS	National Air Pollution Surveillance Network
NATA	National Air Toxics Assessment
NCDC	National Clean Diesel Campaign
NCO-P	Nepal Climate Observatory Pyramid
NEI	National Emissions Inventory (EPA)
NESCAUM	Northeast States for Coordinated Air Use Management
NESHAP	National Emissions Standards for Hazardous Air Pollutants
NGO	Non-Governmental Organization
NIOSH	National Institute for Occupational Safety and Health
NMIM	National Mobile Inventory Model
NOAA	National Oceanic and Atmospheric Administration
NO_x	Nitrogen Oxides
NRC	National Research Council
NSPS	New Source Performance Standard
Nss-S	Non-Sea-Salt Sulfur
O_3	Ozone
OC	Organic Carbon
OCa	Apparent Organic Carbon
OC/BC	Organic Carbon to Black Carbon Ratio (also OC:BC)

OECD	Organisation for Economic Co-operation and Development
OM	Organic Matter
OMI	Ozone Monitoring Instrument
PAH	Polycyclic Aromatic Hydrocarbon
PBL	Planetary Boundary Layer
PCFV	Partnership for Clean Fuels and Vehicles
PCIA	Partnership for Clean Indoor Air
PDPF	Partial Diesel Particulate Filters
PIC	Product of Incomplete Combustion
PM	Particulate Matter (for related terms, such as $PM_{2.5}$ and PM_{10}; see Glossary)
POM	Polycyclic Organic Matter
RCP	Representative Concentration Pathway
RHR	Regional Haze Rule
RICE	Reciprocating Internal Combustion Engine
RM	Raman Microspectroscopy
RPO	Regional Planning Organization
RWC	Residential Wood Combustion
SAGE	System for Assessing Aviation's Global Emissions (FAA)
SCC	Social Cost of Carbon
SCR	Selective Catalytic Reduction
SEARCH	SouthEastern Aerosol Research and Characterization
SEM	Scanning Electron Microscopy
SFP	Specific Forcing Pulse
SLCF	Short-Lived Climate Forcer
SNV	Netherlands Development Organization
SO_2	Sulfur Dioxide
SOA	Secondary Organic Aerosols
SPARC	Stratospheric Processes and their Role in Climate (World Climate Research Programme)
SRES	Special Report on Emissions Scenarios (IPCC)
STN	Speciation Trends Network
STRE	Surface Temperature Response per unit continuous Emission
TEM	Transmission Electron Microscopy
TERP	Texas Emissions Reduction Plan
TF HTAP	Task Force on Hemispheric Transport of Air Pollution
Tg	Teragram (10^{12} g = 1 megatonne)
TOA	Top of the Atmosphere
TOMS	Total Ozone Mapping Spectrometer
TOR	Thermal/Optical Reflectance
TOT	Thermal/Optical Transmittance
$\mu g/m^3$	Micrograms per cubic meter
ULSD	Ultra-Low-Sulfur Diesel
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
USAID	United States Agency for International Development
USDA	United States Department of Agriculture
USGCRP	United States Global Climate Change Research Program

UV	Ultraviolet
VA	Vanillic Acid
VMT	Vehicle Miles Traveled
VOC	Volatile Organic Compound
VSBK	Vertical Shaft Brick Kiln
VSL	Value of a Statistical Life
W m^{-2}	Watts per square meter (also W/m^2)
WAIS	West Antarctic Ice Sheet
WHO	World Health Organization
WMO	World Meteorological Organization
WRAP	Western Regional Air Partnership
WTP	Willingness to Pay

Glossary

Aerosol

A mixture of gases and suspended solid and/or liquid particles, with a typical size between 0.01 and 10 micrometers and residing in the atmosphere for at least several hours. Aerosols may be of either natural or anthropogenic origin. Often the term is used interchangeably with "particle" or "particulate matter."

Aerosol Absorption Optical Depth

A quantitative measure of light extinction within a vertical column of atmosphere due to absorption by aerosols.

Aerosol Optical Depth

A quantitative measure of light extinction within a vertical column of atmosphere due to aerosol absorption or scattering. Pollution and cloud-free portions of the atmosphere have a low aerosol optical depth, while highly polluted or densely cloudy skies have a high optical depth.

Aging

The changes that occur to a particle over the course of its atmospheric lifetime, including changes in size or chemical composition.

Agricultural Burning

The planned burning of vegetative debris from agricultural operations; or, the use of fire as a method of clearing land for agricultural use or pastureland.

Albedo

The fraction of solar radiation reflected by a surface or object, often expressed as a percentage. Light-colored surfaces (such as those covered by snow and ice) have a high albedo; dark surfaces (such as dark soils, vegetation and oceans) have a low albedo.

Arctic Haze

A persistent reddish-brown haze visible in the atmosphere at high latitudes in the Arctic due to air pollution, including black carbon, organic carbon, and sulfate particles.

Atmospheric Brown Clouds

Pollution clouds consisting of combinations of black carbon, brown carbon, sulfates, organics, dust, and other components. Atmospheric brown clouds are more common in Asia, southern Africa, and the Amazon Basin. They have been linked to surface dimming and a decrease in vertical mixing (which exacerbates air pollution episodes), and they contribute to changes in the pattern and intensity of rainfall (particularly with respect to monsoon circulation in South Asia).

Atmospheric Lifetime

The approximate amount of time it would take for the atmospheric concentration of a pollutant to return to its natural level (assuming emissions cease) as a result of either being converted to another chemical compound or being taken out of the atmosphere via a sink. This time depends on the pollutant's sources and sinks as well as its reactivity. Average lifetimes for air pollutants can vary from days to weeks (black carbon and ozone) to more than a century (e.g., chlorofluorocarbons and carbon dioxide).

Atmospheric Residence Time
See *atmospheric lifetime*.

Atmospheric Transport
The movement of chemical species through the atmosphere as a result of large-scale atmospheric motions. Transport distances are a function of atmospheric lifetimes, emission location, and overall meteorological activity.

Baghouse
See *fabric filter*.

Biofuels
Biofuels are non-fossil carbon-based fuels derived from organic materials (biomass), including plant materials and animal waste.

Biomass
In the context of energy, the term biomass is often used to refer to organic materials, such as wood and agricultural wastes, which can be burned to produce energy or converted into a gas and used for fuel.

Black Carbon
A solid form of mostly pure carbon that absorbs solar radiation (light) at all wavelengths. Black carbon is the most effective form of particulate matter, by mass, at absorbing solar energy, and is produced by incomplete combustion.

Black Smoke
The term used since the 1950s to describe carbon-containing particulate matter resulting from incomplete combustion (e.g., from coal); also refers to a measurement that quantified the concentration of ambient particulate matter. Term has been used as a synonym for *soot*.

Bottom-Up Inventory
Emissions inventory based on emissions as measured or computed directly by concentration, mass flow, or stream velocity observations at the source, or calculated (using specific emission factors and activity levels) on a source-by-source basis.

Boundary Layer
The bottom layer of the atmosphere that is directly influenced by contact with the surface of the Earth.

Brown Carbon
A class of particulate organic carbon compounds that absorb ultraviolet and visible solar radiation. BrC can be directly emitted as a product of incomplete combustion, or it can be formed in the atmosphere as pollutants age.

Carbon Dioxide
A naturally occurring gas which arises from a variety of human activities such as burning fossil fuels and biomass, land-use changes and other industrial processes. Carbon dioxide is the principal anthropogenic greenhouse gas that affects the Earth's radiative balance. It is used as the reference gas against which other greenhouse gases are measured. See also *climate change*, *global warming, and global warming potential*.

Carbon Dioxide Equivalent
A metric used to compare the emissions from various greenhouse gases based upon their global warming potential. It is the calculated equivalent amount of carbon dioxide emissions that would result in the same radiative effect as a pulse of emissions of another greenhouse gas. Carbon dioxide equivalents are commonly expressed as "million metric tons of carbon dioxide equivalents" (MMTCO$_2$eq).

Carbon Mass Ratio
The ratio of the mass of different components of carbonaceous particles (e.g., the ratio of organic carbon to black carbon, or the ratio of black carbon to total particulate matter).

Carbonaceous Particulate Matter
A general term for carbon-based compounds found in particles, including black carbon and organic carbon. Primary combustion particles are largely composed of carbonaceous particulate matter.

Climate Change
Climate change refers to any significant change in measures of climate (such as temperature, precipitation, or wind) lasting for an extended period (decades or longer). Climate change may result from natural factors, such as changes in the Sun's intensity or slow changes in the Earth's orbit around the Sun; natural processes within the climate system (e.g., changes in ocean circulation); and human activities that change the atmosphere's composition (e.g., through burning fossil fuels) and the land surface (e.g., deforestation, reforestation, urbanization, desertification, etc.).

Cloud Albedo Effect
The process by which aerosols increase the reflectivity of clouds, leading to negative radiative forcing.

Cloud Brightening
See *cloud albedo effect*.

Cloud Burn-off
A subset of the *semi-direct effect*, related specifically to absorbing aerosols being embedded within a cloud and resulting in decreased cloud cover.

Cloud Lifetime Effect
The process by which aerosols reduce the size of cloud droplets, resulting in changes to precipitation patterns and increases in cloud lifetime that lead to cooling.

Coating
Atmospheric process by which a particle can become encased by a shell of a different substance; this often alters the particle's light absorption qualities.

Co-Emitted Pollutants
Gases and particles that are emitted with black carbon, such as organic carbon, sulfates, nitrates, sulfur dioxide and nitrogen oxides.

Co-Pollutants
See *co-emitted pollutants*.

Condensable Particulate Matter
Particles formed as an emissions plume cools in the atmosphere.

Contained Combustion
Closed combustion including internal combustion, reciprocating diesel engines, closed burning. Differentiated from open burning and open combustion.

Deposition
The transfer of atmospheric gases and particles to the Earth's surface. Wet deposition refers to deposition that occurs as a result of precipitation. Dry deposition occurs in the absence of precipitation.

Diesel Particulate Filter
Exhaust emissions control device used to reduce diesel particulate matter; also called a diesel particulate trap. The diesel particulate filter consists of a porous honeycomb structure that physically captures and oxidizes the diesel particulate matter.

Diesel Particulate Matter

The particulate component of diesel exhaust, which includes a mixture of black carbon, organic carbon, sulfates, metals, and trace elements. Black carbon is a major constituent of diesel particulate matter.

Diesel Retrofit

Any technology or system that achieves emissions reductions beyond those required by new engine regulations, including the replacement of high-emitting vehicles/equipment with cleaner vehicles/equipment, repowering or engine replacement, rebuilding the engine to a cleaner standard, installation of advanced emissions control after-treatment technologies such as DPFs, or the use of a cleaner fuel.

Direct Effect

The direct scattering or absorption of solar and terrestrial radiation by atmospheric particles.

Direct Forcing

The change in incoming and outgoing solar and terrestrial radiation due to the *direct effect* of atmospheric pollutants.

Disability Adjusted Life Years

A measure of overall disease burden, expressed as the number of life years lost due to ill-health, disability or early death.

Elemental Carbon

A descriptive term for carbonaceous particles that is based on chemical composition rather than light-absorbing characteristics. Often used as a synonym for *black carbon*.

Externally Mixed

When each particle in a collection of atmospheric particles is assumed to be composed of only one chemical compound for purposes of modeling and study.

Fabric Filters

Fabric filters are one of the most widely used devices for controlling emissions of particulate matter. A fabric filter system typically consists of multiple filter elements (bags) enclosed in a compartment (housing). When the process stream enters the housing and passes through the filter elements, particulate matter accumulates as a dust cake on the surface of the bag.

Filter-Based Techniques

One of several ways to quantify the amount of particulate matter in ambient (outdoor) air. Filter-based measurement methods use samplers that consist of a vacuum pump calibrated to draw in a fixed volume of air per minute through a filter that captures particles. The average concentration of particulate matter in the air can be calculated by weighing the filter before and after the run, and correlating the particulate weight to the volume of air drawn through the pump.

First Indirect Effect

See *cloud albedo effect*.

Flaming

The stage of combustion when fuel gases are rapidly oxidized. Under oxygen-limited and relatively low-temperature conditions, soot is emitted.

Fossil Fuels

Fuels derived from coal, oil, and natural gas.

Gasoline Direct Injection

A fuel injection system for gasoline vehicles which introduces fuel directly into each cylinder, which results in improved fuel economy with higher engine compression rations. Gasoline direct injection is projected to be used on almost all new model year vehicles starting in 2016.

Glaciation Indirect Effect
A warming effect which occurs in certain mixed-phase clouds when black carbon aerosols (and some other particles such as mineral dust) serve as ice nuclei in a super-cooled liquid water cloud, thereby enabling precipitation rather than delaying it.

Global Cost Potential
A metric that compares the relative marginal abatement costs for two climate forcers when a given climate change target is achieved at least cost.

Global Damage Potential
A metric that compares the relative damage resulting from an equal mass of emissions of two climate forcers.

Global Temperature Potential
A physical metric that compares the global average temperature change at a given point in time resulting from equal mass of emissions of two climate forcers.

Global Warming Potential
An index, based upon radiative properties of well-mixed greenhouse gases, measuring the radiative forcing of a unit mass of a given pollutant in the present-day atmosphere integrated over a chosen time horizon (often 100 years, or GWP100), relative to that of carbon dioxide (CO_2 always has a global warming potential of 1). The global warming potential represents the combined effect of the differing times these pollutants remain in the atmosphere and their relative effectiveness in absorbing radiation.

Greenhouse Gas
Any gas that absorbs infrared radiation in the atmosphere. Greenhouse gases include, but are not limited to, water vapor, carbon dioxide, methane, nitrous oxide, chlorofluorocarbons, hydrochlorofluorocarbons, ozone, hydrofluorocarbons, perfluorocarbons, and sulfur hexafluoride.

Hazardous Air Pollutant
Pollutants that are known or suspected to cause cancer or other serious health effects, such as reproductive effects or birth defects, or adverse environmental effects.

Incomplete Combustion
Combustion where only a partial burning of a fuel occurs. Combustion in practice is almost always incomplete due to insufficient oxygen or low temperature during the combustion process preventing the complete oxidation of the fuel to CO_2.

Indirect Effects
The various types of absorption or scattering of solar or terrestrial radiation that occur as a result of anthropogenic aerosol interaction with clouds. These include changes in cloud lifetime, reflectivity, and composition.

Indirect Forcing
The change in incoming and outgoing solar and terrestrial radiation due to the various *indirect effects* resulting from impacts on clouds, including changes in cloud lifetime, reflectivity, and composition. This forcing can be either positive (warming) or negative (cooling), depending on the specific cloud interaction.

Infrared Radiation
Radiation emitted by the Earth's surface, the atmosphere, and the clouds. It is also known as terrestrial or long-wave radiation.

Instantaneous Radiative Forcing
The difference between the amount of radiation coming into the Earth's system and leaving the system, as measured at the tropopause at a specific instant, due to a change in atmospheric concentrations. Unlike other *radiative forcing* measures, instantaneous radiative forcing calculations do not allow any part of the system to adjust prior to estimating net forcing.

Intergovernmental Panel on Climate Change
Established in 1988 by the World Meteorological Organization and the United Nations Environment Programme, the Intergovernmental Panel on Climate Change is responsible for providing the scientific and technical foundation for the United Nations Framework Convention on Climate Change, primarily through the publication of periodic assessment reports.

Internally Mixed
When individual atmospheric particles are treated as mixtures of chemical components for purposes of modeling and study.

Kyoto Basket
The set of greenhouse gases covered under the Kyoto Protocol: carbon dioxide, methane, nitrous oxide, sulfur hexafluoride, hydrofluorocarbons, and perfluorocarbons.

Laser Induced Incandescence
A technique in which a high-energy laser is used to heat soot particles to high temperatures; measurement of the resulting incandescent light emitted by the soot particles indicates the amount of soot (soot volume fraction) and its location within the combustion event.

Light-Absorbing Carbon
Carbonaceous particles that absorb light, including black carbon plus brown carbon.

Light-Absorbing Particulate Matter
Refers to particles that tend to absorb light, which represents energy added to the Earth's system and leads to climate warming.

Light-Scattering Particulate Matter
Refers to particles that tend to reflect or scatter light, which generally leads to increased reflection of light back to space, causing climate cooling.

Long-Lived Climate Forcer
A pollutant like CO_2 that has a positive radiative forcing effect on climate and a long atmospheric lifetime (decades to centuries).

Maximum Achievable Control Technology (MACT) Standards
U.S. federal emissions standards for stationary sources of hazardous air pollutants requiring the maximum emissions reductions, taking cost and feasibility into account. Under the Clean Air Act Amendments of 1990, the MACT must not be less than the average emission level achieved by controls on the best performing 12 percent of existing sources, by category of industrial and utility sources. See also *National Emissions Standards for Hazardous Air Pollutants (NESHAP).*

Metric
An analytical measurement intended to quantify the state of a system. In climate assessments, metrics are used to quantify the impact of a pollutant relative to a common baseline.

Mixed-Phase Clouds
Clouds with both ice and water.

Mixing
The movement of pollutants through the atmosphere that often leads to chemical and physical transformations of the pollutants.

National Ambient Air Quality Standards (NAAQS)

Standards required under the Clean Air Act for widespread pollutants from numerous and diverse sources considered harmful to public health and the environment. Primary standards are designed to protect public health, including the health of "sensitive" populations such as asthmatics, children, and the elderly. Secondary standards are designed to protect public welfare, including protection against decreased visibility, damage to animals, crops, vegetation, and buildings. EPA has set NAAQS for six principal pollutants, which are called "criteria" pollutants: particulate matter, ozone, lead, oxides of nitrogen, oxides of sulfur, and carbon monoxide.

National Emissions Inventory

EPA's national emissions database containing information about sources that emit criteria air pollutants and their precursors, and hazardous air pollutants. The database includes estimates of annual air pollutant emissions from point, nonpoint, and mobile sources in the 50 states, the District of Columbia, and Puerto Rico.

National Emissions Standards for Hazardous Air Pollutants (NESHAP)

Stationary source standards for hazardous air pollutants. The NESHAPs promulgated after the 1990 Clean Air Act Amendments require application of technology-based emissions standards referred to as *Maximum Achievable Control Technology (MACT)* standards. Consequently, these post-1990 NESHAPs are also referred to as MACT standards.

Net Radiative Forcing

The total *radiative forcing* due to the presence of a pollutant in the atmosphere, accounting for both the positive (warming) and negative (cooling) forcing associated with different radiative effects of the pollutant. For particles, this includes accounting for direct, indirect (cloud), and snow/ice albedo effects.

New Source Performance Standards (NSPS)

U.S. federal emissions standards for certain air pollutants that are emitted from new, modified, or reconstructed stationary emissions sources which reflect the use of best available control technology.

Nitrogen Oxides

A generic term for a group of highly reactive gases, including nitrogen oxide (NO) and nitrogen dioxide (NO_2). Nitrogen oxides result from combustion of fossil or biofuels, especially at high temperatures.

Open Biomass Burning

Open burning of vegetative material; includes agricultural burning, prescribed burning, and wildfires.

Organic Carbon

The mix of compounds containing carbon bound with other elements; e.g., hydrogen and oxygen. Organic carbon may be a product of incomplete combustion, or formed through the oxidation of VOCs in the atmosphere. Both primary and secondary organic carbon possess radiative properties that fall along a continuum from light-absorbing to light-scattering.

Organic Carbon to Black Carbon Ratio

See *carbon mass ratios*.

Organic Matter

The total mass of organic material in a compound.

Oxidation

The chemical reaction of a substance with oxygen or a reaction in which the atoms in an element lose electrons and its valence is correspondingly increased.

Particle Coagulation

The process by which particles collide and stick together. Part of the internal mixing process, coagulation reduces the overall particle number and reduces the differences in chemical composition among the individual particles in the emissions plume.

Particulate Matter

A complex mixture of extremely small particles and liquid droplets suspended in the atmosphere. Particulate matter (PM) is made up of a number of components, including acids (such as nitrates and sulfates), organic chemicals, metals, and soil or dust particles. For purposes of air quality and health studies, PM is typically measured in two size ranges: PM_{10} and $PM_{2.5}$.

PM_{10}

Particulate matter with an aerodynamic diameter less than or equal to a nominal 10 micrometers. PM_{10} includes $PM_{2.5}$.

$PM_{2.5}$

Fine particulate matter with an aerodynamic diameter less than or equal to 2.5 micrometers.

Photoacoustic

A black carbon measurement technique where light from a source is absorbed by the aerosol resulting in the heating and subsequent expansion of the surrounding air. The expansion results in a sound wave which is then detected with a microphone.

Polycyclic Aromatic Hydrocarbon

A group of organic contaminants formed from incomplete combustion. These aromatic compounds comprise two or more benzene rings arranged in various configurations (polycyclic). Many polycyclic aromatic hydrocarbons (PAHs) are known to be toxic to humans and ecosystems, and EPA has classified seven PAH compounds as probable human carcinogens: benzo[a]pyrene, benz[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, dibenz[a,h]anthracene, and indeno[1,2,3cd]pyrene). PAHs are precursors (building blocks) in the formation of black carbon and brown carbon.

Prescribed Fire

Any fire intentionally ignited by management under an approved plan to meet specific objectives.

Primary Particle

A particle that is emitted directly from a source.

Pulse

See *specific forcing pulse*.

Pyrolysis

The heating of solid fuels in the absence of oxygen. Pyrolysis induces the evaporation of volatile gases from the solid fuel needed to support combustion. Thermal breakdown of portions of the solid fuel provide additional fuel gases. Pyrolysis is used to produce charcoal and biochar, a residual form of carbon in solid form.

Radiation

Energy in the form of electromagnetic waves (or photons). Photons absorbed by solid materials, such as light-absorbing particles, are transformed into other forms of energy. Most notably, solar radiation absorbed by a particle converts, in part, into heat energy that warms the surrounding atmosphere or the surfaces upon which the particles are deposited (e.g., snow and ice).

Radiative Forcing
The change in the energy balance between incoming solar radiation and exiting infrared radiation, typically measured in watts per square meter (W m-2), due to a change in concentration (generally the change since preindustrial conditions in 1750). Positive radiative forcing tends to warm the surface of the Earth, while negative forcing generally leads to cooling. A pollutant that increases the amount of energy in the Earth's climate system is said to exert "positive radiative forcing," which leads to warming. In contrast, a pollutant that exerts "negative radiative forcing" reduces the amount of energy in the Earth's system and leads to cooling.

Second Indirect Effect
See *cloud lifetime effect*.

Secondary Organic Aerosols
Carbonaceous aerosols that are produced in the atmosphere rather than being directly emitted. Precursor gases (such as aromatic hydrocarbons, monoterpenes) undergo chemical reactions and condensation to form secondary organic aerosols.

Secondary Particle
A particle (e.g., sulfate or nitrate) that is formed in the atmosphere from the oxidation of gaseous precursors like sulfur dioxide, nitrogen oxides, and volatile organic compounds or through the transformation of directly emitted particles. The acids resulting from the oxidation of these compounds attract water vapor to form tiny droplets (fine particles).

Semi-Direct Effect
Localized heating of the atmosphere by absorbing aerosol particles, affecting the relative humidity and stability of the troposphere, which in turn affect cloud formation and lifetime.

Short-Lived Climate Forcer
A pollutant, such as black carbon, ozone, or methane, that has a positive radiative forcing effect on climate but a relatively short atmospheric lifetime (days to years).

Single-Scattering Albedo
The ratio of scattering optical depth to the total optical depth (scattering plus absorption) of the atmosphere; indicates how much of the light extinction in the atmosphere is due to scattering vs. absorption. If single-scattering albedo equals 1, all particle extinction is due to scattering; if single-scattering albedo equals 0, all extinction is due to absorption.

Smoldering
A non-flaming phase of the combustion process that involves a slower, cooler form of combustion which occurs as oxygen attacks the surface of heated solid fuel directly.

Snow/Ice Albedo Effect
Decrease in reflectivity (and increase in absorption) of solar radiation that occurs as a result of the darkening of snow and ice through aerosol deposition.

Snow/Ice Albedo Forcing
Positive radiative forcing resulting from the deposition of black carbon on snow and ice, which darkens the surface and decreases reflectivity (albedo), thereby increasing absorption of *solar radiation* and accelerating melting.

Social Cost of Carbon
An estimate of the monetized damages resulting from an incremental increase in CO_2 emissions in a given year. It can be thought of as the monetized benefit to society of reducing one ton of CO_2.

Solar Radiation
Radiation emitted by the Sun. It is also referred to as short-wave radiation. Solar radiation has a distinctive range of wavelengths (spectrum) determined by the temperature of the Sun. In the context of this report, the term refers to the portions of the solar spectrum which reach the lower atmosphere, including the ultraviolet (>280 nm), visible and infrared.

Solar Zenith Angle
The angle between a point directly above any location on the Earth's surface (the zenith) and the Sun, as measured at the location. These angles relate to the elevation of the Sun above the horizon (in degrees).

Soot
A complex mixture of mostly black and organic carbon that is the primary light-absorbing pollutant emitted by the incomplete combustion of fossil fuels, biofuels, and biomass.

Source Apportionment
The use of ambient and/or emissions data along with statistical modeling to determine the contribution of a specific emissions source category to measured ambient concentrations of air pollutants like $PM_{2.5}$.

SPECIATE database
EPA's repository of speciation profiles characterizing the composition of emissions from specific air pollution sources.

Specific Forcing Pulse
A metric based on the amount of energy added to the Earth's system due to the radiative forcing caused by a given mass of a pollutant.

Surface Dimming Effect
The reduction of solar radiation at Earth's surface due to high concentrations of particles, especially light absorbing particles, in the atmosphere, above. This results in cooling at the Earth's surface (even though net forcing measured at the top of the atmosphere might be positive).

Surface Temperature Response per Unit Continuous Emission
A metric that compares the change in surface temperature due to an assumed continuous emission of equal masses a climate forcer and CO_2.

Tar Balls
Liquid aerosol droplets observed in biomass burning plumes that appear to be formed entirely from brown carbon.

Thermal Techniques
A variety of approaches used for the measurement of organic and elemental carbon. The techniques involve the heating of particulate matter with the subsequent detection of the evolved carbon with a variety of techniques.

Thermal-Optical Techniques
A variety of approaches used for the measurement of organic and elemental carbon. These techniques are similar to the thermal techniques defined above, with the addition of an optical measurement to improve the separation of elemental carbon from organic carbon.

Thermodynamic Effect
The process by which freezing of droplets in mixed-phase clouds is delayed because droplet size is reduced. This effect ultimately changes the characteristics of the cloud, but whether it leads to warming or cooling is unclear.

Third Pole
Refers to the Hindu Kush-Himalayan-Tibetan region.

Tier 2 Standards
U.S. EPA emissions standards for emissions of hydrocarbons, carbon monoxide, oxides of nitrogen, and PM from light-duty vehicles (automobiles and light-duty trucks) phased in with the 2004 through 2006 model years. Standards represent roughly 99% emissions reduction compared to pre-control (pre-1968 model year) vehicles.

Top-Down Inventory
Emissions inventory based on ambient air quality data. Individual emissions source estimates are based upon the relative magnitude of tracer compounds in the ambient air.

Top of the Atmosphere
The location between the troposphere and the stratosphere. Measuring radiative forcing at this altitude is best for determining net energy balance.

Top of the Atmosphere Radiative Forcing
Net radiative forcing measured (or modeled) at the top of the atmosphere to capture the total change in incoming and outgoing radiation due to the presence of atmospheric pollutants.

Transport
See *atmospheric transport*.

Troposphere
The lowest part of the atmosphere from the Earth's surface (ranging from 9 km in high latitudes to 16 km in the tropics on average) where clouds and "weather" phenomena occur. In the troposphere, temperatures generally decrease with height.

Ultra-Low Sulfur Diesel Fuel
Diesel fuel that has a maximum of 15 parts per million of sulfur content. Ultra-low sulfur diesel enables advanced pollution control technologies such as *diesel particulate filters* and urea selective catalytic reduction systems for NO_x.

Ultraviolet Radiation
The energy range just beyond the violet end of the visible spectrum, with wavelengths between 10-400 nm. Ultraviolet radiation constitutes only about 5 percent of the total energy emitted from the Sun, and most ultraviolet radiation is blocked by Earth's atmosphere. The ultraviolet radiation that does reach the Earth aids in plant photosynthesis and helps produce vitamin D in humans. Too much ultraviolet radiation can burn the skin, cause skin cancer and cataracts, and damage vegetation.

Value of a Statistical Life
A summary measure for the dollar value of small changes in mortality risk experienced by a large number of people. The total estimated value of a statistical life is derived from aggregated estimates of individual values for small changes in mortality risks.

Volatile Organic Compounds
Organic chemical compounds whose composition makes it possible for them to evaporate under normal atmospheric conditions of temperature and pressure. These carbonaceous pollutant gases are emitted by both anthropogenic and natural processes and often serve as precursors for the formation of aerosol particles and ozone. Examples of volatile organic compounds include benzene, toluene, methylene chloride, and methyl chloroform.

Wildfire
An unplanned ignition caused by lightning, volcanoes, unauthorized activity, accidental human actions, and escaped prescribed fires.

Willingness to Pay
The maximum amount a person would be willing to pay, sacrifice, or exchange in order to receive a good or to avoid something undesired, such as pollution.

Executive Summary

Black carbon (BC) emissions have important impacts on public health, the environment, and the Earth's climate. BC is a significant component of particle pollution, which has been linked to adverse health and environmental impacts through decades of scientific research. Recent work indicates that BC also plays an important role in climate change, although there is more uncertainty about its effects on climate than for greenhouse gases (GHG), such as carbon dioxide and methane. BC has been linked to a range of climate impacts, including increased temperatures, accelerated ice and snow melt, and disruptions to precipitation patterns. Importantly, reducing current emissions of BC may help slow the near-term rate of climate change, particularly in sensitive regions such as the Arctic. However, BC reductions cannot substitute for reductions in long-lived GHGs, which are necessary for mitigating climate change in the long run.

Despite the rapidly expanding body of scientific literature on BC, there is a need for a more comprehensive evaluation of both the magnitude of particular global and regional climate effects due to BC and the impact of emissions mixtures from different source categories. To advance efforts to understand the role of BC in climate change, on October 29, 2009, Congress requested the U.S. Environmental Protection Agency (EPA) 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. As requested by Congress, EPA has consulted with other federal agencies on key elements of this report, including inventories, health and climate science, and mitigation options. The report draws from recent BC assessments, including work under the United Nations Environment Programme (UNEP) and the World Meteorological Organization (WMO), the Convention on Long Range Transboundary Air Pollution (CLRTAP), and the Arctic Council. Each of these individual efforts provides important information about particular sectors, regions, or issues. The task outlined for EPA by Congress is broader and more encompassing, requiring a synthesis of currently available information about BC across numerous bodies of scientific inquiry. The results are presented in this *Report to Congress on Black Carbon*. The key messages of this report can be summarized as follows.

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

BC can be defined specifically as a solid form of mostly pure carbon that absorbs solar radiation (light) at all wavelengths. BC is the most effective form of PM, by mass, at absorbing solar energy;

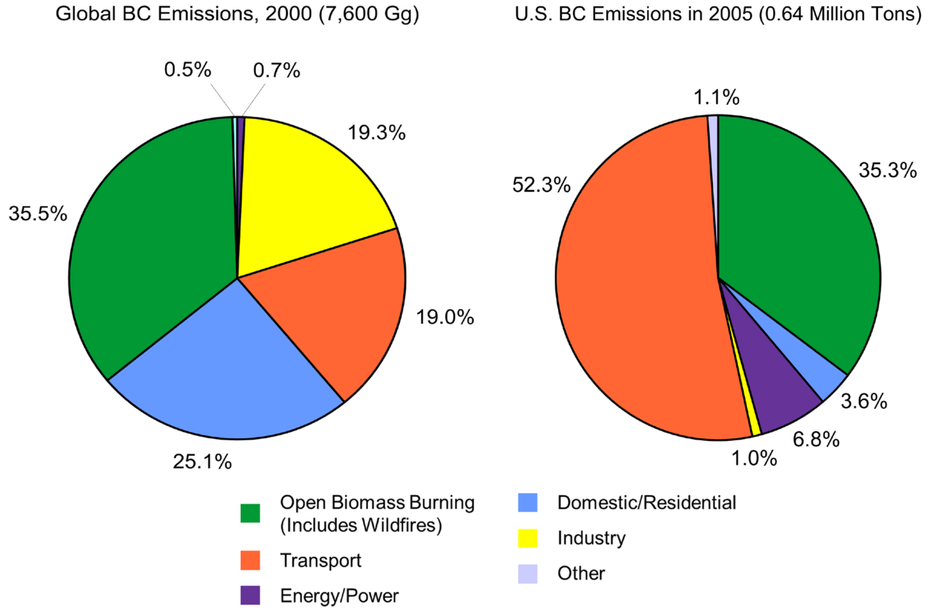


Figure A. BC Emissions by Major Source Category. (Source: Lamarque et al., 2010 and U.S. EPA)

other types of particles, including sulfates, nitrates and organic carbon (OC), generally reflect light. BC is a major component of "soot," a complex light-absorbing mixture that also contains organic carbon. Recent estimates of BC emissions by source category in the United States and globally are shown in Figure A.

2. ***BC is emitted directly into the atmosphere in the form of fine particles ($PM_{2.5}$). The United States contributes about 8% of the global emissions of BC. Within the United States, BC is estimated to account for approximately 12% of all direct $PM_{2.5}$ emissions in 2005. Many countries have significantly higher $PM_{2.5}$ emissions than the United States, and countries with a different portfolio of emissions sources might have a significantly higher percentage of BC.***

3. ***BC contributes to the adverse impacts on human health, ecosystems, and visibility associated with $PM_{2.5}$.***

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. $PM_{2.5}$, both ambient and indoor, is estimated to result in millions of premature deaths worldwide, the majority of which occur in developing countries. The World Health Organization estimates that indoor smoke from solid fuels is the 10th major mortality risk factor globally, contributing to approximately 2 million deaths annually. Women and children are particularly at risk. Ambient air pollution is also a significant health threat: according to the WHO, urban air pollution is among the top ten risk factors in medium- and high-income countries. Urban air pollution is not ranked in the top ten major risk factors in low-income countries since other risk factors (e.g., childhood underweight and unsafe water, sanitation and hygiene) are so substantial; however, a much larger portion of the total deaths related to ambient $PM_{2.5}$ globally are expected to occur in developing regions, partly due to the size of exposed populations in those regions. $PM_{2.5}$ 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.

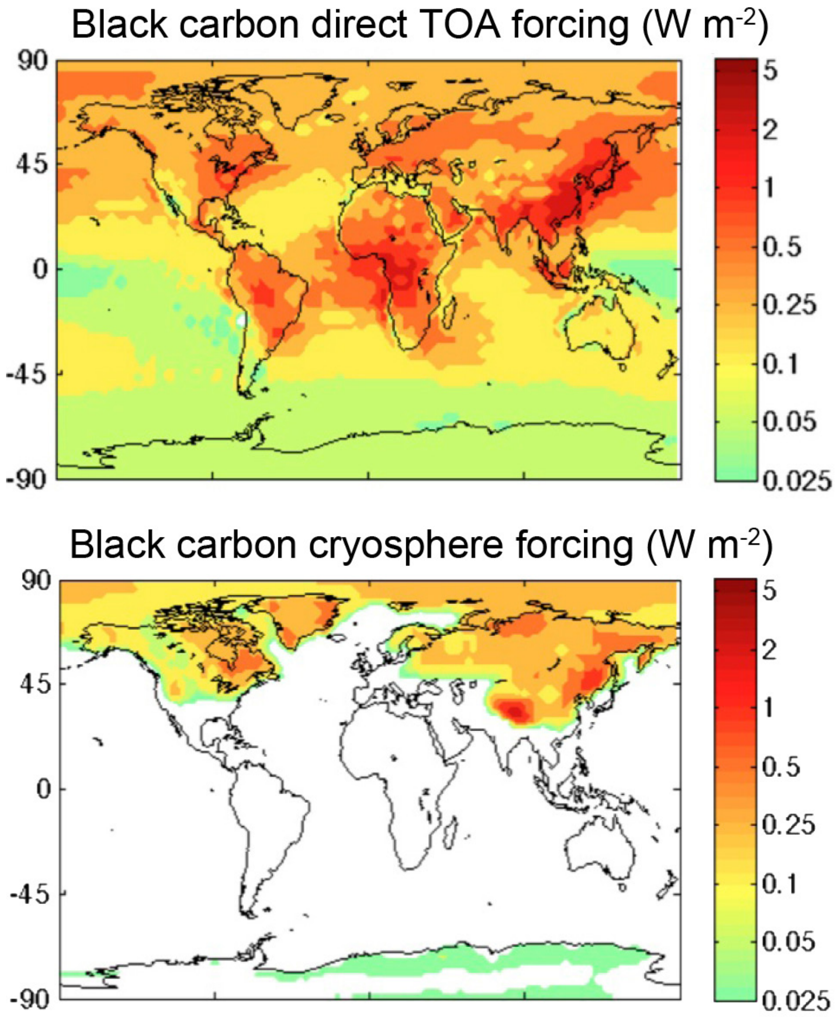
Over the past decade, the scientific community has focused increasingly on trying to identify the health impacts of particular $PM_{2.5}$ constituents, such as BC. However, EPA has determined that there is insufficient information at present to differentiate the health effects of the various constituents of $PM_{2.5}$; thus, EPA assumes that many constituents are associated with adverse health impacts. It is noteworthy that emissions and ambient concentrations of directly emitted $PM_{2.5}$ are often highest in urban areas, where large numbers of people live.

4. ***BC influences climate through multiple mechanisms:***

- *Direct effect:* BC absorbs both incoming and outgoing radiation of all wavelengths, which contributes to warming of the atmosphere and dimming at the surface.
- *Snow/ice albedo effect:* BC deposited on snow and ice darkens the surface and decreases reflectivity, thereby increasing absorption and accelerating melting.
- *Other effects:* BC also alters the properties of clouds, affecting cloud reflectivity and lifetime ("indirect effects"), stability ("semi-direct effect") and precipitation.

5. ***The direct and snow/ice albedo effects of BC are widely understood to lead to climate warming. However, the globally averaged net climate effect of BC also includes the effects associated with cloud interactions, which are not well quantified and may cause either warming or cooling. Therefore, though most estimates indicate that BC has a net warming influence, a net cooling effect cannot be ruled out. It is also important to note that the net radiative effect of all aerosols combined (including sulfates, nitrates, BC and OC) is widely understood to be negative (cooling) on a global average basis.***

The direct radiative forcing effect of BC is the best quantified and appears to be positive and significant


Black carbon direct TOA forcing (W m^{-2})

Black carbon cryosphere forcing (W m^{-2})

Figure B. Regional Variability in Direct Radiative Forcing and Snow/Ice Albedo Forcing for BC from All Sources, simulated with the Community Atmosphere Model. (Source: Bond et al., 2011) *on both global and regional scales. This warming effect is augmented by deposition of BC on snow and ice.* These effects are shown in Figure B. The central estimates of global average direct forcing by BC from surveyed studies range from +0.34 to +1.0 Watts per square meter (W m^{-2}). A recent UNEP/WMO assessment presented a narrower central range of +0.3 to +0.6 W m^{-2}. These estimates are generally higher than the 2007 Intergovernmental Panel on Climate Change (IPCC) estimate of +0.34 (±0.25) W m^{-2}.

The snow/ice albedo effect from BC has been estimated in recent studies to add about +0.05 W m^{-2}, generally less than the +0.1 (±0.1) W m^{-2} estimated by the IPCC; however, UNEP/WMO found that when the snow/ice albedo forcing estimates are adjusted to account for the greater warming efficacy of the snow/ice deposition mechanism, the snow/ice albedo effect could add +0.05 to +0.25 W m^{-2} of forcing. The sum of the direct and snow/ice albedo effects of BC on the global scale is likely comparable to or larger than the forcing effect from methane, but less than the effect of carbon dioxide;[1] however, there is more uncertainty in the forcing estimates for BC.

The climate effects of BC via interactions with clouds are more uncertain, and their net climate influence is not yet clear. All aerosols (including BC) affect climate indirectly by changing the reflectivity (albedo) and lifetime of clouds. The net indirect effect of all aerosols is very uncertain but is thought to have a net cooling influence. The IPCC estimated the global average cloud albedo forcing from all aerosols as -0.7 W m^{-2} (with a 5 to 95% confidence range of -0.3 W m^{-2} to -1.80 W m^{-2}). The IPCC did not provide quantitative estimates of the effect of aerosols on cloud lifetime, and the contribution of BC to these indirect effects has not been explicitly quantified to date. BC has additional effects on clouds—including changes to cloud stability and enhanced precipitation from colder clouds—that can lead to either warming or cooling. However, few quantitative estimates of these effects are available, and significant uncertainty remains. Due to all of the remaining gaps in scientific knowledge, it is difficult to place quantitative bounds on the forcing attributable to BC impacts on clouds at present; however, UNEP/WMO have provided a central forcing estimate of -0.4 to +0.4 W m^{-2} for all of the cloud effects of BC combined.

The sign and magnitude of the net climate forcing from BC emissions are not fully known at present, largely due to remaining uncertainties regarding the effects of BC on clouds. There is inconsistency among reported observational and modeling results, and many studies do not provide quantitative estimates of cloud impacts. In the absence of a full quantitative assessment, the current scientific basis for understanding BC climate effects is incomplete. Based on a limited number of modeling studies, the recent UNEP/WMO assessment estimated that global average net BC forcing is likely to be positive and in the range of 0.0 to +1.0 W m^{-2}, with a best estimate of +0.6 W m^{-2}; however, further work is needed to refine these estimates.

6. ***Sensitive regions such as the Arctic and the Himalayas are particularly vulnerable to the warming and melting effects of BC.***

Studies have shown that BC has especially strong impacts in the Arctic, contributing to earlier spring melting and sea ice decline. All particle mixtures reaching the Arctic are a concern, because even emissions mixtures that contain more reflective (cooling) aerosols can lead to warming if they are darker than the underlying ice or snow. Studies indicate that the effect of BC on seasonal snow cover duration in some regions can be substantial, and that BC deposited on ice and snow will continue to have radiative effects as long as the BC remains exposed (until the snow melts away or fresh snow falls). BC has also been shown to be a significant factor in the observed increase in melting rates of some glaciers and snowpack in parts of the Hindu Kush-Himalayan-Tibetan (HKHT) region (the "third pole").

7. ***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 cause 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, 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.

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[1] The IPCC's radiative forcing estimates for elevated concentrations of CO_2 and methane are +1.66 W m^{-2} and +0.48 W m^{-2}, respectively.

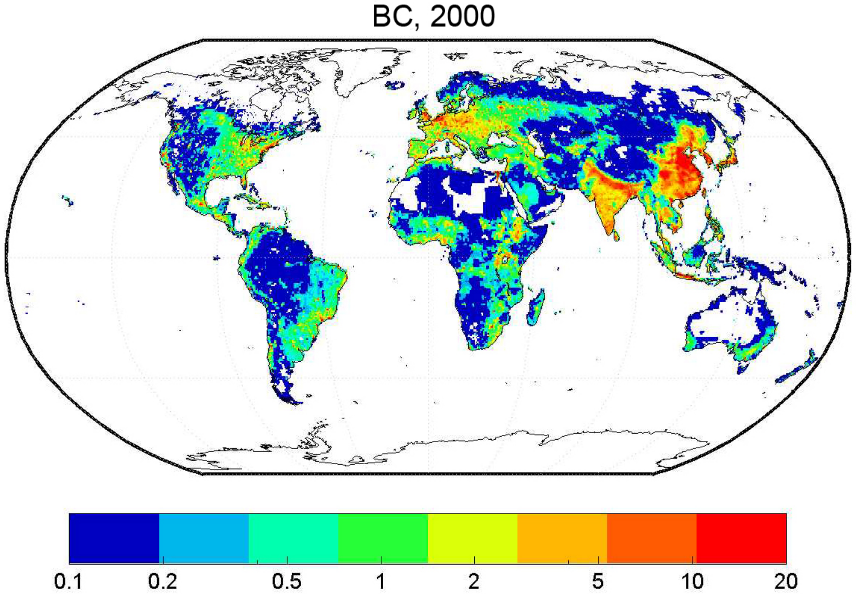

Figure C. BC Emissions, 2000, Gg. (Courtesy of Tami Bond, produced based on data from Bond et al., 2007)

8. ***BC is emitted with other particles and gases, many of which exert a cooling influence on climate. Therefore, estimates of the net effect of BC emissions sources on climate should include the offsetting effects of these co-emitted pollutants. This is particularly important for evaluating mitigation options. Some combustion sources emit more BC than others relative to the amount of co-pollutants; reductions from these sources have the greatest likelihood of providing climate benefits.***

The same combustion processes that produce BC also produce other pollutants, such as sulfur dioxide (SO_2), nitrogen oxides (NO_x), OC and CO_2. 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 sign and magnitude of the forcing resulting from particular emissions mixtures depend on their composition. For example, the particles emitted by mobile diesel engines are about 75% BC, while particle emissions from biomass burning are dominated by OC. Sources rich in BC have a greater likelihood of contributing to climate warming, and this may affect climate-related mitigation choices. Although OC generally leads to cooling, some portion of co-emitted OC, notably brown carbon (BrC), partially absorbs solar radiation. The net contribution of BrC to climate is presently unknown.

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

9. ***BC's short atmospheric lifetime (days to weeks), combined with its strong warming potential, means that targeted strategies to reduce BC emissions can be expected to provide climate benefits within the next several decades.***

Because the duration of radiative forcing by BC is very limited, the climate will respond quickly to BC emissions reductions, and this can help slow the rate of climate change in the near term. In contrast, long-lived GHGs may 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 the largest contributor to current and future climate change, and because GHGs accumulate in the atmosphere, deep reductions in these pollutants are necessary for limiting climate change over the long-term.

Emissions sources and ambient concentrations of BC vary geographically and temporally (Figure C), resulting in climate effects that are more regional and seasonal than the more uniform effects of long-lived, well-mixed GHGs. Likewise, mitigation actions for BC will produce different climate results depending on the region, season, and sources in the area where emissions reductions occur.

10. The different climate attributes of BC and long-lived GHGs make it difficult to interpret comparisons of their relative climate impacts based on common metrics.

Due in large part to the difference in lifetime between BC and CO_2, a comparison between the relative climate impacts of BC and CO_2 (or other climate forcers) is very sensitive to the metric used. There is currently no single metric (e.g., Global Warming Potential or GWP) that is widely accepted by the science and research community for this purpose. However, new metrics designed specifically for short-lived climate forcers like BC have recently been developed, and these metrics may enable better prioritization among mitigation options with regard to potential net climate effects.

11. Based on recent emissions inventories (2000 for global and 2005 for the United States), the majority of global BC emissions come from Asia, Latin America, and Africa. The United States currently accounts for approximately 8% of the global total, and this fraction is declining. Emissions patterns and trends across regions, countries and sources vary significantly.

Though there is significant uncertainty in global BC emissions inventories, recent studies indicate that global BC emissions have been increasing for many decades. 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 use of cleaner fuels. Elsewhere, BC emissions have been increasing, with most of the increase coming from developing countries in Asia, Africa and Latin America. 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.

12. Control technologies are available to reduce BC emissions from a number of source categories.

BC emissions reductions are generally achieved by applying technologies and strategies to improve combustion and/or control direct $PM_{2.5}$ emissions from sources. Though the costs of such mitigation approaches vary, many reductions can be achieved at reasonable costs. Controls applied to reduce BC will help reduce total $PM_{2.5}$ and other co-pollutants.

13. BC mitigation strategies, which lead to reductions in fine particles, can provide substantial public health and environmental benefits.

Strategies to reduce BC generally lead to reductions in emissions of all particles from a particular source. Thus, while it is not easy to reduce BC in isolation from other constituents, most mitigation strategies will provide substantial benefits in the form of $PM_{2.5}$ reductions. 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 $290,000 to $1.2 million per ton $PM_{2.5}$ in 2030 (2010$). 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 about $14,000 per ton $PM_{2.5}$ (2010$). BC reduction strategies implemented at the global scale could provide very large benefits: the $PM_{2.5}$ reductions resulting from BC mitigation measures could potentially result in hundreds of thousands of avoided premature deaths each year.

14. Mitigating BC can also make a difference in the short term for climate, at least in sensitive regions.

Benefits in sensitive regions like the Arctic, or in regions of high emissions such as Asia, may include reductions in warming and melting (ice, snow, glaciers) and reversal of changes in precipitation patterns. 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).

15. Selecting optimal BC mitigation measures requires taking into account the full suite of impacts and attempting to maximize co-benefits and minimize unintended consequences across all objectives (health, climate, and environment).

With a defined set of goals, policymakers can evaluate the “mitigation potential” within each country or region. The mitigation potential depends on total BC emissions and key emitting sectors, and also depends on the availability of control technologies or alternative mitigation strategies.

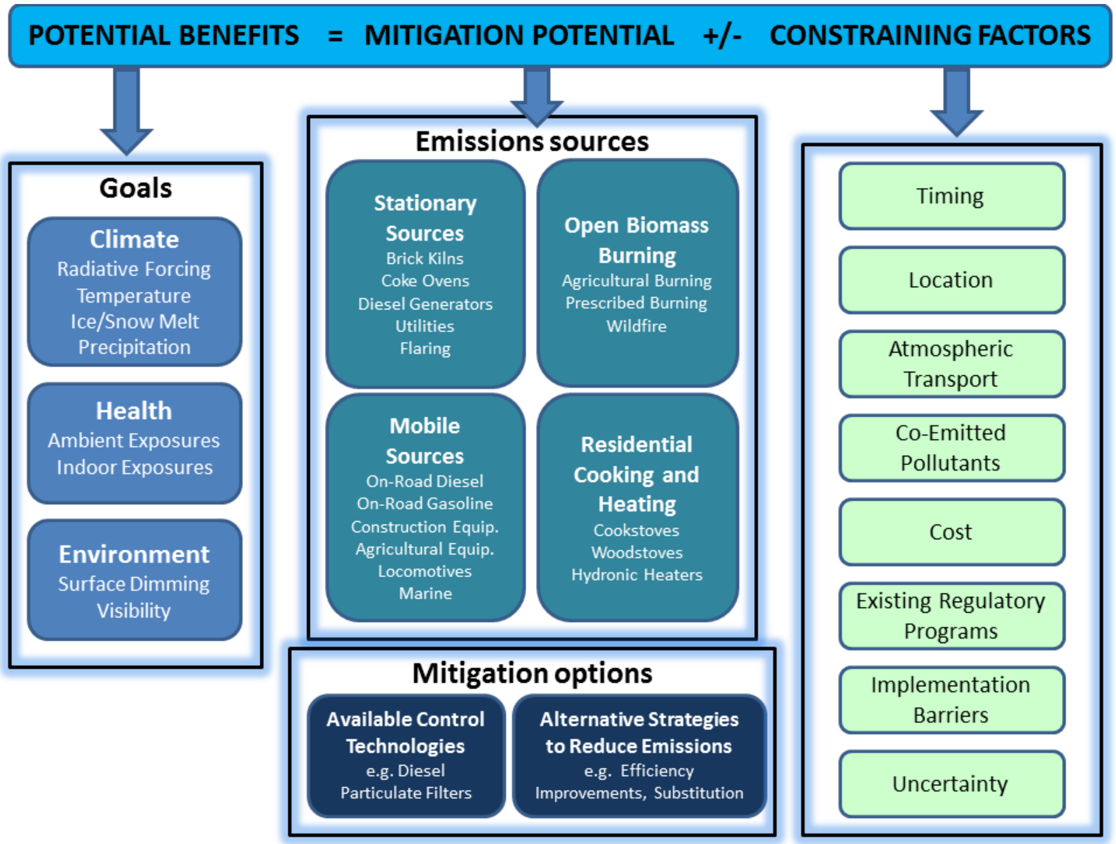

Figure D. Policy Framework for Black Carbon Mitigation Decisions. (Source: U.S. EPA.)

As illustrated in Figure D, the ideal emissions reduction strategies will also depend on a range of constraining factors, including:

- Timing
- Location
- Atmospheric Transport
- Co-emitted Pollutants
- Cost
- Existing Regulatory Programs
- Implementation Barriers
- Uncertainty

16. Considering the location and timing of emissions and accounting for co-emissions will improve the likelihood that mitigation strategies will be properly guided by the balance of climate and public health objectives.

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. Some of the most significant climate benefits of BC-focused control strategies may come from reducing emissions affecting the Arctic, Himalayas and other ice and snow-covered regions.

The effect of BC emissions reductions on human health is 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.

17. Achieving further BC reductions, both domestically and globally, will require adding a specific focus on reducing direct $PM_{2.5}$ emissions to overarching fine particle control programs.

BC reductions that have occurred to date (largely in developed countries) are mainly due to control programs aimed at $PM_{2.5}$, not targeted efforts to reduce BC per se. Greater attention to BC-focused strategies has the potential to help protect the climate (via the BC reductions achieved through direct $PM_{2.5}$ controls) while ensuring continued improvements in public health (via control of direct $PM_{2.5}$ in highly populated areas). Even if such controls are more costly than controls on secondary PM precursors, the combined public health and climate benefits may justify the expense.

18. *The most promising mitigation options identified in this report for reducing BC (and related "soot") emissions are consistent with control opportunities emphasized in other recent assessments.*

- **United States:** *The United States will achieve substantial BC emissions reductions by 2030, largely due to controls on new mobile diesel engines. Diesel retrofit programs for in-use mobile sources are a valuable complement to new engine standards for reducing emissions. Other source categories in the United States, including stationary sources (industrial, commercial and institutional boilers, stationary diesel engines, uncontrolled coal-fired electric generating units), residential wood combustion (hydronic heaters and woodstoves), and open biomass burning also offer potential opportunities but have more limited mitigation potential due to smaller remaining emissions in these categories, or limits on the availability of effective BC control strategies.*
  - Total **mobile source** BC emissions are projected to decline by 86% by 2030 due to regulations already promulgated. BC emissions from mobile diesel engines (including on-road, non road, 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. Substantial future reductions in mobile diesel emissions are anticipated through new engine requirements and diesel retrofit programs**.**
  - 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 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. For wildfires, expanding domestic fire prevention efforts may help to reduce BC emissions.
- **Global:** *The most important BC emissions reduction opportunities globally include residential cookstoves in all regions; brick kilns and coke ovens in Asia; and mobile diesels in all regions. A variety of other opportunities may exist in individual countries or regions.*
  - 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 does 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. Mitigation in this sector represents the area of largest potential public health benefit of any of the sectors considered in this report. 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. Setting BC emissions reductions as a policy priority would drive cookstove efforts toward solutions that achieve this goal. A number of factors point to much greater potential to achieve large-scale success in this sector today.

- The largest **stationary sources** of BC emissions 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 (including potentially light absorbing BrC) 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. Expanded wildfire prevention efforts may help to reduce BC emissions globally. 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.

- **Sensitive Regions:** To address impacts in the Arctic, other assessments have identified the transportation sector (land-based diesel engines and Arctic shipping); residential heating (wood-fired stoves and boilers); and forest, grassland and agricultural burning as primary mitigation opportunities. In the Himalayas, studies have focused on residential cooking; industrial sources (especially coal-fired brick kilns); and transportation, primarily on-road and off-road diesel engines.

19. A variety of other options may also be suitable and cost-effective for reducing BC emissions, but these can only be identified with a tailored assessment that accounts for individual countries' resources and needs.

Some potential sectors of interest for further exploration include agricultural burning, oil and gas flaring, and stationary diesel engines in the Arctic far north.

20. Despite some remaining uncertainties about BC that require further research, currently available scientific and technical information provides a strong foundation for making mitigation decisions to achieve lasting benefits for public health, the environment, and climate.

Chapter 1

Introduction

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 climate change over the long term, BC offers a promising mitigation opportunity to address climate effects in the near-term and to 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 fine particles ($PM_{2.5}$) and is therefore associated with an array of respiratory and cardiovascular health impacts. This makes it ripe for emissions reduction approaches that bring both climate and public health benefits.

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 (IPCC, 2007). 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.

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:

> *"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*. This report has been peer reviewed by a special panel of experts appointed under the Council on Clean Air Compliance Analysis, a Federal Advisory Committee chartered under the Federal Advisory Committee Act (FACA), 5 U.S.C., App 2. The Black Carbon Review Panel concluded that "the draft report is comprehensive and well-written; summarizes relevant scientific literature; and successfully convey[s] a wealth of complex information" (Pope, 2011). This final report reflects numerous additions and improvements suggested by the peer review panel. In addition, the final report reflects input from other federal agencies, many of which have a great deal of expertise in this area and a range of programs relevant to BC. As requested by Congress, EPA has consulted with these other federal agencies while developing this report, and has incorporated material supplied by the agencies into key sections of the Report, including chapters on emissions inventories, health and climate science, and sector-specific mitigation options.

1.1 Key Questions Addressed in this Report

In evaluating the climate impacts and mitigation opportunities for BC, it is essential to recognize from the outset that BC presents a different kind of climate challenge than CO_2 and other long-lived GHGs. BC's short atmospheric lifetime (days to weeks) and heterogeneous distribution around the globe result in regionally concentrated climate impacts. Thus, the location of emissions releases is a critical determinant of BC's impacts, which is not the case for long-lived and more homogeneously distributed GHGs like CO_2. The composition of the total emissions mixture is also key: since many co-emitted pollutants such as sulfur dioxide, oxides of nitrogen, and most organic carbon particles tend to produce a cooling influence on climate, the amount of BC relative to these other constituents being emitted from a source is important. Furthermore, BC is linked to a whole variety of effects beyond warming. These include the darkening of ice and snow, which reduces reflectivity and accelerates melting; changes in the formation and composition of clouds, which affect precipitation; and impacts on human health.

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

1. What is BC, and how does it lead to climate warming?
2. What is the net effect of atmospheric BC on global and regional temperature change in terms of both magnitude and time scale?
3. What is known about the magnitude of BC's effect on snow and ice, and its impacts on precipitation?
4. What is known about BC's contribution to $PM_{2.5}$-related human health impacts and other, non-climate environmental impacts?
5. What kind of real-world BC data exists from monitoring networks and other observational research?
6. How large are U.S. and international emissions of BC currently, which sectors are the main contributors, and how are emissions projected to change in the future?
7. What is the potential value of BC reductions as a component of a broader climate change mitigation program, taking into account both co-pollutant emissions reductions and the public health co-benefits?
8. What specific considerations will determine preferred mitigation strategies in different national and regional contexts?
9. What technologies and approaches are available to address emissions from key sectors, and at what cost?
10. Which mitigation options represent potential top-tier opportunities for key world regions, including the United States?

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

1.2 Other Recent Assessments of BC

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

In the *Integrated Assessment of Black Carbon and Tropospheric Ozone* (UNEP and WMO, 2011a), UNEP/WMO conclude that BC mitigation may offer near-term climate benefits. This study was designed to assess the role of BC and ozone in climate and air quality, and to recommend mitigation measures that could be expected to provide benefits in both the climate and air quality arenas. Out of roughly two thousand potential mitigation measures, the UNEP/WMO analysis has identified a small subset of measures as providing the largest mitigation potential. The *Assessment* finds that full implementation of the targeted measures (which included methane reductions for ozone mitigation, as well as BC reductions) could greatly reduce global mean warming rates over the next few decades. Specifically, the analysis suggests that warming anticipated to occur during the 2030s based on emissions projections could be reduced by half through application of these BC and methane measures. In contrast, even a fairly aggressive strategy to reduce CO_2 would do little to mitigate warming over the next 20-30 years. The UNEP/WMO *Assessment* concludes that while CO_2 measures clearly are the key to mitigating long-term climate change out to 2100, BC and methane measures could reduce warming and slow the rate of change in the next two decades. The *Assessment* also recognizes the substantial benefits to air quality, human health, and world food supplies that would result from reductions in BC and tropospheric ozone.

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

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

The Expert Group concluded that because of the public health benefits of reducing BC, as well as the location of the countries across the Convention regions in relation to the Arctic, the Executive Body should consider taking additional measures to reduce BC. The report included information about key sectors and emphasized the need to develop emissions inventories, ambient monitoring and source measurements in an effort to improve the understanding of adverse effects, efficacy of control measures and the costs and benefits of abatement. Based in part on the findings of the Expert Group, the CLRTAP Executive Body decided to include consideration of BC as a component of PM in their ongoing process of revising the Gothenburg Protocol.[1] This decision marks the first time an international agreement has attempted to address the issue of short lived climate forcers in the context of air pollution policy. Revisions to the Gothenburg Protocol are expected to be completed in 2012.

The Arctic Council Task Force on Short-Lived Climate Forcers was formed following the issuance of the

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[1] The Protocol to Abate Acidification, Eutrophication and Ground-level Ozone was adopted in Gothenburg, Sweden in 1999. See *http://www.unece.org/env/lrtap/multi_h1.html*.

Tromsø Declaration at the Arctic Council Ministerial in April 2009.[2] This declaration formally noted the role that short-lived forcers may play in Arctic climate change, and recognized that reductions of emissions of these compounds and their precursors have the potential to slow the rate of Arctic snow, sea ice, and sheet ice melting in the near term. The Task Force, which is being co-chaired by the United States and Norway, is charged with identifying existing and new measures to reduce emissions of short-lived climate forcers (BC, ozone and methane) and recommending further immediate actions that can be taken. The Task Force focused its initial efforts on BC, and presented a menu of mitigation options to Arctic Nations at the Arctic Council Ministerial in May of 2011 (Arctic Council, 2011). The key findings of this Task Force include:

- Addressing short-lived climate forcers such as black carbon, methane and ozone offers unique opportunities to slow Arctic warming in the near term.
- Black carbon emitted both within and outside of the Arctic region contributes to Arctic warming. Per unit of emissions, sources within Arctic Council nations generally have a greater impact.
- Controls on black carbon sources that reduce human exposure to particulate pollution improve health, and in that regard many measures can be considered no regrets.
- To maximize climate benefits, particulate matter control programs should aim to achieve maximum black carbon reductions.

In making these recommendations, the Task Force recognized that not all measures to control ambient PM necessarily reduce BC, and encouraged countries to consider BC-focused strategies as a complement to existing PM control programs in order to ensure climate as well as health and environmental benefits. Based on its findings, the Task Force recommended that Arctic Council nations take action to reduce BC, and laid out a menu of specific mitigation options in key sectors such as land-based transportation, stationary diesel engines, residential wood combustion, agricultural and forest burning, and shipping. The Task Force has also encouraged Arctic nations to develop and share domestic inventories of BC emissions, which can be used to further define—and refine—global inventories.

The Task Force is collaborating with the Arctic Monitoring and Assessment Programme (AMAP) working group, which established an Expert Group on Short-Lived Climate Forcers to prepare a detailed technical report, *The Impact of Black Carbon on Arctic Climate*. This report (Quinn et al., 2011) contains climate modeling results highlighting the significance of BC emissions sources from different sectors and different regions for the Arctic climate. The AMAP report indicates that emissions both from within the Arctic and from the rest of the world affect Arctic warming, with sources near to or within the Arctic having particularly significant impacts per unit of emissions.

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

1.3 Organization of this Report

This Report is organized into twelve chapters and seven technical appendices. Each of the chapters that follow this *Introduction,* and the appendices, are described briefly below:

Chapter 2 describes how particles, including BC, absorb and scatter light, and identifies the factors that influence the direction and magnitude of their effect on the Earth's climate. The chapter defines "black carbon," describes how BC relates to other types of particles, and discusses how these substances affect climate. Next, the chapter provides detailed information on the range of direct and indirect impacts of BC on global and regional climate. It summarizes available estimates of BC's global and regional radiative forcing and related temperature effects, snow and ice albedo effects, cloud effects, and precipitation effects.

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[2] 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.

The chapter also considers how best to compare the effects of BC to the impacts of other climate forcers, particularly CO_2. It evaluates the applicability of traditional metrics developed for CO_2 to BC, and presents alternative metrics designed specifically for evaluating the climate impacts of short-lived climate forcers like BC.

Chapter 3 outlines EPA's current scientific understanding of the health and non-climate environmental effects of BC. This chapter discusses the large body of scientific evidence regarding the adverse human health impacts of $PM_{2.5}$ in general, and provides a summary of health research related to BC as a component of the overall $PM_{2.5}$ mix. It also describes BC's role in visibility impairment and ecological effects.

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

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

Chapter 6 considers the potential climate and human health benefits of BC emissions reductions. The chapter describes the findings of existing studies on the global and regional benefits of BC mitigation, including specific strategies aimed at reducing emissions from key sectors. The chapter acknowledges the large remaining uncertainties with regard to evaluating the climate benefits of BC mitigation in some sectors, but notes that controls on BC and co-emitted pollutants are generally associated with significant public health benefits, through reductions in $PM_{2.5}$ and its precursors. The chapter also discusses approaches for valuing health and climate impacts.

Chapter 7 provides a framework for evaluating mitigation options. The chapter describes the different considerations, including benefits, costs, technologies, and other factors that affect decisionmaking, and the impact of these considerations on choices among BC mitigation options. It provides some illustrative examples of which mitigation options might be preferred depending on the weight policymakers assign to different factors in the mitigation framework.

Chapters 8-11 describe existing control programs and technologies that have been demonstrated to be effective in reducing BC emissions from source categories of regional and/or global importance. These include **Mobile Sources** (Chapter 8), **Stationary Sources** (Chapter 9), **Residential Heating and Cooking** (Chapter 10), and **Open Biomass Burning** (Chapter 11). For each sector, the chapter recaps current and projected emissions estimates (accounting for control programs currently in place but not yet implemented), describes key control technologies and other mitigation strategies that can help control BC emissions from specific source types, and provides available information regarding control costs. The chapters also discuss how alternative strategies, such as changes in land-use policy or energy systems, could impact emissions from the sectors. Control options, costs, and known or potential barriers to mitigation are described separately for U.S. domestic emissions and international emissions. In some cases, there are considerable differences in mitigation approaches, cost, and feasibility between the United States and other countries. Also, there are gaps in available information on these factors for many sectors.

The conclusion, **Chapter 12**, focuses on identifying important BC mitigation opportunities for the United States and other world regions. Drawing on earlier chapters and the findings of other recent assessments, this chapter clarifies some of the key mitigation options that can clearly be expected to provide near-term climate and health benefits. The chapter acknowledges the diversity of approaches for BC mitigation and the need to tailor mitigation strategies to specific national and local contexts. The chapter also identifies key gaps in current scientific understanding, and provides a list of high-priority research needs. Additional research in these areas is essential for improving the current scientific understanding of the impact of BC and other light-absorbing particles on climate, and for estimating the full impact of mitigation approaches in different sectors and regions on both climate and public health. These research needs may stimulate further work on BC by EPA and other organizations.

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

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

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

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

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

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

Appendix 7 discusses a variety of research needs related to BC. Though the highest priority research needs are discussed in Chapter 12, this appendix provides more detail regarding specific gaps in the currently available information on BC that are important both from a scientific perspective and for informing BC mitigation decisions.

Chapter 2

Black Carbon and Its 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 the incomplete combustion of fossil fuels, biofuels, and biomass.
  - BC can be defined specifically as a solid form of mostly pure carbon that absorbs solar radiation (light) at all wavelengths. BC is the most effective form of PM, by mass, at absorbing solar energy. BC is a major component of "soot", a complex light-absorbing mixture that also contains organic carbon (OC).
  - Other carbon-based PM may also be light-absorbing, particularly brown carbon (BrC), which is a class of OC compounds that absorb light within the visible and ultraviolet range of solar radiation and that can exist within the same particles as BC. The net contribution of BrC to climate is presently unknown.
- BC is always emitted with other particles and gases, such as sulfur dioxide (SO_2), nitrogen oxides (NO_x), and OC. Some of these co-emitted pollutants exert a cooling effect on climate. Therefore, estimates of the net effect of BC emissions sources on climate should include the offsetting effects of these co-emitted pollutants.
- 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 (days to weeks) and the mechanisms by which it affects climate distinguish it from long-lived greenhouse gases (GHGs) like carbon dioxide (CO_2).
  - Targeted strategies to reduce BC emissions can be expected to provide climate responses within the next several decades. In contrast, reductions in GHG emissions will take longer to influence atmospheric concentrations and will have less impact on climate on a short timescale, but deep reductions in GHG emissions are necessary for limiting climate change 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 emissions category.
- BC influences climate through multiple mechanisms:
  - *Direct effect*: BC absorbs both incoming and outgoing radiation of all wavelengths, which contributes to warming of the atmosphere and dimming at the surface. 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 (albedo), thereby increasing absorption and accelerating melting. GHGs do not directly affect the Earth's albedo.
  - *Other effects*: BC also alters the properties and distribution of clouds, affecting cloud reflectivity and lifetime ("indirect effects"), stability ("semi-direct effect"), and precipitation. 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. Based on the studies surveyed for this report, the direct and snow/ice albedo effects of BC together likely contribute more to current warming than any GHG other than CO_2 and methane (CH_4).
- The climate effects of BC via interaction with clouds are more uncertain, and their net climate influence is not yet clear.

  - All aerosols (including BC) affect climate indirectly by changing the reflectivity and lifetime of clouds. The net indirect effect of all aerosols combined is very uncertain but is thought to have a net cooling influence. The contribution of BC to this cooling has not been quantified.
  - BC has additional effects on clouds—including changes to cloud stability and enhanced precipitation from colder clouds—that can lead to either warming or cooling.
  - The net climate influence of these cloud interaction effects of BC is not yet clear. There is inconsistency among reported observational and modeling results, and many studies do not provide quantitative estimates of cloud impacts.
- The sign and magnitude of the net climate forcing from BC emissions are not fully known at present, largely due to remaining uncertainties regarding the effects of BC on clouds. Though most estimates indicate that BC has a net warming effect, a net cooling influence cannot be ruled out. Further research and quantitative assessment are needed to reduce remaining uncertainties.
- Regional climate impacts of BC are highly variable, and sensitive regions such as the Arctic and the Himalayas are particularly vulnerable to the warming and melting effects of BC. Estimates of snow and ice albedo forcing in key regions also exceed global averages.
- BC also contributes to the formation of Atmospheric Brown Clouds (ABCs) and resultant changes in the pattern and intensity of precipitation.
- Due in large part to the difference in lifetime between BC and CO_2, the relative weight given to BC as compared to CO_2 (or other climate forcers) in terms of its impact on climate is very sensitive to the formulation of the metric used to make the comparison.
- There is currently no single metric 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 global warming potential (GWP) and global temperature potential (GTP). 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 specific forcing pulse (SFP) and the surface temperature response per unit continuous emission (STRE).
- There is significant controversy regarding the use of metrics for direct comparisons between the long-lived GHGs and the short-lived particles for policy purposes; however, these comparisons are less controversial when used for illustrative purposes.
  - There are a number of factors that should be considered when deciding which metric to use, or whether comparisons between BC and CO_2 are useful given a particular policy question. These include: the time scale (e.g., 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.
  - If the primary goal is reducing long-term change, then a metric like a 100-year GWP or GTP would be more appropriate. If the rate of near-term climate change and near-term damages to sensitive regions like the Arctic are also a consideration, there is no single existing metric that adequately weights impacts over both time periods, and a multi-metric approach may be more appropriate than developing a single metric that attempts to serve all purposes.

2.2 Introduction

There is a general consensus within the scientific community that BC is contributing to climate change at both the global and regional levels. Like CO_2, BC is produced through the burning of carbon-based fuels, including fossil fuels, biofuels and biomass. BC is part of the mix of PM released during the incomplete combustion of these fuels. BC influences climate by absorbing sunlight when suspended in the atmosphere or when deposited on the Earth's surface. The energy absorbed by BC is then released as heat and contributes to atmospheric warming and the accelerated melting of ice and snow. In addition, BC is capable of altering other atmospheric processes, such as cloud formation and evaporation, and precipitation patterns.

The strong absorption, short atmospheric lifetime, and other characteristics of BC make its impacts on climate different from those of long-lived GHGs like CO_2 (see Figure 2-1). Because BC is involved in complex atmospheric physical and chemical processes, it is difficult to disentangle all associated impacts and to evaluate its net effect on climate. In addition, the combustion processes that produce BC also produce other pollutants, such as SO_2, NO_x, and OC. Since many of these compounds have a cooling effect, BC's impacts are mixed with—and sometimes offset by—these co-emitted substances. This must be considered when evaluating the net effect of emissions sources.

This chapter focuses on how and to what extent BC influences the Earth's climate. Specifically, this chapter discusses approaches for defining BC and other light-absorbing particles, highlights the differences between BC and GHGs, and addresses the role of co-emitted pollutants. Further, this chapter summarizes recent scientific findings regarding the processes by which BC affects climate and the magnitude of BC's impacts on global and

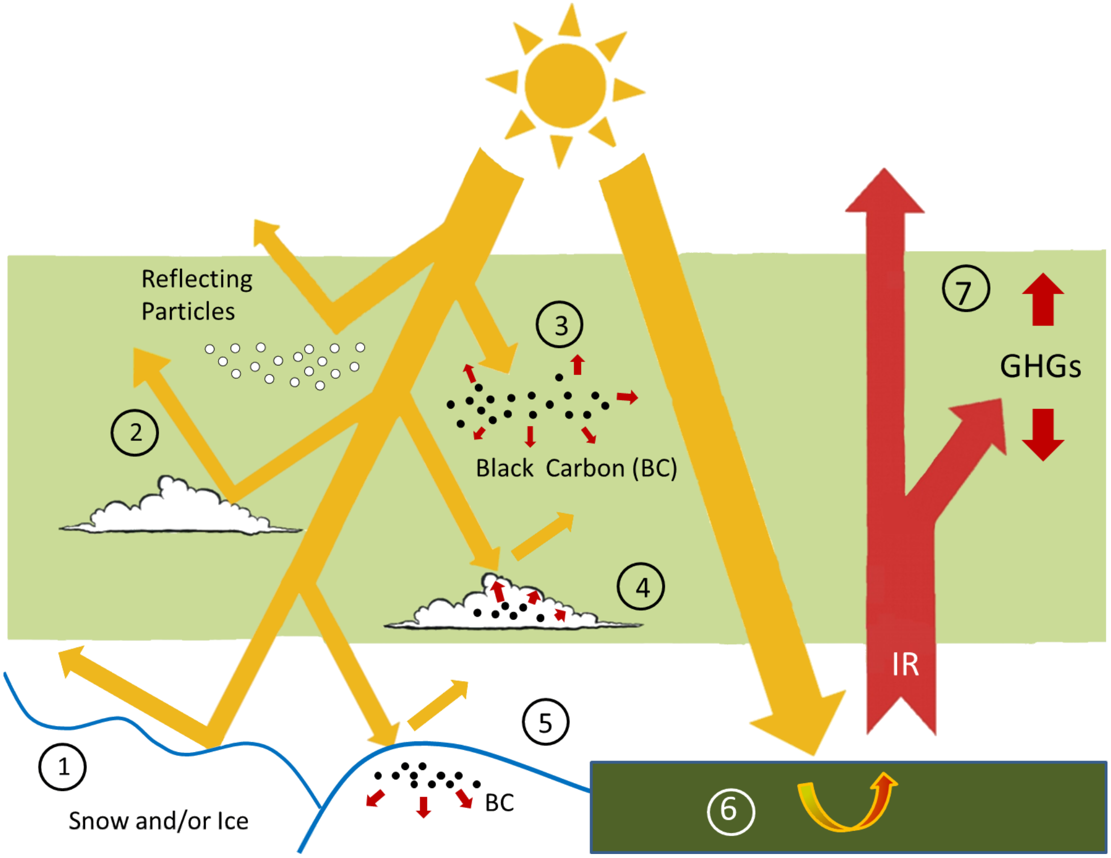

Figure 2-1. Effects of BC on Climate, as Compared to GHGs. (Source: U.S. EPA)

1. Sunlight that penetrates to the Earth's surface reflects off bright surfaces, especially snow and ice.
2. Clean clouds and non-light-absorbing (transparent) particles scatter or reflect sunlight, reducing the amount of solar energy that is absorbed by the surface.
3. BC suspended in the atmosphere absorbs some incoming solar radiation, heating the atmosphere.
4. Clouds containing BC inclusions in drops and BC interstitially between drops can absorb some incoming solar radiation, reducing the quantity that is reflected. Clouds warmed by the absorbed energy have shorter atmospheric lifetimes and may be less likely to precipitate compared to clean clouds.
5. BC deposited on snow and/or ice absorbs some of the sunlight that would ordinarily be reflected by clean snow/ice, and increases the rate of melting.
6. Most solar radiation is absorbed by the Earth's surface and warms it. Part of the absorbed energy is converted into infrared radiation that is emitted into the atmosphere and back into space.
7. Most of this infrared radiation passes through the atmosphere, but some is absorbed by GHG molecules like CO_2, methane, ozone and others. These gases re-emit the absorbed radiation, with half returning to the Earth's surface. This GHG effect warms the Earth's surface and the lower atmosphere.

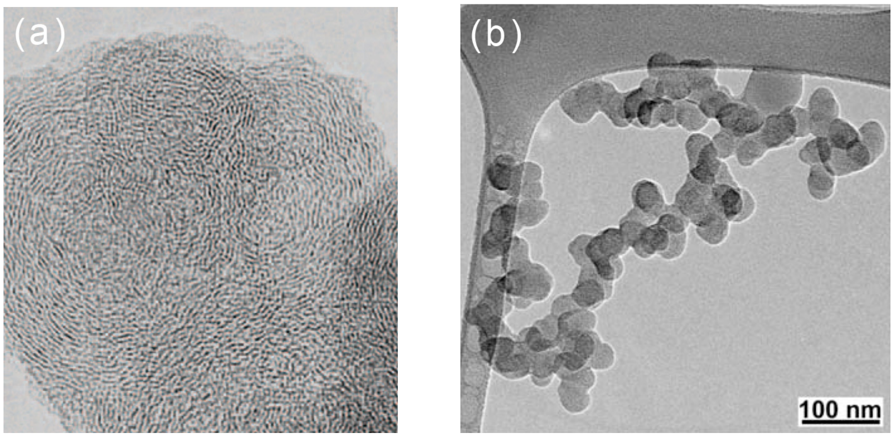

Figure 2-2. BC Images. (a) High resolution transmission electron microscopy (TEM) image of a BC spherule (Pósfai and Buseck, 2010). (b) TEM image of a representative soot particle. Freshly emitted soot particles are aggregates of soot spherules (Alexander et al., 2008).

regional climate, highlighting the effect of BC in sensitive regions such as the Arctic and other snow- and ice-covered regions. The chapter discusses the significant remaining uncertainties about BC's effects on climate, and the need for further research in key areas. The final section of this chapter introduces several metrics that can be used to quantify the climate impacts of BC and other pollutants (such as CO_2 and CH_4) relative to a common baseline. The section highlights the fact that there is no one "best" metric for comparing BC to other pollutants and that the utility of each metric depends on the policy objective.

2.3 Defining Black Carbon and Other Light-Absorbing PM

All PM in the atmosphere can affect the Earth's climate by absorbing and scattering light. Sunlight absorbed by PM increases the energy in the Earth's climate system, leading to climate warming. Conversely, light scattered by PM generally leads to increased reflection of light back to space, leading to climate cooling (Charlson, 1992; Moosmüller et al., 2009; Seinfeld and Pandis, 2006; Forster et al., 2007). Carbonaceous PM, a class of material found in primary and secondary particles, has typically been divided into two classes: BC and OC (see text box on "Terminology"). Neither BC nor OC has a precise chemical definition. The term BC generally includes the solid forms of carbon emitted by incomplete combustion while OC refers to the complex mixtures of different carbon compounds found in both primary and secondary carbonaceous particles. Carbonaceous PM includes an array of organic compounds that, along with BC, possess radiative properties that fall along a continuum from light-absorbing to light-scattering. Both BC and OC are part of the broader category of suspended particles and gases known as aerosols, all of which have light-absorption and light-scattering properties.

In this report, BC is defined as the carbonaceous component of PM that absorbs all wavelengths of solar radiation.[1] For this reason, among the many possible forms of PM, BC absorbs the most solar energy. Per unit of mass in the atmosphere, BC can absorb a million times more energy than CO_2 (Bond and Sun, 2005), making it a significant climate warming pollutant in regions affected by combustion emissions.

BC forms during combustion, and is emitted when there is insufficient oxygen and heat available for the combustion process to burn the fuel completely (see text box on "Products of Incomplete Combustion"). BC originates as tiny spherules, ranging in size from 0.001 to 0.005 micrometers (μm), which aggregate to form particles of larger sizes (0.1 to 1 μm) (Figure 2-2). Particles in this range are similar in size to the wavelengths emitted by the sun, making them especially effective in scattering or absorbing these wavelengths (Horvath, 1993). The characteristic particle size range in which fresh BC is emitted also makes it an important constituent of the ultrafine (<100 nanometers (nm)) subclass of $PM_{2.5}$.

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[1] The spectrum of solar radiation striking Earth's atmosphere ranges from high energy UV with wavelengths shorter than 280 nm down to infrared radiation as long as 1000 nm. However, UV wavelengths shorter than 280 nm are substantially absorbed by the stratosphere. For the purposes of this discussion, the term "all wavelengths of solar radiation" corresponds to the solar wavelengths present in the troposphere (e.g., in the range 280 - 2500 nm).

Terminology

Black carbon (BC) is a solid form of mostly pure carbon that absorbs solar radiation (light) at all wavelengths. BC is the most effective form of PM, by mass, at absorbing solar energy, and is produced by incomplete combustion.

Organic carbon (OC) generally refers to the mix of compounds containing carbon bound with other elements like hydrogen or oxygen. OC may be a product of incomplete combustion, or formed through the oxidation of VOCs in the atmosphere.[2] Both primary and secondary OC possess radiative properties that fall along a continuum from light-absorbing to light-scattering.

Brown carbon (BrC) refers to a class of OC compounds that absorb ultraviolet (UV) and visible solar radiation. Like BC, BrC is a product of incomplete combustion.[3]

Carbonaceous PM includes BC and OC. Primary combustion particles are largely composed of these materials.

Light absorbing carbon (LAC) consists of BC plus BrC.

Soot, a complex mixture of mostly BC and OC, is the primary light-absorbing pollutant emitted by the incomplete combustion of fossil fuels, biofuels, and biomass.

BC is emitted directly from sources, making it a form of *primary* PM. This distinguishes it from *secondary* PM such as sulfates, nitrates and some forms of OC that are formed in the atmosphere from gaseous precursors like SO_2, NO_x and volatile organic compounds (VOCs).

When BC is emitted directly from sources as a result of the incomplete combustion of fossil fuels, biofuels and biomass, it is part of a complex particle mixture called *soot* which primarily consists of BC and OC. This mixture is the light-absorbing component of these air pollution emissions.

Soot mixtures can vary in composition, having different ratios of OC to BC,[2] and usually include inorganic materials such as metals and sulfates. For example, the average OC:BC ratio among global sources of diesel exhaust is approximately 1:1. For biofuel burning, the ratio is approximately 4:1 and for biomass burning it is approximately 9:1 (Lamarque et al., 2010). As expected, very dark soot indicates the presence of low OC:BC ratios. As the OC fraction begins to dominate, the color of the soot mixture shifts to brown and yellow. A brown soot sample is dominated by a form of OC known, as might be expected, as "brown carbon" (BrC).[3] BrC, another product of incomplete combustion, absorbs portions of the visible spectrum, but is less effective in capturing solar energy than BC (Alexander et al., 2008; Novakov and Corrigan, 1995b). The mixture shifts in color toward yellow when the emissions source is no longer producing BC and BrC. Yellow carbon, another form of OC, is also able to absorb visible radiation, but to a lesser extent than BrC (Bond, 2001; Gelencsér, 2004; Andreae and Gelencsér, 2006). Figure 2-3 illustrates the variance in soot composition resulting from different fuels and stages of fuel combustion. The stages of fuel combustion responsible for producing BC and the various forms of OC observed in soot are described in the text box on this page.

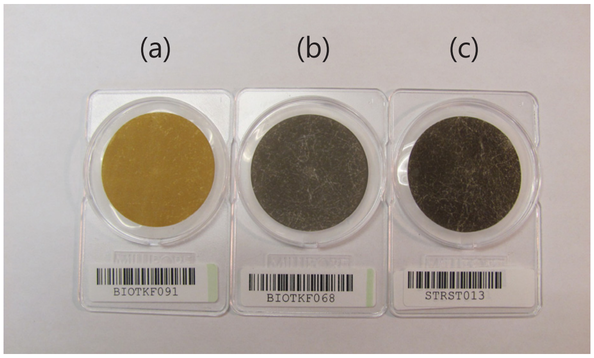

Figure 2-3. Representative Examples of Filter Samples Collected from Different Sources, including: (a) Smoldering Biomass, (b) Flaming Biomass, and (c) Diesel Exhaust. (Photo courtesy of Desert Research Institute)

In general, light absorption by carbonaceous PM can be described as a continuum from light-absorbing to light-scattering with BC at one end, most OC at the other, and BrC occupying the partially absorbing

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[2] When referring to emissions and measurements, OC denotes the total carbon associated with the organic compounds, while organic mass (OM) refers to the mass of the entire carbonaceous material, including hydrogen and oxygen. Similarly, measurements and emissions reported as elemental carbon (EC) denote the non-organic, refractory portion of the total carbon and is an indicator for BC. For more details, see Chapter 5 and Appendix 1.

[3] During solid fuel combustion, BrC forms during the preheating (pyrolysis) phase, and during both flaming and smoldering combustion. The light-colored smoke characteristic of the pyrolosis and smoldering combustion phases is primarily OC, including both BrC and other forms of OC, and does not include soot. Secondary BrC can also form during reactions, similar to polymerization, that take place in primary particles as emissions plumes age. BrC of this type is known as "humic-like substances" (HULIS).

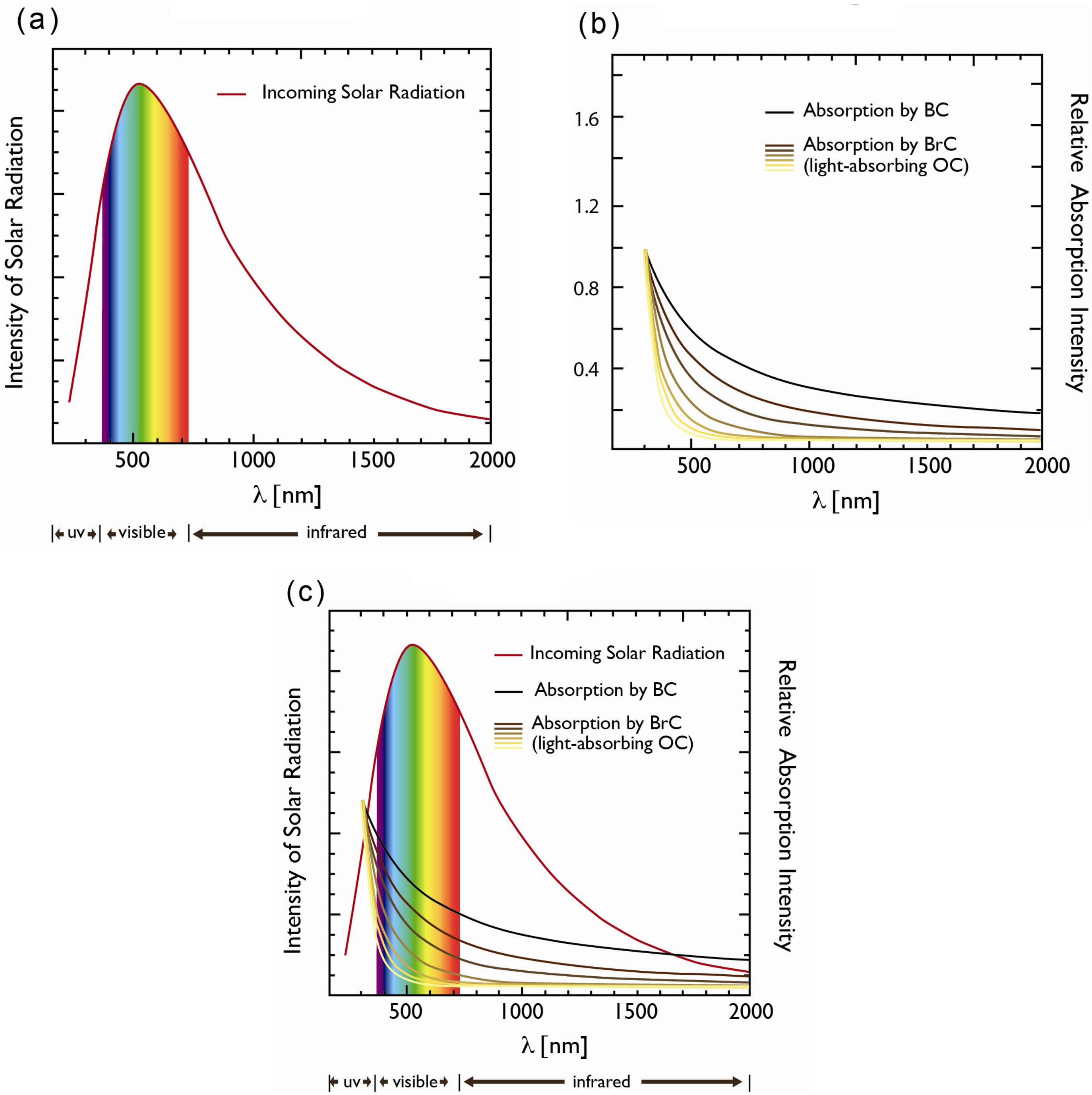

Figure 2-4. Light Absorption by BC, BrC, and Ambient Mixtures. (Source: U.S. EPA)

a. The radiation wavelengths emitted by the Sun that reach the Earth begin around 280 nm (UV-B), peak in the mid-visible range, and reach out past 2000 nm (Infrared). The shorter the wavelength, the higher its energy.

b. The extent to which BC and BrC absorb solar radiation depends upon the wavelength of incoming light. This plot shows idealized examples of those dependencies, assuming that both forms of carbon absorb to the same extent at 280 nm. BC is more effective in absorbing solar energy across the entire solar spectrum than any form of BrC. BrC increasingly declines in its capacity to absorbing longer wavelengths as the mixture moves from larger light-absorbing compounds to smaller compounds, as indicated by the change in color from dark brown to yellow.

c. When the light absorption curves for BC and BrC are superimposed upon the solar spectrum, the significance of the different absorption efficiencies between BC and BrC becomes evident. BC will, all else being equal, absorb more total solar radiation than BrC. In practice, the mass ratios of BC and BrC, along with the specific composition of the BrC mixture, determine the degree to which each form of carbon absorbs the solar energy penetrating the emissions plume.

Products of Incomplete Combustion

Most combustion occurring on Earth (both anthropogenic and natural) involves carbon-based fuels, including fossil fuels (e.g., coal, oil, and natural gas), biomass (e.g., wood and crop residues), and biofuels (e.g., ethanol). Complete combustion of a carbon-based fuel means all carbon has been converted to CO_2. Once ignited, the fuel must be well mixed with oxygen at a sustained high temperature for this to occur. Incomplete combustion emits various materials in both gas and particle form, depending on the combustion conditions (e.g., oxygen availability, flame temperatures, and fuel moisture) and the type of fuel burned (e.g., gas, liquid, or solid). The PM emissions are generically known as soot.

BC is formed during the **flaming** phase of the combustion process. The quantity of BC emitted depends largely on combustion conditions. If there is sufficient oxygen and high temperatures, the soot will be completely oxidized, and BC emissions will be minimal. To increase fuel efficiency and reduce soot emissions, closed combustion systems (e.g., furnaces, combustors, reactors, boilers, and engines) are engineered to increase the mixing of air with the fuel and are insulated to ensure temperatures remain high. Open and uncontrolled burning produces large quantities of BC because oxygen availability and temperatures within the fire can vary widely.

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

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

Thermal breakdown of high molecular weight fuels, known as **pyrolysis,** produces a wide array of BrC compounds. When sustained, pyrolysis converts solid fuels such as coal and biomass into char, while releasing volatile gases that can fuel flaming combustion. There is also a non-flaming process known as **smoldering** that is a slower, cooler form of combustion which occurs as oxygen directly attacks the surface of heated solid fuel. The smoke that appears is light-colored, consisting of a variety of organic particles composed of BrC. BC does not form under these conditions, since temperatures are too low to sustain flaming combustion. During open or uncontrolled burning of solid fuels, all stages of the burning process—pyrolysis, smoldering, and flaming combustion—occur simultaneously, in different parts of the fuel pile, resulting in emissions of both BC and BrC.

middle ground.[4] The characteristic light absorption spectra vary significantly among individual BrC compounds, but are almost entirely limited to the UV to visible portion of the solar spectrum (Jacobson, 1999). Mixtures of these compounds range in color from yellow to brown, roughly corresponding to the average molecular weight of the light-absorbing compounds present (see Figure 2-4). Emissions dominated by smaller BrC compounds will appear yellow, while plumes containing high concentrations of heavy polycyclic aromatic hydrocarbons (PAHs) will appear darker brown.

BC and different mixtures of BrC show different patterns of light absorption versus wavelength. Light absorption by BC tends to decline more slowly with increasing wavelength, while the falloff in absorption by BrC is always faster than that of BC, tending to vary depending on the composition of the BrC mixture. Absorption by the aforementioned yellow BrC mixtures, dominated by lower molecular weight compounds, falls off very quickly. Dark brown BrC mixtures containing large PAHs or HULIS (see footnote 3) continue absorbing to a significant degree across the solar spectrum.

BrC typically accompanies BC in soot particles. However, independent BrC particles may form in the uncontrolled burning conditions typical of biomass burning, or during inefficient combustion of biofuels. These independent particles, labeled "tar

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[4] 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., 2008a).

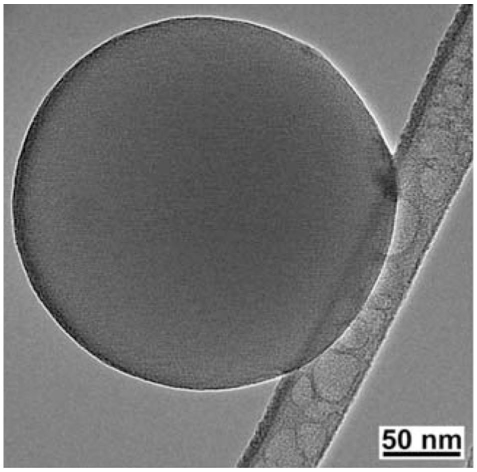

Figure 2-5. TEM Image of a BrC Particle. These particles are referred to as "tar balls" or "carbon spheres" in the literature. (Alexander et al., 2008)

Figure 2-6. Coarse Urban PM (Diameter > 2.5 microns) with a Black Surface Coating, shown on a Filter from a Harvard Impactor Run Approximately 100 feet Above Street Level in Boston. (Photo courtesy of NESCAUM)

balls" or "carbon spheres," range in diameter from 0.03 and 0.5 μm (Pósfai et al., 2004; Chakrabarty et al., 2010; Adachi and Buseck, 2008). Alexander et al. (2008), in their study of East Asian pollution plumes, found numerous examples of these particles (see Figure 2-5). They noted that, on the basis of their statistical analysis, these particles are very abundant.

Fuel type and burning conditions determine the quantity of BrC produced from a particular combustion source. Some sources, such as open biomass burning, can produce substantially more BrC than BC. On the basis of quantity alone, BrC may lead to greater total solar energy absorbed than BC for those sources, despite the fact that BrC absorbs less energy than BC per unit mass.

Until recently, most measurements of light absorbing carbonaceous PM focused on BC, classifying all other carbon as OC. Growing awareness of the presence of light-absorbing BrC in biomass emissions has prompted the recent suggestion from the scientific community that absorption by BrC should be explicitly included in the evaluation of the role of LAC in climate warming, by accounting for emissions of BrC, along its known radiative properties, in climate modeling studies. Unfortunately, those data do not yet exist.

Combustion emissions, depending upon the purity of the fuel and burning conditions, can also contain a number of inorganic pollutants. For example, combustion of high-sulfur coal is a well-known source of SO_2 and sulfuric acid emissions. Mineral salts, such as potassium chloride, are emitted during biomass burning due to the presence of potassium in wood and other plant materials. Localized regions of very high heat within a combustion mixture produce NO_x, which further reacts to form nitrates. These co-pollutants in a combustion plume can subsequently form light-scattering particles. The light scattered by these particles may offset the warming effect due to the light absorbed by BC and BrC in the emissions plume. This effect is discussed further later in this chapter in sections 2.5 and 2.6.1.5.

The focus of the discussion up to this point has been on the light absorbing properties of carbonaceous PM (i.e., of the classes of material that appear in particle form in the atmosphere). The chemical composition and physical structure of *individual particles* are also factors determining the overall radiative properties of an emissions plume. Size and morphology influence the efficiency of light absorption by a particle. Particle size and chemical composition change as a fresh emissions plume begins mixing with the ambient atmosphere. Light absorption by BC can be enhanced by 30% to 100% when chemical processing in the atmosphere creates a transparent coating[5] on the surface of the particle (Fuller et al., 1999; Shiraiwa et al., 2009; Bond et al., 2006a). These effects and the role they play in the overall radiative effect of a combustion plume on regional climate are discussed in section 2.5.

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[5] Coatings effective in enhancing light absorption by BC include sulfuric acid from the oxidation of SO_2, water from cloud-processing and secondary OC that does not absorb solar radiation.

There is also a significant BC component in (or rather on) coarse particles ($PM_{10-2.5}$ and larger), especially in urban areas where coarse-mode particles (such as from re-entrained road dust) are often coated with BC, as shown in Figure 2-6. This coarse urban PM is black, not earth-colored, and likely results from a BC surface coating of coarse mode particles, rather than from a uniform BC composition. While coarse particles will have a very limited effect on climate, they represent a means of human exposure to BC. Chapter 3 discusses the currently understood effects of coarse particles and BC on human health.

BC has been studied as a component of soils and sediments, in addition to its role in air pollution and climate. It plays an important role in various biological, geochemical processes, and has been used as a marker for local vegetation fire histories (Schmidt and Noack, 2000). BC has also been the subject of intense study in the combustion science and engineering fields (Frenklach, 2002). Given the number of distinct scientific disciplines that have studied BC, and the different chemical and environmental contexts in which it appears, many different measurement and estimation approaches exist in the literature, each with their own operational definitions of BC. In the atmospheric sciences alone, the terms "graphitic carbon", "apparent elemental carbon" (ECa), "equivalent black carbon" (BCe), "light absorbing carbon", "carbon black", "soot" and "black smoke" are used interchangeably with or as surrogates for BC materials (Bond and Bergstrom, 2006; Andreae and Gelencsér, 2006).

Although these commonly used terms are not strictly equivalent, we believe that the validity of our analyses and conclusions are not materially compromised by our adopting the convention of using surrogate measurements for BC and soot. EPA has traditionally used surrogate or indicator measurements for many pollutants, including $PM_{2.5}$ whose current regulatory characterization by EPA is based on stable, historically available, consistent and reproducible measurements (Watson et al., 1995). For this report, we believe that BC or elemental carbon (EC) measurements are the best available indicators of BC and soot as these particles are only directly emitted from incomplete combustion, whereas OC can be derived from several sources (e.g., pollens, spores, condensed vapors, secondary aerosols). Nevertheless, the most commonly used measurements may not fully capture the light absorption by BrC, and thus current emissions estimates (and observations on which they are based) may underestimate the positive radiative forcing associated with these particles. The connections between highly correlated BC and EC measurements and their physical properties are discussed further in Chapter 5.

For purposes of regional air quality management (e.g., human health studies related to air quality, the evaluation of modeled estimates, and the attribution of emissions to sources), BC is measured as a constituent of ambient $PM_{2.5}$, and is expressed in units of mass. Moreover, BC emissions inventories, as discussed in Chapter 4, are also generally expressed in mass units (e.g., tons/year).

Thus, light-absorption measurements are often converted into estimates of carbon mass. This practice, however, may also contribute to some of the uncertainty in reported ambient concentrations and emissions estimates. Due to reliance on these mass-based indicators, BC is frequently labeled EC due to the long-standing use of carbon measurement methods from which the air quality and emissions estimates were derived. This issue is also evident in health studies that sometimes make a distinction between BC and the measurement on which it is based.

These issues are discussed more fully in Chapter 5 and Appendix 1, which provide a brief description of the various BC and LAC absorption- and mass-related measurement approaches.

2.4 Key Attributes of BC and Comparisons to GHGs

The net impact of BC on climate depends on a number of other factors in addition to its powerful light-absorption capacity. These include atmospheric lifetime, the geographic location of emissions, altitude, interactions with clouds, the presence of co-emitted pollutants, and the influence of aging and mixing processes in the atmosphere. In many of these aspects, BC differs substantially from long-lived GHGs, as summarized in Table 2-1. These differences have implications for how BC influences climate and the climate benefits of BC mitigation as compared to CO_2 mitigation. Each of these dimensions is explored further below.

Particles in general have relatively **short atmospheric lifetimes** in comparison to GHGs. Particles of any type, including BC, are removed from the atmosphere within days to weeks by precipitation and/or dry deposition to surfaces. This short atmospheric lifetime curtails their total contribution to the Earth's energy balance, even for those particles like BC that have strong absorptive capacity. The efficiency with which particles are removed is influenced by their size and chemical

Table 2-1. Comparison of BC to CO_2 on the Basis of Key Properties that Influence the Climate. (Source: U.S. EPA)

Property	BC	CO_2
Atmospheric lifetime	Days to weeks	Up to millennia[e]
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	+0.34 to 1.0 W m^{-2} direct forcing[a] +0.05 W m^{-2} (snow/ice albedo forcing)[b] ± ? (cloud interactions)[c] Net effect: uncertain, but likely warming	+1.66 (±0.17) W m^{-2} (f)
Cloud interactions	Multiple cloud interactions that can lead to warming or cooling (typically cooling), as well as effects on precipitation	Increases cloud droplet acidity
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 third largest contributor (after CO_2 and CH_4), but large uncertainty[d]	Largest contributor
Dimming	Contributes to surface dimming	No direct effects on surface dimming
Acidification/fertilization	No ocean acidification/fertilization effects	Main contributor to ocean acidification and fertilization

[a] UNEP and WMO (2011a) estimate narrower central range of +0.3 to +0.6 W m^{-2}.

[b] Some adjustment to this value may be appropriate to account for the greater warming efficacy of BC deposited on snow and ice. UNEP and WMO (2011a) suggest a range of +0.05 to +0.25 W m^{-2} for BC snow/ice albedo forcing (adjusted for efficacy).

[c] Values are highly uncertain. The IPCC estimated that the cloud albedo effect of all aerosols combined was -0.7 W m^{-2}, but did not include other cloud effects and did not estimate the albedo effect of BC alone. UNEP and WMO (2011a) provided a central forcing estimate for all of the cloud impacts of BC of -0.4 to +0.4 W m^{-2}.

[d] Based on the IPCC forcing estimates (shown in Figure 2-10), the central estimate of the UNEP assessment (UNEP and WMO, 2011), and this report's assessed range of BC forcing (from Figure 2-11), it is likely that the net BC forcing will be less than that of CH_4 but it is possible that BC might be the second largest contributor to warming, depending on uncertainties in the direct and indirect contributions of BC to warming.

[e] The lifetime of CO_2 is more complicated than for most other GHGs. The carbon in CO_2 cycles between the atmosphere, oceans, ecosystems, soil, and sediments. Carbon added to the carbon cycle is removed very slowly (over thousands of years) through processes such as weathering and calcium carbonate formation (Archer et al., 2009). However, even if it is not removed from the carbon cycle, carbon added to the atmosphere can also cycle to other media: approximately three quarters of the added carbon will, over a time scale of decades or centuries, move out of the atmosphere into the ecosystem or oceans.

[f] Forster et al. (2007).

composition. For example, atmospheric aging can increase the size of a particle or alter its chemical composition in a way that makes it an efficient nucleus for cloud droplet formation, facilitating its removal by precipitation.

By contrast, GHGs have longer atmospheric lifetimes. This enables them to become well mixed in the atmosphere and to continue to absorb energy over many decades or centuries. Gases such as nitrous oxide (N_2O), CH_4, or hydrofluorocarbons (HFCs) have lifetimes that range from as short as a year for some of the HFCs to as long as 50,000 years for tetrafluoromethane (CF_4), a perfluorocarbon (Forster et al., 2007).[6] The carbon in CO_2 cycles between the atmosphere, oceans, ecosystems, soil, and sediments; therefore, CO_2 does not have a single defined lifetime. Computer models have indicated that about half of an emissions pulse of

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[6] Several months are required in order for a gas to mix throughout a hemisphere, and one to two years are required for a gas to become well-mixed globally. Some gases, such as CH_4, are well-mixed but are included in the category of "short-lived climate forcer" because a decadal lifetime has different implications for mitigation decisions than the lifetimes of a century or more for many other GHGs. On the other hand, ozone and water vapor have short lifetimes and, like black carbon and other aerosols, are not well-mixed in the atmosphere.

CO_2 will disappear within 30 years, 30% within a few centuries, and the last 20% may remain in the atmosphere for thousands of years (Denman et al., 2007).

BC's short atmospheric lifetime means that atmospheric concentrations are highest near significant emissions sources and during time periods and seasons of emissions releases. This **high spatial and temporal variability** affects BC's impacts on climate. BC is a regional pollutant. CO_2 and other GHGs with lifetimes longer than a year are global pollutants with relatively uniform concentrations around the globe. It is generally assumed that CO_2 and other well-mixed GHGs have essentially the same effect on climate regardless of the location or season of emissions. The same is not true for BC.

Geographic location and altitude are important determinants of the impact of BC on climate. Fine combustion particles including BC can be transported up to thousands of miles from sources. Particles have a greater effect on the net absorption of solar radiation by the atmosphere when they are emitted or transported over light-colored, reflective (i.e., high "albedo") surfaces such as ice, snow, and deserts. In the absence of PM, a high percentage of sunlight would reflect off these surfaces and return to space. Therefore, any absorption of either incoming or reflected light by PM above these surfaces is more likely to lead to warming than absorption of light by PM above darker surfaces. Even PM that is typically classified as reflecting can darken these bright surfaces and contribute to warming (Quinn et al., 2011). This mechanism explains why studies have found the effects of BC to be magnified in the Arctic and other alpine regions, as discussed in sections 2.6.4 and 2.6.5. In addition, the net radiative effects of BC can be sensitive to altitude. A modeling study by Ban-Weiss et al. (2011) suggests that while BC at low altitudes (where most BC is indeed located) warms the surface considerably, BC at stratospheric or upper-tropospheric altitudes may decrease surface temperature. In addition, as with particles suspended above a bright desert or glacier, particles suspended above bright cumulus clouds can absorb both incoming and outgoing solar radiation, increasing the net radiative effect of the light absorbing particle. When suspended between cloud layers or beneath a cloud, the particle may be shielded from incoming light, therefore lessening its potential radiative impact (Schulz et al., 2006).

Other key distinguishing features of BC include the wide **range of mechanisms** through which it influences climate and its association with other adverse, non-climate related **public health and welfare effects**. In addition to the direct radiative forcing characteristic of both BC and GHGs, BC has significant interactions with clouds that can result in both warming and cooling effects. It can also cause melting and warming via deposition to snow and ice. BC and other particles are also directly associated with a host of other environmental effects, such as changes in precipitation patterns and surface dimming. All of these effects are discussed in greater detail later in this chapter. GHGs, on the other hand, influence climate mainly through direct radiative forcing effects. GHGs do not directly interact with clouds, snow and ice, though the warming of the atmosphere due to GHGs does influence cloud formation, snow melt, and many other climate properties. In addition, CO_2 has a fertilization effect on plants and an acidification effect in the ocean, and CH_4 emissions lead to increased ozone concentrations and changes in the lifetime of other atmospheric pollutants. Finally, as a constituent of $PM_{2.5}$, BC is directly linked to a range of public health impacts (see Chapter 3). This, too, distinguishes it from long-lived GHGs, which affect public health and welfare primarily via climate change effects.

An important implication of BC's strong absorptive capacity, coupled with its short atmospheric lifetime, is that when emissions of BC are reduced, atmospheric concentrations of BC will decrease immediately and the climate, in turn, will respond relatively quickly. The **potential for near-term climate responses** (within a decade) is one of the strongest drivers of the current scientific interest in BC. Mitigation efforts that reduce BC emissions can halt the effects of BC on temperature, snow and ice, and precipitation almost immediately. This means that reductions of BC may have an immediate and important benefit in slowing the near-term rate of climate change, especially for vulnerable regions such as the Arctic and the HKHT region. In contrast, when long-lived GHG emissions are reduced, the climate takes longer to respond because atmospheric GHG concentrations—the result of cumulative historic and present-day emissions—remain relatively constant for longer periods (see, for example, Figure 2-7). It is important to recognize, however, that the short atmospheric lifetime of BC also means that reductions in current BC emissions will have much less impact on temperature many

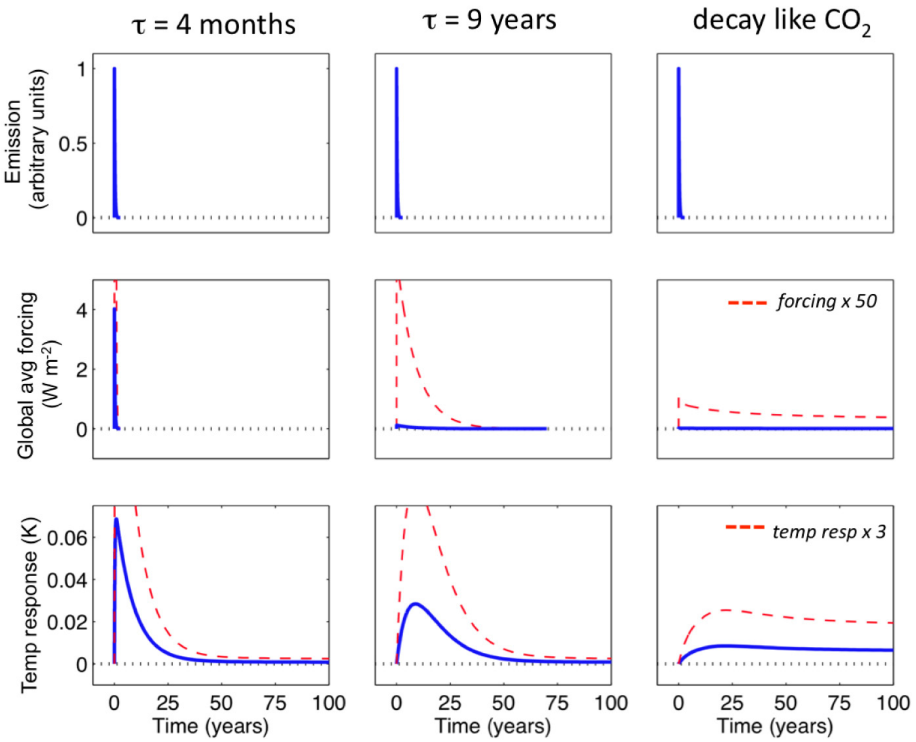

Figure 2-7. Climate Response to Emissions of Pollutants with Different Lifetimes. Figure illustrates forcing and temperature response to pulse emissions of three hypothetical substances with different atmospheric lifetimes (Column 1 illustrates short-lived species like BC, Column 2 illustrates species with medium atmospheric lifetime like methane, and Column 3 illustrates a long-lived species like CO_2). In the middle row, which illustrates global average forcing, the dashed red trace shows forcing value times 50 to show more detail in the time dependence. (Bond et al., 2011)

decades from now,[7] which is mainly driven by past, current, and future emissions of persistent species like CO_2 combined with emissions of BC at that future date. Therefore, only sustained reductions in long-lived GHGs can avert long-term climate change.

In order to avoid (rather than delay) reaching temperature thresholds, long-lived GHG emissions reductions are necessary, but future BC reductions in conjunction with these GHG reductions can "shave the peak" off the temperature change and help avoid crossing these thresholds (possibly avoiding key non-linear climate impacts, often referred to as "tipping points") (Lenton et al., 2009). Figure 2-8 demonstrates this visually: reductions in emissions of CH_4 and BC [the specific measures involved are described in more detail in the UNEP report (UNEP and WMO, 2011a)] lead to a near-term reduction in the rate of warming, but in the long run serve only to delay any given temperature change. While CO_2 reduction measures do little to slow near-term warming (in this case, in part because they are accompanied by simultaneous reductions of cooling aerosols), they are necessary for long-term stabilization of the climate. Reducing short-lived forcers in conjunction with CO_2 enables both a reduction in near-term warming (due to near-term BC and CH_4 emissions reductions) and a decrease in peak warming (due to future BC and CH_4 reductions combined with CO_2 reductions over the entire time period).

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

2.5 The Role of Co-Emitted Pollutants and Atmospheric Processing

As discussed in section 2.3, BC is never emitted into the atmosphere in isolation. Rather, it is always emitted as part of a mixture of particles and gases during the combustion process. The composition of this mixture can vary significantly, depending on combustion conditions and fuel type. BC is generally

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[7] As shown in Figure 2-7, the temperature response to an immediate reduction in BC can last a couple of decades – due mainly to inertia in the climate because of how the ocean takes up heat. However, not included in this figure is the possibility that these temperature changes can have feedbacks which might persist for even longer – for example, higher temperatures reduce the rate at which CO_2 is mixed into the ocean, so a change in BC emissions today could change CO_2 concentrations, and therefore temperatures, for many decades.

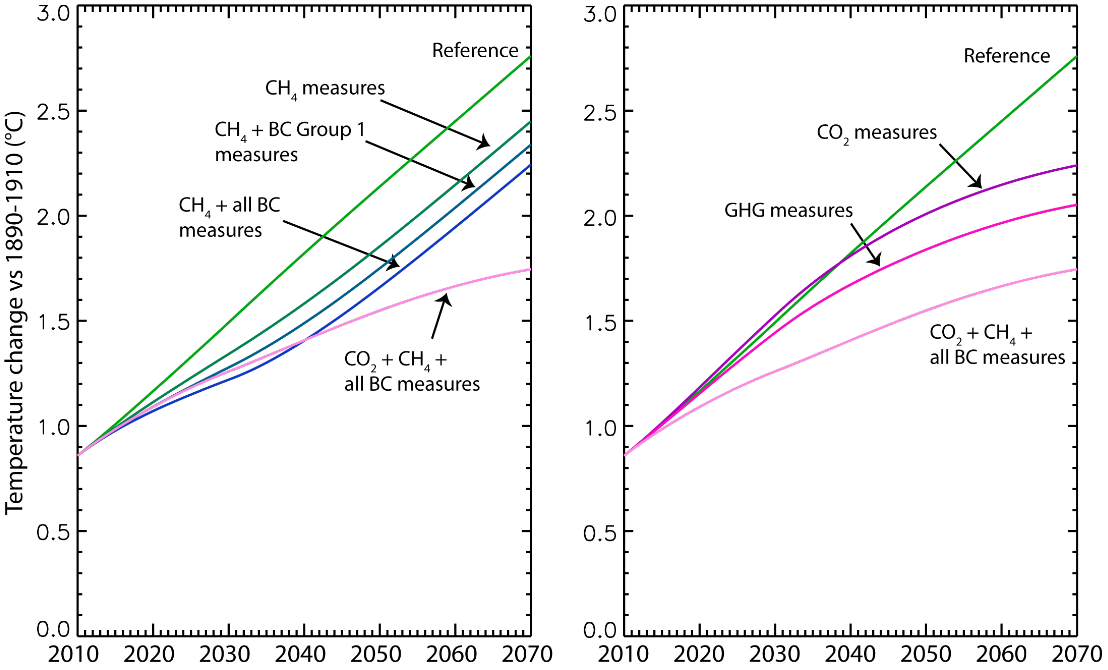

Figure 2-8. Projected Global Mean Temperatures under Various Scenarios Relative to the 1890-1910 Average. This figure is adapted from Figure 6.2 from the UNEP/WMO Assessment. The left panel shows a reference scenario, and then a sequence of emissions reduction scenarios in which first a set of methane reductions are implemented, then one set of BC measures, then a second set of BC measures, and then a set of CO_2 measures. In the right panel, the same reference scenario is shown, followed by a sequence of emissions reduction scenarios in which first the same set of CO_2 measures is applied, followed by some additional GHG reductions (mainly methane), followed by the full set of reductions from the left panel. More details about these measures can be found in the UNEP/WMO report. (Note that modeled effects of all measures include reductions of co-emissions, including reductions in cooling aerosols, which in the case of CO_2 measures resulted in a slight warming effect in the early years.) (UNEP and WMO, 2011a)

accompanied by OC, including BrC and other carbonaceous materials. In addition, an emissions plume may contain water, inorganic potassium and sodium salts, ammonium nitrate and sulfate, gaseous constituents (e.g., SO_2, NO_x and VOCs), various hazardous air pollutants (e.g., metals), and even soil particles.

The absorptive properties of an emissions plume from a specific source will depend on all of the ***co-emitted pollutants***, and on how these constituents interact with one another and other atmospheric constituents in the atmosphere. As described above, BC is co-emitted with OC and/or sulfate, nitrate and gaseous constituents (SO_2, NO_x, and VOCs). Since OC and sulfate and nitrate particles generally exert a net cooling influence, these pollutants play an important role in determining the net absorptive capacity of the emissions plume. These other constituents, however, may be emitted in greater volume than BC, counteracting the warming influence of BC. Thus, estimating the climate impact of BC quantitatively requires accounting for the impact of these co-emitted pollutants. Emissions from a single source can also vary over time. For example, the flaming phase of a wildfire produces much more BC than its smoldering phase. Also, when diesel trucks are under load, they produce more BC than during other parts of their driving cycle. Total ***particle number*** also impacts scattering and absorption: the more particles present in a portion of the atmosphere, the greater the probability that light rays will be scattered or absorbed by some of these particles.

Emissions from particular sources are often characterized in terms of their OC to BC ratio. Sources whose emissions mixtures are richer in BC relative to the amount of OC emitted (i.e., with lower OC:BC ratios) are more likely to contribute to climate warming; therefore, mitigation measures focusing on these sources are more likely to produce climate benefits. These ratios are useful in that they take the emissions mixture into account;

however, they rely on crude accounting methods and cannot provide precise measures of a particular source's climate impacts. A particular concern is the common presumption that all OC is cooling, when in fact some components (especially BrC) are light-absorbing and may contribute to the warming associated with an emissions mixture. (The use of OC:BC ratios is discussed further in Chapter 7.)

A fresh emissions plume contains particles of many different chemical compositions. Atmospheric scientists refer to an emissions plume with this kind of high inter-particle chemical variability as "***externally mixed***." The externally mixed plume, however, undergoes rapid chemical and physical transformations. The coagulation of particles, assuming no secondary particles form through atmospheric chemical reactions, reduces the overall particle number. This process, combined with thermodynamically-driven mixing processes such as water condensation and the redistribution of semi-volatile PM compounds, reduces the differences in chemical composition among the individual particles in the plume. Over time, the chemical composition of the particles within a given plume approaches uniformity. A plume is said to be "***internally mixed***" when it is near this theoretical end-point. *In situ* measurements indicate that emissions become internally mixed within a few hours (Moffet and Prather, 2009b).

BC is an insoluble material, thus a well-mixed or "aged" emissions plume will contain particles where BC appears as an inclusion in an otherwise liquid particle. These particles are often described as "coated" BC particles. As described in section 2.3, common coatings include sulfuric acid, water, and transparent OC. The degree of mixing, or more specifically, the fraction of BC particles that are coated, influences the absorptive properties of the particle. Internal mixtures of particles that include BC have been observed to absorb light more strongly than pure BC alone, by 30 to 100%. Whether BC is modeled as an externally or internally mixed particle can have a large effect on resulting estimates of radiative forcing (see section 2.6.1.2).

Emissions plumes from different sources interact with each other as well as with the surrounding atmosphere. As a combustion emissions plume rises into the atmosphere, it is diluted by ambient air (see Figure 2-9). The open atmosphere contains

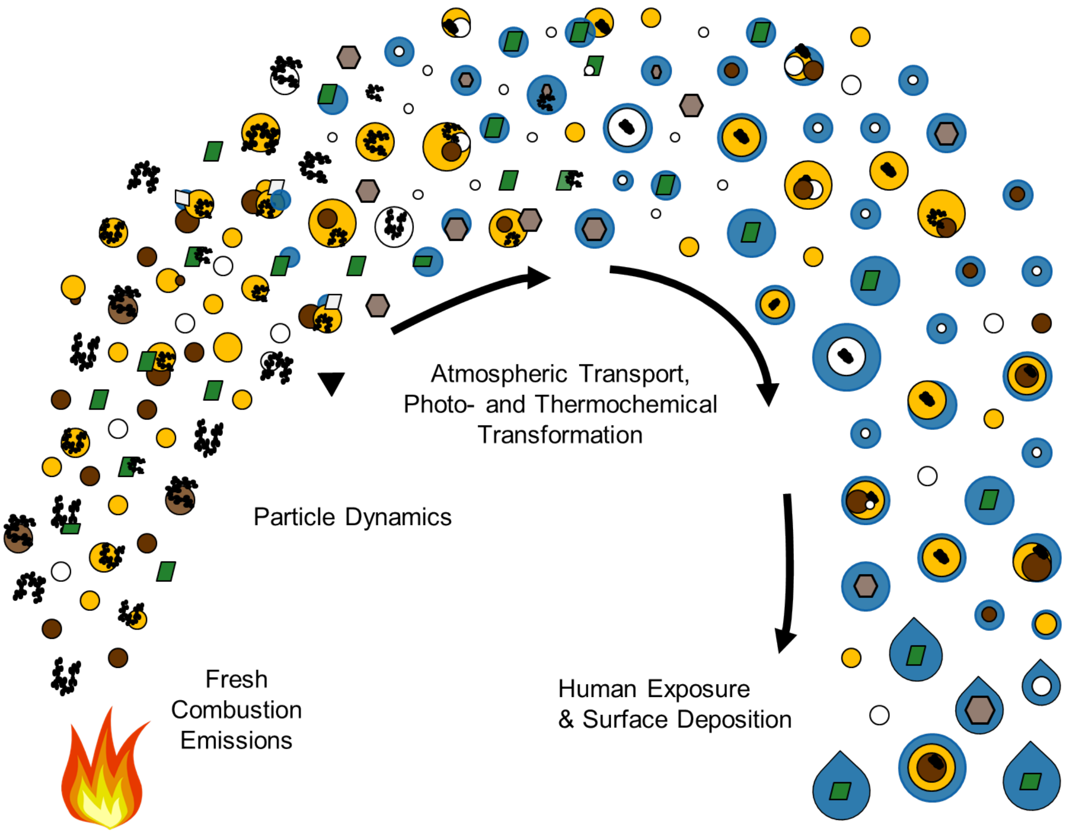

Figure 2-9. Particle Transformation in the Atmosphere, from Point of Emission to Deposition. A variety of physical and chemical processes contribute to changing the light-absorption capacity of a fresh plume. (Source: U.S. EPA)

a number of reactive gases and particles originating from a wide variety of anthropogenic and natural sources. Physical and chemical changes resulting from coagulation, condensation, and other photochemical and atmospheric processes can alter the climate-forcing impact of a given emissions plume (Lauer and Hendricks, 2006) (see Table 2-2). These mixing and aging processes are complex. However, excluding them from models of the climate impacts of BC may yield incomplete or erroneous estimates. For example, coating of a BC particle by a clear (light-scattering) shell has been shown to enhance light absorption because the shell acts as a lens that directs more light toward the core (Ackerman and Toon, 1981; Jacobson, 2000; Lack and Cappa, 2010; Jacobson, 2001). Other authors have also found that light-absorption by BC is enhanced when BC particles are coated by sulfate or other light scattering materials (Shiraiwa et al., 2009; Sato et al., 2003; Moffet and Prather, 2009a; Bond et al., 2006a). Other atmospheric processes, however, such as further chemical processing or

Table 2-2. Examples of Particle Types and Mixtures Present in Combustion Plumes. The size, shape, 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

[a] Fresh BC is produced primarily during flaming combustion.

[b] Particles condense within a fresh combustion plume from pyrolytic BrC and yellow OC. Oxidation of anthropogenic and biogenic VOCs produces non-light absorbing carbon particles, and may also produce BrC and yellow carbon PM.

[c] Emitted directly as a byproduct of combustion, or formed through the oxidation of SO_x or NO_x.

[d] In the exhaust gases of solid fuel fires, low volatility BrC and other organic compounds can condense on BC particles. In the ambient atmosphere, low volatility organics produced by oxidation of VOCs can also condense on BC.

[e] Forms when high particle concentrations lead to the coagulation of multiple particles.

[f] Forms by condensation of water vapor onto acidic organic (carbon-based) and inorganic particles.

[g] Forms when complex particles undergo the humidification and drying cycles characteristic of cloud formation and evaporation.

particle growth through coagulation, may off-set this enhancement. Observations by Chan et al. (2011) at a rural site in Ontario, where BC particles were assumed to be coated, did not show enhanced light absorption. Furthermore, coated particles are more easily removed by cloud droplets and precipitation, decreasing their atmospheric lifetime (Stier et al., 2006).

2.6 Global and Regional Climate Effects of Black Carbon

BC affects climate through both direct and indirect mechanisms. The most extensively studied of these mechanisms is radiative forcing, which is directly linked to temperature change. Radiative forcing is a measure of how a pollutant affects the balance between incoming solar radiation and exiting infrared radiation, generally calculated as a change relative to preindustrial conditions defined at 1750. A pollutant that increases the amount of energy in the Earth's climate system is said to exert "positive radiative forcing," which leads to warming.[8] In contrast, a pollutant that exerts negative radiative forcing reduces the amount of energy in the Earth's system and leads to cooling. The net radiative impact of a pollutant since preindustrial times can be averaged over the Earth's surface and is expressed in Watts per square meter (W m^{-2}). Global average radiative forcing is a useful index because in general it is related linearly to the global mean temperature at the surface (Forster et al., 2007) and is approximately additive across pollutants. Radiative forcing also provides a consistent measure for comparing the effects of past and projected future emissions. As a result, it has become a standard measure for organizations like the IPCC and the U.S. Global Change Research Program (National Research Council of the National Academies, 2005).

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

This section mainly addresses the climate impacts of BC, GHGs, and other substances based on the radiative forcing resulting from the change in concentrations of these substances since preindustrial times. There are, however, other ways to address climate impacts, and the decision of which approach to use depends on at least two key issues.

Radiative Forcing

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

The first issue is that an analysis based on changes in concentrations since preindustrial times is in some ways a historical measure. An alternative approach would be to analyze the effect of current day emissions on radiative forcing into the future [e.g., Figure 2-19 in this report, Figure 2.22 in Forster et al. (2007), or the analysis by Shindell et al. (2009)]. The benefit of such an approach is that an emissions-based analysis is more policy-relevant, because policies directly control emissions and not concentrations. There are several disadvantages to using this approach. First of all, fewer studies have used this approach. Second, the approach requires choosing a time frame of integration (see section 2.7 on metrics for more discussion of time frames). Third, there are increased uncertainties because the results depend on decisions regarding background concentrations and are more sensitive to model factors such as atmospheric chemistry and carbon cycles. In contrast, analyses based on existing concentrations are less uncertain because for the most part they depend on measured concentrations. In general, this is more important for the long-lived GHGs than for short-lived substances such as BC, and thus becomes an issue for purposes of comparisons.

The second issue in addressing climate impacts is whether to consider radiative forcing as an endpoint, or whether to actually calculate impacts on temperatures. The advantage to examining temperatures is that they are more immediately relevant to human experience than the more abstract "radiative forcing". Also, though radiative forcing is approximately additive in most cases, there are some exceptions: as discussed in section 2.6.1.4, temperatures are actually much more sensitive to snow albedo forcing than to other forcings.

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[8] In general, radiative forcing in this document refers to "top-of-the-atmosphere" radiative forcing unless otherwise specified. Measuring at the top of the atmosphere (TOA) (in this context, between the troposphere and the stratosphere) is the best location for determining net energy balance.

However, there are two disadvantages to the temperature approach. The first is that, again, fewer studies use this approach, in part because the models used to calculate temperature are more computationally intensive than those that calculate radiative forcing. The second is that the relationship between radiative forcing and temperature, while generally linear, is very model dependent. Every model has an inherent "climate sensitivity", and therefore when comparing the impacts of BC from one model to the next, differences in this climate sensitivity can make it difficult to understand if differences between the models are due to BC physics or to the general model response to any forcing.

2.6.1 Global and Regional Radiative Forcing Effects of BC: Overview

This section provides an overview of the different effects of BC on radiative forcing based on the best estimates in the current literature. These effects include **direct forcing** (direct absorption of solar or terrestrial radiation), **snow/ice albedo forcing** (forcing that results from the darkening of snow and ice), and **indirect forcing** (a range of forcing effects resulting from impacts on clouds, including changes in cloud lifetime, reflectivity, and composition). Section 2.6.1.1 describes the overall net effects of BC on radiative forcing, when the effects of these different types of forcing are accounted for. The subsequent sections present more detailed information regarding the state of knowledge regarding the specific effects of BC on direct forcing, snow/ice albedo forcing, and indirect forcing at the global and regional scales. Each section provides a summary of the findings of recent studies, explanations of the differences among estimates, and characterizations of key remaining uncertainties.

In evaluating estimates of the effects of BC on radiative forcing, it is necessary to consider several caveats. For example, it is important to differentiate among estimates with respect to the baseline time period used to define the radiative forcing estimates. The radiative forcing estimates (and other climate effects) are often expressed as a comparison to a given historical level rather than with respect to present day or in terms of the anthropogenic influence compared to total forcing. However, these assumptions are not always stated clearly. Also, it is important to differentiate with respect to the types of BC emissions included. For example, many studies exclude open biomass burning. The inclusion or exclusion of BC from wildfires and other sources of open biomass burning will affect the estimates of net BC effects. In addition, because some studies evaluate the climate effects of BC as it co-occurs with other aerosol chemical species, such as OC, sulfates and nitrates, while others do not, it is important to distinguish studies where BC is estimated individually from studies where BC is estimated as part of an aerosol mixture. In the following sections, it is indicated whether the radiative forcing estimates include co-occurring OC and other species and how these other pollutants influence estimates of BC's global and regional climate impacts, when possible.

2.6.1.1 Net Forcing

As is described in more detail in the following sections, the different kinds of forcing involve different mechanisms of action and can have offsetting climate effects. For example, direct effects are associated with positive forcing, while most (but not all) indirect effects are thought to result in negative radiative forcing. This section provides an overview of estimates of the direction and magnitude of the net effect of BC on radiative forcing when the direct forcing, snow/ice albedo forcing, and indirect forcing effects of BC are summed, and identifies key factors that contribute to variability in these estimates

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

The most widely utilized estimates of forcings for GHGs come from the IPCC's Fourth Assessment Report, which was issued in 2007. The IPCC also estimated forcing due to BC based on a review of the scientific studies available at the time, though estimates from more recent studies surveyed in this EPA report differ somewhat as detailed further below. The IPCC estimated a direct radiative forcing of +0.34 W m^{-2} for BC, making BC third only to CO_2 and CH_4. In addition, the IPCC estimated BC's snow/ice albedo forcing to be +0.1 W m^{-2} (see Figure 2-10). Other aerosols were generally shown to have a cooling influence on climate. The IPCC estimates of negative direct radiative forcing due to OC and sulfates are also shown in Figure 2-10. Indirect effects for *all* aerosols, including BC, are also estimated to result in net negative forcing due to increased reflectivity of clouds ("cloud albedo effect"). The IPCC did not provide quantitative estimates of the effect of aerosols on other properties of clouds (such as lifetime, stability, etc.),

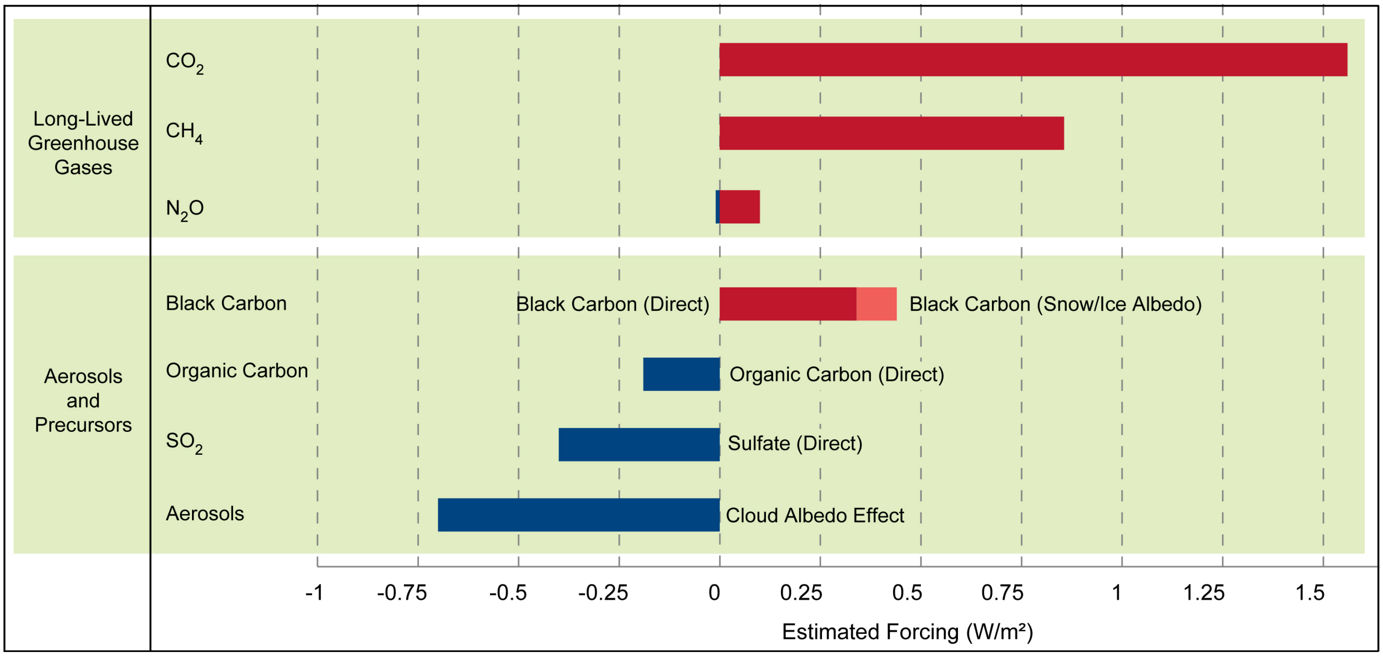

Figure 2-10. Components of Global Average Radiative Forcing for Emissions of Principal Gases, Aerosols, and Aerosol Precursors, based on IPCC estimates. Values represent global average radiative forcing in 2005 due to emissions and changes since 1750. Total radiative forcing for CH_4 includes the effects of historical CH_4 emissions on levels of tropospheric O_3 and stratospheric H_2O, and the CO_2 oxidation product of CH_4 from fossil sources. Similarly, total radiative forcing for N_2O includes the effect of historical N_2O emissions on levels of stratospheric O_3. The IPCC does not report an overall uncertainty for the net contribution to forcing of individual GHG emissions. However, based on the uncertainties provided for the individual components of these contributions, the uncertainty in forcing from CO_2 and N_2O emissions is extrapolated as being approximately 10% and approximately 20% from CH_4 emissions. Uncertainty in direct forcing is ±0.25 W m^{-2} for BC and ±0.20 W m^{-2} for both OC and SO_2. The range of forcing for the cloud albedo effect is -1.8 to -0.3 W m^{-2}. (Adapted from Figure 2.21 of Forster et al., 2007)

or quantify the indirect effects of individual aerosol species (such as BC) separately. As a result, there is substantial uncertainty in the IPCC's estimates of net forcing for BC.

In addition to the estimates compiled by the IPCC (2007), many other studies have attempted to estimate the global average radiative forcing attributable to BC. An examination of the results of these studies, as summarized in Figure 2-11, indicates that the direct effect and the snow/ice albedo effect of BC are positive, though the magnitude of these effects is uncertain. The figure shows the range of central estimates from the included studies (solid box) as well as the highest and lowest uncertainty estimates from those studies (error bars) for both the direct effect and the snow and ice albedo effect. As discussed further below, a number of studies have estimated BC's direct radiative forcing to be higher than the IPCC estimate (Sato et al., 2003; Ramanathan and Carmichael, 2008).

The biggest source of uncertainty about the net forcing effect of BC is the magnitude of the cloud effects of BC. (Cloud effects are discussed in detail in section 2.6.1.3.) The limited number of studies in the literature allow for statements on the direction (e.g., warming or cooling) of some of these forcings, but not their magnitude, as shown in Figure 2-11. The impact on cloud lifetime and albedo is likely cooling. The interactions with mixed-phase and ice clouds are likely to be warming. Semi-direct effects are uncertain, and existing studies differ on the definition and net influence of the effect. The cloud absorption effect is positive.

In light of the large remaining uncertainties about the magnitude (and in some cases the sign) of the different forcing effects of BC, particularly with regard to the cloud interaction effects, this EPA report does not assign a range to the magnitude of the net effect beyond noting that it is very likely to be positive (however, a net negative effect cannot be excluded). As indicated in Figure 2-11, the

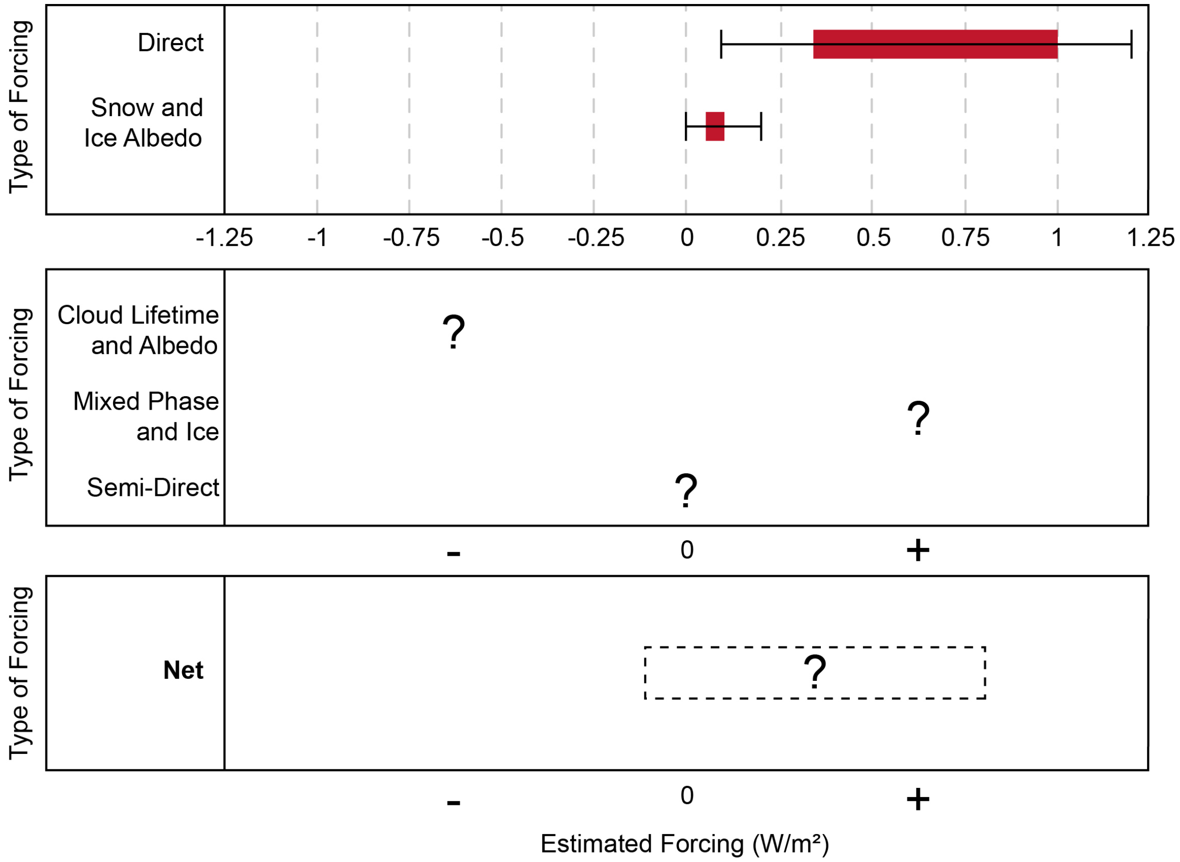

Figure 2-11. Estimates of Radiative Forcing from BC Emissions Only. The boxes indicate ranges of central estimates from the papers identified in this report, with error bars indicating the highest and lowest uncertainty estimates from those papers. Estimates are based on a synthesis of results from eleven studies that considered the direct forcing effects of BC emissions and six studies that considered reduction in snow and ice albedo from BC emissions. The range for the snow and ice albedo bar does not include the effects of the higher efficacy of the snow albedo effect on temperature change (forcing efficacy is described in section 2.6.1.4). The studies of indirect and semi-direct radiative forcing effects due to BC emissions are not sufficiently comparable in scope and approach to combine the estimates. As a result, only the likely direction of forcing is presented. Two other warming effects, the cloud absorption effect and the water vapor effect, have recently been discussed in the literature but are not included here. (Source: U.S. EPA)

estimates of direct and snow/ice albedo forcing are likely to be positive, but additional work is needed to determine the extent to which these positive forcing effects are offset by indirect effects, semi-direct effects, and other effects on clouds.

The recent UNEP/WMO assessment (UNEP and WMO, 2011a) evaluated a number of recent studies to investigate the net effect of BC on climate. Based on estimates of BC forcing due to direct, indirect, and snow/ice albedo effects, UNEP/WMO estimated that the global average net forcing was likely to be positive and in the range of 0 to 1 W m^{-2}, with a central value of 0.6 W m^{-2} (this estimate included an enhanced efficacy factor for the snow/ice albedo effect: forcing efficacy is described in section 2.6.1.4.). The UNEP/WMO assessment estimates for the different forcing effects of BC are shown in Table 2-3. In selecting a central estimate for net BC forcing, the authors of the UNEP/WMO assessment noted the very strong negative forcing due to total aerosols,[9] and the current lack of quantitative

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[9] The IPCC estimated the net forcing due to all aerosols as -1.2 W m^{-2} (including a direct forcing effect of -0.5 W m^{-2} (-0.9 to -0.10 W m^{-2}) and a cloud albedo effect of -0.7 W m^{-2} (-1.8 to -0.3 W m^{-2}) (IPCC, 2007). However, these estimates are assigned a low level of scientific understanding by IPCC. More recent estimates in the literature continue to span a large range: Bauer and Menon (2012) estimate the net aerosol effect as -0.6 W m^{-2} (including direct -0.5 W m^{-2} and indirect -0.1 W m^{-2}), while Hansen et al. (2011) argue the effect is more strongly negative (-1.6 W m^{-2} ±0.3)

Table 2-3. Summary of UNEP/WMO Assessment Estimates of Radiative Forcing Effects of BC. (Adapted from UNEP and WMO, 2011a, Table 3-1)

Forcing	Range (W/m^2)	Central Values (W/m^2)	Mean Value (W/m^2)
Direct	0.2 to 0.9	0.3 to 0.6	0.45
Semi-direct and indirect	-0.6 to 0.5	-0.4 to 0.4	0.0
Deposition	(0.01 to 0.08) corrected for efficacy (x5) 0.05 to 0.4	(0.01 to 0.05) 0.05 to 0.25	0.03 0.15
Total BC	-0.3 to 1.8	0.0 to 1.3 0.0 to 1.0 with constraints on total aerosol forcing	0.5 0.6 including efficacy of deposition

estimates for all processes. In light of these constraints, they argued that net effective BC forcing is unlikely to exceed 1 W m^{-2}.

There remains a strong need for further research to provide better estimates of the forcing effects of BC and to reduce remaining uncertainties. Work in progress by a consortium of researchers under IGAC/SPARC (currently being prepared for submission to an academic journal with expected publication later in 2012) will likely provide more definitive quantitative bounds on the BC cloud interaction effects, and the net effects overall.

2.6.1.1.1 Factors that Contribute to Variability in Estimates

There are a number of factors that may contribute to the lack of consensus among modeled estimates of net global average radiative forcing from BC. Koch et al. (2009) attributed the range of estimates to differences in the aerosol microphysical calculations in the models (i.e., different estimates of how much solar radiation each unit of BC absorbs). The authors also pointed out key differences in models, such as the assumed values of various physical properties, and differences in the representation of vertical transport and cloud effects.

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

2.6.1.1.2 Regional Dynamics

Studies focusing on global average radiative forcing may overlook key regional dynamics associated with BC as a spatially heterogeneous pollutant. Many studies have found that BC's regional climate impacts are more pronounced than the contributions of BC to global average temperature change. In addition, certain regions of the world are more sensitive to or more likely to be affected by BC forcing, either due to transport and deposition (e.g., the Arctic) or high levels of aerosol pollution in the region (e.g., Asia). Global average radiative forcings for BC hide much of the regional variability in the concentrations and impacts. Note, however, that regional variability of BC forcing may exaggerate the regional variability of impacts, as temperature impacts usually occur over a larger area and longer time period than the forcing effects.

2.6.1.2 Direct Forcing

The direct effect of BC is to absorb solar radiation. As mentioned above and as shown in Figure 2-12, the IPCC (2007) estimated the global average radiative forcing of BC from all sources at +0.34 (±0.25) W m^{-2}. A subset of this forcing due to BC from fossil fuel combustion (mainly coal, petroleum and gas fuels) was estimated to be +0.2 (±0.15) W m^{-2}. Most studies published since the IPCC report have reported higher direct forcing values. The UNEP/WMO assessment (UNEP and WMO, 2011a) estimated that direct forcing of BC is most likely to be within the range 0.3 to 0.6 W m^{-2}. Additional work

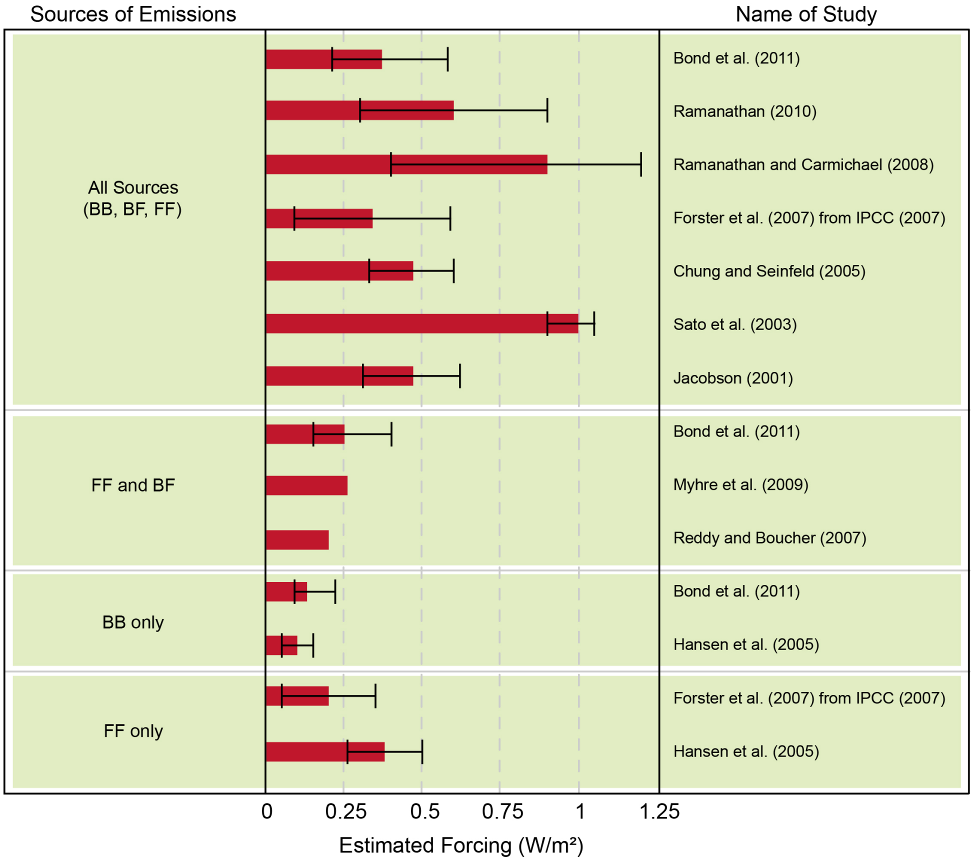

Figure 2-12. Estimates of Direct Radiative Forcing from BC Emissions Only. These values represent the range of estimates in the peer reviewed literature; however, they are not all directly comparable. Some are based on different estimates of BC emissions, include different sectors, and present the forcing with respect to different baseline time periods (e.g., 1750, post-industrial, present day). Note: BB = open biomass burning; BF = biofuels; and FF = fossil fuels. (Source: U.S. EPA)

is underway to try to develop a new central estimate for these direct impacts (IGAC/SPARC, forthcoming).

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

In addition to mixing state, other factors that can lead to variation in modeled radiative forcing effects of BC include particle size and removal (e.g., by precipitation). Vignati et al. (2010) describe how a modeled reduction in wet removal of BC from the atmosphere by 30% results in a 10% increase in BC's atmospheric lifetime. In some cases, mainly in work based on observational constraints from the Aerosol Robotic Network (AERONET) ground-based sunphotometer network, much higher values have been reported. Sato et al. (2003) inferred a forcing of 1 W m^{-2} based on these observational constraints. Chung et al. (2005) and Ramanathan and Carmichael (2008) combined the AERONET results with satellite data and report 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}. While most recent studies find global forcing higher than the IPCC, a discrepancy remains between the very high observationally constrained results and model results (even those

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[10] Ramanathan (2010) suggests that Bond et al. (2011) underestimate the total forcing, and suggests a central value of +0.55 with a 90% confidence interval of +0.2 to +0.9 W m^{-2}.

that include internal mixing and therefore produce higher values). Bond et al. (2011) hypothesized that the higher forcing in the observationally constrained results could result from higher emissions than in the model work. The exact cause of these differences, however, has not been isolated.

Compared to global radiative forcing, fewer studies have reported regional direct radiative forcing by BC. Studies such as Bond et al. (2011) show the geographic distribution of direct forcing from all sources of BC emissions (Figure 2-13). They found the largest forcing over South and East Asia and parts of Africa. Myhre et al. (2009), who considered only fossil fuel and biofuel BC, also found the largest forcing over South and East Asia. Other work such as Chung and Seinfeld (2005) showed similar patterns with higher forcing in Central and South America (the Amazon basin) and sub-Saharan Africa due to the inclusion of biomass burning emissions. Chung and Seinfeld (2005) report a range of +0.52 to +0.93 W m^{-2} for externally and internally mixed BC respectively, averaged over the Northern Hemisphere. Their earlier work also suggests a strong seasonal cycle which peaks in May at +1.4 W m^{-2} (Chung and Seinfeld, 2002). For the Southern Hemisphere, Chung and Seinfeld (2005) estimate a range of +0.15 to +0.23 W m^{-2}. Reddy and Boucher (2007) calculated the influence of regional BC emissions on the global average radiative forcing. The largest contribution to global TOA BC radiative forcing came from East Asia (+0.08 W m^{-2}). The global average forcing due to North American BC emissions in this study was +0.02 W m^{-2}.

2.6.1.3 Cloud-related Forcings

The net effect of particles on climate via impacts on clouds is highly uncertain (IPCC, 2007). There are several different kinds of cloud effects that are important for radiative purposes, as summarized in Table 2-4. These cloud effects contribute to changes in the radiative balance of the atmosphere, and also influence climatic factors such as precipitation and dimming (section 2.6.3).

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

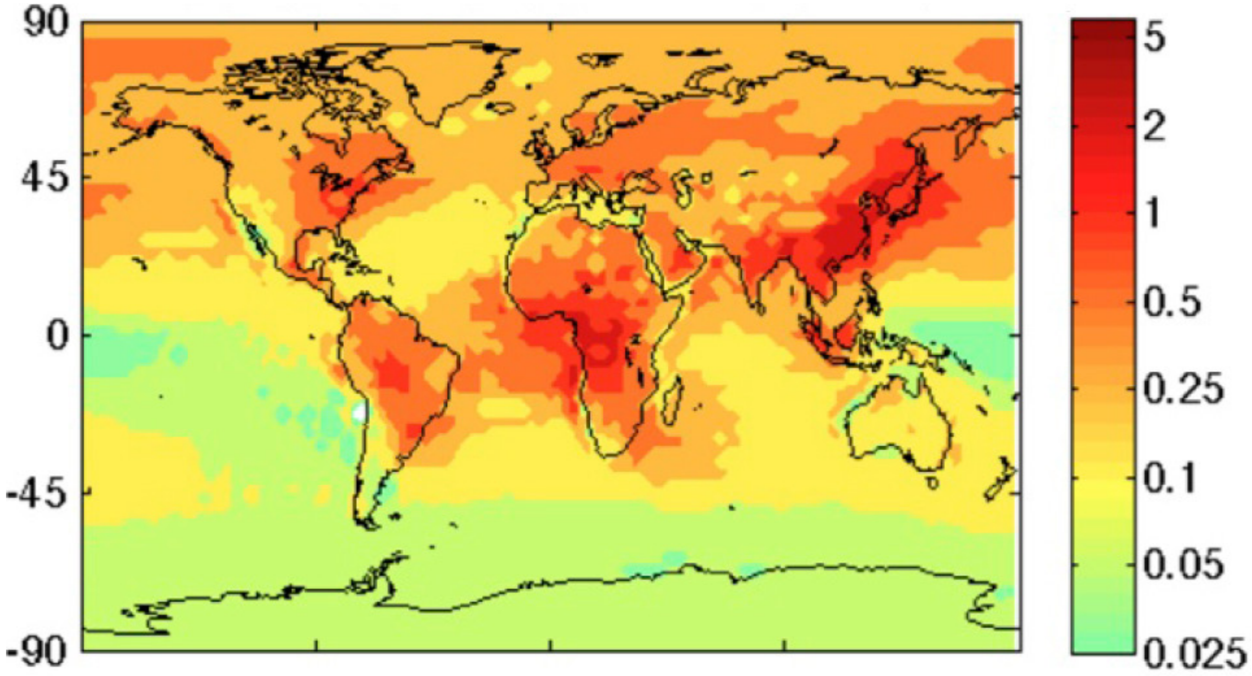

Figure 2-13. Direct Radiative Forcing (W m^{-2}) of BC from All Sources, simulated with the Community Atmosphere Model. (Bond et al., 2011)

Table 2-4. Overview of the Different Aerosol Cloud Effects. This summary applies to all aerosols and is not BC-specific, though BC is expected to participate to some extent in all of these mechanisms. Net effect refers to top-of-the-atmosphere radiative forcing. Scientific understanding is based on IPCC terminology. (Adapted from IPCC, Denman et al., 2007, Table 7.10a)

Effect	Cloud Types Affected	Process	Net Effect	Potential Magnitude	Scientific Understanding
Cloud albedo effect (first indirect effect)	All clouds	Smaller cloud particles reflect more solar radiation	Cooling	Medium	Low
Cloud lifetime effect (second indirect 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

cloud droplets are less likely to aggregate sufficiently to form rain drops, which changes precipitation patterns and increases cloud lifetime (Albrecht, 1989) (see also section 2.6.3.2 on precipitation impacts). This has been called the "second indirect effect" or the "**cloud lifetime effect**". In general, the cloud albedo effect and the cloud lifetime effect are estimated to lead to cooling. However, Ramanathan (2010) asserts that the empirical evidence shows a positive forcing (warming) over land regions.

The magnitude and sign of the radiative effects depend on whether the clouds are composed of liquid droplets, ice particles, or a mix of ice and liquid droplets, and on the composition of the aerosol particles. In certain kinds of "mixed-phase clouds" (clouds with both ice and water), smaller droplets cause a delay in the freezing of the droplets, changing the characteristics of the cloud; however, the IPCC was not able to determine whether this "**thermodynamic effect**" would result in overall warming or cooling (Denman et al., 2007).

The "**semi-direct effect**" is specific to BC and other absorbing aerosols, while the **"glaciation indirect effect"** appears to be important for aerosols, such as BC and mineral dust, which are not especially hydrophilic. The semi-direct effect refers to the heating of the troposphere by absorbing aerosols, affecting the relative humidity and stability of the troposphere, which in turn affects cloud formation and lifetime (IPCC, 2007; Ackerman et al., 2000).

Older literature refers to the semi-direct effect as cloud burn-off (i.e., a decrease in cloud formation) from BC within the cloud layer. The definition was extended to include all effects on cloud formation and lifetime as other studies have found that humidity and stability effects from BC above and below clouds can cause both increases and decreases in clouds (Koch and Del Genio, 2010). The IPCC did not assign a sign to the net forcing of the semi-direct effect (Denman et al., 2007).

More recently, Koch and Del Genio (2010) found in their review of the literature that most model studies generally indicate a global net negative semi-direct effect (i.e., the effect of atmospheric heating by absorbing aerosols on cloud formation and lifetime causes net cooling). This was observed despite regional variation in the cloud response to absorbing aerosols (such as BC), and resulting regional differences in warming and cooling from the semi-direct effect. In contrast to Koch and Del Genio, Jacobson (2010) found that the semi-direct effect is positive. This difference may be due in part to the more inclusive definition used by Koch and Del Genio. Isaksen et al. (2009) reported a range of -0.25 to +0.50 W m^{-2}.

While the sign of the semi-direct effect is therefore in question, the glaciation effect is very likely a warming effect, though it occurs only in some mixed-phase clouds. This indirect effect is caused by BC aerosols (and some other particles such as mineral dust) serving as ice nuclei in a supercooled liquid water cloud, thereby enabling precipitation rather than delaying it (Denman et al., 2007; Lohmann and Hoose, 2009). However, some preliminary work (e.g., Penner et al., 2009) suggests that the effect of soot on ice nuclei and their subsequent effect on cirrus clouds could offset some of the warming resulting from the glaciation effect.

Most estimates of the forcing from aerosol indirect effects are based on all aerosol species (e.g., total PM) and are not estimated for individual species (e.g., BC alone). The net indirect effect of all aerosols is estimated as a negative value. The IPCC (Forster et al., 2007) estimated the change in cloud albedo due to all aerosols to have a 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 scientific understanding.[11] A recent study by Bauer and Menon (2012) found a smaller net cloud interaction effect due to aerosols of only -0.1 W m^{-2}.[12]

There are some BC-specific effects that do not fall into the IPCC categories from Table 2-4, which have been identified in a limited number of studies. The first is called the "cloud absorption effect." This effect is the result of BC particles being included between and within cloud particles, increasing absorption (or decreasing reflectivity) of the clouds. Jacobson (2010) suggests that the inclusion of this cloud absorption effect may increase warming from BC by as much as 75%. Additionally, recent experimental work above the Amazon indicates that cloud albedo will increase with increasing aerosol loading (as predicted by the cloud albedo effect) only as long as the aerosol loading is smaller than a critical threshold. Above this threshold, additional aerosols from biomass burning actually lead to a decrease in cloud albedo, attributed to inhibition of cloud formation by absorbing aerosols (Ten Hoeve et al., 2011; Koren et al., 2008). The second effect that does not fall into the more common IPCC categories is called the "BC-water vapor effect" (Jacobson, 2006, 2010). This effect is based on an increase in water vapor due to inhibition of precipitation from clouds, related to the cloud absorption effect. However, BC can also lead to a decrease in relative humidity in some regions where the increase in atmospheric temperature at altitude does not lead to increased evaporation at the surface. It is unclear what the magnitude of this effect is.

To summarize, it is unclear to what extent BC contributes to the overall aerosol indirect effect. As a result, this report does not assign any central estimate or even a range of possible values for the role of BC in the overall indirect aerosol effect. BC's role in the first and second indirect effects (cloud albedo and cloud lifetime effects) is likely to be cooling, but possibly to a lesser extent than for other aerosols. Although freshly emitted, externally mixed BC particles are hydrophobic and would be less active cloud condensation nuclei (CCN), aging may increase their ability to serve as CCN (Dusek et al., 2006). Recent work (e.g., Bauer et al., 2010) using models with a more explicit representation of aerosol mixing than older models suggests that the role of BC in the indirect effect may be greater than previously thought. Similarly, BC may also participate in the thermodynamic indirect effect for mixed-phase clouds, but whether this effect is net warming or cooling is still uncertain. BC has a primary role in the semi-direct effect, but this effect may produce warming or cooling depending on conditions. Finally, BC particles may contribute to warming from the glaciation indirect effect in mixed-phase clouds, the cloud absorption effect, and the water vapor effect, but the magnitude of these effects are uncertain. A comprehensive, quantitative estimate of the net effect of BC would require an assessment of the likely bounds of these cloud effects. While this EPA report does not assess those bounds, the UNEP/WMO assessment (UNEP and WMO, 2011a) found that the most likely case was that the cloud interaction effects resulting from BC alone would cancel out (mean value of 0.0 W m^{-2}), but with a central range of -0.4 to +0.4 W m^{-2}. The forthcoming IGAC/SPARC study will present a more definitive bound on these effect estimates. Studies are just beginning to estimate indirect or semi-direct radiative effects for BC at a regional level (e.g., Bauer and Menon, 2012).

2.6.1.4 Snow and Ice Albedo Forcing

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

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[11] The IPCC definition of "level of understanding" is a qualitative measure based on a combination of the quantity of evidence available and the degree of consensus in the literature.

[12] Bauer and Menon estimated the indirect cloud albedo effect as -0.17 W m-2 and the semidirect effect as -0.10 W m-2, but noted these effects "can be isolated on a regional scale, and they often have opposing forcing effects, leading to overall small forcing effects on a global scale."

There are a number of estimates of the magnitude of radiative forcing due to the snow albedo effect (see Figure 2-14). In a modeling study, Hansen and Nazarenko (2004) estimated the global average radiative forcing of BC on snow and ice to be +0.16 W m^{-2} for what they considered to be the most realistic of the four cases that were simulated in their study. In later work, Hansen et al. (2007b) lowered this estimate to +0.05 W m^{-2}, with a probable range of 0 to +0.1 W m^{-2} (Hansen et al., 2007b). Relying on these studies, the IPCC (Forster et al., 2007) adopted a best estimate for the global average radiative forcing of deposited BC on snow and ice of +0.10 (±0.10) W m^{-2}, although the authors acknowledged a low level of scientific understanding regarding this effect. In more recent work, Flanner et al. (2007) estimated the average forcing of BC on snow and ice (from fossil fuels and biofuels) at +0.043 W m^{-2}, of which +0.033 W m^{-2} was attributed to BC from fossil fuels. When biomass burning was included in the calculation, the forcing of BC on snow and ice was estimated to be approximately +0.05 W m^{-2}. Bond et al. (2011) estimated a global forcing of +0.047 W m^{-2}, of which 20% was calculated to occur in the Arctic (defined as north of 60 degrees), and suggested that more mechanistic studies in general yield estimates lower than the central IPCC estimate of +0.1 W m^{-2}. In line with these more recent studies, the UNEP/WMO assessment (UNEP and WMO, 2011a) estimated 0.05 W m^{-2}, with a range of 0.01 to 0.10 W m^{-2}, for the snow and ice albedo effect.

Hansen et al. (2007b) also investigated the "effectiveness" (or "efficacy") of the snow albedo forcing. This is a relative measure of positive feedback effects that occur with BC, compared to the feedbacks that occur with warming due to CO_2 forcing. They calculated that the radiative forcing from decreases in surface albedo is 2.7 times more effective at warming than radiative forcing from CO_2. This is a result of the energy absorption from the BC being directly applied to melting snow rather than spread throughout the height of the atmosphere. BC particles left behind in melting surface snow can concentrate and further reduce the surface albedo (see section 5.6). Furthermore, BC deposited on ice and snow will continue to have radiative effects as long as the BC remains exposed (i.e., until the snow melts away or fresh snow falls). Melting snow can expose a dark surface, leading to a positive feedback. Flanner et al. (2007) found a larger efficacy of 3.2, with an uncertainty range of 2.1 to 4.5. Based on the more recent work of Flanner (2009) and Koch (2009), the UNEP/WMO assessment (UNEP and WMO, 2011a) adopted an even larger effective forcing of a factor of five. Flanner et al. (2011) also found that observed Northern Hemisphere snow retreat between 1979 and 2008 (from all causes)

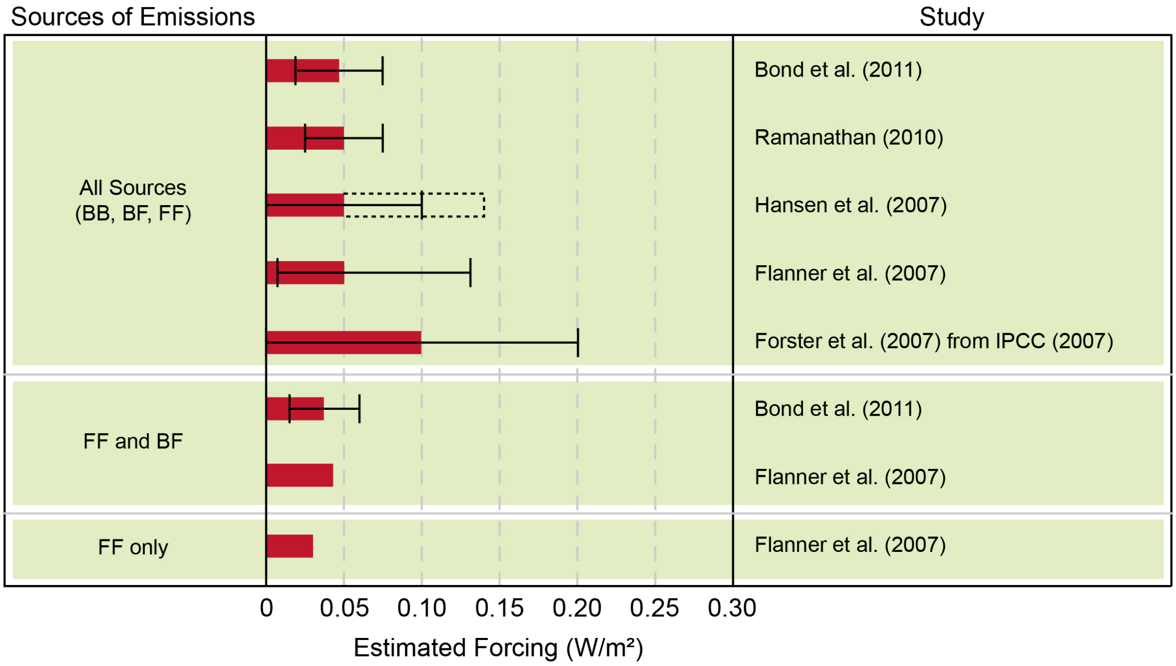

Figure 2-14. Estimates of Snow and Ice Albedo Radiative Forcing Effects from BC Emissions Only. Note: BB = open biomass burning; BF = biofuels; and FF = fossil fuels. Dashed lines indicate the estimated range of snow and ice albedo radiative forcing when forcing efficacy is considered. Hansen et al.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

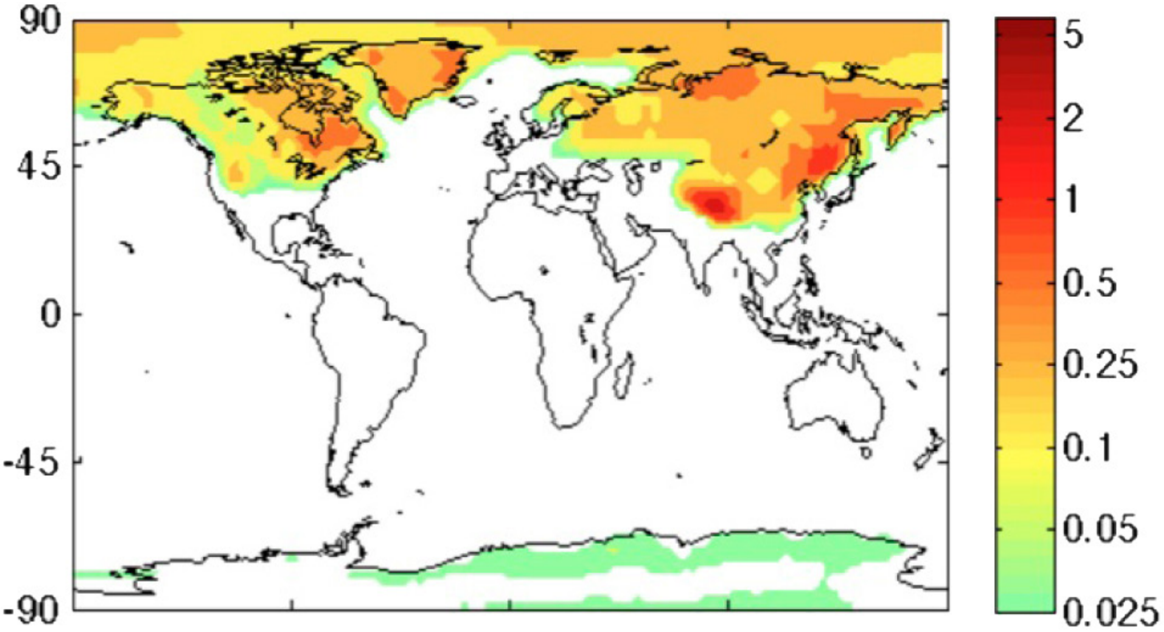

Figure 2-15. Snow and Ice Albedo Forcing by BC, simulated with the Community Atmosphere Model. (Bond et al., 2011)

would be consistent with a total albedo feedback on the order of +0.45 W m^{-2}. This suggests that albedo feedback is a larger process than represented in most climate models.

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

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

2.6.1.5 The Radiative Forcing Effects of OC and other Co-Pollutants

Although BC is mixed with other pollutants, both at the point of emission and in the atmosphere, most studies examine the impact of different types

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[13] Instantaneous radiative forcing refers to the flux at the tropopause, rather than forcing averaged over a longer time period.

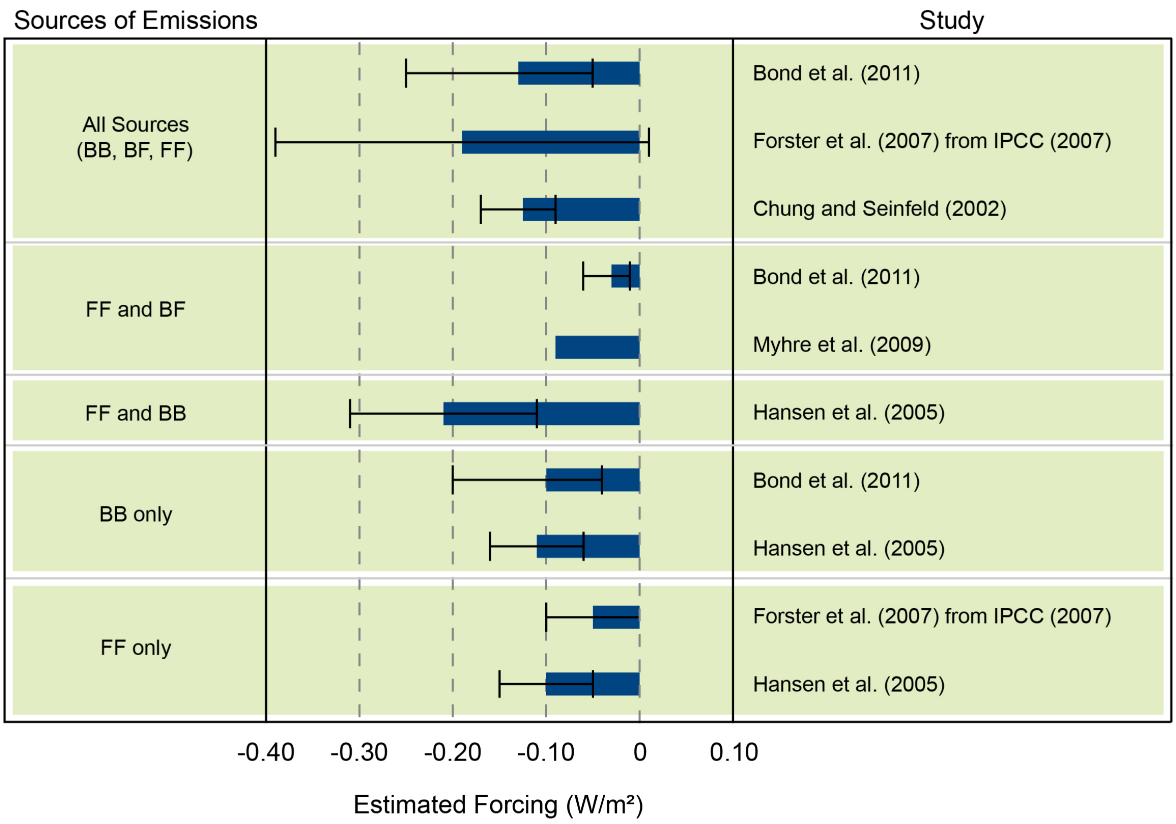

Figure 2-16. Estimates of Direct Radiative Forcing from OC Emissions Only. Note: BB = open biomass burning; BF = biofuels; and FF = fossil fuels. (Source: U.S. EPA)

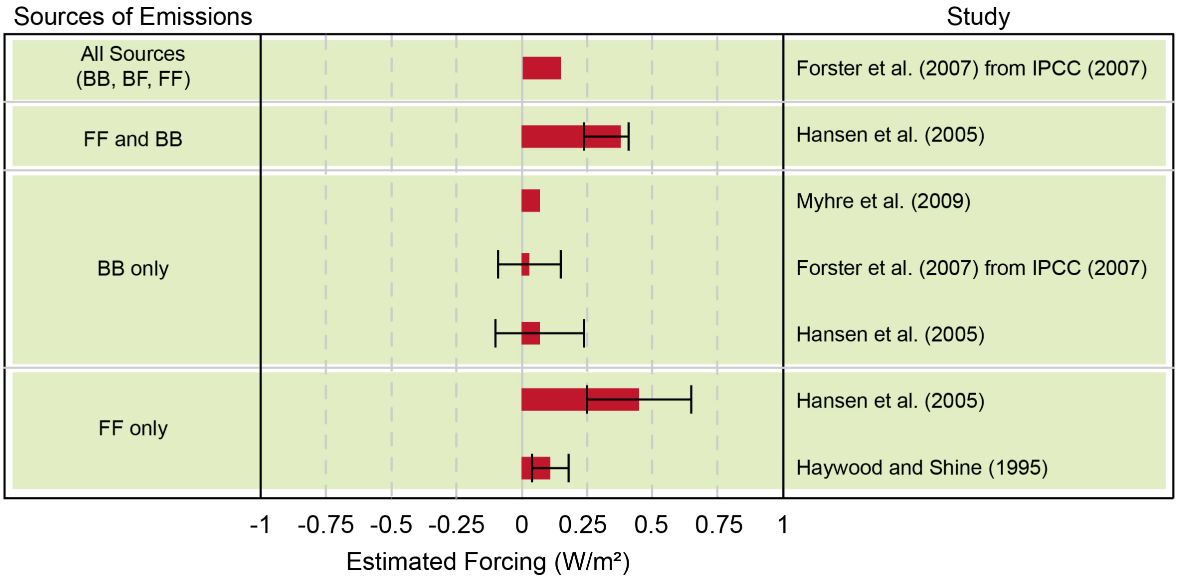

Figure 2-17. Estimates of Direct Radiative Forcing from BC and OC Emissions. Note: BB = open biomass burning; BF = biofuels; and FF = fossil fuels. Forster et al. (2007) – from IPCC (2007) – estimate the uncertainty surrounding estimates of direct radiative forcing from BC and OC independently. For this reason, the uncertainty surrounding the combined estimated direct radiative forcing from BC and OC emissions from all sources according to Forster et al. (2007) is omitted. (Source: U.S. EPA)

of aerosols in isolation. Only a limited number of studies consider the impacts of co-pollutants, and most of these studies have focused on OC rather than all aerosol species. Figure 2-16 shows estimates of direct radiative forcing for OC from a number of studies. As indicated in Figure 2-16, OC emissions from all sources are estimated to have net cooling impacts. For example, the IPCC (2007) estimated the negative direct radiative forcing of OC aerosols from all sources at -0.19 (±0.20) W m^{-2} and from fossil fuel alone at -0.05 (±0.05) W m^{-2}.

When OC and BC emissions are combined, the estimates of global average direct radiative forcing are generally positive. Figure 2-17 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 W m^{-2} (global average), and even biomass burning aerosols were estimated to have a positive net forcing of +0.03 (±0.12) 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 W m^{-2} (Bond et al., 2011). The UNEP/WMO assessment reports a central net value of BC and OC forcing (including the snow albedo efficacy factor) of 0.41 W m^{-2} (UNEP and WMO, 2011a).

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

The inclusion of other species, mainly nitrate and sulfate aerosols, also tends to reduce the estimate of net forcing. In particular, the presence or absence

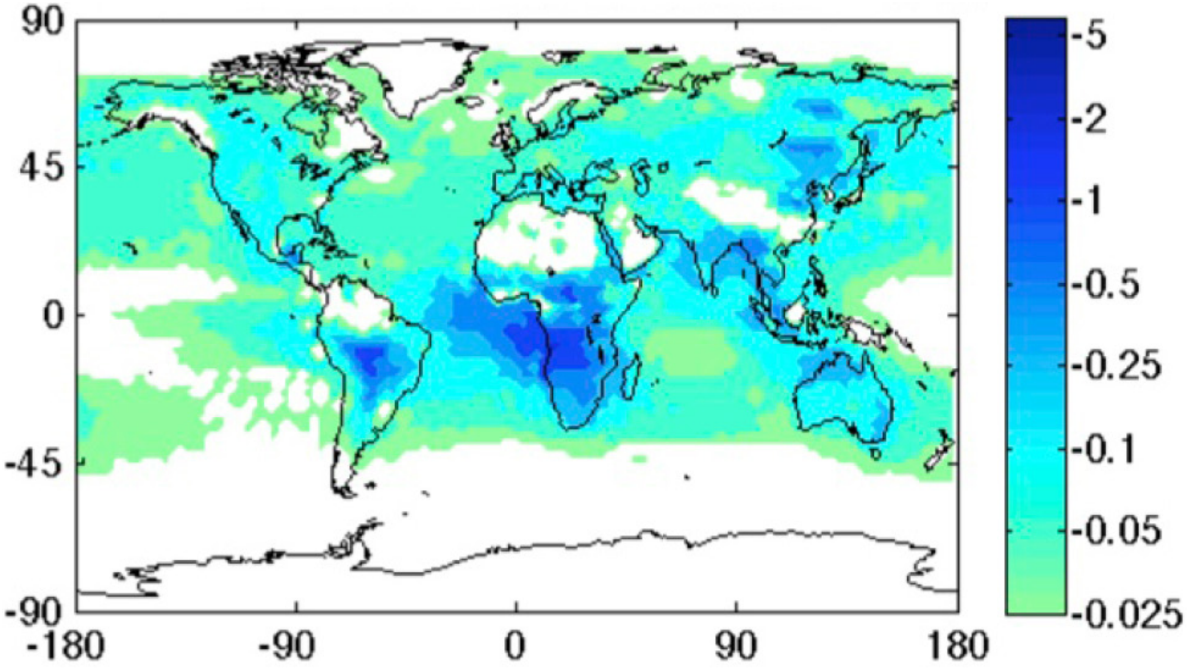

Figure 2-18. Direct Forcing by OC from All Sources, simulated with the Community Atmosphere Model. (Bond et al., 2011)

of sulfates and nitrates, which together comprise a large fraction of aerosol mass, in calculations of indirect effects can dominate radiative forcing calculations. Inclusion of both direct and indirect effects of aerosol species in the review by Ramanathan and Carmichael (2008) led to an estimate of the total aerosol effect including direct and indirect effects of -1.4 W m^{-2}, in contrast to a calculated BC direct forcing of +0.9 W m^{-2}. However, because much of the nitrate and sulfate precursor emissions come from sectors that are not rich in BC, the net global effect of aerosols can be less important than the estimates of the net effects of aerosols from a specific sector or measure (discussed further in section 2.6.1.6). These aerosols also play a role in the mixing state and therefore the direct radiative forcing effect of BC, as discussed in sections 2.5 and 2.6.1.2. Therefore, ambient concentrations of these other aerosols can be important in determining the influence of BC reductions. Using surface and aircraft measurements, Ramana et al. (2010) found that the ratio of BC to sulfate was important in determining the net warming or cooling impact of pollution plumes in China.

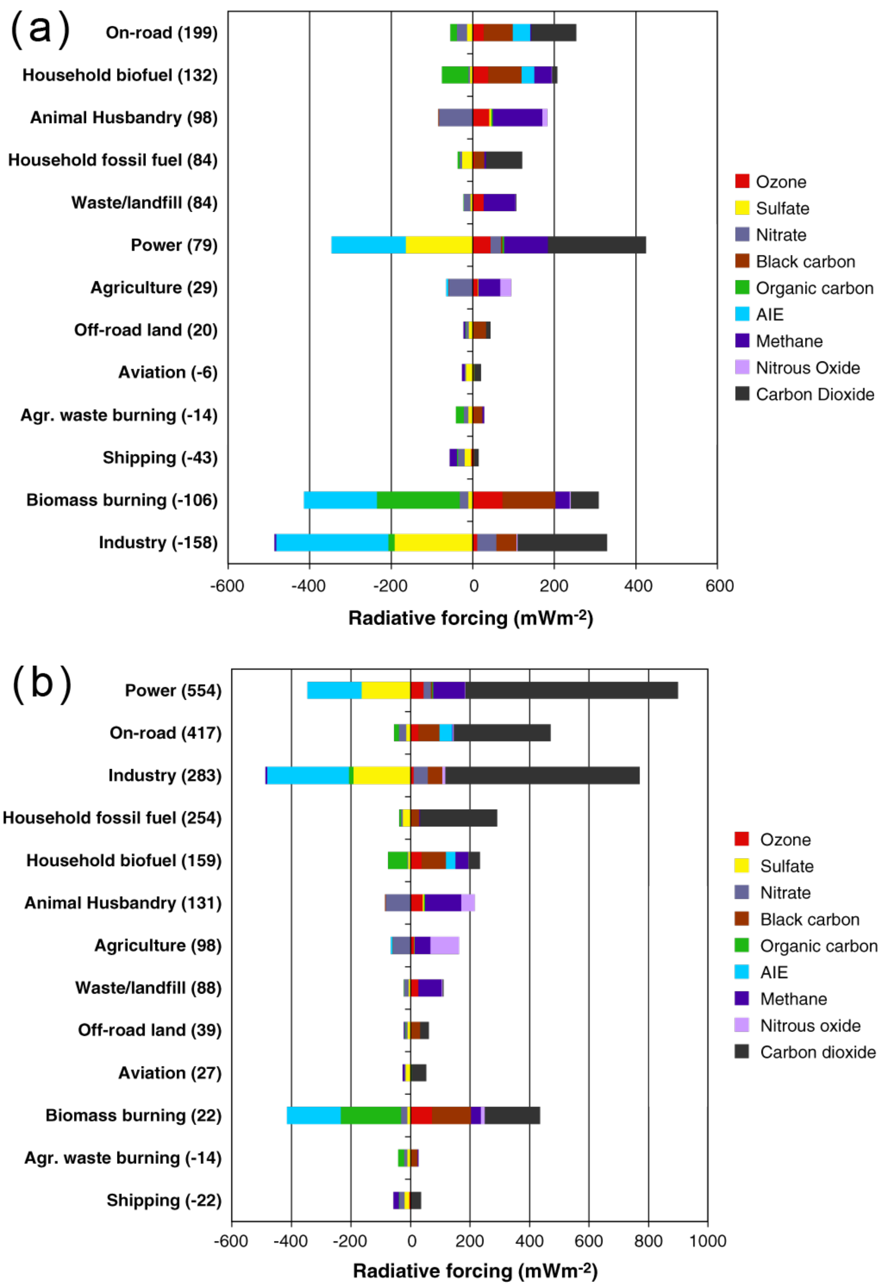

Figure 2-19. Global Radiative Forcing Due to Perpetual Constant Year 2000 Emissions, Grouped By Sector, at (a) 2020 and (b) 2100 and Showing the Contribution from Each Species. The sum is shown on the title of each bar, with a positive radiative forcing means that removal of this emissions source will result in cooling. AIE is the aerosol indirect effect. (Unger et al., 2010)

2.6.1.6 Sector-Based Contributions to Radiative Forcing

As described in Chapter 4, BC emissions can be attributed to a wide range of sectors (e.g., transportation, residential, industrial, and biomass burning). Some studies have attempted to quantify the radiative forcing effect of emissions mixtures containing BC and other co-pollutants by estimating the radiative forcing of defined emissions sectors. Comparisons among studies, however, is hindered in part by variation in several parameters, including estimates of the sector-level contributions, the relative fraction of warming and cooling aerosols, and the microphysical properties of these aerosols.

Unger et al. (2010) examined the impacts of sector-specific emissions on the short- and long-term radiative forcing from a range of pollutants. Figure 2-19 shows that the mixture of emissions from some of the largest BC emissions sources contributes considerably to total radiative forcing. On-road transportation emissions are the largest contributor to radiative forcing in the short term (by 2020), due to a combination of GHG and BC emissions. On-road transportation is also seen to be the second largest contributor in the long term (by 2100), but this is largely the result of the significant GHG emissions from this sector. Residential biofuel combustion is the second largest contributor in the short term due to the contribution from BC and CH_4. Since these sources have lower net GHG emissions, they contribute less to total global radiative forcing in the long term. However, these calculations have substantial uncertainties owing to the details of aerosol physics and chemistry, the interactions of aerosols and clouds, and the regional nature of the radiative forcing, as discussed earlier in this chapter. Bauer and Menon (2012) focused on regional differences in the impact of emissions from different source categories, and concluded that the largest opportunities to reduce positive forcing due to all aerosols included transportation in all regions, agricultural burning in Europe and Asia, and residential cooking and heating ("domestic sector") in Asia.

There is significant disagreement regarding the net impact of aerosol emissions from open biomass burning on radiative forcing. As noted in the previous section, the IPCC estimated the net direct radiative forcing impact from open biomass burning aerosols to be small, but positive at +0.03 (±0.12) W m^{-2} (Forster et al., 2007). However, because of uncertainties regarding the extent and composition of emissions from this source category, and the indirect radiative forcing effects of biomass burning aerosols, it is not clear if this sector has an overall global warming or cooling effect. Kopp and Mauzerall (2010) developed probability distributions from multiple studies to examine the likelihood of warming from individual sectors. Based on existing evidence, they concluded that open burning in forests and savannas is unlikely to contribute to warming, while the effect of open burning of crop residues remains uncertain. The results of current analyses are sufficiently different that there is no consensus on the likelihood of warming. Stohl et al. (2007) concluded that biomass burning has a "significant impact on air quality over vast regions and on radiative properties of the atmosphere" and in particular "has been underestimated as a source of aerosol and air pollution for the Arctic, relative to emissions from fossil fuel combustion." As discussed further in section 5.6, surface snow records indicate that biomass burning is currently a major source of BC in Greenland and the North Pole (Hegg et al., 2010). Bauer and Menon (2012) point to agricultural burning in Europe and Asia as contributing to positive net forcing in those regions. Additional work is needed to improve scientific understanding of the radiative forcing impacts of open biomass burning.

Several modeling experiments, such as Jacobson (2002, 2005, 2010), Hansen et al. (2005), and Schulz et al. (2006), and observationally constrained studies such as Ramanathan and Carmichael (2008), have found that carbonaceous aerosols from biofuel combustion and fossil fuel combustion both contribute to warming. Among fossil fuels, diesel combustion for transportation is the largest contributor to global BC emissions and several studies suggest these emissions may contribute to warming (Jacobsen 2002, 2005; Hansen et al., 2005). Kopp and Mauzerall (2010) concluded that carbonaceous PM emissions from gasoline combustion are unlikely to contribute to warming,[14] while diesel combustion and residential coal combustion are very likely to contribute to warming.

Kopp and Mauzerall (2010) also found mixed results with respect to the contribution of residential biofuel combustion for the models included in their assessment. More recent work by Rehman et al. (2011) has reported higher BC concentrations due to burning biomass for cooking than previously reported, both indoors and outdoors, in the study region in northern India. Moreover, Rehman et al. found that the albedo of the particles in the study villages indicated high absorption. Wavelength analysis suggested that though OC concentrations were a factor of five higher than BC concentrations, the OC included significant absorbing BrC.

Aviation is also a source of BC emissions. While the amount of BC emitted by aircraft at cruise altitudes is subject to large uncertainties (see Appendix 2-10), BC particles at these altitudes absorb not only the downward radiation but also the reflected upward radiation. In addition, intercontinental flight tracks are concentrated in the arctic stratosphere and particles emitted in this region may be deposited in the Arctic ice and snow. Research is needed to quantify radiative forcing due to BC of aviation origin.

A few studies highlight the substantial uncertainties regarding the contribution of biofuel combustion

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[14] 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 8 for more discussion of this issue.

and fossil fuel combustion to warming, given our limited understanding of how carbonaceous aerosols affect cloud processes. In the modeling experiments by Chen et al. (2010), reductions in fossil fuel carbonaceous aerosols (BC and OC) lead to decreases in CCN, leading to a decrease in cloud albedo, causing an increase in radiative forcing. The impact of these cloud changes equal or exceed the direct radiative forcing impacts. This result contrasts with that of Jacobson (2010) and Bauer et al. (2010) in which estimated warming from indirect effects did not exceed the direct and other radiative forcing from fossil fuel emissions. The distinction between biofuel combustion and fossil fuel combustion, in terms of the effect on radiative forcing, is particularly noticeable in the residential sector. Aunan et al. (2009) estimate that the global annual mean radiative forcing from BC from residential fuel consumption in Asia is positive for both biofuel and fossil fuel consumption, but that the net effects on radiative forcing from residential biofuel consumption in Asia (accounting for BC and the range of other co-emitted pollutants) is negative.

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

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

As mentioned in section 2.6.1, global average radiative forcing is linearly related to the global mean temperature at the surface (Forster et al., 2007). Radiative forcing from agents such as BC has similar effects on global mean temperature as radiative forcing from CO_2 and other GHGs (Hegerl et al., 2007), though the efficacy of the forcing may differ slightly (especially for the snow/ice albedo effect). Temperature itself has already been linked to a range of climate impacts as identified in, for example, the 2009 USGCRP report, "Global Climate Change Impacts in the United States." This and other recent climate change assessments describe the risks and impacts associated with climate change, including degradation of air quality, temperature increases, changes in extreme weather events, effects on food production and forestry, effects on water resources, sea level rise, disruption to energy consumption and production, and potential harm to ecosystems and wildlife. Though few studies explicitly link BC to all of these outcomes, to the extent that BC increases temperature it will contribute to these impacts, especially impairment of air quality and sea level rise (via melting of ice, snow, and glaciers).

Work by Jacobson often uses temperature change as an endpoint, rather than radiative forcing. As discussed earlier in section 2.6, there are advantages and disadvantages to using temperature change as an endpoint. Jacobson also includes a more complete suite of BC and co-pollutant effects than most other models. Jacobson (2010) found that the net effect of existing fossil fuel BC plus OC emissions was to warm the climate by 0.3 to 0.5°C at equilibrium compared to a case without those emissions.

There have been some efforts to translate regional direct radiative forcing estimates into regional changes in temperature. For example, Chung and Seinfeld (2005) used the GISS GCM model with BC emissions from Bond et al. (2004) to predict annually averaged changes in regional temperatures due to BC direct radiative forcing, based on simulations of 100 years of future emissions and temperature responses. They predict that externally mixed BC leads to an increase in average annual surface temperatures of 0.29°C in the Northern Hemisphere and 0.11°C in the Southern Hemisphere, when model results for the latter 75 years of simulations are averaged (the first 25 years of simulations were omitted from the results, to allow for the model to settle). Internally mixed BC is predicted to result in an increase in average annual surface temperatures of 0.54°C in the Northern Hemisphere and 0.20°C in the Southern Hemisphere over the same period. Despite the uncertainties regarding mixing state, the authors show that surface temperature response to regional direct radiative forcing is more concentrated in the Northern Hemisphere, especially in locations of high latitude.

Few studies have evaluated the North America-specific temperature impacts associated with BC emissions. However, Qian et al. (2009) found that BC emissions lead to warming of a tenth to a full degree Celsius over snow in the western United States. Simulations show that BC absorption of solar radiation in the atmosphere leads to as much as 0.6˚C of warming in the lower and mid troposphere over most of North America, including the Arctic region (Ramanathan and Carmichael, 2008). Additional estimates of regional temperature effects associated with BC emissions in the Arctic and the Himalayas are discussed in sections 2.6.4 and 2.6.5, respectively.

The snow/ice albedo effects of BC have been linked to accelerated melting of snow and ice. While many glaciers around the world and Arctic sea ice have receded in recent decades, attribution of this melt to BC is challenging due to other global and local contributions to warming and precipitation changes. Regardless of the deposited BC, other factors – including the solar zenith angle, cloud cover, snow grain size, and depth of the snow – also influence the snow and ice albedo (Wiscombe and Warren, 1980). The most common method of determining the contribution of BC to accelerated snow and ice melt has been to compare model runs with and without BC influences, and evaluate with observations. Direct measurements are generated by melting and then filtering samples of snow and ice. The filters provide an estimate of BC concentration by comparing their observed optical transmissions to optical transmissions of known amounts of BC (Noone and Clarke, 1998; Warren and Clarke, 1990). The mass is then used to estimate or compare to measured snow albedo, calculating the influence of BC. Another approach has been to apply a known amount of soot to an area, and then compare the measured albedo and melting rate to a nearby clean plot of snow.

In the western United States, deposition of BC on mountain glaciers and snow packs produces a positive snow and ice albedo effect, contributing to the melting of snowpack earlier in the spring and reducing the amount of snowmelt that normally would occur later in the spring and summer (Hadley et al., 2010; Koch and Del Genio, 2010). This has implications for freshwater resources in regions of the United States that are dependent on snow-fed or glacier-fed water systems. In the Sierra Nevada mountain range, Hadley et al. (2010) found BC at different depths in the snowpack, deposited over the winter months by snowfall. In the spring, the continuous uncovering of the BC contributed to the early melt. A model capturing the effects of soot on snow in the western United States shows significant decreases in snowpack between December and May (Figure 2-20, Qian et al., 2009). Snow water equivalent (the amount of water that would be produced by melting all the snow) is reduced by 2-50 millimeters (mm) in mountainous areas, particularly over the Central Rockies, Sierra Nevadas, and western Canada. In addition, dust deposition on snow, at high concentrations, can have similar effects to BC. A study done by Painter et al. (2007) in the San Juan Mountains in Colorado observed a decrease in snow cover duration of 18-35 days as a result of dust transported from non-local desert sources. As the authors note, in the future, exacerbated dryness in desert and arid regions—

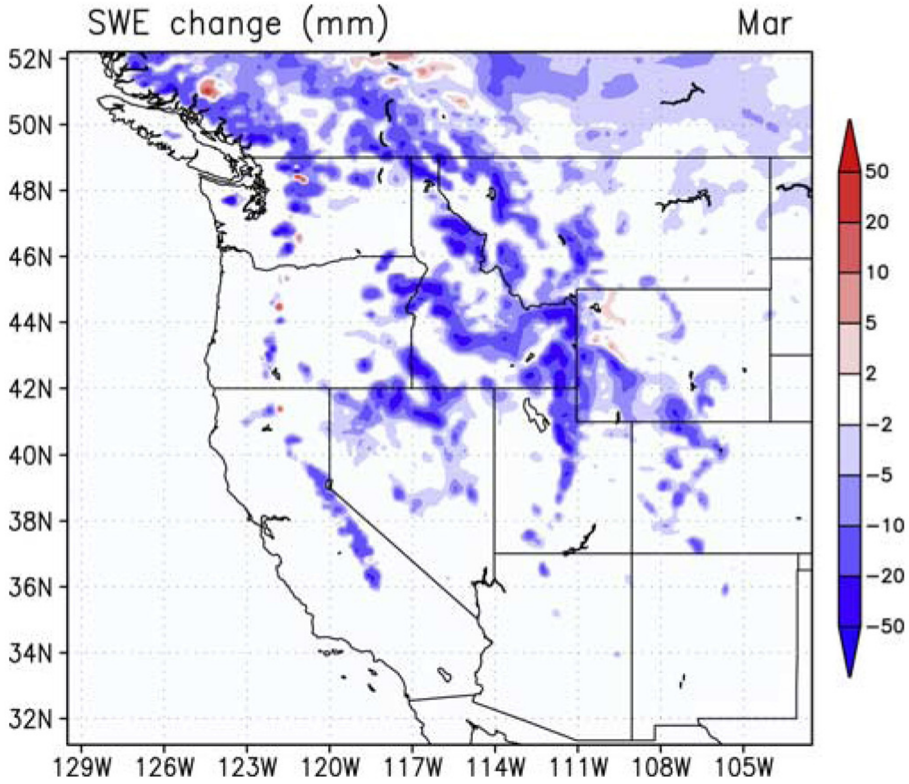

Figure 2-20. Spatial Distribution of Change in Mean Snow Water Equivalent (SWE, mm) for March. (Qian et al., 2009)

especially in the Southwest United States, the Middle East, and the Sahel—could cause increased deposition of dust on mountain snow cover in areas such the Rocky Mountains, the HKHT, and the Alps, respectively, which could lead to considerable reductions in snow cover duration over large areas.

Changes in snow and ice melt due to BC can 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 melting of these glaciers. Figure 2-21 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.

It is important to note that the impacts of BC on snow and ice albedo are not constrained to regions of high elevation or high latitude. BC deposition can also contribute to accelerated melting of seasonal, non-mountain snow, especially in mid-latitude regions due to the additional exposure to sunlight these regions experience (compared to polar regions). In addition, these regions are generally closer to BC emissions sources than polar regions are, so concentrations of BC on seasonal snow accumulations can be considerable (Huang et al., 2011).

2.6.3 Other Impacts of BC

In addition to warming and cooling effects due to absorption and reflection of light both directly and through cloud interactions, BC and other aerosols contribute to climate change through surface dimming and changes in precipitation patterns. The following sections provide information on BC's contributions to these two types of impacts. It is important to note that unlike BC and other aerosols, 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 TOA 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

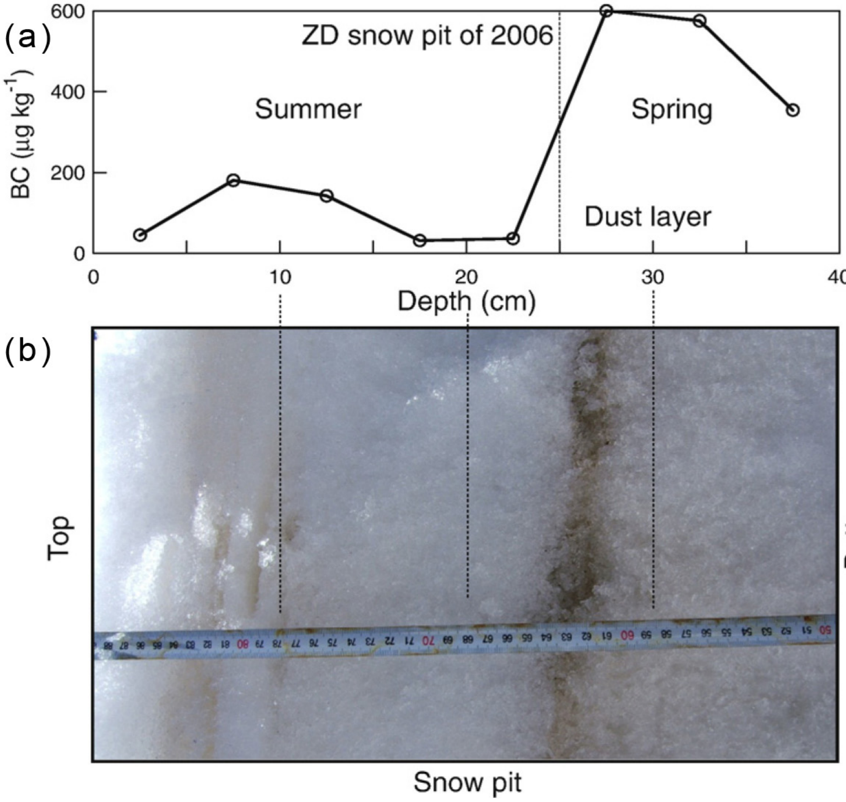

Figure 2-21. BC Concentrations in the ZD Glacier. (a) Measured BC concentration (b) the snow pit of the ZD Glacier. Dust layer at 30 cm indicated the spring (melting season) of 2006. (From Ming et al., 2009, Figure 4)

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 emissions over time and the interaction of aerosol direct and indirect radiative forcing (Stanhill and Cohen, 2001; Wild et al., 2005; Streets et al., 2006; Ruckstuhl et al., 2008). See Table 2-5 for a summary of how aerosol interactions affect surface dimming.

Estimating the magnitude of the surface dimming effect is complicated, largely because the vertical distribution of BC in the atmospheric column affects the impacts at the surface. Isolating the effect of BC is also difficult due to the interactions of non-BC aerosols. For example, many non-BC aerosols (primarily sulfates) scatter incoming solar radiation, reducing the energy reaching the surface (Dwyer et al., 2010; Ramanathan and Carmichael, 2008). In addition, the indirect effect of aerosols on cloud albedo and cloud lifetime may decrease solar radiation at the surface (Ramanathan and Carmichael, 2008). Furthermore, surface cooling combined with atmospheric heating from BC may increase the stability of the boundary layer (e.g., the bottom layer of the troposphere that is in contact with the surface of the earth) and reduce vertical mixing. This increase in atmospheric stability reduces natural removal processes for air pollutants, resulting in worse air pollution episodes (Ramanathan and Carmichael, 2008).

In some regions, BC, BrC, sulfates, organics, dust and other components combine to form pollution clouds known as ABCs, which have been linked to global dimming (Ramanathan and Carmichael, 2008; Ramanathan et al., 2007). Ramanathan and Carmichael (2008) estimate the total global dimming effect from ABCs to be -4.4 W m^{-2}, with about -3.4 W m^{-2} from the direct effect of aerosols (roughly half of which is attributed to BC) and the remaining -1 W m^{-2} from the indirect effect. In the Ramanathan and Carmichael study (2008), the -1.7 W m^{-2} of surface dimming from BC was found to be offset by +2.6 W m^{-2} of heating in the atmosphere. This resulted in a net TOA forcing estimate from this study of +0.9 W m^{-2}, as cited in section 2.6.1.2.

The dimming effects due to BC and the other aerosols are not spatially uniform (see Figure 2-22). A number of studies have found that dimming effects are particularly acute in certain regions associated with high aerosol pollution levels and the presence of ABCs. These include certain major urban areas (Ramanathan and Feng, 2009; Trenberth et al., 2007), primarily in South Asia (Ramanathan and Feng, 2009; Ramanathan et al., 2005; 2007). The ABCs which cover large areas in the North Indian Ocean and South Asia can reduce energy at the surface by 5-10% (Ramanathan et al., 2007; Ramanathan and Carmichael, 2008). Some studies have estimated that the dimming associated with ABCs can mask approximately half of the warming that would occur at the surface in the absence of ABCs due to GHGs alone, especially over Asia (Ramanathan et al., 2007; UNEP, 2008a). Another study estimates that surface dimming causes a reduction of approximately 6% in solar radiation at the surface over China and India when compared to pre-industrial values (UNEP, 2008a). The U.S. Global Change Research Program (CCSP, 2009) estimated surface forcing values as low as -10 W m^{-2} over China, India, and sub-Saharan Africa due to elevated optical depth from aerosol emissions in that region.

2.6.3.2 Precipitation Effects

Aerosols affect the processes of cloud and rain droplet formation. Some studies have linked aerosols to reductions in rainfall, but these interactions are not well understood. A summary of the aerosol interactions with clouds that cause

Table 2-5. 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"). For descriptions of the effects, see section 2.6.1.3. (Adapted from Denman et al., 2007, Table 7.10b.)

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

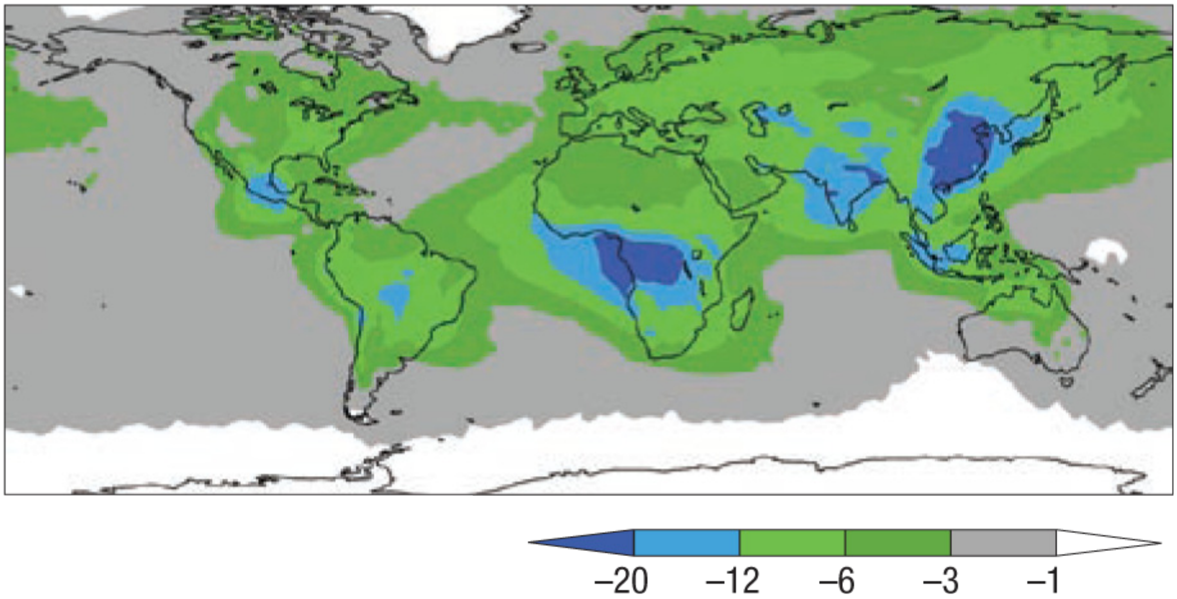

Figure 2-22. Surface Dimming by Anthropogenic Aerosols (W m^{-2}). (Adapted from Chung et al., 2005, and Ramanathan and Carmichael, 2008)

changes in precipitation is provided in Table 2-5. According to the IPCC (2007), the precipitation effects on a global scale attributed primarily to BC are from the semi-direct effect (described in section 2.6.1.3) and the increased atmospheric stability resulting from that effect (Ramanathan et al., 2005; Chung and Zhang, 2004; Menon et al., 2002). The increased stability inhibits convection, affecting both rainfall and atmospheric circulation. As discussed in section 2.6.1.3, increased availability of CCN increases cloud lifetime, thereby inhibiting rainfall for a time period, which may be more important for shifting the location of rainfall than changing net global precipitation. There may also be increases in precipitation: BC in particular can stimulate precipitation from ice clouds. However, because of the dependence of precipitation on complex and localized conditions, scientific understanding of these effects is low and models often disagree on the magnitude or sometimes even the sign of changes in precipitation due to factors such as warming or aerosol emissions.

Surface dimming due to all types of aerosols may reduce precipitation by reducing the energy available for evaporation from the Earth's surface (Liepert et al., 2004; Ramanathan et al., 2001). Bauer and Menon (2012) estimate that the global mean change in precipitation due to all aerosols is -0.3%. Because rain is a major removal mechanism for BC from the atmosphere, large decreases in rainfall could result in higher atmospheric concentrations of BC and other aerosols (Ramanathan and Carmichael, 2008; Ramanathan et al., 2005).

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 United States, 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 BC or any other 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 United States (Bell et al., 2008). Similar results have also been found for the East Coast of the United States (Cerveny and Balling, 1998).

There is stronger evidence linking aerosols to reduced precipitation in the tropics. Studies have indicated that surface dimming in this region reduces evaporation (Feingold et al., 2005; Yu et al., 2002a; Hansen et al., 2007b). 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 20° N) and decreased precipitation just south of the equator (between 0° and 20° S). 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 Southern Hemispheres (also a result of global warming from GHGs), which induces a change in circulation and convection in the tropics. Aerosols have also been linked to impacts on regional precipitation in the Amazon basin (Martins et al., 2009; Bevan et al., 2009). This is a region of high biomass burning emissions in the dry season. Further, seasonal biomass emissions have been linked to larger changes in atmospheric circulation patterns by affecting the global distribution of high-level clouds and convection precipitation (Jeong and Wang, 2010). Jeong and Wang (2010) also found that the climate response extends outside of the biomass burning season. The effects of BC aerosols on precipitation may also extend beyond areas of high concentrations. Wang (2007) found the largest change in precipitation occurs in the tropical Pacific region which is far from the regions of largest BC forcing. The effect may be very similar to the pattern of precipitation anomalies associated with the El Niño/Southern Oscillation.

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

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

2.6.4 BC Impacts in the Arctic

BC emissions that are transported to the Arctic are strongly linked to local warming (Reddy and Boucher, 2007), even if the globally averaged net climate impact of the total particulate emissions from individual sources is uncertain. For example, Quinn et al. (2008) calculated that the contribution of short-lived climate forcers (i.e., CH_4, tropospheric ozone, and tropospheric aerosols, including BC) to Arctic warming is about 80% that of CO_2. BC can have significant snow albedo effects and the magnitude of the cooling effect over snow from co-emitted aerosols is reduced in the Arctic. Because temperature in the Arctic has warmed at twice the global rate over the past 100 years (IPCC, 2007) and because of the dramatic retreat of summer sea ice extent during the satellite observation period (see Figure 2-23), there is interest in mitigation strategies that may slow the near-term rate of climate change in this region.

The estimated radiative forcing 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 GHGs (Quinn et al., 2008). Looking at forcing from fossil fuel and biofuel BC emissions, Quinn et al. (2008) calculated a radiative forcing in the Arctic of +1.2 W m^{-2} in the spring, +0.66 W m^{-2} in the summer, +0.16 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 W m^{-2} in the spring, +0.21 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.

Due to the frequency of strong temperature inversions that inhibit atmospheric mixing, and the prevalence of dry conditions that impede wet deposition, the lifetime of aerosol particles in the Arctic is longer than other regions – sometimes weeks, rather than days (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 of particles via deposition. Over a highly reflective surface like the Arctic, BC particles absorb solar radiation and warm the atmosphere above

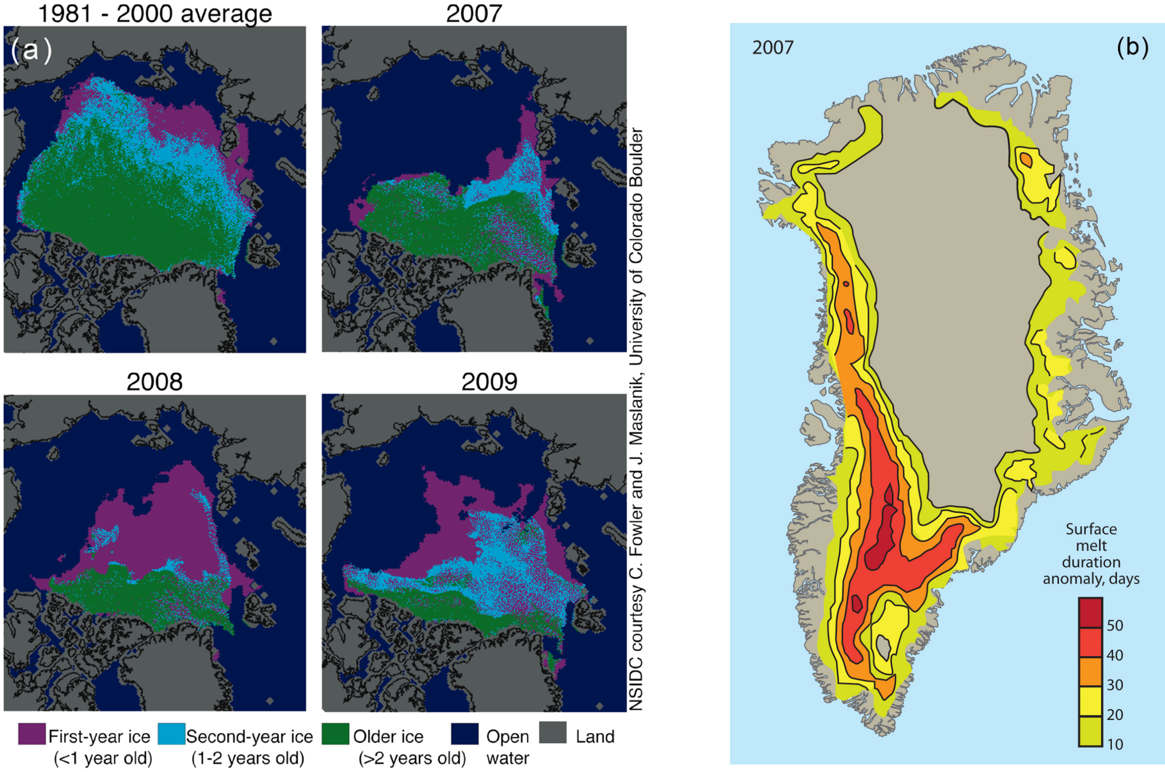

Figure 2-23. Evidence of Arctic Ice Melt. (a) Extent of summer Arctic sea ice for 2007-2009 compared to 1981-2000 average. (Source: National Snow and Ice Data Center, *http://nsidc.org/news/press/20091005_minimumpr.html*) (b) Duration of summer surface melt on Greenland in 2007 relative to 1973-2000 average (Source: Arctic Monitoring and Assessment Programme, 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 toward 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, increasing sunlight absorption. Recent work suggests a link between Arctic sea-ice melt and increased glacier runoff in Greenland (Rennermalm et al., 2009).

and within the haze layer, while simultaneously contributing to surface dimming. Rather than a cooling effect from surface dimming, however, the atmospheric heating increases the downward longwave radiation and causes warming at the surface (Shaw and Stamnes, 1980; Quinn et al., 2008; Mauritsen et al., 2011). Any warming particle above a highly reflective surface can lead to heating of the entire surface–atmosphere aerosol column. In addition, the stable atmosphere above the Arctic prevents rapid heat exchange with the upper troposphere, increasing surface warming in the Arctic (Hansen and Nazarenko, 2004; Quinn et al., 2008).

Radiative forcing from both atmospheric concentration and deposition on the snow and ice has contributed to the surface temperature warming in the Arctic (Quinn et al., 2008). Simulations by Flanner et al. (2007) suggest that the deposition of BC from sources in North America and Europe on Arctic sea ice may have resulted in a surface warming trend of as much as 0.5 to 1°C. Similarly, Shindell and Faluvegi (2009) found 0.5 to 1.4°C of warming from BC in the Arctic since 1890. For the BC snow albedo effect, Quinn et al. (2008) estimated a warming of 0.24 to 0.76°C, varying by season. Warming due to BC heating in the atmosphere is estimated to be a further 0.24°C in spring, 0.15°C in summer, and nearly zero in autumn and winter. In Table 2-6, we show estimates of temperature increases in the Arctic from various BC emissions sources (Shindell and Faluvegi, 2009; Flanner et al., 2007; Jacobson, 2010). Part of these increases in temperature may also have been "unmasked" in recent years from reductions in sulfate aerosols and its gaseous precursor, SO_2 (Shindell and Faluvegi, 2009). While sulfate aerosols have a negative

Table 2-6. Arctic Temperature Impacts from Emissions of BC 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 globally (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 as above plus CO_2, CH_4	Same as above	Jacobson (2010)	
Arctic	Indirect > 0	100% of All sources	All aerosols	Idealized calculations, observations	Mauritsen et al. (2010)	Indirect only, no direct or snow albedo effect.

FF = fossil fuel; BF = biofuel; BB = biomass burning.

radiative forcing, the reductions in sulfate aerosols have been strongly justified by improvements in air quality, acid rain, visibility, public health, and lessening of direct effects of sulfates on ecosystems. It has also been suggested that the potential cooling effects of BC—such as indirect radiative forcing and the ratio of BC to cooling components (e.g., OC)—may not be as important in the Arctic since the snow and ice albedo darkening is so dominant (Mauritsen et al., 2010).

While there are strong qualitative indications of Arctic snow and ice melt from BC, quantitative studies have only recently entered the peer-reviewed literature. Some studies have linked the local warming measured on the Greenland Ice Sheet to observations of a gradual loss of ice, and modeled the overall impact on the mass balance of the ice sheet. Box et al. (2004), for example, estimated the modeled ice sheet mass balance at -76 km^3 per year, leading to a 0.24 mm sea level rise per year (contributing 15% of global sea level rise) during 1991-2000. Hanna et al. (2005) considered a longer time period, and estimated that the overall mass balance declined at a rate of -22 km^3 per year in 1961-1990 and -36 km^3 per year for 1998-2003, with melting during the past six years contributing 0.15 mm per year to global sea level rise. Finally, Thomas et al. (2006) reported accelerating mass loss between an earlier period (between 4 and 50 Gigatons (Gt) per year, depending on the model, from 1993-1999) and a more recent period (between 57 and 105 Gt per year, from 1999-2004). In a modeling study by Flanner et al. (2007), land snowmelt rates north of 50°N latitude (about 70 miles north of the U.S./Canada border in Minnesota) increased by 28% in 1998 and 19% in 2001 in the month preceding maximum melt when compared to control runs that did not include BC from large boreal fires that occurred in 1998 and 2001. Strack et al. (2007) found soot deposition in the Alaskan Arctic tundra created snow-free conditions five days earlier than model runs without BC deposition. Ongoing studies will help evaluate and constrain modeling simulations. Importantly, American, Norwegian, Russian, and Canadian research groups collaborated under the International Polar Year (2007-2008) program to survey BC concentrations in snow and ice north of 65°N latitude in both the Eastern and Western Arctic (Doherty et al., 2010).

The location of the emissions also matters for the magnitude of the effects in the Arctic, which has important implications for mitigation decisions. A recent study by the Arctic Monitoring and Assessment Program (Quinn et al., 2011) analyzed the radiative forcing impacts of BC emissions from different regions on the Arctic (direct and snow/ice albedo only). This study found that compared to the average emissions of BC from regions between 40°N and 50°N latitude, emissions of BC from between 50°N and 60°N latitude had about three times as much forcing impact in the Arctic on a per-ton basis. In addition, emissions from north of 60°N had seven times as much impact per ton. However, because total emissions are much larger in the southern regions, almost half of the total impact on Arctic forcing due to BC in this study was derived from emissions from regions below 50°N, and most of the remainder was derived from emissions from regions between 50°N and 60°N.

BC emissions from near-Arctic countries have decreased since their peak in the early 20th century. This is supported by a downward trend in the observed concentrations of ambient and snow/ice BC in the Arctic (see section 5.6). However, BC deposited on areas covered permanently with ice and snow, such as the Greenland Ice Sheet, can remain in the ice sheet for hundreds of years, as seen in historical ice core records (Quinn et al., 2008). Although the sunlight that reaches the snow surface typically only penetrates between 10 and 20 cm deep, with the topmost 5 cm of snow receiving the most sunlight (Galbavy et al., 2007), as the Arctic warms and snow and ice melt, deeper, hidden BC that was deposited over decades may become exposed, enhancing the melting of snow and ice. The effect of BC on the snow and ice albedo in the Arctic thus can involve historical—in addition to present-day—BC deposition in the Arctic region.

An important uncontrolled source of BC in some near-Arctic countries is open biomass burning. Several recent studies have looked at the effect of these emissions on the Arctic. For example, Stohl et al. (2006) found that North American boreal forest fires lead to elevated concentrations of light absorbing aerosols including BC throughout the entire Arctic, with substantial implications for Arctic warming and enhanced snow albedo effects. Analyses conducted by Hegg et al. (2010) suggest that the dominant source of light-absorbing aerosols (including BC) in the Arctic region is biomass burning. Other studies have linked open biomass burning to reduced surface albedo and accelerated melting (Hegg et al., 2009; Generoso et al., 2007; Kim et al., 2005). Following agricultural fires in Eastern Europe in spring 2006, Stohl et al. (2007) measured record high air pollution levels and BC concentrations in parts of the Arctic above Europe. Similarly, in a series of studies, Warneke et al. (2009; 2010) found that spring fires in Russia (Siberia) and Kazakhstan can more than double the Arctic haze that builds up during the winter months.

2.6.5 BC Impacts in the Himalayas

The world's third largest snowpack after Antarctica and the Arctic is found in the Hindu Kush-Himalayan-Tibetan (HKHT) region. The mountain ranges that define this region fall primarily along the borders of Pakistan, Afghanistan, India, Nepal, and China (UNEP, 2008a). It is often referred to as the Earth's "third pole." Atmospheric warming associated with BC is believed to be a significant factor in the observed increases in melting rates of glaciers and snowpack in the HKHT (Barnett et al., 2005; Lau and et al., 2010; UNEP, 2008a; Thompson, 2010). Ramanathan and Carmichael (2008) and Ramanathan et al. (2007) suggested that the advection of air warmed by BC over the Himalayas has played a role comparable to that of GHGs in the observed retreat of Himalayan glaciers. A recent study by Carmichael et al. (2009) also shows that BC throughout Asia has an atmospheric warming potential of about 55% of that attributed to CO_2.

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

BC can alter snowpack and glacier extent and retreat through two mechanisms, the first being increasing and decreasing precipitation as discussed in section 2.6.1.4, and the second being local warming, especially through deposition, increasing the rate of melt. Lau et al. (2010) found that heating of the atmosphere by dust and BC leads to widespread enhanced warming over the Tibetan Plateau and accelerated snowmelt in the western Tibetan Plateau and Himalayas. Menon et al. (2010) show observed trends in snow cover in the Himalayas, with a spatially heterogeneous pattern of decreases and increases of up to 17% from 1990 to 2001, where the area of decreases is much larger than the area of increases. Menon et al. (2010) simulated similar heterogeneous snow cover changes due to aerosol emissions, showing that the influence of the aerosols was larger than the influence of changing sea surface temperatures over that time period. Over Eurasia, Flanner et al. (2009) conducted a modeling study that found the combination of strong snow albedo feedback and large fossil fuel and biofuel emissions of BC and organic matter from Asia induce 95% as much springtime snow cover loss as anthropogenic CO_2 alone. The effects on glaciers are not well quantified, but Xu et al. (2009a) found evidence that soot deposited on Tibetan glaciers has been a significant contributing factor to observed rapid glacier retreat. Changes in the timing and extent of melting may adversely affect regional freshwater resources in this region, which relies heavily on this melt (Carmichael et al., 2009).

2.6.6 Summary of BC Impacts in Key Regions

As described in the previous sections, the climate-related effects of BC can vary considerably across regions. Table 2-7 provides an overview of the regional variability in terms of BC's effects on radiative forcing, temperature, precipitation, and snow and ice across the United States, Asia, and the Arctic. In addition, Figures 2-13 and 2-15 are useful for understanding the regional variability of BC's radiative forcing effects.

2.7 Metrics for Comparing Black Carbon Impacts to Impacts of Other Climate Forcers

In response to Congress's request for an assessment of potential comparative metrics, this section summarizes a number of different approaches to comparing the effects of BC to CO_2 and other GHGs, but cautions that there is no one "best" metric; rather, the utility of a metric depends on the desired environmental outcome and policy objective. Therefore, this section begins by introducing the concept of using metrics for comparing BC-related

Table 2-7. Climate Effects of BC in the United States, Asia, and the Arctic (Summary).

Effects	U.S.	Asia	Arctic
Radiative Forcing Effects	• Estimates of direct radiative forcing of BC over the United States range from 0.1 to 0.7 W m^{-2}.	• 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}.	• Springtime Arctic forcing has been estimated to be 1.2 W m^{-2} (direct) and 0.53 W m^{-2} (snow albedo).
Temperature Effects	• No studies were identified for U.S. temperature effects from BC. All global modeling studies include the temperature effects over the U.S., but results are difficult to extract. • Estimates of average warming from BC in the Northern Hemisphere range from 0.29°C to 0.54°C.	• Over the Himalayan region, atmospheric BC was estimated to result in up to 0.6°C of warming.	• 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	• One study found little change in the amount of precipitation in the western United States as a result of BC effects. • Other studies have found that rainfall patterns in the eastern United States match PM emissions, but not specifically those of BC.	• 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.	• No studies were identified for Arctic precipitation effects.
Snow and Ice Effects	• In the western United States, BC deposition on mountain glaciers and snow produces a positive snow and ice albedo effect, contributing to the snowmelt 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.	• BC atmospheric warming is believed to be a significant factor in the melting of the HKHT glaciers and snowpack. • The deposition of BC on glaciers and snowpack 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.	• BC may increase snowmelt rates north of 50°N latitude by as much as 19-28%. • Soot deposition in the Alaskan Arctic tundra created snow free conditions five days earlier than model runs without BC deposition.

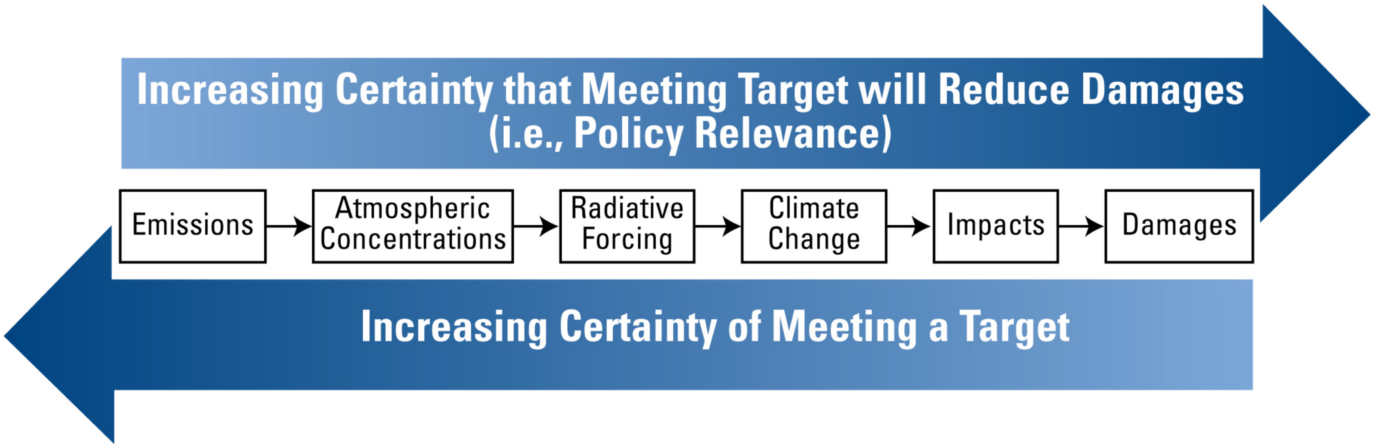

Figure 2-24. Cause and Effect Chain from Emissions to Climate Change, Impacts, and Damages. (Adapted from Fuglestvedt et al., 2003.) The arrows indicate that a policy could focus on different elements along the causal chain and, depending on whether the policy focuses on the emissions or damages end of the chain, can determine the certainty of meeting the stated policy target versus the certainty of reducing damages at issue.

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., CO_2 versus CH_4), 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 (e.g., in a market-based climate program) so that multi-pollutant emitters can compose mitigation strategies (Forster et al., 2007).

Climate metrics are often defined relative to a baseline pollutant (usually CO_2) 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 GWP metric is used to convert all GHGs into "CO_2-equivalent" units. Importantly, metrics such as GWP have been used as an exchange rate in multi-pollutant 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. That is to say, 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 (Fuglestvedt et al., 2010). 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 et al., 2007).

Metrics can also be used to prioritize among mitigation measures designed to control emissions of similar compounds from different sources. As described previously in this chapter, aerosols are composed of numerous components, and these different components can contribute to both warming (BC) and cooling. A metric can aggregate these effects in order to determine the relative contribution of a given source or measure.

2.7.1 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 2-24). Some of the links in this cause and effect chain may be simultaneous rather than sequential. For example, the atmospheric loading of aerosols affects dimming and precipitation directly, rather than mediated through radiatively induced temperature changes. Nor is the chain always unidirectional. Climatic changes can lead to changes in atmospheric concentrations of climate-forcing pollutants (e.g., changes in precipitation will change aerosol lifetimes) or even emissions of those pollutants (e.g., changes in temperature affect fossil fuel consumption for heating and cooling needs, which affects emissions of particles and precursors). There are uncertainties at each stage of the cause and effect chain, and these uncertainties compound over multiple steps of the chain. The uncertainties for BC are generally larger at all stages of the causal chain compared to the long-lived GHGs (for reasons discussed in this and other chapters of this report).

Within the climate change field, metrics have been calculated for changes in radiative forcing, global mean temperature, and monetized damages. The closer the metric is to the emissions end of the chain, the less uncertainty there is in how to calculate the metric; it is easier to determine how a change in emissions will change concentrations than it is to determine how a change in emissions will change temperature (a calculation which requires several intermediate steps). Additionally, the further along the chain, the more physical systems (and economic systems) need to be included in order to calculate the metric. However, if a reduction in damages is considered the ultimate objective of the policy, then a metric that focuses explicitly on impacts or damages best represents that objective. Since the economic value of damages (expressed in dollars) is one of the easiest metrics for the public and policymakers to place in context, there has been a great deal of interest recently in calculating the monetary value of climate change impacts associated with different pollutants (see Chapter 6). The choice of a metric can be considered in part a choice about how to allocate uncertainty between calculation of the metric and the representativeness of the metric for the ultimate impacts of interest.

Fuglestvedt (2009) identified the following considerations for developing a metric for climate forcers (see Table 2-8 for examples of how commonly used metrics address these considerations):

1. What climate impact is of interest for the policy being considered?
2. What climate forcer will be used as the baseline for comparison?
3. What is the temporal frame for emissions? Is it an instantaneous pulse or a sustained change in emissions?
4. What is the temporal frame for the impact? 10 years, 50 years, 100 years? Is the impact considered only at the end point of the time frame, or integrated over the period?
5. Does the metric address the magnitude of change or the rate of change or both?
6. What is the spatial dimension of the metric for both emissions and impacts? Is it global or regional?
7. What economic considerations should be taken into account? How are damages in the far future weighed compared to damages in the near term?

Table 2-8. 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-pulse (Global Temperature Potential)	Temperature	CO_2	Pulse	Global	No
GTP-sustained	Temperature	CO_2	Sustained	Global	No
STRE (Surface Temperature Response per unit continuous Emission)	Temperature	CO_2	Sustained	Global	No
SFP (Specific Forcing Pulse)	Energy	Joules/gram	Pulse	Global or regional	No
Cost-effectiveness Metrics (e.g., Manne and Richels, 2001, 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

First, the climate impact must be identified because the effectiveness of a given metric is dependent on the primary policy goal. Considerations 2 through 7 are then framed by the selected climate impact. This is important because choosing an inappropriate metric could lead to policy decisions that ultimately result in undesirable climate or economic impacts.

2.7.2 Commonly-Used Metrics for GHGs

Article 2 of the United Nations Framework Convention on Climate Change (UNFCCC) calls for a policy that addresses the magnitude and the rate of climate change as well as the cost effectiveness of controlling emissions (IPCC, 2009). Therefore, appropriate metrics could cover either the physical or economic dimensions of climate change, or both. A number of metrics have been developed and refined for application to CO_2 and other long-lived GHGs. These metrics are summarized in Table 2-8 and described further below. Their potential applicability to BC is considered in the next section. Note that two of the metrics listed in the table (SFP and STRE) were developed specifically for application to short-lived climate forcers like BC, and are discussed only in section 2.7.3.4 below.

Five considerations are listed in Table 2-8. The first, climate impact, refers to where the metric falls on the cause-effect chain shown in Figure 2-24. The second, baseline forcer, lists whether the metric is measured in comparison to CO_2, or in absolute units (whether dollars or energy). The third column notes what kind of emissions change is being considered. A "pulse" of emissions refers to an effectively instantaneous release of that pollutant (though sometimes that release is considered to be spread out over a year). A pulse analysis is appropriate for a one-time trading of emissions permits, but may not be as realistic for analyzing investment decisions which spread reductions out over time (though a longer term reduction can be approximated as a series of pulses). Therefore, other analyses consider the possibility that an emission reduction (or increase) will be permanent (i.e., sustained over time). The third temporal option is to calculate the optimal emissions path, which is discussed in more detail in section 2.7.2.3 (cost-effectiveness metrics). The fourth column shows that most metrics have been designed to be used on a global scale, though some of these might be adaptable for regional impacts. Finally, most metrics consider temperature change or damages either at a single point in time or summed over time: only a few consider that there may be value in limiting the rate of change in addition to reducing the absolute magnitude of the change.

Table 2-8 is also ordered in a rough approximation of the transparency of the metric. Metrics which are transparent and easy to calculate are likely to be more readily accepted for policy use than those which require complex modeling. The GWP is in widespread use and can be calculated based only on knowing the average lifetime of a molecule of a gas and the radiative forcing caused by that molecule. The remaining metrics require the use of computer models of more or less complexity in order to calculate, and if the metric is sensitive to assumptions involved in the modeling then that could have potential for controversy.

2.7.2.1 Global Warming Potential

To date, the most widely established and well-defined metric is the GWP. The definition of the GWP by the IPCC (2007) is

> *"An index, based upon radiative properties of well-mixed greenhouse gases, measuring the radiative forcing of a unit mass of a given well-mixed greenhouse gas in the present-day atmosphere integrated over a chosen time horizon, relative to that of carbon dioxide. The GWP represents the combined effect of the differing times these gases remain in the atmosphere and their relative effectiveness in absorbing outgoing thermal infrared radiation. The Kyoto Protocol is based on GWPs from pulse emissions over a 100-year time frame."*

The identified climate impact the GWP addresses is globally averaged change in radiative forcing and its baseline climate forcer is CO_2 (e.g., the GWP of CO_2 is defined to be 1).[15] The temporal frame for emissions is a pulse. The GWP provides the magnitude, but not the rate of change, of the integrated radiative forcing over a given time frame. The time frame is usually 100 years, but 20-year and 500-year GWPs are sometimes presented to show how GWPs would differ if short-term or long-term impacts are given more weight. Finally, the GWP, which addresses only radiative forcing, a physical metric, does not take into account any economic dimension.

As discussed below, there have been a number of criticisms of the GWP in the peer-reviewed literature (e.g., O'Neill, 2000; Shine, 2009), mainly focused on either the inability of the GWP to capture key differences between gases (such as different lifetimes) or the failure of the GWP to incorporate economic considerations. Despite such criticisms, at the time of the Kyoto Protocol in 1997, the GWP was

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[15] The GWP is calculated as the ratio of the Absolute Global Warming Potential (AGWP) of a given gas to the AGWP of CO_2. The AGWP has units of W m^{-2} yr g^{-1}.

adopted as the metric used in climate negotiation. While acknowledging that there are shortcomings involved in using GWPs even for comparisons among the long-lived gases, a recent IPCC Expert Meeting on the topic found that GWPs were still a useful measure for these gases (IPCC, 2009). It remains the most accepted metric due to its simplicity, the small number of input parameters, the relative ease of the calculation, and a lower level of uncertainty compared to some alternatives (Shine et al., 2005). The GWPs as calculated by the IPCC Second Assessment Report (Schimel et al., 1996) currently remain the standard GWPs used for the official U.S. GHG emissions inventory compiled annually by EPA, as required by UNFCCC reporting guidelines.[16]

2.7.2.2 Global Temperature Potential

One alternative metric that has received recent attention is the GTP. Like the GWP, the GTP is a physical metric. Whereas the GWP considers change in globally averaged radiative forcing, the GTP compares the globally averaged temperature change at a given point in time resulting from the emission of two climate forcers of equal mass (Shine et al., 2005). The GTP moves one step further down the cause and effect chain and addresses a climate response to radiative forcing, the global-mean surface temperature change. The GTP therefore includes more physical processes, such as the heat exchange between the atmosphere and ocean, than the GWP. This introduces more uncertainty to the metric, and can require the use of more complex models in order to calculate the GTP value. In addition, while the GWP represents the integrated radiative forcing of a pulse of emission over a given time period, the GTP is evaluated at a given point in time (IPCC, 2009). Like GWPs, the GTP can be calculated over a variety of timescales, with 20, 100, and 500 years being the timescales most commonly presented. There are advantages and disadvantages to using either the GWP or a GTP, and they may each address different policy goals and may be more relevant to different climate forcers and time frames, depending upon the policy need. To date, however, the GTP has not been used as a metric for trading gases in international, national, or regional accords.

There are two versions of the GTP: one that involves the effects of a pulse of emissions, and another that involves a sustained reduction of emissions. The latter version of the GTP results in comparative values between different gases that are similar to the values calculated using GWPs. The former version of the GTP, by contrast, leads to longer-lived gases being given more relative weight because a pulse of a short-lived gas has very little impact on temperatures many years in the future.

The GTP can also be calculated as a function of a future global temperature stabilization target. One criticism of a number of metrics is that they are not compatible with a goal of stabilization because the target is not part of the metric. Manne and Richels (2001) developed a methodology to calculate a time-dependent metric (referenced below as a cost-effectiveness metric) that would change as a target level was approached. For example, if the target is not to exceed a 2 degree temperature change above preindustrial, then when global temperatures are still only 1 degree above preindustrial, and therefore the target temperature is still decades away, the metric will place weight on long-lived gases like CO_2. But as the target temperature is approached, the time to reach that target becomes short, and the metric places weight on the strong, short-lived forcers like CH_4 and BC.

Shine et al. (2007) used a similar approach to develop a time-dependent GTP, the GTP(t). Shine et al. applied the GTP(t) to BC using a target of 2^{o}C, and found that for a low emissions scenario, GTP(t) starts at about 2 in 2010, rising to 1,000 by 2080. But for a high emissions scenario, GTP(t) can start at 200, reach 1,000 in 2030, and reach 20,000 in about 2045. While this approach is one of the few approaches that are truly compatible with a stabilization target, there are some drawbacks. Drawbacks include the dependence on assumptions about future emissions scenarios, the undefined nature of the metric *after* reaching the stabilization target, and the dependence of the metric on computer modeling, which reduces transparency. In addition, policymakers might not desire a metric whose value can change by orders of magnitude over several decades and without a transparent and predictable schedule. One advantage of the GTP(t) and related metrics is that they can easily be adapted to include a rate of change goal; for example, rather than just constraining the metric to reach a 2^{o} C target, it is also possible to value the rate of change as well by adding on a constraint that the temperature not increase more than a given amount in any given decade. Such an additional constraint would increase the value of short-lived substances like BC.

2.7.2.3 Cost-Effectiveness Metrics

Manne and Richels (2001) examined relative tradeoffs between different gases that vary over time and are calculated to optimally achieve a given target using a computer model that included economic considerations. Similarly, the Global Cost

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[16] See *http://www.epa.gov/climatechange/emissions/usgginventory.html*.

Potential (GCP), compares the relative marginal abatement costs for two climate forcers when a given climate change target is achieved at least cost (IPCC, 2009). These approaches define a temperature or radiative forcing target and calculate the relative (or absolute) dollar value that should be imposed on different gases in order not to exceed that target.

2.7.2.4 GHG Metrics for Measuring Economic Impacts

Two metrics, the Global Damage Potential (GDP) and the social cost of a pollutant, involve monetization of the damages of climate change (see detailed discussion in Chapter 6). The GDP compares the relative damage resulting from an equal mass of emissions of two climate forcers (IPCC, 2009). The social cost calculation has most commonly been used for CO_2 alone, where it is referred to as the Social Cost of Carbon (SCC). However, even where risks and impacts can be identified and quantified with physical metrics, it may be difficult to monetize these risks and impacts (e.g., such as ecosystem damage or the potential to increase the probability of an extreme weather event) such that an accurate cost-benefit comparison could be undertaken. Both the GDP and the social cost calculation depend on the physical aspects of the climate system as well as the economic linkages between climate change impacts and the economy (IPCC, 2009). Therefore, the GDP and the social cost require calculations of the entire cause and effect chain, but as a result contain a large amount of uncertainty.

2.7.3 Applicability of Climate Metrics to BC

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 discussed earlier in this chapter, BC influences the climate differently than the warming effects of GHGs. These differences have important implications for identifying appropriate metrics to compare climate impacts (and reductions thereof). Table 2-1 compared some of BC's climate attributes and effects to those of CO_2. The implications of these differences with respect to metrics are discussed here.

As described in detail below, the significant differences between BC and CO_2 make applying the metrics introduced in the previous section difficult and, for some purposes, 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 heavily on where and when it is emitted. In comparison, the shortest-lived GHG 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 variation in atmospheric concentrations of BC among regions contrasts with the well-mixed nature of most GHGs. This distinction 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 decisions about BC. Conversely, focusing on short-term or regional impacts may be inappropriate for decisions involving long-lived GHGs. The following sections discuss how different physical (GWP, GTP, SFP, and STRE) and economic metrics have been used to compare BC to other substances.

2.7.3.1 Global Warming Potential

While a GWP can be calculated for BC, there are reasons that GWPs may be less applicable for this purpose due to the different nature of BC compared to GHGs, in terms of various physical properties and the fact that unlike GHGs, BC is not well mixed in the atmosphere. However, because GWPs are the most commonly used, and only official, metric in climate policy discussions, many studies have calculated GWPs for BC. One-hundred-year GWPs for BC in the literature range from 330 to 2,240. That is to say, 330 to 2,240 tons of CO_2 would be required to produce the same integrated radiative effect over 100 years as one ton of BC. Some of the factors that account for the range in these estimates include the use of different and uncertain indirect and snow/ice albedo effects estimates, use of a different CO_2 lifetime for the baseline, and recognition of the dependence of a GWP for BC on emissions location.

Using time periods shorter than 100 years has also been explored for determining the GWPs of BC. Those who are concerned with near-term impacts (such as Arctic ice retreat) sometimes suggest 20-year GWPs as more appropriate for short-lived forcers such as BC (CATF, 2009b). Jacobson (2007) estimates a 20-year GWP for BC of 4,470. However, for those concerned about the long-term problems of climate change, even 100-year GWPs may be considered too short (IPCC, 2009). Because BC is a short-lived species, the shorter the policy-relevant time horizon considered, the greater the relative importance of BC compared to CO_2 (and vice versa: the longer the relevant time horizon, the less important BC is compared to CO_2). If the focus is on achieving immediate climate benefits within a 10- to 20-year time period, the 20-year GWP provides a more realistic picture of the impact of reductions in different species in the near-term. On the other hand, if the concern is to identify measures that will help avert climate change at a broad scale, over a longer time frame, as the problem is generally conceptualized, a 20-year time horizon is insufficient, and the 100-year GWP is a more relevant metric.

2.7.3.2 Global Temperature Potential

GTPs, as described previously, evaluate the impact on temperature at a given time. Studies have applied the GTP using approaches that differ with respect to how the emissions are reduced and how the impacts are calculated. Boucher and Reddy (2008) use a short, pulse-like (1-year) reduction of emissions and find that the 100-year GTPs are about a factor of 7 smaller than the corresponding GWPs. Berntsen et al. (2006) reduced BC emissions for a 20year time span (approximately the lifetime of a given investment in abatement technology) and found that the 100-year GTP of BC was about 120 to 230 (i.e., reducing 120 to 230 tons of CO_2 has the same impact on temperatures in 100 years as reducing 1 ton of BC).

Several papers have recently summarized different BC GWP and GTP estimates (Sarofim, 2010; California Air Resources Board, 2010; Fuglestvedt et al., 2010). However, of the studies surveyed by these three papers, only Hansen et al. (2007a) considered indirect cloud interactions of BC and only a few included estimates for metrics of co-emitted OC. If co-emissions are not included, then any metric will likely overestimate the globally averaged climate benefits of reducing BC. Inclusion of indirect effects could either increase or decrease the calculated value of the metric.

Figure 2-25, based on Fuglestvedt et al. (2010), summarizes a number of studies that attempted to develop metrics for comparing CO_2 and BC. This figure shows how the GWP metric depends on the time horizon used (20 years, 100 years, and 500 years). Additionally, for the first four studies, the range of values results from a dependence of the GWP on the region in which the emission occurs. The difference between the studies is the result of differences in the climate models used to link the emissions to the temperature change. Figure 2-26 shows a similar analysis from Fuglestvedt et al. (2010) which evaluates the equivalent GTP for these different models.

Fuglestvedt et al. (2010) show that the metric for comparing BC to CO_2 can range from a ton of BC being equivalent to 48 tons of CO_2 based on a 100-year GTP (which measures the temperature change 100 years after a pulse of emissions) in one model, to 4,900 tons of CO_2 based on a 20-year GWP (which integrates the total radiative forcing impact of a pulse of emissions over the 20-year time span) in another model. The variation between GWPs or GTPs for emissions from different

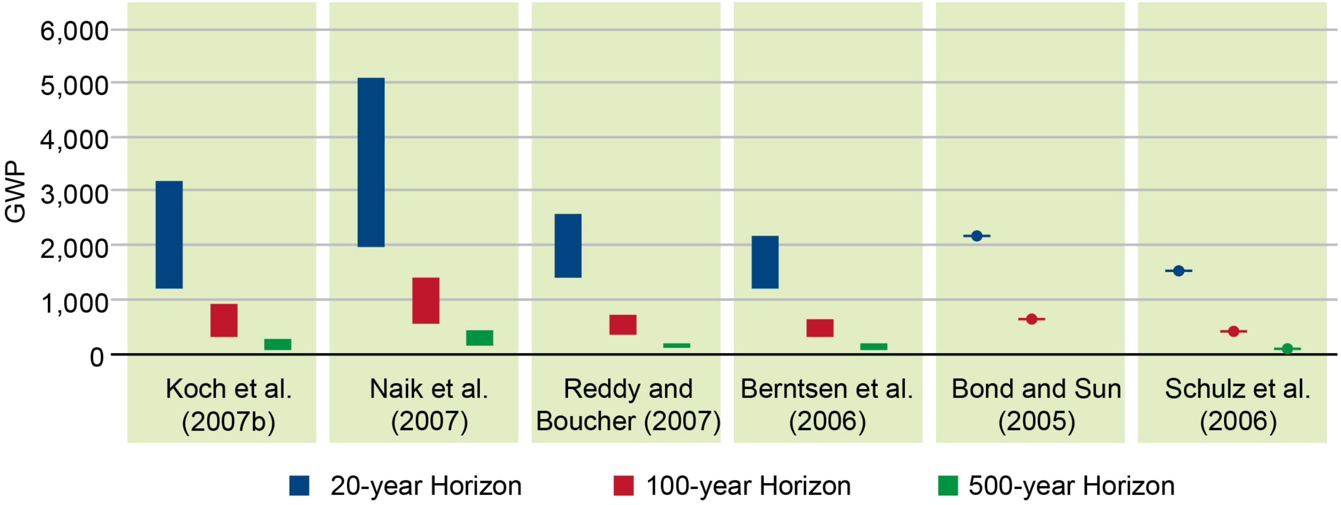

Figure 2-25. Ranges and Point Estimates for Regional Estimates of GWP Values for One-Year Pulse Emissions of BC for Different Time Horizons. The GWP values in the Y axis of the figure refer to the number of tons of CO_2 emissions which are calculated to be equivalent to one ton of BC emissions based on the particular metric. (Adapted from Fuglestvedt et al., 2010.) Note that the first four studies referenced evaluated GWP values for different sets of regions; Bond and Sun (2005) and Schulz et al. (2006) produced global estimates only.

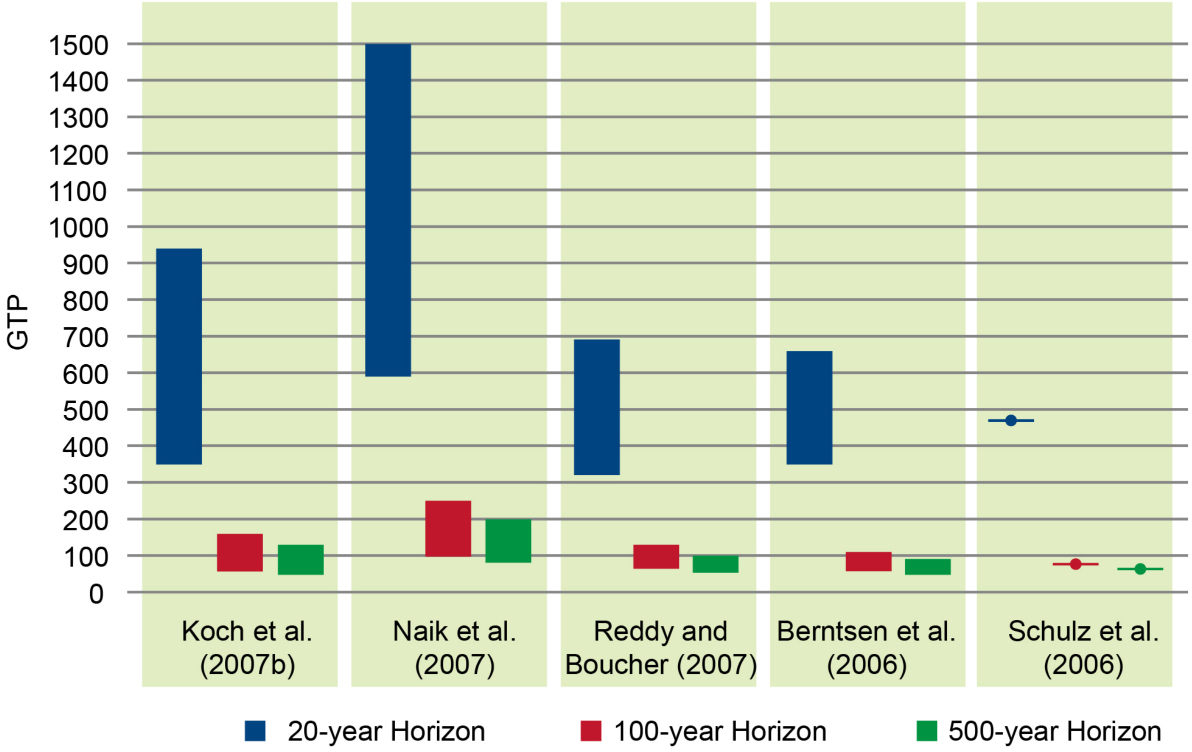

Figure 2-26. Ranges and Point Estimates for Regional Estimates of GTP Values for One-Year Pulse Emissions of BC for Different Time Horizons. The GTP values in the Y axis of the figure refer to the number of tons of CO_2 emissions which are calculated to be equivalent to one ton of BC emissions based on the particular metric. (Adapted from Fuglestvedt et al., 2010) Note that the five studies referenced evaluated radiative forcing estimates for different sets of regions (which were translated into GTP values by Fuglestvedt et al.); Schulz et al. (2006) produced global estimates only.

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. 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 OC 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 and the assessment of a larger range of sectors and co-emission types. Additionally, because most metrics use CO_2 as a baseline forcer, the use of different carbon cycle models can significantly influence the metric values for BC. Some researchers may report metric values in carbon equivalents, rather than CO_2 equivalents, which leads to a factor of 3.7 difference.

2.7.3.3 Specific Forcing Pulse

The SFP is a relatively new metric proposed by Bond et al. (2011) to quantify climate warming or cooling from short-lived substances (i.e., substances with lifetimes of less than four months). This metric is based on the amount of energy added to the Earth system by a given mass of the pollutant. The rationale for developing this new metric was that short-lived substances contribute energy on timescales that are short compared to time scales of mitigation efforts, and therefore can be considered 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 the snow albedo effect included is 1.15±0.53 GJ/g. They also find that the SFP for OC is -0.064 (from -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 to BC is close to climate neutral. However, this does not include cloud indirect effects or co-emissions of substances other than OC. Bond et al. also find that the SFP varies by 45% depending on where the BC is emitted. While the paper notes that fundamental differences in temporal and spatial scales raise concerns about equating the impacts of GHGs and short-lived aerosols, they do use the SFP to calculate a GWP for the direct effect of BC of 740 (±370), for both the direct and the snow albedo effect of BC of 830 (±440), and for organic matter of -46 (from -18 to -92).

This metric is mathematically similar to the Absolute Global Warming potential (see footnote in GWP section), but is applied somewhat differently. Additionally, the use of this metric for regional impacts is interesting, though as discussed earlier, the regional pattern of radiative forcing (or energy input) is not necessarily the same as the regional pattern of temperature response to that forcing.

2.7.3.4 Surface Temperature Response per Unit Continuous Emission

Another new metric, the STRE has been proposed by Jacobson (2010). The STRE is similar to the sustained version of the GTP. Jacobson found that the STRE (which he compares to GWPs) for BC on the 100 year time scale is 2,900 to 4,600 for BC in fossil fuel soot and 1,060 to 2,020 for BC in solid-biofuel soot. The uncertainty ranges presented by Jacobson depend on his assumption that CO_2 will decay exponentially with either a 30- or a 50-year lifetime. The use of a more sophisticated carbon cycle model or the Bern carbon cycle approximation from the IPPC (which is a sum of 4 exponentials rather than a single exponential as in the Jacobson calculations) would result in a lower STRE and would be more comparable with other approaches. Jacobson also presents estimates of the combined BC plus OC STRE, finding that the STRE for emissions of BC plus OC from fossil fuel soot ranges from 1,200 to 1,900 and for emissions from biofuel soot the STRE ranges from 190 to 360.

2.7.3.5 Economic Valuation Metrics

Economic valuation approaches for BC that focus on valuing climate damages from a comprehensive, societal standpoint are discussed in detail in Chapter 6. For reasons discussed in that chapter, techniques used to value the climate damages associated with long-lived GHGs are not directly transferrable to BC or other short-lived forcers. In fact, most such approaches have focused exclusively on valuing the climate impacts of CO_2, and may not even be transferrable to other GHGs. Additional work is needed to design approaches for valuing the climate impacts of BC directly, and to incorporate those approaches into metrics comparable to the SCC.

2.7.4 Using Metrics in the Context of Climate Policy Decisions

There is currently no single metric widely accepted by the research and policy community for comparing BC and long-lived GHGs. In fact, some question whether and when such comparisons are useful. For example, there are concerns that some such comparisons may not capture the different weights placed on near-term and long-term climate change. However, there are multiple reasons to compare BC to other short-lived and long-lived climate substances, including offsets, credit trading, evaluation of net effects of a mitigation option, or illustrative analyses.

The choice of a metric depends greatly on the policy goal. No single metric will accurately address all the consequences of emissions of all the different climate forcers, and all of the differences between BC and the well-mixed gases must be considered. The appropriate metric to use depends on a range of factors, such as: the time scale (20 years, 100 years, or more), the nature of the impact (radiative forcing, temperature, or more holistic damages), concern over different processes (indirect effects, snow albedo changes, co-emissions), and whether sources and impacts should be calculated regionally or globally. It is important to note that different climate models will yield different results even if the same metric definition is chosen. Taking several of these factors into account, especially the use of different time scales, a ton of BC has been calculated to be equivalent to anywhere from 48 tons of CO_2 to 4,600 tons of CO_2. For comparison, the UNEP/WMO assessment, looking only at the 100 year timescale, estimated that BC could be 100 to 2,000 times as potent as CO_2 per ton (UNEP and WMO, 2011a). Certainly, the appropriateness of the comparison depends on the policy question at hand, and the differences in lifetime, uncertainties, co-emissions, modes of interaction with the climate system, and non-climatic effects such as human health should be evaluated when choosing a metric. This section highlights how these differences affect the metric choice.

The tradeoff between capturing short-term and long-term impacts is not strictly a scientific consideration but also a policy question. Much like the original choice of 100 years for the GWP was a policy compromise between long-term and short-term impacts; policymakers may consider whether using a GWP or GTP metric is an acceptable compromise given a desire to compare BC and the long-lived GHGs. A key question is how the metric is used to inform the policy decision. The NRC has warned against delaying CO_2 reductions in favor of short-lived forcer mitigation, suggesting that CO_2 emissions control and control of short-term forcing agents could be thought of instead as "two separate control knobs that affect entirely distinct aspects of the Earth's climate" (National Research Council, 2011). The results of the UNEP/WMO assessment suggest that the two strategies are complementary and should be pursued simultaneously, with BC reductions forming part of a larger strategy for near-term climate change and CO_2 programs influencing climate over the longer term (UNEP and WMO, 2011a). Such an approach could incorporate separate metrics for short-lived and long-lived species. One metric would be appropriate for guiding global emissions of climate forcers to achieve stabilization of GHG concentrations in the long-term, while another metric would focus on mitigating near-term warming and could be used to guide regional emissions reductions in short-lived climate forcers to reduce the impacts on regional forcing, precipitation, and ice/snow melt. It is important to recognize that long-term stabilization of CO_2 concentrations requires limiting total cumulative emissions of CO_2 and that CO_2 reductions today are necessary to achieve climate goals decades and centuries from now (National Research Council, 2011).

Reductions of BC today do little to achieve climate goals in the next century: however, they are important for climate goals in the near future, which can include reducing impacts on vulnerable regions such as the Arctic and reducing the rate of near-term climate change. In addition, if and when we approach climate stabilization, sustained reductions in emissions of BC will be important to keep those peak temperatures lower than they would otherwise be. Along these lines, the IPCC found that the complexity of climate change may indicate that a basket of metrics approach would best capture the variety of spatial, temporal and uncertain features (IPCC, 2009). Such a basket approach to addressing short-lived and long-lived forcers separately (though not BC specifically) has also been supported by Jackson (2009) and Daniel et al. (2011).

Outside of the policy context, the use of multiple metrics can be valuable for illustrative purposes. For example, Figure 2-26 shows the impact of BC relative to CO_2 on different timescales. Such a figure could be combined with an analysis such as the Unger et al. (2010) figure replicated in Figure 2-19 to show the GTP (or GWP) weighted impact of a set of proposed mitigation options at 20 years and 100 years (or some other timescale).

2.7.4.1 Considering the Full Range of BC Effects

As discussed in section 2.6, BC is associated with complex indirect effects and a number of hydrological effects that are unrelated to radiative forcing and that—along with the health effects discussed in Chapter 3—distinguish it from long-lived GHGs. These effects include impacts on the water cycle, inhibition of photosynthesis due to deposition on plants (Kozlowski and Keller, 1966), enhancement of soil productivity due to deposition on soil (Laird, 2008), and effects such as surface dimming. Capturing these additional effects in a single global metric is challenging. Even the current GWP metric continues to see widespread use despite not capturing the ecosystem effects of CO_2-driven ocean acidification or the health and agricultural impacts of CH_4-induced ozone production.

For most GHGs, relative radiative forcing is a reasonable approximation of temperature impacts: a given W m^{-2} of CO_2 has similar impacts to a W m^{-2} of N_2O. By contrast, BC forcing includes a combination of surface dimming and absorption of both incoming and outgoing radiation at many wavelengths, while GHGs mainly absorb outgoing thermal infrared radiation. As discussed in section 2.6.1.4, the temperature change resulting from a given W m^{-2} of forcing from the snow albedo effect may be much greater than the temperature change resulting from a W m^{-2} of CO_2 forcing, whereas the result of forcing from BC-related direct effects may depend on the pattern of BC loading. Inclusion of the cloud effects of BC makes this metric even more uncertain.

Further complicating the use of existing metrics for BC are the significant remaining uncertainties in estimates of BC forcing, especially regarding the indirect cloud effects [which can be compared to the uncertainty in forcing from changes in well-mixed GHG concentrations, estimated to be only 10% of 2.63 W m^{-2} (Forster et al., 2007)]. However, even if BC forcing is at the low end of the range, a consequence of the globally averaged nature of common metrics is that the right mix of BC and OC emissions might have little net global radiative forcing impact and yet still have significant impacts on regional precipitation, dimming, and snow melt as well as possibly on regional patterns of warming and cooling.

2.8 Key Gaps in Understanding and Expressing the Climate Impacts of BC

This chapter has summarized key findings from a wide range of peer-reviewed studies related to BC and its effects on climate. The complex atmospheric chemistry of BC and its regional nature make it a challenging subject for study. The chapter attempts to identify where the strength of the evidence suggests that reasonable conclusions can be drawn (such as for BC's direct forcing impact, which is widely understood to lead to warming), and also highlights those areas where such conclusions may be premature (such as the net effect of BC, considering its impacts on clouds and also the impact of co-emitted pollutants). Despite rapidly advancing science, there is clearly the need for additional research, particularly with regard to BC's effects on clouds and its impacts on radiative forcing, melting and precipitation in specific regions.

Recent studies have begun to apply more rigorous modeling and estimation approaches to try to provide better centralized estimates of BC's direct forcing impact, its impacts on snow and ice, and its effects on clouds. Further work is needed to improve these quantitative estimates and to ensure that the full range of BC effects on climate is considered.

Key research needs include continued investigation of basic microphysical and atmospheric processes affecting BC and other co-pollutants, particularly with regard to the climate effects of BC-cloud interactions and aerosol mixing state. In addition, there is a dearth of research on other types of light-absorbing carbon, such as BrC, which may also contribute to climate impacts especially in sensitive regions such as the Arctic. In general, further investigation of impacts of aerosols in snow- and ice-covered regions would be fruitful, along with additional research on the climate impacts of emissions mixtures from particular source categories.

It is also difficult to compare BC directly to CO_2 or other long-lived GHGs. This chapter has explored some of the metrics that are currently available to determine how well they perform for purposes of expressing the climate effects of BC and comparing BC to CO_2. However, there are clear limitations to using these metrics. In general, there is a strong need for further refinement of policy-relevant metrics for BC and other short-lived climate forcers. Appropriately tailored metrics for BC are needed in order to quantify and communicate BC's impacts and properly characterize the costs and benefits of BC mitigation.

Chapter 3

Black Carbon Effects on Public Health and the Environment

3.1 Summary of Key Messages

- BC is a component of both fine and coarse particulate matter (PM), though because of its small size, it is most strongly associated with the fine particle ($PM_{2.5}$) fraction. Most of the literature evaluating the potential impacts of BC on human health (and the health benefits of BC mitigation) has focused on BC as part of $PM_{2.5}$.

- Short-term and long-term exposures to $PM_{2.5}$ are associated with a broad range of adverse human health effects including respiratory and cardiovascular effects, as well as premature death.

- Over the past decade, the scientific community has focused increasingly on trying to identify the health impacts of particular $PM_{2.5}$ constituents, such as BC. However, EPA has determined that there is insufficient information at present to differentiate the health effects of the various constituents of $PM_{2.5}$; thus, EPA assumes that many constituents are associated with adverse health impacts.

    - The limited scientific evidence that is currently available about the health effects of BC is generally consistent with the general $PM_{2.5}$ health literature, with the most consistent evidence for cardiovascular effects. However, study results for BC are variable, and further research is needed to address remaining uncertainties.

- $PM_{2.5}$, both ambient and indoor, is estimated to result in millions of premature deaths worldwide, the majority of which occur in developing countries.

    - The World Health Organization (WHO) estimates that indoor smoke from solid fuels is among the top ten major risk factors globally, contributing to approximately 2 million deaths annually. Women and children are particularly at risk.

    - Ambient air pollution is also a significant health threat: according to the WHO, urban air pollution is among the top ten risk factors in medium- and high-income countries. Urban air pollution is not ranked in the top ten major risk factors in low-income countries since other risk factors (e.g. childhood underweight and unsafe water, sanitation and hygiene) are so substantial; however, a much larger portion of the total deaths related to ambient $PM_{2.5}$ globally are expected to occur in developing regions, partly due to the size of exposed populations in those regions. It is noteworthy that emissions and ambient concentrations of directly emitted $PM_{2.5}$ are often highest in urban areas, where large numbers of people live.

- $PM_{2.5}$, including BC, is 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.

3.2 Introduction

This chapter assesses the current scientific knowledge relating to the public health and non-climate welfare effects associated with short-term and long-term exposure to BC. The magnitude of these impacts in the U.S. and globally is also addressed.

3.3 Health Effects Associated with BC

3.3.1 Key Health Endpoints Associated with Exposure to PM

BC is a component of both fine and coarse PM. Since 1997, EPA has recognized the need to regulate fine and coarse-fraction particles separately. Current national ambient air quality standards (NAAQS) use $PM_{2.5}$ as the indicator for fine particles, and PM_{10} as the indicator for thoracic coarse particles. At present, EPA is undertaking another periodic review of these standards. As part of this review, EPA has completed an Integrated Science Assessment for Particulate Matter (ISA) (U.S. EPA, 2009b) providing a concise evaluation and integration of the policy-relevant

Table 3-1. Summary of Causal Determinations for Exposure to $PM_{2.5}$ from 2009 PM ISA. (Source: U.S. EPA)

Exposure	Outcome	Causality Determination
Short-term exposure to $PM_{2.5}$	Cardiovascular Effects	Causal
	Respiratory Effects	Likely to be causal
	Mortality	Causal
Long-term exposure to $PM_{2.5}$	Cardiovascular Effects	Causal
	Respiratory Effects	Likely to be causal
	Mortality	Causal
	Reproductive and Developmental Effects	Suggestive
	Cancer, Mutagenicity, and Genotoxicity	Suggestive

science pertaining to the health and environmental effects of ambient particles. The ISA presents causal determinations by PM size fraction and exposure duration (i.e., short-term [days to weeks] or long-term [months to years]) for the health effects for which sufficient evidence was available to conclude a causal, likely to be causal, or suggestive relationship (Table 3-1). The discussion below is focused on the health effects with the strongest weight of evidence (i.e., cardiovascular effects, respiratory effects, and mortality) and conclusions drawn for these effects in the ISA. A more limited subset of studies has evaluated reproductive and developmental outcomes and cancer effects, but the weight of evidence for these effects is less substantial.[1]

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

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

The body of scientific evidence linking exposures to coarse particles (i.e., ambient $PM_{10-2.5}$ mass

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[1] See Sections 7.4 and 7.5 of the PM ISA for an in-depth characterization of the evidence for an association between $PM_{2.5}$ and reproductive and developmental effects and cancer, respectively. (*http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546*)

concentrations) to health effects is much smaller than the body of evidence for $PM_{2.5}$ (U.S. EPA, 2009b). Similar to $PM_{2.5}$, the chemical composition of $PM_{10-2.5}$ can vary considerably by location, but city-specific speciated $PM_{10-2.5}$ data are limited. However, $PM_{10-2.5}$ may contain iron, silica, aluminum, and base cations from soil, plant and insect fragments, pollen, fungal spores, bacteria, and viruses, as well as fly ash, brake lining particles, debris, and automobile tire fragments. The last four of these components (fly ash, brake lining particles, debris, and automobile tire fragments) are associated with urban or industrial ambient mixes of coarse PM, which are often coated with BC (See Figure 2-3). Urban or industrial ambient mixes of coarse PM are dominated by high density vehicular, industrial, and construction emissions, and are likely to be associated with adverse health effects (U.S. EPA, 2006b). While there are no studies that specifically examine the association between BC as a component of $PM_{10-2.5}$ and health effects, the current evidence, primarily from epidemiological studies, indicates that short-term exposure to $PM_{10-2.5}$ is associated with effects on both the cardiovascular and respiratory systems. However, variability in the chemical and biological composition of $PM_{10-2.5}$, limited evidence regarding effects of the various components of $PM_{10-2.5}$, and lack of clearly defined biological mechanisms for $PM_{10-2.5}$-related effects are important sources of uncertainty (U.S. EPA, 2009b).

3.3.2 Health Effects Related to Ambient BC Concentrations

Some community epidemiological studies have included BC[2] as one of several indicators of fine particulate air pollution. Of $PM_{2.5}$ components, BC is one of the larger contributors to $PM_{2.5}$ total mass. For example, Bell et al. (2007) examined levels of PM components on a national basis, and identified EC as one of the seven main contributors. The effects observed with BC in health studies are similar to those observed for $PM_{2.5}$ and some other PM constituents (e.g., nickel, vanadium), suggesting that these effects are not attributable solely to BC. Indeed, it would be difficult to separate the contribution of BC to these associations from that of co-emitted OC and other correlated and co-emitted primary pollutants in such studies. Still, these studies provide generally consistent evidence for an association between cardiovascular morbidity and BC concentrations.

A number of studies have reported associations between short-term exposure to BC and cardiovascular effects (See Table 3-2). Telomere length attrition, an indication of biological age that is inversely associated with risk of cardiovascular disease, was associated with ambient BC concentrations in the Boston, MA, area (McCracken et al., 2010). A series of analyses found that changes in blood pressure (Delfino et al., 2010; Mordukhovich et al., 2009; Wilker et al., 2010) and heart rate variability (HRV) (Adar et al., 2007; Chuang et al., 2008; Gold et al., 2005; Huang et al., 2003; Park et al., 2005; Schwartz et al., 2005) were associated with increases in mean ambient BC concentration. The ST-segment of an electrocardiograph represents the period of slow repolarization of the ventricles and ST-segment depression can be associated with adverse cardiac outcomes, including ischemia. Delfino et al. (2011) found positive associations between ST-segment depression and BC concentrations. Homocysteine, a sulfur-containing amino acid formed during metabolism of methionine, is a risk factor for atherosclerosis, myocardial infarction (MI), stroke, and thrombosis. Similarly, lower blood DNA methylation content is found in processes related to cardiovascular outcomes, such as oxidative stress and atherosclerosis. Several studies observed an association between ambient BC concentration and elevated plasma total homocysteine (Park et al., 2008; Ren et al., 2010). An additional study (Baccarelli et al., 2009) observed an association between lower blood DNA methylation content and BC concentrations. Cardiac arrhythmia (a broad group of conditions where there is irregular electrical activity in the heart) was associated with increased concentrations of BC in studies conducted in Boston (Rich et al., 2005; Rich et al., 2006; Zanobetti et al., 2009; Baja et al., 2010; Dockery et al., 2005), but not in Vancouver, Canada (Rich et al., 2004). Another series of analyses has reported inconsistent associations between BC and blood markers of coagulation and inflammation, with some studies finding an effect (Dubowsky et al., 2006; Rückerl et al., 2006; Delfino et al., 2009; 2008; O'Neill et al., 2007), and others finding no effect for a blood marker with large intra-individual variability (i.e., B-type natriuretic peptide or BNP) (Wellenius et al., 2007) or no effects for acute lag periods (i.e., 48 hours or 1 week) (Zeka et al., 2006). Ambient concentrations of BC (Peters et al., 2001; Zanobetti and Schwartz, 2006) and EC (Bell et al., 2009; Peng et al., 2009; Sarnat et al., 2008; Tolbert et al., 2007; Ito et al., 2011) were also found to be associated with hospital admissions and emergency department visits for cardiovascular outcomes.

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[2] The monitoring methods used to estimate BC vary, and include various surrogate measurements such as optical BC and thermal-optical EC (see Chapter 5 and Appendix 1). Categorization of studies according to the indicator measurements used should be the focus of future research.

Table 3-2. Summary of Epidemiological Studies of BC and Cardiovascular Health Outcomes. The study reference numbers listed in the left-hand column are for purposes of cross-reference with Figure 3-1. (Source: U.S. EPA)

Reference Number in Figure 3-1	Study	Location	Health Outcome	Metric	Representative Concentration (µg/m^3)	Exposure Assessment[a]	Selected Effect Estimates (95% CI)[b]
Short-term Exposure Studies							
—	McCracken et al. (2010)	Boston, MA	Telomere Length	BC	Mean (annual avg): 0.32	Annual outdoor home concentration estimates from spatiotemporal model	Association with leukocyte telomere length: -7.6% (-12.8%, -2.1%)
1.	Delfino et al. (2010)	Los Angeles, CA	Blood Pressure	BC	Mean (24-h avg): 1.67	Hourly outdoor home air-pollutant concentrations	Change in BP: SBP: 0.22 (-0.65, 1.09) DBP: 0.36 (-0.11, 0.83)
2.	Mordukhovich et al. (2009)	Boston, MA	Blood Pressure	BC	Mean (7-day moving average): 1.10	Continuous measurements from single monitor averaged by hour before BP measurement	Change in BP: SBP: 1.46 (0.10, 2.82) DBP: 0.87 (0.15, 1.59)
3.	Wilker et al. (2010)	Boston, MA	Blood Pressure	BC	Mean (7-day moving average): 0.98	Continuous measurements from single monitor averaged by hour before BP measurement	Change in BP: SBP: 3.52 (2.77, 4.26) DBP: 2.72 (2.31, 3.12)
4.	Adar et al. (2007)	St. Louis, MO	HRV	BC	Median (5-min avg periods): 0.285-2.911 (range across microenvironments)	Two portable carts containing continuous sampling instrumentation	Percent change in HRV: SDNN: -5.3 (-6.5, -4.1) RMSSD: -10.7 (-11.9, -9.5) PNN50+1: -13.2 (-15.0, -11.4) LF: -11.3 (-13.7, -8.8) HF: -18.8 (-21.1, -16.5) LF/HF: 9.3 (7.2, 11.4) HR: 1.0 (0.8, 1.3)
5.	Chuang et al. (2008)	Boston, MA	HRV	BC	Median (24-h avg): 0.79	Continuous measurements from single monitor	Relative risk for ST-segment Depression ≤1 mm: 1.50 (1.19-1.89)
6.	Gold et al. (2005)	Boston, MA	HRV	BC	Median (12-h avg): 1.14	Continuous measurements from single monitor	Relative risk for ST-segment Depression ≤0.5 mm: First rest: 3.8 (0.7, 21.3) Blood pressure: 5.7 (0.6, 56.3) Standing: 8.3 (0.8, 81.9) Exercise: 0.6 (0.1, 3.1) Second rest: 2.8 (0.5, 14.3) Paced breathing: 3.5 (0.5, 23.6)

Reference Number in Figure 3-1	Study	Location	Health Outcome	Metric	Representative Concentration (μg/m^3)	Exposure Assessment[a]	Selected Effect Estimates (95% CI)[b]
7.	Park et al. (2005)	Boston, MA	HRV	BC	Mean (24-h avg): 0.92	Continuous measurements from single monitor	Percent change in HRV: SDNN: -3.4 (-10.2, 3.9) HF: -13.8 (-28.9, 4.4) LF: -2.4 (-16.2, 13.6) LF/HF: 13.2 (-1.1, 29.6)
8.	Schwartz et al. (2005)	Boston, MA	HRV	BC	Mean (24-h avg): 1.2	Continuous measurements from single monitor	Percent change in HRV: SDNN: -5.1 (-1.5, -8.6) r-MSSD: -10.1 (-2.4, -17.2) PNN50: -16.9 (-6.0, -26.6) LF/HF: 7.2 (0.7-14.1)
9.	Delfino et al. (2011)	Los Angeles, CA	HRV	BC	Mean (24-h avg): 1.67	Hourly outdoor home air-pollutant concentrations	Odds ratio for ST-segment Depression ≤1 mm: 2.07 (1.30, 3.29)
10.	Park et al. (2008)	Boston, MA	Plasma Total Homocysteine	BC	Mean (24-h avg): 0.99	Continuous measurements from single monitor	Percent change in total homocysteine: 3.13 (0.76, 5.55)
11.	Ren et al. (2010)	Boston, MA	Plasma Total Homocysteine	BC	Mean (7-day moving average): 0.99	Continuous measurements from single monitor	Percent change in total homocysteine: 0.68 (-0.46, 1.81)
—	Baccarelli et al. (2009)	Boston, MA	Blood DNA Methylation	BC	Mean (24-h avg): 0.89	Continuous measurements from single monitor	Coefficient for effect on methylation: LINE-1: -0.09 (-0.15, -0.02) Alu: -0.02 (-0.08, 0.05)
12.	Baja et al. (2010)	Boston, MA	Ventricular Repolarization	BC	Mean (1-h avg): 1.08	Continuous measurements from single monitor	Change in mean QTc (msec): 1.89 (-0.16, 3.93)
13.	Dockery et al. (2005)	Boston, MA	Arrhythmia	BC	Median (2-day avg): 0.98	Continuous measurements from single monitor	Association with ventricular arrhythmias: Recent arrhythmia > 3 days: 1.02 (0.83, 1.24) Recent arrhythmia <3 days: 1.74 (1.28, 2.37)
14.	Rich et al. (2005)	Boston, MA	Arrhythmia	BC	Median (24-h avg): 0.94	Continuous measurements from single monitor	Odds ratio for ventricular arrhythmias: 0.93 (0.74, 1.18)
15.	Rich et al. (2006)	Boston, MA	Arrhythmia	BC	Median (24-h avg): 0.94	Continuous measurements from single monitor	Odds ratio for paroxysmal atrial fibrillation: 1.46 (0.67, 3.17)

Reference Number in Figure 3-1	Study	Location	Health Outcome	Metric	Representative Concentration (µg/m^3)	Exposure Assessment[a]	Selected Effect Estimates (95% CI)[b]
16.	Zanobetti et al. (2009)	Boston, MA	Arrhythmia	BC	Median (6-h avg): Ambient: 0.72 Indoor: 0.41 Outdoor: 0.50	Continuous ambient measurements from single monitor; indoor and outdoor measured continuously at participants' homes	Odds ratio for maximum T-wave alternans ≤ 26 µV: 1.42 (1.19, 1.69)
17.	Rich et al. (2004)	Vancouver, Canada	Arrhythmia	EC	Mean (24-h avg): 0.8	Continuous measurements from single monitor	Odds ratio for defibrillator discharge: 1.06 (0.87, 1.33)* *estimated from graph
18.	Sorensen et al. (2003)	Copenhagen, Denmark	Blood Markers of Coagulation and Inflammation	Carbon Black	Median (24-h avg): 8.1 (10-6/m)	Personal exposure	Association with plasma proteins: 4.1% increase in plasma proteins per 1 x 10^{-5}/m increase in personal CB exposure
19.	Dubowsky et al. (2006)	St. Louis, MO	Blood Markers of Coagulation and Inflammation	BC	Mean (24-h avg): 0.9	Continuous measurements from single monitor	Association with markers of inflammation: IL-6: -0.8 (-8.9, 8.0) CRP: 13 (-0.34, 28) WBC: 1.3 (-2.1, 4.8)
20.	Ruckerl et al. (2006)	Erfurt, Germany	Blood Markers of Coagulation and Inflammation	EC	Mean (24-h avg): 2..6	Continuous measurements from single monitor	Odds ratio for increase in blood marker above 90th percentile: CRP: 1.3 (0.7, 2.4) ICAM-1: 2.6 (1.7, 3.8)
21.	Delfino et al. (2008)	Los Angeles, CA	Blood Markers of Coagulation and Inflammation	BC	Mean (24-h avg): 2.00	Outdoor home measurements	Coefficient for association: CRP (ng/ml): 585.61 IL-6 (pg/ml): 0.48 sTNF-RII (pg/ml): 135.15 sP-selectin (ng/ml): 1.99 Cu, Zn-SOD (U/g Hb): -187.95
22.	Delfino et al. (2009)	Los Angeles, CA	Blood Markers of Coagulation and Inflammation	BC	Mean (24-h avg): 1.59 – 1.76	Hourly outdoor home air pollutants	Coefficient for association CRP (ng/ml): 252 (-54, 558) IL-6 (pg/ml): 0.16 (0.01, 0.31) sTNF-RII (pg/ml): 38 (-26, 102) sP-selectin (ng/ml): 1.19 (-0.52, 2.90) Cu, Zn-SOD (U/g Hb): -114 (-229, 1) TNF-α (pg/ml): 0.02 (-0.06, 0.10) GPx-1 (U/g Hb): -0.47 (-0.97, 0.03)
23.	O'Neill et al. (2007)	Boston, MA`	Blood Markers of Coagulation and Inflammation	BC	Mean (24-h avg): 1.1	Continuous measurements from single monitor	Percent change in inflammatory marker: ICAM-1 (ng/ml): 5.84 (0.87, 11.05) VCAM-1 (ng/ml): 9.26 (2.98, 15.91) vWF (proportion): 7.96 (-4.34, 21.84)

Reference Number in Figure 3-1	Study	Location	Health Outcome	Metric	Representative Concentration ($\mu g/m^3$)	Exposure Assessment[a]	Selected Effect Estimates (95% CI)[b]
24.	Wellenius et al. (2007)	Boston, MA	Blood Markers of Coagulation and Inflammation	BC	Mean (24-h avg): 0.73	Continuous measurements from single monitor	"No significant associations observed between [BC] and BNP levels at any of the lags examined."
25.	Zeka et al. (2006)	Boston, MA	Blood Markers of Coagulation and Inflammation	BC	Mean (2-day avg): 0.77	Continuous measurements from single monitor	Percent increase in inflammatory marker: Fibrinogen: 0.84 (-0.63, 2.31) CRP: 4.51 (-2.03, 11.06) Sediment rate: -4.56 (-25.55, 16.43) WBC count: -0.63 (-2.45, 1.19)
26.	Peters et al. (2001)	Boston, MA	ED visits and hospital admissions for CVD	BC	Mean (24-h avg): 1.35	Continuous measurements from single monitor	Odds ratio for MI: 1.21 (0.87, 1.70)
27.	Zanobetti and Schwartz (2006)	Boston, MA	ED visits and hospital admissions for CVD	BC	Median (24-h avg): 1.15	Continuous measurements from single monitor	Percent change in MI admissions: 8.34 (0.21, 15.82)
28.	Bell et al. (2009)	106 U.S. Counties	ED visits and hospital admissions for CVD	EC	Mean (24-h avg): 0.72	County-wide averages	Percent increase in health effect estimate: 25.8 (4.4, 47.2)
29.	Peng et al. (2009)	119 U.S. Counties	ED visits and hospital admissions for CVD	EC	Median (24-h avg): 0.58	County-wide averages	Percent increase in CVD hospital admissions: 0.72 (0.43, 1.01)
30.	Sarnat et al. (2008)	Atlanta, GA	ED visits and hospital admissions for CVD	EC	Mean (24-h avg): 1.4-1.7	Positive matrix factorization applied to measurements from single monitor	Relative risk of ED visit for CVD: 1.025 (1.014, 1.036)
31.	Tolbert et al. (2007)	Atlanta, GA	ED visits and hospital admissions for CVD	EC	Mean (24-h avg): 1.6	Continuous measurements from single monitor	Relative risk of ED visit for CVD: 1.015 (1.005, 1.025)
32.	Ito et al. (2011)	New York, NY	ED visits and hospital admissions for CVD	EC	Mean (24-h avg): 1.13	Average of continuous measurements from three monitors	Percent excess risk: 1.4 (0.1, 2.7)

Reference Number in Figure 3-1	Study	Location	Health Outcome	Metric	Representative Concentration (μg/m³)	Exposure Assessment[a]	Selected Effect Estimates (95% CI)[b]
Long-term exposure studies							
33.	O'Neill et al. (2005)	Boston, MA	Endothelial dysfunction	BC	Mean (24-h avg): 1.0	Continuous measurements from single monitor	Percent change in vascular reactivity among those with (a) diabetes and (b) all subjects: Flow-mediated dilation: (a) -12.6 (-21.7, -2.4); (b) -9.3 (-17.8, 0.2) Nitroglycerin-mediated dilation: -(a) 6.6 (-14.0, 1.5); (b) -5.4 (-12.0, 1.7)
34.	Madrigano et al. (2010)	Boston, MA	Endothelial dysfunction	BC	Mean (24-h avg): 0.84	Continuous measurements from single monitor	Percent change in blood markers: sVCAM-1: 4.52 (1.09, 7.96) sICAM-1: -1.37 (-3.89, 1.15)
35.	Alexeeff et al. (2011)	Boston, MA	Markers of inflammation and endothelial response	BC	Mean (24-h avg): 0.42	Validated spatio-temporal (land use regression) model	Percent change in blood markers: 4 wk exposure, sVCAM-1: 1.00 (-0.65, 2.67) 4 wk exposure, sICAM-1: 1.50 (0.22, 2.80) 8 wk exposure, sVCAM-1: 1.20 (-0.58. 3.02) 8 wk exposure, sICAM-1: 1.58 (0.18, 3.00) 12 wk exposure, sVCAM-1: 1.26 (-0.58, 3.14) 12wk exposure, sICAM-1: 1.49 (0.04, 2.95)
36.	Gan et al. (2011)	Vancouver, BC	CHD hospitalizations	BC	Mean (24-h avg): 1.19	High resolution land use regression model	Relative Risk of CHD hospitalization: 1.01 (1.00, 1.03)

[a] A more complete description or evaluation of the BC monitoring method is beyond the scope of this summary, and may include surrogate measurements.

[b] SBP=systolic blood pressure; DBP=diastolic blood pressure; SDNN=standard deviation of normal intervals measured between consecutive sinus beats; RMSSD=root mean square successive difference, a measure of heart period variability; PNN50=number of times per hour in which the change in consecutive normal sinus intervals exceeds 50 milliseconds; LF=low frequency component of heart rate variability; HF=high frequency component of heart rate variability; HR=heart rate; Line-1=long interspersed nuclear element-1; Alu=short stretch of DNA; IL-6=interleukin 6; CRP=c-reactive protein; WBC=white blood cells; ICAM=intercellular adhesion molecule; sTNF-RII=extracellular domain of the tumor necrosis factor receptor; sp-Selectin=plasma soluble P-selectin; Cu, Zn-SOD=copper/zinc superoxide dismutase; TNF-α=tumor necrosis factor-alpha; GPx-1=glutathione peroxidase; VCAM=vascular cell adhesion molecule; vWF=vonWillebrand Factor; BNP=B-type natriuretic peptide; MI=myocardial infarction; CVD=cardiovascular disease; ED=emergency department; CHD=coronary heart disease.

The most noteworthy new cardiovascular-related revelation in recent years with regard to long-term PM exposure is that the systemic vasculature may be a target organ (U.S. EPA, 2009b). Endothelial dysfunction is a factor in many diseases and may contribute to the origin and/or exacerbation of MI or ischemic heart disease, as well as hypertension. Endothelial dysfunction is also a characteristic feature of early and advanced atherosclerosis. New evidence supports an association of ambient BC with decrements in the systemic vasculature. O'Neill et al. (2005) reported that increases in mean BC concentration were associated with decreased vascular reactivity among diabetics, but not among subjects at risk for diabetes. Several recent studies (Madrigano et al., 2010; Alexeeff et al., 2011; Gan et al., 2011) observed that ambient BC was associated with a marker of endothelial function and inflammation, and that genes related to oxidative defense might modify this association. Consistent with these findings, animal toxicological studies have shown that BC can affect heart rate variability (Tankersley et al., 2007; 2004), cardiac contractility (Tankersley et al., 2008) and oxidative stress response (Tankersley et al., 2008), providing biological plausibility for a long-term effect on cardiovascular health.

Overall, the limited body of evidence suggests that ambient BC may be associated with a continuum of effects ranging from more subtle subclinical measures (e.g. changes in blood pressure, heart rate variability) to emergency department visits and hospital admissions for cardiovascular outcomes (Figure 3-1). Generally, this is consistent with the association observed for $PM_{2.5}$ and cardiovascular outcomes (Janssen et al., 2011), as described above (Section 3.3.1).

Fewer studies have examined the effects of BC with respiratory effects (Table 3-3). Clark et al. (2010) investigated the effect of exposure to ambient air pollution in utero and during the first year of life on risk of subsequent asthma diagnosis (incident asthma diagnosis up to age 3-4) and reported that BC exposure was associated with a 14% (1-29%) increase in asthma risk. Delfino et al. (2006) found associations between airway inflammation and ambient EC concentrations among asthmatic children, while Jansen et al. (2005) reported an association with a marker of pulmonary inflammation and BC concentrations among older adults. These results are supported by toxicological studies reporting evidence of airway inflammation (Godleski et al., 2002; Saldiva et al., 2002). There is consistent evidence from a number of studies that report associations of respiratory symptoms among both asthmatic and non-asthmatic children and ambient BC or EC (Kim et al., 2004; Mann et al., 2010; McConnell et al., 2003; Patel et al., 2010; Spira-Cohen et al., 2011). Additionally, Suglia et al. (2008) reported that ambient BC was associated with decreased lung function among urban women. Recent studies evaluated the effect of ambient BC or EC on respiratory hospital admissions and found statistically significant associations between the county-average ambient concentrations of BC or EC and respiratory hospital admissions (Zanobetti and Schwartz, 2006; Bell et al., 2009; Ostro et al., 2009). However other studies found less consistent evidence (Peng et al., 2009; Mohr et al., 2008) or no evidence (Sarnat et al., 2008; Tolbert et al., 2007) for an association between ambient EC and respiratory emergency department visits. Overall, there is inconsistent evidence for an association between ambient BC concentrations and respiratory effects. Similar to what was observed in studies of $PM_{2.5}$, studies examining ambient BC report increased respiratory symptoms in asthmatic children, but less consistent evidence for an association with emergency department visits and hospital admissions.

Several recent epidemiological studies have examined the association between mortality and short-term ambient exposure to components of $PM_{2.5}$, including BC or EC (Table 3-4). Lippmann et al. (2006) reported that nickel, vanadium, and EC were the best predictors, respectively, of PM_{10} risk estimates for mortality. Cakmak et al. (2011) reported an association between increased exposure to concentrations of EC and increases in all cause mortality, while Ito et al. (2011) and Ostro et al. (2007) found positive associations between EC and cardiovascular mortality. These associations (Ostro et al., 2007) were higher in individuals with lower educational attainment and of Hispanic ethnicity (Ostro et al., 2008). Studies of long-term exposure to EC (Lipfert et al., 2006; 2009) and BC (Gan et al., 2011) also report associations with mortality. Overall, the limited body of evidence examining the association of ambient BC with mortality has reported associations with mortality, especially cardiovascular mortality. This association is consistent with the evidence for a causal relationship between $PM_{2.5}$ and mortality.

3.3.2.1 Health Effects Related to Indicators of Ambient BC Concentrations

Concentrations of many traffic-generated air pollutants are elevated for up to 300-500 meters downwind of roads with high traffic volumes (Zhou and Levy, 2007). Numerous sources on roads contribute to elevated roadside concentrations, including exhaust and evaporative emissions,

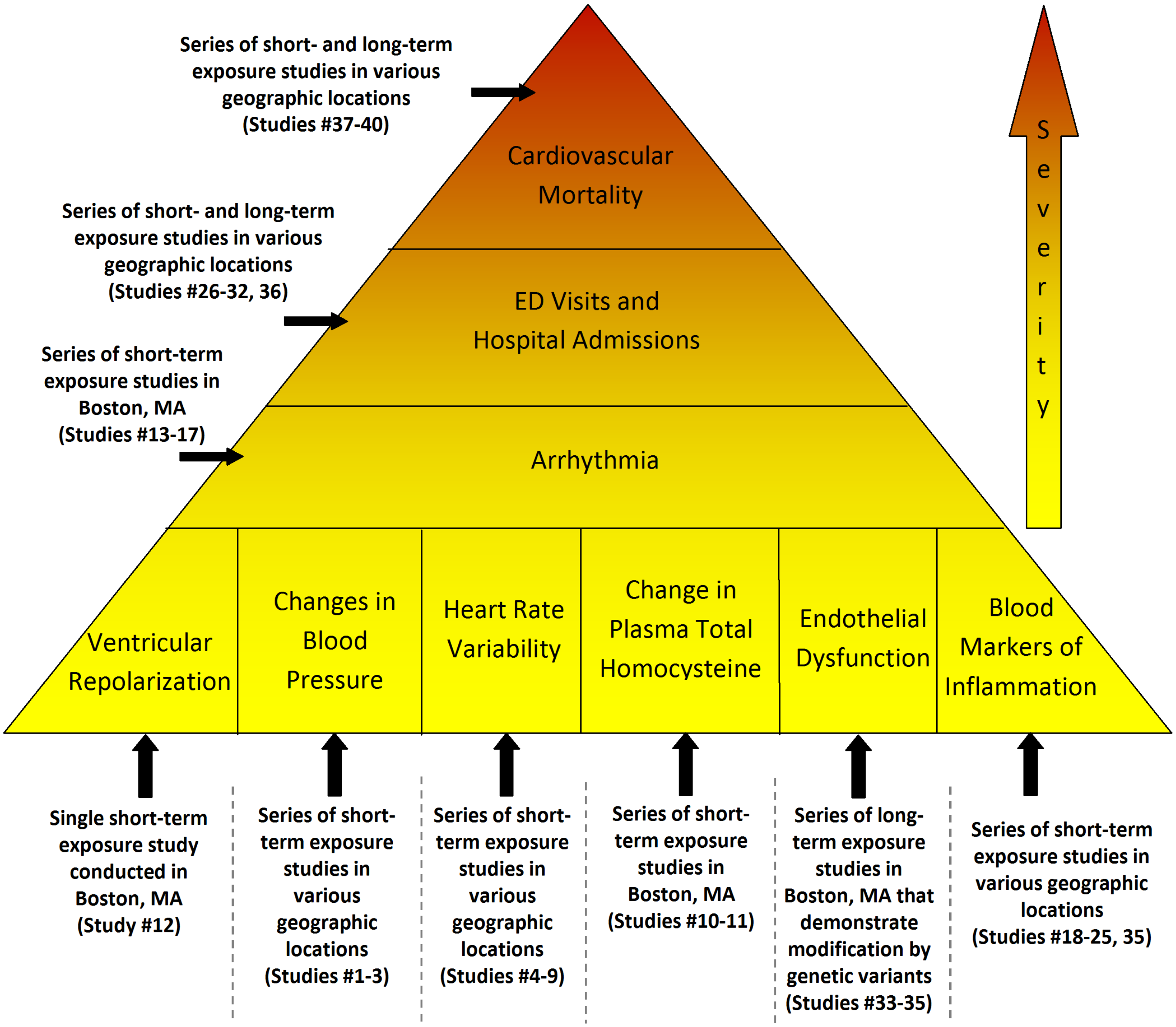

Figure 3-1. Conceptual Diagram of the Epidemiological Evidence for the Association of BC with the Continuum of Cardiovascular Effects, including sub-clinical effects (bottom level of the pyramid) and clinical effects, increasing in severity moving up the pyramid. It is important to note that the body of evidence describing the association between BC and cardiovascular effects is much smaller and less consistent than the one characterizing $PM_{2.5}$ and cardiovascular effects. The study reference numbers listed in parentheses correspond to the reference numbers assigned to individual studies in the left-hand column of Table 3-2 and Table 3-4. For study-specific details, please see Table 3-2 and Table 3-4. (Source: U.S. EPA)

and resuspension of road dust and tire and brake wear. Concentrations of several criteria and hazardous air pollutants are elevated near major roads. Furthermore, different semi-volatile organic compounds and chemical components of PM, including BC, organic material, and trace metals, have been reported at higher concentrations near major roads. While this document is focused on the health effects associated with BC specifically, this section discusses the mixture of different pollutants near major roadways, of which BC is a component. As such, this section emphasizes traffic-related air pollution, in general, as the relevant indicator of exposure to BC.

Table 3-3. Summary of Epidemiological Studies of BC and Respiratory Health Outcomes. (Source: U.S. EPA)

Study	Location	Health Outcome	Metric	Representative Concentration (µg/m³)	Exposure Assessmenta	Selected Effect Estimates (95% CI)
Clark et al. (2010)	British Columbia	Development of childhood asthma	BC	Mean (24-h avg): 1.34	Land Use Regression (LUR) Model	Asthma risk due to average exposures In utero exposure: 1.08 (1.02, 1.15) First-year exposure: 1.14 (1.01, 1.29)
Delfino et al. (2006)	Los Angeles, CA	Exhaled nitric oxide (biomarker of airway inflammation)	EC	Mean (24-h avg): 0.71-1.61	Personal exposure and continuous measurements from central site monitors	Association between EC and exhaled NO: Personal exposure: 0.72 (0.32, 1.12) Central site exposure: 1.38 (0.15, 2.61)
Jansen et al. (2005)	Seattle, WA	Exhaled nitric oxide (biomarker of airway inflammation)	BC	Mean (24-h avg): Central site: 2.01 Indoor: 1.34 Personal: 1.64	Personal exposures, indoor monitors, and central outdoor monitoring site	Association between BC and exhaled NO among asthmatics: Central site exposure: 2.3 (1.1, 3.6) Indoor Exposure: 4.0 (2.0, 5.9) Personal exposure: 1.2 (0.2, 2.2)
Kim et al. (2004)	San Francisco, CA	Bronchitis symptoms and asthma	BC	Mean (24-h avg): 0.8	BC measured at 10 school sites	Odds ratios of respiratory illness by school-based BC concentration: Bronchitis: 1.04 (1.00, 1.08) Asthma: 1.07 (0.98, 1.17)
Mann et al. (2010)	Fresno, CA	Wheeze among asthmatic children	EC (estimated from BC)	Median (24-h avg): 1.3	Continuous measurements from single monitor	Odds ratio of EC and wheeze: 1.12 (0.97, 1.30)
McConnell et al. (2003)	12 Southern California communities	Bronchitic symptoms among asthmatic children	EC	Mean (24-h avg): 0.71	Annual averages computed from 2-week averages measured in each community	Bronchitic symptoms as a function of the 4-yr avg EC concentration: 1.64 (1.06, 2.54)
Patel et al. (2010)	New York City, NY	Respiratory symptoms among asthmatics and non-asthmatics	BC	Median (24-h avg): 0.49-2.4	BC measured at 4 high schools	Odds ratios for respiratory symptoms and use of asthma medication: Wheeze: 1.11 (1.00, 1.22) Cough: 0.95 (0.87, 1.03) Shortness of Breath: 1.26 (1.14, 1.38) Chest Tightness: 1.11 (1.01, 1.24) Use of Asthma Medication: 1.09 (0.89, 1.33)
Spira-Cohen et al. (2011)	South Bronx, NY	Lung function and respiratory symptoms among asthmatic children	EC	Mean (24-h avg): 1.9	Personal and outdoor school site monitoring at 4 elementary schools	Effect estimates for lung function decrements: Personal EC, PEF: -9.13 (-19.13, 0.86) Personal EC, FEV1: -0.02 (-0.09, 0.04) School-site EC, PEF: -4.58 (-14.01, 4.85) School-site EC, FEV1: 0.01 (-0.04, 0.07) Relative Risks for respiratory symptoms: Personal EC, cough: 1.61 (1.17, 2.21) Personal EC, wheeze: 1.67 (1.05, 2.66) Personal EC, shortness of breath: 1.41 (1.01, 1.99) Personal EC, total symptoms: 1.54 (1.13, 2.10)

Study	Location	Health Outcome	Metric	Representative Concentration (µg/m³)	Exposure Assessment[a]	Selected Effect Estimates (95% CI)
Suglia et al. (2008)	Boston, MA	Lung function among women	BC	Mean (predicted annual): 0.62	Local BC levels estimated by spatiotemporal land-use regression model	Effect estimates for change in lung function: FEV_1: -1.09 (-2.5, 0.3) FVC: -0.62 (-1.9, 0.6) $FEF_{25\text{-}75\%}$: -3.03 (-5.8, -0.3)
Bell et al. (2009)	106 U.S. Counties	ED visits and hospital admissions for respiratory effects	EC	Mean (24-h avg): 0.72	County-wide averages	Percent increase in health effect estimate: 511 (80.7, 941)
Ostro et al. (2009)	6 CA Counties	ED visits and hospital admissions for respiratory effects	EC	Mean (24-h avg): 1.0	County-wide averages	Excess risk per IQR: Ages <19: 5.4 (0.8, 10.3) Ages <19, Cool season only: 6.8 (-0.2, 14.2)
Peng et al. (2009)	119 U.S. Counties	ED visits and hospital admissions for respiratory effects	EC	Median (24-h avg): 0.58	County-wide averages	Percent increase in respiratory hospital admissions: 0.43 (-0.02, 0.85)
Mohr et al. (2008)	St. Louis, MO	ED visits and hospital admissions for respiratory effects	EC	Median (24-h avg): ~0.6-0.7	Continuous measurements from single monitor	Relative risk for asthma ED visits: Ages 2-5, summer: 1.01 (0.93, 1.09) Ages 6-10, summer: 0.98 (0.77, 1.26) Ages 11-17, summer: 1.09 (1.02, 1.17) Ages 2-17, summer: 1.05 (1.00, 1.11)
Sarnat et al. (2008)	Atlanta, GA	ED visits and hospital admissions for respiratory effects	EC	Mean (24-h avg): 1.4-1.7	Positive matrix factorization applied to measurements from single monitor	Relative risk of ED visit for Resp: 0.996 (0.988, 1.003)
Tolbert et al. (2007)	Atlanta, GA	ED visits and hospital admissions for respiratory effects	EC	Mean (24-h avg): 1.6	Continuous measurements from single monitor	Relative risk of ED visit for respiratory effects: 0.996 (0.989, 1.004)
Zanobetti and Schwartz (2006)	Boston, MA	ED visits and hospital admissions for respiratory effects	BC	Median (24-h avg): 1.15	Continuous measurements from single monitor	Percent change in pneumonia ED visits: 11.71 (4.79, 17.36)

[a] A more complete description or evaluation of the BC monitoring method is beyond the scope of this summary, and may include surrogate measurements.

[b] NO=nitric oxide; PEF=peak expiratory flow; FEV_1=forced expiratory volume in 1 second; FVC=forced vital capacity; $FEF_{25\text{-}75\%}$=difference between the 25th and 75th percentile of forced expiratory flow; ED=emergency department.

Table 3-4. Summary of Epidemiological Studies of BC and Mortality. The study reference numbers listed in the left-hand column are for purposes of cross-reference with Figure 3-1. (Source: U.S. EPA)

Reference Number in Figure 3-1	Study	Location	Health Outcome	Metric	Representative Concentration (μg/m³)	Exposure Assessment[1]	Selected Effect Estimates (95% CI)
Short-term Exposure Studies							
—	Cakmak et al. (2011)	3 urban areas in Chile	All-cause mortality	EC	Mean (24-h avg): 2.69 – 5.37	Continuous measurements from 3 monitors (one in each urban area)	Mortality risk ratio: All-cause: 1.084 (1.067, 1.100) ≤64 yrs old: 1.052 (1.019, 1.085) ≥85 yrs old: 1.137 (1.103, 1.173)
37.	Ito et al. (2011)	New York City, NY	CVD mortality	EC	Mean (24-h avg): 1.13	Continuous measurements from 3 monitors	Percent increase in mortality: All year: 2.0 (0.8, 3.3) Warm season: 2.3 (0.3, 4.3) Cold season: 1.6 (-0.1, 3.2)
—	Lippmann et al. (2006)	90 U.S. cities and Hong Kong	All-cause mortality	EC	(Not reported)	Speciation data from U.S. EPA AQS from 2000-2003	"Elevated but nonsignificant increases >0.21 [in health effect estimate] were associated with EC"
38.	Ostro et al. (2007)	6 California counties	All-cause, CVD and respiratory mortality	EC	Mean (24-h avg): 0.966	Continuous measurements across monitors	Percent excess risk: All-cause mortality: 0.7 (-0.6, 1.9) Cardiovascular mortality: 2.1 (0.3, 3.9) Respiratory mortality: 1.2% (-2.2, 4.7)
39.	Zhou et al. (2011)	Seattle, WA and Detroit, MI	All-cause, CVD and respiratory mortality	BC (as a surrogate index of EC in the filter samples)	Median (24-h avg): 0.52-0.71	Continuous measurements from 1 monitor in each city	Percent excess risk: Detroit, warm season: All-cause mortality: 1.359 (-1.027, 3.804) Cardiovascular mortality: 1.227 (-2.276, 8.499) Respiratory mortality: 0.426 (-7.046, 4.062) Seattle, warm season: All-cause mortality: -1.652 (-7.029, 4.035) Cardiovascular mortality: -0.539 (-9.764, 9.629) Respiratory mortality: 10.579 (-8.382, 33.465)
Long-term Exposure Studies							
—	Lipfert et al. (2006)	187 U.S. Counties	All-cause mortality	EC	Mean (24-h avg): 0.79	Speciation data from U.S. EPA AQS from 2002	Coefficient (SE) for association with mortality: 0.1664 (0.05884)
—	Lipfert et al. (2009)	3,065 U.S. Counties	All-cause mortality	EC	Mean (24-h avg): 0.82	Estimates from Atmospheric and Environmental Research, Inc (AER) plume-in-grid air quality model	Cumulative mortality risks: All subjects: 1.07 (1.05, 1.10) Subjects in counties with high traffic density: 1.15 (1.13, 1.16) Subjects in counties with low traffic density: 1.04 (1.01, 1.07)
40.	Gan et al. (2011)	Vancouver, BC	CHD mortality	BC	Mean (24-h avg): 1.19	High resolution land use regression model	Relative Risk of CHD mortality: 1.06 (1.03, 1.09)

1. A more complete description or evaluation of the BC monitoring method is beyond the scope of this summary, and may include surrogate measurements.
2. SE=standard error; CHD=coronary heart disease.

Populations near major roads experience greater risk of certain adverse health effects. The Health Effects Institute (HEI) published a report on the health effects of traffic-related air pollution (Health Effects Institute, 2010). It concluded that evidence is "sufficient to infer the presence of a causal association" between traffic exposure and exacerbation of childhood asthma symptoms. The HEI report also concludes that the evidence is either "sufficient" or "suggestive but not sufficient" for a causal association between traffic exposure and new childhood asthma cases. A review of asthma studies by Salam et al. (2008) reaches similar conclusions. The HEI report also concludes that there is "suggestive" evidence for pulmonary function deficits associated with traffic exposure, but concluded that there is "inadequate and insufficient" evidence for causal associations with respiratory health care utilization, adult-onset asthma, chronic obstructive pulmonary disease (COPD) symptoms, and allergy. A review by Holguin (2008) notes that the effects of traffic on asthma may be modified by nutrition status, medication use, and genetic factors.

The HEI report also concludes that evidence is "suggestive" of a causal association between traffic exposure and all-cause and cardiovascular mortality. There is also evidence of an association between traffic-related air pollutants and cardiovascular effects such as changes in heart rhythm, heart attack, and cardiovascular disease. The HEI report characterizes this evidence as "suggestive" of a causal association, and an independent epidemiological literature review by Adar and Kaufman (2007) concludes that there is "consistent evidence" linking traffic-related pollution and adverse cardiovascular health outcomes.

Some studies have reported associations between traffic exposure and other health effects, such as birth outcomes (e.g., low birth weight) and childhood cancer. The HEI report concludes that there is currently "inadequate and insufficient" evidence for a causal association between these effects and traffic exposure. A review by Raaschou-Nielsen and Reynolds (2006) concluded that evidence of an association between childhood cancer and traffic-related air pollutants is weak, but noted the inability to draw firm conclusions based on limited evidence.

Investigators have attempted to trace PM health effects back to specific sources (e.g., traffic) using source apportionment techniques. A number of these studies have linked BC-rich sources, including motor vehicles and traffic, with adverse cardiovascular and respiratory health outcomes (U.S. EPA, 2009b, Section 6.6.2). For example, Sarnat et al. (2008) found consistent positive associations between cardiorespiratory morbidity and sources related to biomass combustion and metal processing. However, in general there are uncertainties associated with source apportionment methods; these have been characterized in a recent review (Stanek et al., 2011). First, the number of components that comprise PM is not only large, but the correlations between them can be high. Some studies identify the resulting groups or factors with named sources of ambient PM (e.g., "traffic") or PM-related processes (e.g., "secondary organic aerosols"), but many do not draw explicit links between factors and actual sources or processes. Second, there is no well-established, objective method for conducting the various forms of factor analysis and source apportionment, leaving much of the model operation and assignment of factors to sources open to judgment by individual investigators. Because of this and differences in composition and correlations among components between studies, the factors identified vary considerably, thus complicating direct comparisons. Likewise, it cannot be ruled out that a seemingly comparable factor across studies may correspond to different sources depending on location. Despite these uncertainties, a number of studies (e.g., Hopke et al., 2006; Thurston et al., 2005; Mar et al., 2006; Ito et al., 2006; Sarnat et al., 2008) have found that effect estimates based on different source apportionment methods were generally in close agreement, and that the variability in relative risks across source apportionment methods was smaller than the variability across source types (Ito et al., 2006).

Overall, source apportionment studies report little agreement for a particular group of components or sources being responsible for cardiovascular or respiratory effects, which may be due in part to the limited number of studies evaluating these endpoints (Stanek et al., 2011). The results of source apportionment studies indicate that many grouped components can be linked with various health effects, but collectively they have not yielded a clear and consistent association with specific health outcomes.

Finally, it is important to note that a variety of hazardous air pollutants (HAPs) including polycyclic aromatic hydrocarbons (PAHs), dioxins and furans, are co-emitted with BC (Allen et al., 1996; Shih et al., 2008; Hedman et al., 2006; Yadav et al., 2010; Amador-Muñoz et al., 2010; Walgraeve et al., 2010). These HAPs are associated with adverse health effects including cancer and respiratory effects, among others. Reductions in HAP emissions occurring in conjunction with BC mitigation programs will help reduce these health risks. Furthermore, these toxic pollutants are generally persistent once they are emitted into the environment, so these co-benefits can be expected to have long-lasting beneficial impacts (Quiroz et al., 2010; Chi et al., 2010).

3.3.2.2 Magnitude of Impacts of Ambient $PM_{2.5}$ in the United States and Globally

$PM_{2.5}$ is a serious detriment to public health, both in the United States and globally. Regulation of $PM_{2.5}$ concentrations in the United States has resulted in significant declines in $PM_{2.5}$ concentrations and $PM_{2.5}$-related mortality over time (Fann and Risley, 2011). However, many areas of the country remain in non-attainment for the $PM_{2.5}$ NAAQS, and 2005 ambient $PM_{2.5}$ concentrations have been associated with 130,000 premature deaths annually, corresponding to 1.1 million years of life lost (Fann et al., 2011). While a portion of these $PM_{2.5}$-related deaths will be reduced by the recently finalized Cross-State Air Pollution Rule aimed at controlling SO_2 and NO_X emissions (U.S. EPA, 2011e), $PM_{2.5}$ remains a significant risk factor for public health in the United States.

Globally, ambient air pollution concentrations are often much higher than those found in the United States, and the public health burden is correspondingly more severe. In 2004, the WHO estimated that ambient $PM_{2.5}$ in urban areas was associated with about 800,000 premature deaths each year globally, based on surface monitor observations which are limited in many locations around the world (Cohen et al., 2004). More recently, Anenberg et al. (2010) estimated about 3.7 million global premature deaths annually due to outdoor anthropogenic $PM_{2.5}$ using a global atmospheric model to isolate the total anthropogenic contribution to $PM_{2.5}$ concentrations (calculated as the difference between simulated present-day concentrations in 2000 and preindustrial concentrations in 1860) with full spatial coverage including both urban and rural populations. This estimate was still considered to be an underestimate since the resolution of the atmospheric model was too coarse to capture fine spatial gradients of both concentration and population, particularly in urban areas. Impacts of outdoor $PM_{2.5}$ were estimated to be an order of magnitude higher than the impacts of outdoor ozone, due both to high $PM_{2.5}$ concentrations, particularly in very populated areas, and a stronger mortality relationship for $PM_{2.5}$ relative to ozone (e.g., Jerrett et al., 2009; Krewski et al., 2009).

The WHO estimates that urban air pollution ranks as the 10th and 8th major risk factor in medium- and high-income countries, respectively (World Health Organization, 2009). Urban air pollution is not ranked in the top 10 of major risk factors in low-income countries since other risk factors (e.g. childhood underweight and unsafe water, sanitation, and hygiene) are so significant; however, a much larger portion of deaths related to ambient $PM_{2.5}$ are expected to occur in developing regions (Cohen et al., 2004; Anenberg et al., 2010). The ongoing Global Burden of Diseases, Injuries, and Risk Factors Study[3] is expected to update these burden estimates leveraging the advantages from air pollution monitors on the ground, satellite observations, and atmospheric models.

Since the literature on differential toxicity of $PM_{2.5}$ components is currently inconclusive, these studies all assume that all $PM_{2.5}$ components are equally toxic, and calculate premature deaths associated with total $PM_{2.5}$ concentrations from the epidemiology literature. Using the same assumption, Anenberg et al. (2011) estimated that halving anthropogenic BC emissions globally avoids 157,000 premature deaths annually. Multiplying this estimate by two for the total anthropogenic BC burden (using a reasonable assumption that $PM_{2.5}$ concentrations respond about linearly to BC emission changes) yields about 314,000 avoided premature deaths annually worldwide.

3.3.3 Health Effects Related to Indoor BC Exposures

BC is a component of indoor air pollution, which has been implicated in an array of adverse health effects for those who rely on solid fuels for everyday cooking and heating, mostly in the form of biomass (e.g., wood, animal dung, or crop wastes) but also coal (mainly in China) (Rehfuess et al., 2006). The use of solid fuels in poorly ventilated conditions results in high levels of indoor air pollution, most seriously affecting women and their youngest children (Bruce et al., 2000; Martin et al., 2011). Recent observational studies have suggested that indoor air pollution from biomass fuel is associated with respiratory morbidity, including acute lower respiratory tract infections in children (Smith et al., 2000a; 2011) and COPD in women (Orozco-Levi et al., 2006; Rinne et al., 2006; Liu et al., 2007; Kiraz et al., 2003; Regalado et al., 2006; Ramirez-Venegas et al., 2006; Ezzati et al., 2004; Smith et al., 2004). Exposure to biomass smoke in Guatemalan women has been shown to increase diastolic blood pressure (McCracken et al., 2007). Evidence also exists that implicates exposure to biomass fuel smoke in adverse effects on

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[3] *http://www.who.int/healthinfo/global_burden_disease/GBD_2005_study/en/index.html.*

different birth outcomes, including low birth weight and stillbirth (Boy et al., 2002; Sram et al., 2005; Pope et al., 2010). Finally, exposure to indoor air pollution from solid fuel use has been linked to mortality (World Health Organization, 2009).

3.3.3.1 Magnitude of Impacts of Indoor Exposures to $PM_{2.5}$ Globally

Globally, more than half of the population burns solid fuels (e.g. coal, wood, straw, agricultural residue, dung, etc.) for cooking and heating, mainly in the developing world (World Health Organization, 2009). Indoor burning of solid fuels results in high exposure concentrations, as emissions are largely uncontrolled, the homes in which they are used often have poor ventilation, and women and children may spend long periods of time in direct exposure to the emissions during cooking activities. Solid fuel combustion emits a mixture of harmful substances, including $PM_{2.5}$. Consistent with the epidemiological literature on indoor air pollution, impact assessments generally relate risk of mortality with household use of solid fuel combustion, including the total mixture of emissions, rather than using a concentration-response function for individual pollutants (e.g., $PM_{2.5}$).

The WHO estimates that exposure to indoor burning of solid fuels is associated with 2 million annual premature deaths worldwide (World Health Organization, 2009; Smith et al., 2004). Globally, indoor smoke from solid fuels ranks as the 10th leading risk factor for premature death and contributes 3.3% of total deaths. In terms of overall disease burden, as measured in Disability Adjusted Life Years (DALYs), indoor smoke from solid fuels ranks as the 9th leading risk factor globally, associated with 2.7% of all DALYs. It is particularly a problem in low-income countries, where indoor smoke from solid fuels ranks as the 6th leading mortality risk factor (4.8% of total deaths) and the 5th leading disease risk factor (4% of all DALYs). Indoor smoke from solid fuels does not rank as a major risk factor for high-income countries, where use is relatively limited and ventilation is generally sufficient to maintain air quality indoors. As for ambient air pollution, the ongoing Global Burden of Diseases, Injuries, and Risk Factors Study[4] is expected to update these burden estimates with improved assumptions and more recent demographic information.

3.4 Non-Climate Welfare Effects of $PM_{2.5}$, Including BC

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

3.4.1 Role of BC in Visibility Impairment

Particles are the dominant air pollutant responsible for visibility impairment, e.g. "haze," in both urban and remote areas. In the same way that particles influence the Earth's radiative balance, by scattering and/or absorbing solar radiation, they influence the quantity and quality of light received by the human eye and, therefore, one's ability to recognize and appreciate the form, contrast detail, and color of near and distant features. Aerosol-based light extinction can be estimated using the Interagency Monitoring of Protected Visual Environments (IMPROVE) algorithm that multiplies the ambient concentration of PM components by typical component-specific light extinction efficiencies.[5] BC and crustal minerals are the only included components that contribute to light absorption. Under low humidity conditions, BC and OC have the greatest effect on visibility among the major PM species. Per unit mass, the algorithm specifies that BC is 2.5 times more effective at absorbing light than organic carbon is at scattering.

Carbonaceous PM is responsible for a large fraction of regional haze, particularly in the Northwest, where annual average concentrations for 2000-2004 account for 40-60% of the aerosol based light extinction. Most of this average carbonaceous visibility impairment throughout the United States is associated with OC (in both rural and urban

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[4] *http://www.who.int/healthinfo/global_burden_disease/GBD_2005_study/en/index.html.*

[5] See *http://vista.cira.colostate.edu/improve*. 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. The original IMPROVE equation included Rayleigh scattering (from natural atmospheric gasses) and factors for particulate sulfates, nitrates, organic carbon, elemental carbon, fine soil and coarse particles, with a hygroscopic growth function for enhanced light scattering from water associates with the sulfates and nitrates. A recently proposed revision to this equation (Pitchford et al., 2007) enhances the scattering from high concentrations of sulfates, nitrates or organics and adds terms for scattering and hygroscopic growth from sea salt and for light absorption from gaseous NO_2.

areas) because of relatively high OC concentrations compared to BC. Regional haze in the eastern United States generally contains even higher concentrations of carbonaceous PM and light-absorbing BC plays a relatively larger but still minor role compared to OC (DeBell, 2006).

As described further in Chapter 5, urban areas have more carbonaceous PM than nearby remote (rural) areas in the same region (U.S. EPA, 2004b). Western urban areas have more than twice the average concentrations of carbonaceous PM than remote areas in the same region (DeBell, 2006). As shown in Figure 5-6, average urban $PM_{2.5}$ is composed of roughly equal proportions of carbonaceous and sulfate components in some eastern areas. In conditions of high relative humidity common in the eastern United States, hydrated sulfate dominates as the constituent responsible for most urban haze on the haziest summer-time days (U.S. EPA, 2009b).

The 1977 Clean Air Act Amendments called for the development of regulations to address regional haze (visibility impairment) in 156 National Parks and wilderness areas in the United States. The EPA promulgated a Regional Haze Rule (RHR) in 1999 in response to this mandate. Implementation of the RHR entails planned emissions reductions to ensure that by 2064, the worst haze days in these protected areas will improve to natural conditions without degrading visibility conditions for the best haze days. In addition to the RHR aimed at achieving visibility improvements in protected National Park areas, the NAAQS program has been successful at achieving visibility improvements in rural areas, as well as in urban areas where people live and work.

3.4.2 Role of BC in Crop Damage and Other Environmental Impacts

Crop yields can be sensitive to the amount of sunlight received. As discussed in detail in Chapter 2, BC and other airborne particles contribute to surface dimming, and crop losses have been attributed to increased airborne particle concentrations in some areas of the world (Chameides et al., 1999). Auffhammer et al. (2006) found that fossil fuel and biomass burning contributes to reduced rice harvests in India. Decreases in rice and winter wheat yields have also been attributed to regional scale air pollution in China (Chameides et al., 1999).

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

Building materials (metals, stones, cements, and paints) undergo natural weathering processes from exposure to environmental elements (wind, moisture, temperature fluctuations, sunlight, etc.). Deposition of PM is associated with both physical damage (materials damage effects) and impaired aesthetic qualities (soiling effects) for building materials. Wet and dry deposition of PM can physically affect materials, adding to the effects of natural weathering processes, by potentially promoting or accelerating corrosion of metals, by degrading paints and by deteriorating building materials (Haynie, 1986; Nazaroff and Cass, 1991). Fine particles may coat building materials, damaging the appearance of homes, public buildings, and historic landmarks (Hamilton and Mansfield, 1991). Studies have been conducted by a number of authors identifying the anthropogenic sources of soiling and materials damages to monuments and historical buildings (Sabbioni et al., 2003; Bonazza et al., 2005). For example, Bonazza evaluated deposition to the London Tower and found that "deposition of elemental carbon darkens surfaces and has importantly aesthetic implications for buildings." Reduction of PM deposition is beneficial in terms of reduced cleaning, maintenance, and restoration expenditures for buildings and structures.

3.5 Key Uncertainties Regarding Health/Environmental Impacts of BC

A review of the literature describing the health effects associated with ambient concentrations of BC indicates that the strongest relationship exists between BC and cardiovascular effects. This evidence includes support for a continuum of cardiovascular effects ranging from subtle subclinical measures to more severe effects on the cardiovascular system, such as emergency department visits and hospital admissions. These associations are generally consistent with the associations observed for $PM_{2.5}$ and cardiovascular effects (Janssen et al., 2011), though the body of evidence describing the association between BC and cardiovascular effects is much smaller and less consistent than the one characterizing $PM_{2.5}$ and cardiovascular effects. It is noteworthy that, among the studies that characterize the association between BC and cardiovascular effects, a large majority have been conducted in the greater Boston, MA area, utilizing BC measurements from a single BC monitor (see Table 3-2). There can be substantial spatial variation in BC concentrations within a single city, and ambient concentrations of BC in any urban area can vary widely from location to location within the city. Thus, the reliance on this single monitor to estimate exposure for a number of studies across the entire greater Boston, MA area may contribute uncertainty to the reported associations. Similarly, the ambient concentration and composition of PM is geographically heterogeneous, with variations due to unique PM sources and from unique formation, transport, transformation, removal, and infiltration processes in different locations. Thus, a body of evidence that is focused on one geographic area, in this case Boston, MA, introduces uncertainty to the characterization of the association between ambient BC and cardiovascular effects, and the generalizability of this association to broader geographic areas.

An additional uncertainty regarding the health impacts of BC is the inconsistency between the results of studies examining ambient concentrations of BC and the results attributed to traffic in the HEI report (Health Effects Institute, 2010). In examining the body of evidence for health effects associated with BC, the strongest relationship was observed for BC and cardiovascular effects, while the evidence for an effect of BC on respiratory effects was observed to be inconsistent. Conversely, the HEI report on traffic (Health Effects Institute, 2010), concluded that evidence is "sufficient to infer the presence of a causal association" between traffic exposure and respiratory effects (i.e., exacerbation of childhood asthma symptoms), while the evidence for an association with cardiovascular effects was "suggestive." Thus, while BC is a known component of the air pollution mixture attributed to traffic sources, it may have a stronger association with some health effects attributed to traffic (i.e., respiratory effects) than others (i.e., cardiovascular effects). Furthermore, this line of reasoning indicates that there are likely additional components to the air pollution mixture attributed to traffic sources (other than BC) that contribute to the health effects associated with exposure to traffic. Additionally, BC could be serving as an indicator for a larger category of primary combustion particles, which, in addition to BC, can include trace metals and hydrocarbons such as PAHS, any or all of which could be acting to cause adverse health effects.

Chapter 4

Emissions of Black Carbon

4.1 Summary of Key Messages

- Emissions of BC from U.S. sources total about 0.64 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. Approximately 93% of the mobile source total is from diesel sources. Open biomass burning is the next largest source 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 (1500 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 8.4 million tons (7,600 Gg) of 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 emissions 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 eight 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 emissions inventories are based on a limited number of existing source measurements. Better information is needed on chemical composition of PM for some critical emissions sources to improve estimates of BC in these inventories.

4.2 Introduction

Emissions inventories provide valuable information about major sources of BC, both domestically and internationally, and the trends in BC emissions over time. This chapter covers domestic and global emissions of BC and OC. In the case of domestic emissions, the discussion begins with source measurements that generate speciated emissions profiles and ends with a description of the current U.S. emissions inventory for BC and OC by source category, with particular attention to mobile sources, open biomass burning, and fossil fuel combustion.[1] This chapter also provides an overview of key emissions estimates from available global inventories as well as inventories for key world regions such as China and India, and evaluates historical trends in global emissions. This chapter includes a comparison of the U.S. portion of the global BC inventory to the EPA developed estimates. In addition, this chapter discusses the implications of long-range transport of aerosols, which contributes to total BC in the column of air above an area. Based

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[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.

on the discussion in this chapter, key emissions research needs for BC and other light absorbing aerosols are incorporated into the recommendations discussed in Chapter 12.

4.3 U.S. Black Carbon Emissions

4.3.1 Summary of Emissions Methodology

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

In general, EPA estimates emissions of BC and OC by appropriately matching $PM_{2.5}$ emissions estimates from EPA's NEI with source profiles contained in EPA's SPECIATE database (see Appendix 2 for details). SPECIATE is the EPA's repository of PM and VOC speciation profiles of air pollution sources. The PM speciation profiles contain weight fractions of chemical species (e.g., BC and OC) for specific sources. Applying these profiles to PM emissions inventories provides estimates of how much BC and OC is emitted by specific source categories. There are about 300 profiles in the SPECIATE database that are of sufficient quality for this purpose. The mapping of how these approximately 300 profiles have been applied to the over 3,400 source categories available in EPA's NEI for $PM_{2.5}$ is described in Appendix 2 and more details are available in the literature (Reff et al., 2009; Simon et al., 2010). For all non-mobile source and non-open biomass emissions estimates, all BC and OC estimates are based on EPA's 2005 modeling inventories (termed "2005CK" inventories), which rely on the 2005 NEI for $PM_{2.5}$.

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

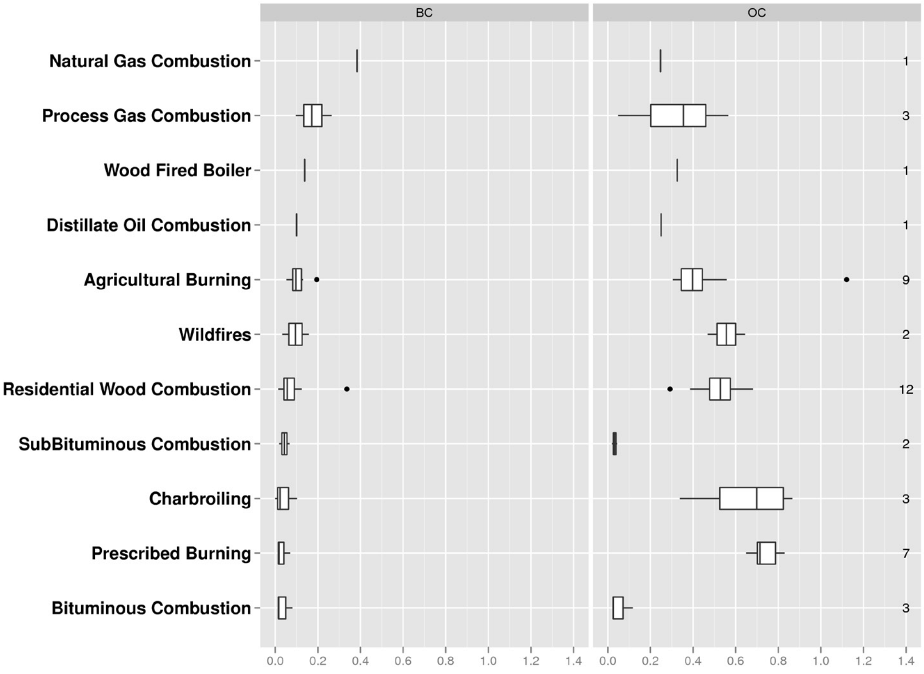

Figure 4-1. BC and OC Fractions of $PM_{2.5}$ Emissions for the Highest BC Emitting Non-Mobile Source Categories in the United States. The box represents the 25th to 75th percentile range and the whiskers represent the 10th and 90th percentile points of the individual source profiles based on emissions source test data as it exists in EPA's SPECIATE database for each source category. The vertical lines within the box represent the median values for that source category. The dots represent outliers. Some of the outliers show a fraction greater than unity; that is due to the statistical procedures used to composite averages. See Appendix 2 for further details. (Source: U.S. EPA)

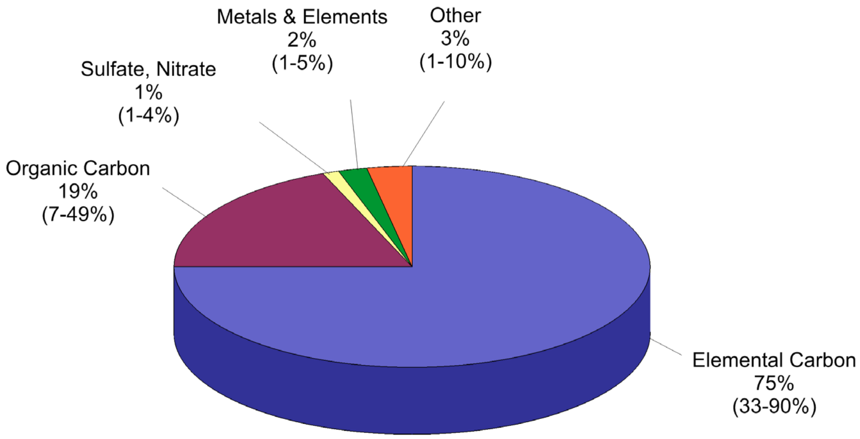

Figure 4-2. Heavy-Duty Diesel $PM_{2.5}$ Emissions Profile.
(Source: U.S. EPA, 2002b)

into EPA's emissions models. There are more data on these issues for gasoline PM than for diesel PM.

$PM_{2.5}$ emissions from open biomass burning (wildfires, agricultural burning, and prescribed burning) come from an emissions inventory compiled by the Regional Planning Organizations (RPOs) for the year 2002 (Regional Planning Organization, 2004a; 2004b; 2005; 2006; 2008). There are five RPOs in the United States which are set up to address regional haze and related issues across the country. Due to the need to accurately represent local/regional fire emissions, each RPO has invested time in including greater regional/local specificity resulting in development of more accurate fire inventories, thereby making them more accurate than national estimates developed by EPA. In addition, these RPO estimates have received more widespread review and acceptance by the states, RPOs and other federal agencies. Though these emissions estimates represent the year 2002, the difference between the year of estimates matters less than the accuracy and review of the estimates; this is because there is very little year-to-year variation in categories besides wildfires. In the case of wildfires, these 2002 estimates are consistent with an average of wildfire activity over a ten year period from 2001 to 2010. BC and OC emissions were then estimated based on these $PM_{2.5}$ estimates using the same methodology explained above. Despite the higher accuracy of RPO emissions estimates as compared to EPA's, it should be noted that biomass burning BC estimates remain more uncertain than engine combustion BC, for example, because of the tremendous year-to-year variability in open burning activity and for other reasons addressed later in this chapter.

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

Figure 4-1 displays the number of resulting profiles (the numbers on the right-hand side of the graphs) and their distribution of BC and OC fractions of $PM_{2.5}$ by source category. Mobile source categories are not shown in Figure 4-1 due to the fact that BC emissions are estimated directly from models for some of those sources (see below and Appendix 2 for more information on all mobile source categories). The number of individual profiles by source category can be quite limited; sometimes only a single value is known. The source categories depicted on the y-axis in Figure 4-1 are shown top-to-bottom in order of maximum BC fraction to minimum BC fraction. Natural-gas combustion (see caveat later in this chapter) has only one profile available but has the highest BC fraction of the source categories shown in Figure 4-1 while some of the burning and coal combustion categories have the lowest BC fractions.

Heavy-duty diesels have the highest BC fraction of all source categories, at an average of about 77% although this percentage varies depending on operating mode and engine technologies. This fact is supported by the EPA's *Health Assessment Document for Diesel Engine Exhaust* (2002b) in which the chemical composition of diesel engine exhaust is identified as shown in Figure 4-2, with BC contributing 75% of the total $PM_{2.5}$ composition. Light-duty gasoline vehicles have a much smaller fraction (about 20-25%) of PM that is BC.

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

In 2005, the United States is estimated to have emitted about 5.5 million tons (or about 5,000 Gg) of primary $PM_{2.5}$ of which about 0.64 million tons (12%) was BC and about 1.7 million tons

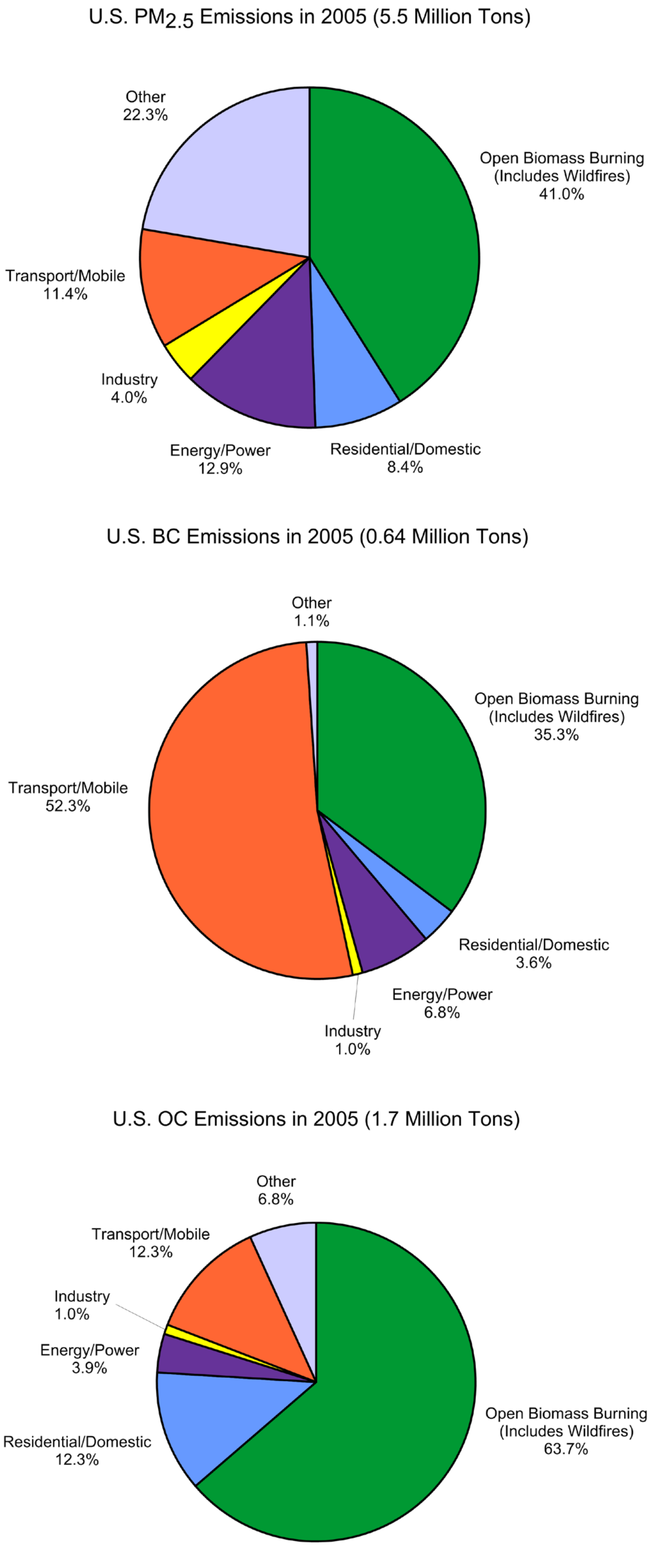

Figure 4-3. Contribution to Primary $PM_{2.5}$, BC, and OC Emissions by Mega Source Categories. (Source: U.S. EPA, 2002a, 2005a)

(30%) was primary OC.[2, 3, 4] Thus at a national level, more than twice as much OC is emitted from domestic sources as BC. The domestic emissions of 0.64 million tons represents about 7% of the world's total BC emissions (i.e., 8.4 million tons) making the United States the 8th 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. In addition, it should be noted that all emissions estimates shown in this chapter are annual averages. There is expected to be some seasonal patterns in BC emissions for some of the source categories. However, that detail is beyond the scope of this chapter. All emissions numbers are only estimates, as detailed in the Appendices. There are uncertainties in all of these estimates that vary from category to category. Systematic quantitative estimates of uncertainty in U.S. emissions estimates are not available at this time, though some qualitative discussion is provided both in this chapter and in the Appendices.

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

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[2] The U.S. emissions estimates presented in this chapter reflect data from EPA's National Emissions Inventory and mobile source models, supplemented with data from U.S. Regional Planning Organizations (RPOs) on open biomass burning (wildfires, agricultural burning, and prescribed burning). Most estimates presented in this chapter are for the year 2005. However, all emissions estimates for open biomass burning are based on a 2002 inventory developed by the RPOs, which are partially funded by EPA. For ease of reference, these various sources are grouped under the label "U.S. EPA" in figures and tables throughout this chapter. More detail on how different portions of the inventory are constructed is provided in other parts of this chapter and in Appendix 2.

[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.

Table 4-1. 2005 U.S. Emissions (tons) and Ratios of Emissions by Mega Source Category. (Source: U.S. EPA)

Mega Source Category	$PM_{2.5}$	BC	OC	OC/BC	$BC/PM_{2.5}$
Open Biomass Burning	2,266,513	224,608	1,058,494	4.7	0.1
Residential	464,063	22,807	204,160	9	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 (Transport)	626,859	333,400	205,172	0.6	0.53
Other	1,232,123	6,743	112,967	16.8	0.01
Totals (Short Tons)	**5,521,456**	**637,167**	**1,662,165**	**2.61**	**0.12**
In GigaGrams (Gg)	**(5,009)**	**(578)**	**(1,508)**		

- Transport/mobile (including on-road, nonroad, locomotives, commercial marine, aircraft and tire/brake wear)
- Other

Table 4-1 shows the actual tons per year of direct $PM_{2.5}$, BC, and OC emissions for these source categories, as well as some key emissions ratios. In the last row, emissions in Gigagrams (Gg) are shown in parenthesis, since metric units are standard for reporting global emissions.[5]

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

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

The mega source categories can be subdivided into more specific categories. Table 4-2 shows the national-level emissions of primary $PM_{2.5}$, BC, and OC emissions for about 90 specific sub-categories

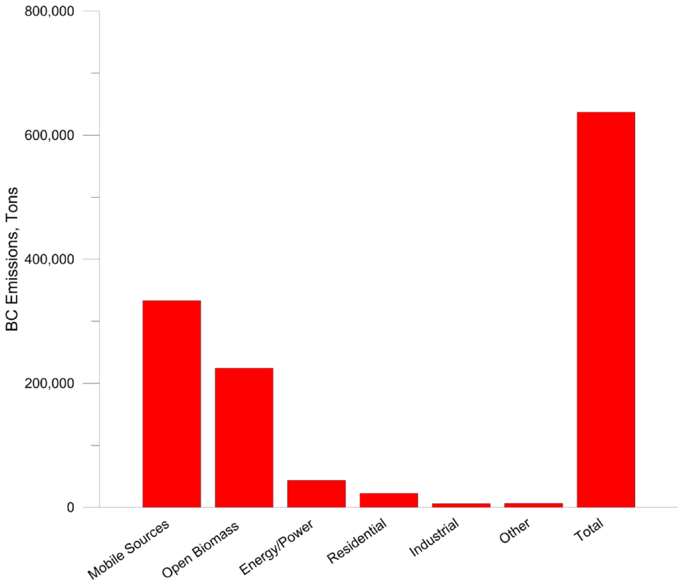

Figure 4-4. U.S. BC Emissions (tons) for Major Source Categories. (Source: U.S. EPA)

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[5] In global inventories, total emissions are often grouped into two main categories, "open" vs. "contained" (or "closed"). To avoid confusion among these terms in this report, the term "contained combustion" is used to refer to all sources except open biomass burning. This is consistent with the global emissions inventory literature. "Contained combustion" is a broadly encompassing term, referring to all combustion sources in which fuel is burned in a chamber or controlled environment (including sources such as industrial/EGU boilers, internal combustion engines, stationary diesel engines, and contained burning of biomass in sources such as wood-fired boilers).

[6] This total includes wildfires. The distinct contributions of wildfires and agricultural/prescribed fires to total domestic emissions of primary $PM_{2.5}$, BC and OC are provided in Table 4-2.

Table 4-2. U.S. Emissions of $PM_{2.5}$, BC, and OC (short tons). (Source: U.S. EPA)

Mega Source Category	Specific Category	Total Primary $PM_{2.5}$	BC	Primary OC	OC/BC	BC/$PM_{2.5}$
Open Biomass	Wildfires	1,600,358	151,855	738,997	4.9	0.09
	Prescribed Burning	535,627	58,525	268,826	4.6	0.11
	Agricultural Burning	130,528	14,228	50,671	3.6	0.11
Residential	Residential Wood Combustion	379,878	21,194	200,645	9.5	0.06
	Residual Oil Combustion	78,672	787	787	1	0.01
	Residential Coal Combustion	2,648	634	1,187	1.9	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	0.64	0.38
	Bituminous Combustion	394,853	6,697	10,387	1.6	0.02
	Sub-Bituminous Combustion	143,383	6,028	4,514	0.7	0.04
	Distillate Oil Combustion	23,718	2,372	5,930	2.5	0.1
	Wood Fired Boiler	56,289	2,088	19,764	9.5	0.04
	Process Gas Combustion	9,457	1,378	2,850	2	0.15
	PMSO2 Controlled Lignite Combustion	20,499	293	5,826	19.9	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

Mega Source Category	Specific Category	Total Primary $PM_{2.5}$	BC	Primary OC	OC/BC	$BC/PM_{2.5}$
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

Mega Source Category	Specific Category	Total Primary $PM_{2.5}$	BC	Primary OC	OC/BC	$BC/PM_{2.5}$
Other	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

Notes:

1. All emissions are for 2005 except those for “open biomass burning,” which are based on 2002 RPO estimates (as referenced earlier).
2. This table represents all emissions in column D as BC; however, they were derived from thermal-optical monitoring techniques and reported as EC.
3. Blank cells indicate that the profiles used showed no BC emissions from these sources.
4. Aircraft inventories only include emissions from landings and take-offs and do not include in-flight emissions.
5. 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).
6. 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). The data used in this report to characterize prescribed burning includes range land and crop residue burning activity as well as other types. Other recent work using satellite-imagery shows the total “agricultural emissions” in the United States (averaged over 5 years) are somewhat lower than the BC emissions estimates shown here (McCarty, 2011). McCarty’s estimates for BC emissions from agriculture burning are based on the inclusion of crop residue burning only and PM emission factors for that type of burning. This is 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 of its own recent work on a 2008 fires inventory that relies on updated remote sensing methods to estimate emissions from agricultural burning.

of sources in the United States. Table 4-2 also shows OC/BC and $BC/PM_{2.5}$ ratios for each of the specific source categories. Some of these data are drawn from the NEI, EPA’s “bottom-up” compilation of estimates of air pollutants discharged on an annual basis and their sources (U.S. EPA, 2005a). As discussed previously, the “open biomass burning” categories shown in yellow come from an emissions inventory compiled by the RPOs for the calendar year 2002 (Regional Planning Organization, 2004a; 2004b; 2005; 2006; 2008).

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 93% of BC. Gasoline vehicles/engines are responsible for most of the remaining BC emissions from the mobile source category. Figure 4-5 shows this more detailed breakout of mobile source BC emissions. In general, diesel $PM_{2.5}$

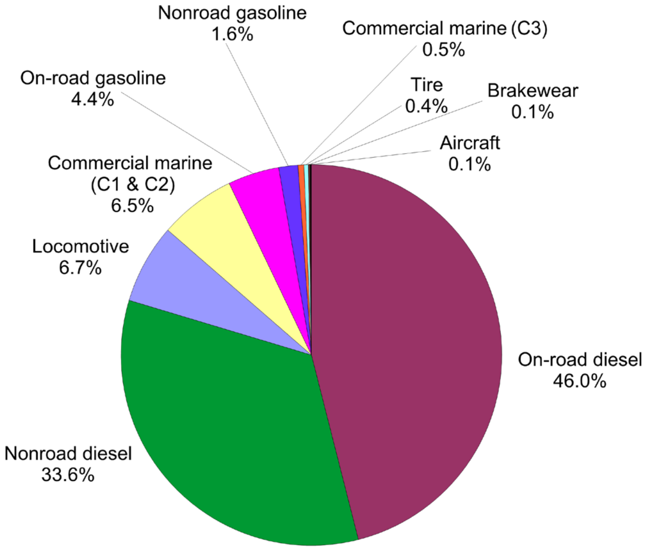

Figure 4-5. U.S. BC Emissions from all Mobile Source Categories (333,400 tons). (Source: U.S. EPA)

consists of about 70-80% BC and about 20% OC.[7] Gasoline $PM_{2.5}$, in contrast, consists of about 20% BC with the remainder being mostly OC. The BC/PM fraction though varies by operating condition such as cold start, high speed/load, and normal driving. Also, as discussed in Appendix 2, diesels used for large (ocean-going) commercial marine vessels have much lower BC/PM fractions due to the high amount of organics and sulfates. Diesel PM is 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, several mobile source measurements suggest 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; Japar and Szkarlat, 1980). 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.

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 8 summarizes mobile source BC inventories for various years from 1990 through 2030, and the control programs that are expected to result in these emissions reductions by 2030.

Numerous source apportionment studies have been done for local areas in the United States including Denver, Los Angeles, Atlanta, Phoenix, and other areas (Watson et al., 1998; Zheng et al., 2007; Brown et al., 2007). These studies show the importance of mobile source emissions to ambient $PM_{2.5}$ (and in some cases, BC) levels, just as various emissions inventories do. The inventories in this report are nationwide inventories and, thus, are not comparable to source apportionment studies which are done in local areas. Also, there are various workshops and studies looking at differing health impacts of PM components including some health studies on source apportionment (Thurston et al., 2005; Marmur et al., 2006).

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, 2007; Bond et al., 2004) 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 included this "biomass combustion" category and their associated emissions.

These biomass combustion sources are estimated to collectively emit a little more than 250,000 tons of BC annually. This represents about 39% of the total amount of BC emitted in the United States, second only to mobile sources in terms of contribution to total domestic BC. The 1.2 million tons of OC emissions from these biomass combustion sources represent about 75% of the total amount of OC emitted in 2005 domestically.[8]

About 90% (roughly 225,000 tons) of total biomass combustion emissions of BC in the United States comes from ''open biomass burning" sources (Figure 4-6). Wildfires contribute about 60% (152,000 tons) to the "biomass combustion" source total with emissions from Alaskan wildfires alone representing about 33% of all biomass combustion emissions in the United States. Emissions from wildfires can vary greatly from year to year (Figure 4-7); however, this single year estimate of 2002 emissions is consistent with an average of wildfire activity in the United States over the ten year period from 2001 to 2010. About 9% (or 21,000 tons) of the national biomass combustion total is emitted by residential wood combustion (from "residential heating/cooking"), and less than 2% (about 5,000 tons) from wood fired boilers and charbroiling (from "other" sources).

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[7] 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.

[8] Often, global inventories define a broad "contained burning" source category that includes the following sources from Table 4-2: all of the sources listed in the "Residential" mega category, "wood fired boilers" in the "Energy/Power" mega category, and "charbroiling" in the "Other" mega category (Bond, 2007; Bond et al., 2004). For the United States, "contained burning" sources defined in this way emit about 27,000 tons of BC combined (in 2005), which represents about 11% of the BC emissions and about 20% of the OC emissions from all biomass combustion (open and contained) that occurs in the United States.

Table 4-3. National Level U.S. Emissions of PM2.5, BC, and OC for Biomass Combustion Sources in 2002/2005 (short tons). (Source: U.S. EPA)

General Category	Specific Source	Total Primary $PM_{2.5}$	BC	OC	OC/BC	$BC/PM_{2.5}$	Mega Source Category in Table 4-2
Open Biomass Burning	Agricultural Burning	130,528	14,228	50,671	4	0.11	Open Biomass
	Wildfires	1,600,358	151,855	738,997	5	0.09	
	Prescribed Burning	535,627	58,525	268,826	5	0.11	
	Subtotal	**2,266,513**	**224,608**	**1,058,494**	**4.7**	**0.1**	
Residential Heating/ Cooking	Residential Wood Combustion	379,878	21,194	200,645	9.5	0.06	Residential
	Subtotal	**379,878**	**21,194**	**200,645**	**9.5**	**0.06**	
Other	Wood Fired Boiler	56,289	2,088	19,764	9	0.04	Energy/Power
	Charbroiling	64,124	2,601	42,975	17	0.04	Other
	Subtotal	**120,413**	**4,689**	**62,739**	**13.4**	**0.04**	
Biomass Consumption Total		**2,766,804**	**250,491**	**1,321,878**	**5**	**0.09**	

Unlike diesel mobile sources, OC/BC ratios for biomass combustion sources are generally much greater than one, indicating a predominance of OC emissions (about 80% on average). Table 4-3 further evidences a smaller OC/BC ratio (on average) for "open biomass burning" than for the other categories of biomass burning; however, the OC/BC ratios are reasonably consistent at about 4 or 5 within the "open burning" categories. While the relatively high OC/BC ratios shown in Table 4-3 for most of these sources may suggest that they do not represent the best mitigation candidates for climate purposes, it should be noted that OC emissions from biomass burning may contain more light-absorbing organic carbon ("brown" carbon) than other sources in general (Hecobian et al., 2010; Moosmüller et al., 2009). Exactly how much of the inventoried OC is light-absorbing is not known at this time.

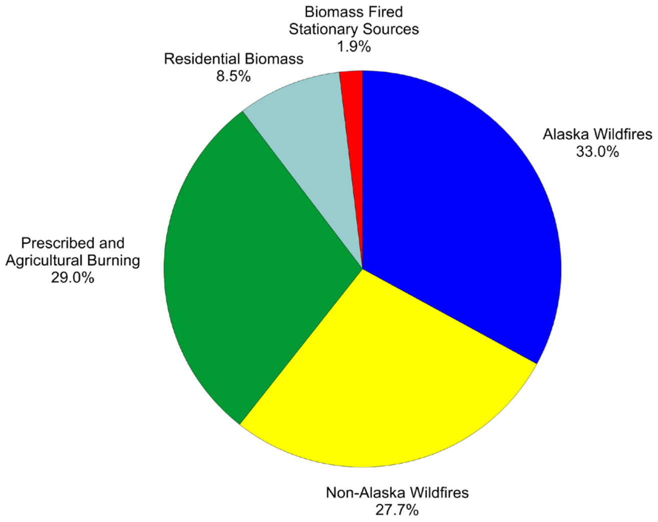

Figure 4-6. U.S. BC Emissions from all Biomass Combustion Source Categories (250,000 short tons). (Source: U.S. EPA)

More than half of U.S. BC emissions from wildfires come from Alaskan fires; due to the close proximity of these emissions to the Arctic, it is likely they would impact the Arctic ice and snow. As discussed in Chapter 2, both BC and OC emissions would be expected to affect Arctic ice melt. However, as noted above, Alaskan wildfire activity is highly variable from year to year and peak emissions occur during the mid-summer season, when they are less likely to influence the Arctic due to prevailing transport patterns during the summer. In addition, fire is a natural ecological process in many ecosystems (see Chapter 11).

4.3.2.3 Emissions from Energy/Power Sector

The energy/power source category contributes approximately 7% of U.S. BC emissions and includes a range of emissions categories, as shown in red in Table 4-2. In general, emissions from these sources are split fairly evenly between BC and OC. The largest fossil fuel combustion source of BC emissions according to the 2005 NEI is natural gas

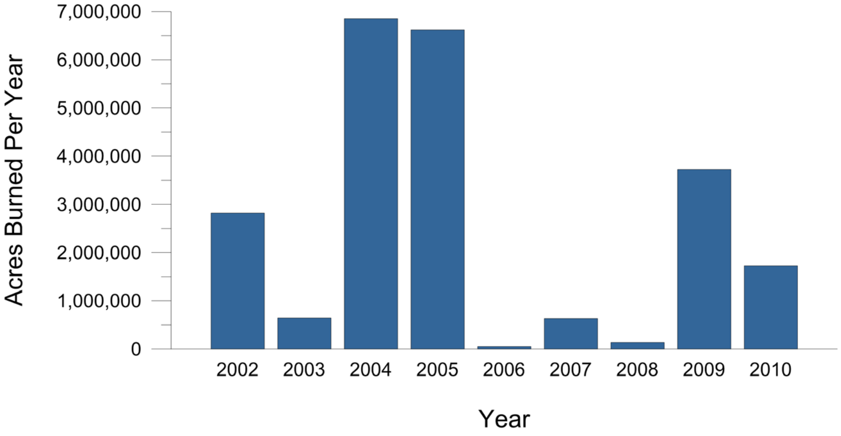

Figure 4-7. Acres Burned per Year in Alaskan Wildfires, 2002-2010. (Source: U.S. Department of the Interior, Wildland Fire Management Information (WFMI) system, *https://www.nifc.blm.gov/cgi.WfmiHome.cgi*)

combustion; however, estimates of the amount of BC compared to OC in direct $PM_{2.5}$ emissions from this source category are highly uncertain.[9] The bituminous and sub-bituminous coal categories, both of which primarily represent electricity generating units (EGUs) but may also reflect small contributions from commercial and institutional sources, represent relatively small contributions to BC emissions in the United States (a little more than 1% each). This small BC contribution is quite different from these sources' contribution to emissions of long-lived GHGs, where they dominate the inventory (e.g., EGUs account for 40% of CO_2 emissions).

4.3.2.4 Emissions from "Other" Source Categories

Table 4-2 shows that the remaining mega categories, "Industrial Sources," and "Other Sources" (in blue and white, respectively), combine to comprise about 2% of total BC emissions domestically. As is explained in more detail in Chapter 9, direct $PM_{2.5}$ emissions from industrial sources in the United States are small compared to emissions of other co-emitted pollutants such as NO_x, HAPs, and CO_2. This is the result of effective control technologies for PM emissions on a variety of stationary/industrial sources. One industrial source of potential interest for additional PM controls is stationary diesel engines (generators, emergency equipment, etc.), which as shown in Table 4-2, has a low OC/BC ratio and contributes more than half of the EC emissions in the "Industrial Sources" category. Existing EPA regulations for new engines in this category are resulting and will continue to result in BC reductions mainly through the use of diesel particulate filters (DPF), although these regulations and resultant controls do not apply to existing engines produced before the model year in which these regulations became effective. Also included in the "other" category are many manufacturing activities as well as fugitive dust emissions sources and charbroiling.

4.4 Global Black Carbon Emissions

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

In general, these global BC inventories are compiled using fuel-consumption data to estimate emissions from particular source categories. A few global inventories are based on a "top-down" concept (Parrish, 2006; Penner et al., 1993) in which emissions are inferred from concentration and ancillary measurements in the ambient air, usually downwind from the source or calculated from generalized emission factors and national or regional activity

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[9] 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.

indicators. Most global inventories, including those of Lamarque et al. (2010) and Bond et al. (2004), used as the basis for this section are based on "bottom-up" type processes. EPA's inventories are also based on "bottom-up" approaches.[10] In this method, emissions are measured or computed directly by concentration, mass flow, and or stream velocity observations at the source, or emissions are calculated (using specific emission factors and activity levels) on a source-by-source or localized basis. Details on methods used to generate both global and domestic BC emissions can be found in Appendix 2, including more details on "bottom-up" approaches.

Global BC inventories have clear advantages when comparing emissions across world-regions, countries, and sectors because the methodology used is consistent across the spatial domain. Global inventories, however, can sometimes overlook important but subtle differences between countries through reliance on default-type information to estimate emissions where actual data are not available. Regional or country-specific inventories, on the other hand, generally contain more accurate emissions information for the domain in question because of the availability of more relevant and more specific data on fuel composition, technology differences in sectors, regulations, emission factors, and activity levels. In this way, the relative importance of certain sources, especially smaller ones, can be incorporated with more accuracy into the final emissions estimates. Unfortunately, each regional inventory tends to employ different methods, making comparisons across regional inventories more difficult. Ideally, regional inventory information could be combined with global inventories to fill in the gaps where global inventories are weakest. While that harmonization has not yet fully occurred, the BC inventories described by Lamarque et al. (2010), below, make an attempt to combine some of the information across global and regional inventories.

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

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

Total global BC emissions for 2000 are estimated to be about 7,600 gigagrams (about 8.4 million tons) for 2000. The spatial distribution of these emissions represented in Figure 4-8 shows Asia, parts of Africa, and parts of Latin America (Central and South America) to be among the regions emitting the largest amounts of BC. Figure 4-9 shows global estimates disaggregated into the these three major world regions responsible for 75% of worldwide BC emissions: (1) Asia (China, India, Southeastern Asia, South Asia, Thailand, Asia-"Stan", Taiwan, Japan, and N. Korea world regions); (2) Africa (Western Africa, Southern Africa rest of, Eastern Africa, Northern Africa, South Africa world regions); and (3) Latin America (South America, Mexico, Central America, Argentina, Venezuela, and Brazil world regions). Asia accounts for about 40% of the global BC emissions, Africa for about 23%, and Latin America for about 12%, as shown in Figure 4-9. Based on these AR5 emissions estimates, the United States accounts for approximately 5% of the global total (i.e., the United States is the 7th most significant region in the world in terms of BC contribution).[11]

Table 4-4 displays total global BC emissions by 37 world regions and by 8 major source categories. Similarly, Table 4-5 shows the distribution of the roughly 35,700 Gg (about 39 million tons) in global OC emissions by these same world regions and source categories. The OC emissions from the United States make up about 3% of the global total.[12] The last column in Table 4-4 shows the ratio of BC emissions from each country or region to those estimated for the United States. For example, China (which comprises nearly all the "East Asia" Region) emits 3.5 times as much BC as the United States.[13] It should be noted that uncertainties/variability in the inventories (both U.S. inventories and global inventories) could confound the ratios presented in the last column in Tables 4-4 and 4-5. The ratios in those tables are simply ratios of the sum total estimate of emissions for the country in question to the U.S. estimate (using the same methods). Source-specific uncertainties could play a role in the overall ranking of a country's contribution to the total global burden of BC emissions (this

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[10] 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.

[11] U.S. EPA estimates of U.S. BC emissions are about 49% higher than those from AR5, which would suggest that the U.S. actually contributes approximately 8% to global BC emissions. These differences are more fully discussed later in this chapter.

[12] EPA estimates OC emissions at about 4% of the global total.

[13] 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.

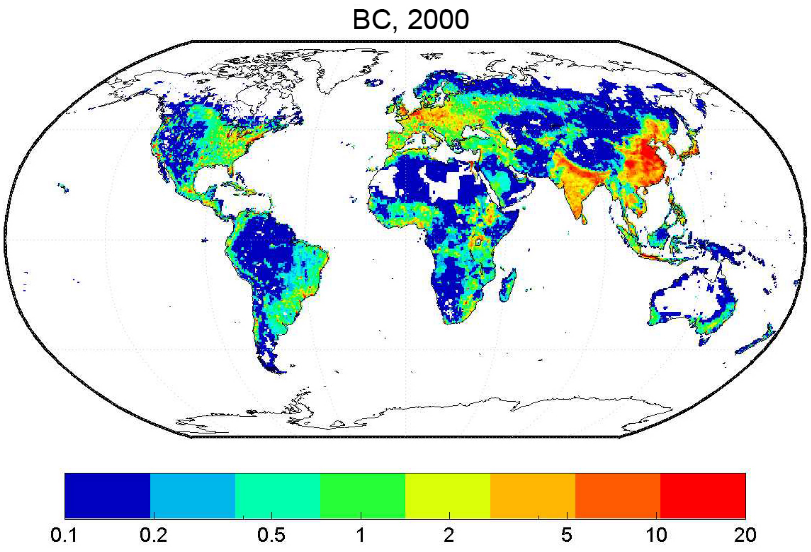


Figure 4-8. Global BC Emissions based on Year 2000 Estimates, in Gigagrams (Gg). (Courtesy of Tami Bond, produced based on data from Bond et al., 2007)

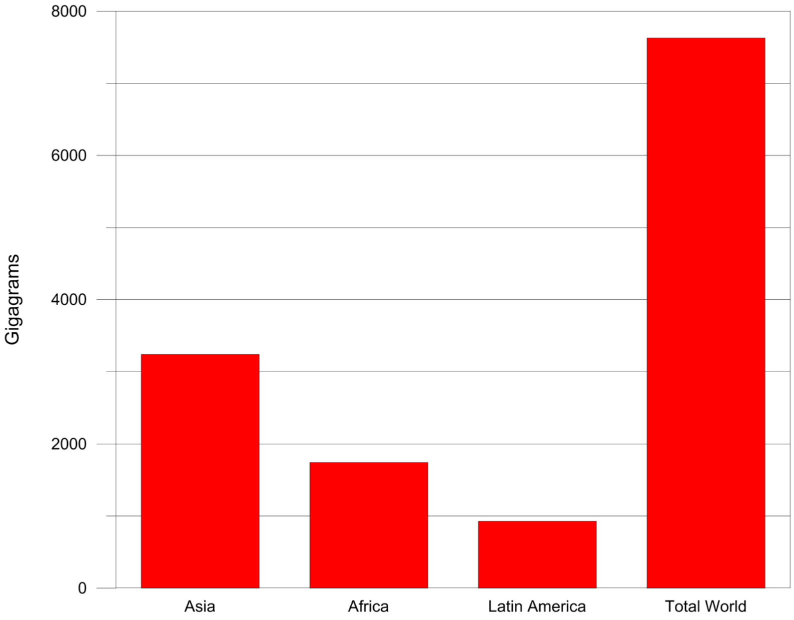


Figure 4-9. BC Emissions by Selected World Region, 2000 (Gg). (Source: Lamarque et al., 2010)

point applies to many of the graphs shown below and is discussed further in section 4.4.4); however, evaluation of how uncertainty plays a role in these ratios and ranking is beyond the scope of this report. While EPA has not done an uncertainty analysis on its inventories, recent work by others (Chow et al., 2011) has looked into the variability in source profiles of BC and OC for the various U.S. source categories to develop a range of emissions estimates instead of just one number. More work along these lines needs to be done in the future for all emissions estimates to have a better understanding of which sectors vary in emissions the most and how that variation plays a role in the estimation of emissions.

Developed world regions like Europe, Japan, and the Middle East have very low BC emissions. In these regions, like in the United States, transportation is the dominant emissions sector. Japan also has significant contributions to BC from industrial sources. In identifying mitigation options, care must be exercised when relying on classifications of world regions/countries as either "developed" or "developing" as surrogates for BC emissions intensity or source to determine how "climate-beneficial" controls might be. China, for example, has the fastest growing economy in the world, yet has a developing country's level of per capita income. While China shares the high BC emissions levels of less developed countries, its sources of BC are not the same as those of less developed areas. A crucial difference between China and other developing areas is China's use of coal in residential combustion, as well as poorly controlled emissions from industry, and apparently a much lower reliance on burning in agriculture than is typical. This makes the contribution to potential warming due to BC emissions greater for China but suggests the most suitable mitigation approaches in China would be different than in other developing countries.

Figure 4-10 groups the global emissions reported in Table 4-5 into six broad source categories, and indicates that global BC totals are dominated by open biomass burning, and residential cooking and heating sources. Roughly 35% of the total global emissions of BC are from open biomass burning, while the domestic (or residential) sector contributes 25% of the global total. In developing countries, most

Table 4-4. Global BC Emissions in 2000 (in Gg). Transport Includes Aircraft and Shipping. (Source: Lamarque et al., 2010)

Region	Energy	Industry	Transport	Residential / Domestic	Ag Waste Burning	Waste	Grasssland Fires	Forest Fires	Total	Country/US Ratio
China	12	669	72	539	44	7	5	9	1,358	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
Russia	5	33	32	102	7	1	35	145	360	0.92
Eastern Africa	0	5	7	119	4	1	210	7	353	0.9
Southeastern Asia (Rest of)	1	30	45	101	3	1	6	166	353	0.9
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	0.3
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	0.2
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

Region	Energy	Industry	Transport	Residential / Domestic	Ag Waste Burning	Waste	Grassland Fires	Forest Fires	Total	Country/US Ratio
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**	**1,497**	**1,340**	**1,947**	**146**	**35**	**1,481**	**1,128**	**7,628**	

Table 4-5. Global OC Emissions in 2000 (in Gg). Transport Includes Only Aircraft. Global OC Emissions Estimates Not Available for Shipping. (Source: Lamarque et al., 2010)

Region	Energy	Industry	Transport	Residential/ Domestic	Ag Waste Burning	Waste	Grasssland Fires	Forest Fires	Total	Country/US Ratio
Western Africa	1	104	43	538	41	3	3,679	882	5,291	6.37
Indonesia	5	34	63	327	57	1	51	3,060	3,595	4.33
Russia	25	23	33	550	34	3	338	2,582	3,588	4.32
China	39	877	72	1,812	208	7	37	122	3,174	3.82
Southern Africa (Rest of)	0	24	9	275	7	0	2,732	34	3,083	3.71
Brazil	8	203	103	85	14	2	487	1,788	2,690	3.24
Eastern Africa	0	22	10	525	20	1	1,461	56	2,095	2.52
Southeastern Asia (Rest of)	8	70	80	428	15	1	41	1,405	2,048	2.46
India	15	260	63	1,301	20	2	38	146	1,846	2.22
Australia	3	8	7	27	19	0	836	165	1,066	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
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	0.5
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	0.3
Middle East	27	14	171	5	28	1	2	0	248	0.3
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

Region	Energy	Industry	Transport	Residential/ Domestic	Ag Waste Burning	Waste	Grasssland Fires	Forest Fires	Total	Country/US Ratio
Baltic States (Estonia, Latvia,	1	1	2	64	0	0	1	11	79	0.1
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**	**2,249**	**1,447**	**7,746**	**696**	**47**	**10,800**	**12,372**	**35,725**	

of the residential (domestic) emissions come from cook stoves that burn biomass, dung or coal resulting in significant emissions of BC. China, India, and Africa to contribute nearly two-thirds of the total BC emissions from this source category, an issue discussed in more detail in Chapter 10.

Table 4-6 displays the global BC, OC, and OC/BC ratios for 6 major source categories. Transportation sources show the lowest OC/BC ratios, while burning categories are seen to be dominated by OC emissions and industrial sources are somewhere in the middle. All these sources also emit CO_2 and other GHGs as well as sulfur emissions that transform into SO_2, NO_x emissions that transform into nitrates and contribute to ozone, and other particles.

Figure 4-11 ranks BC emissions estimates for the 37 world regions shown in Table 4-4, highlighting the relative contribution of open biomass (grassland and forest fires) and anthropogenic sources. With this AR5 BC inventory, regions like Africa, Brazil, and Australia are dominated by open biomass burning sources whereas countries like the United States, China, and India are dominated by anthropogenic sources.

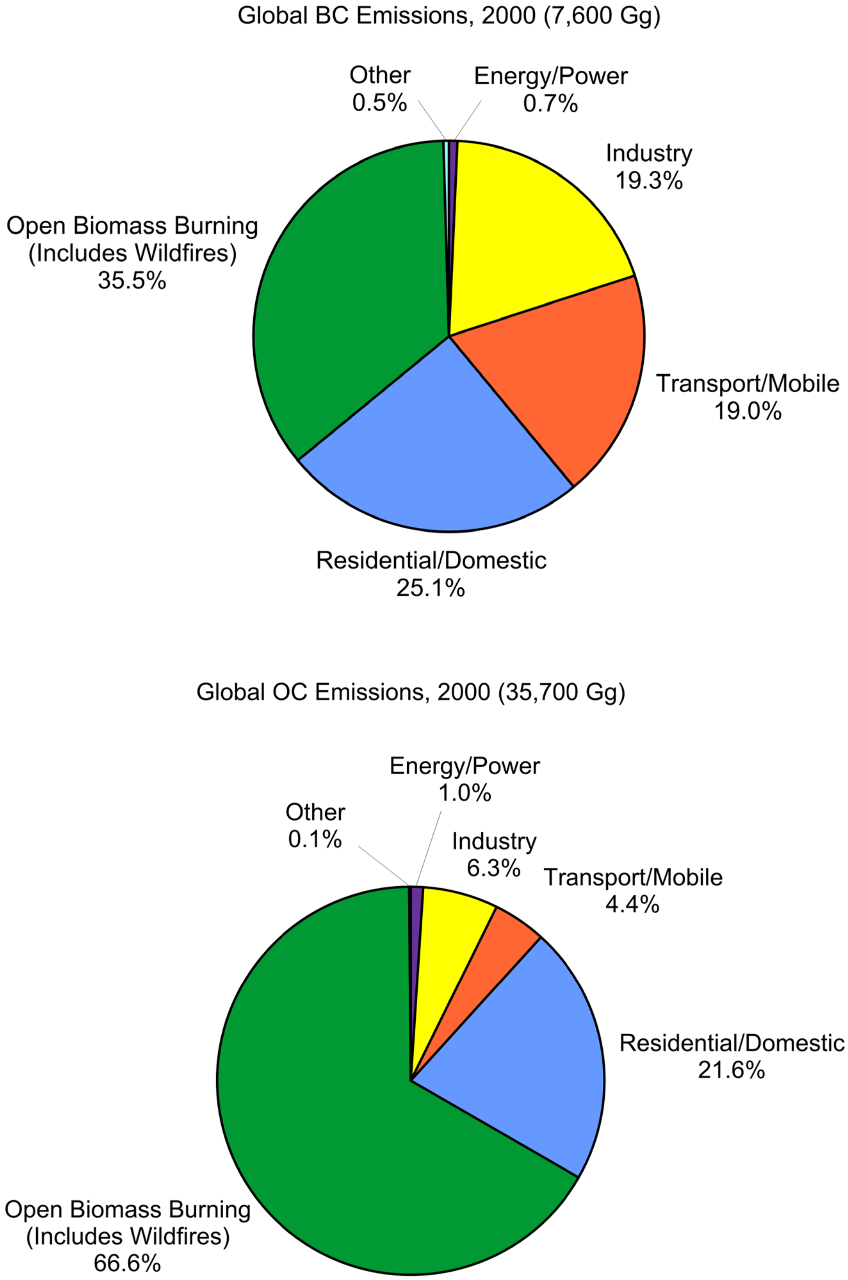


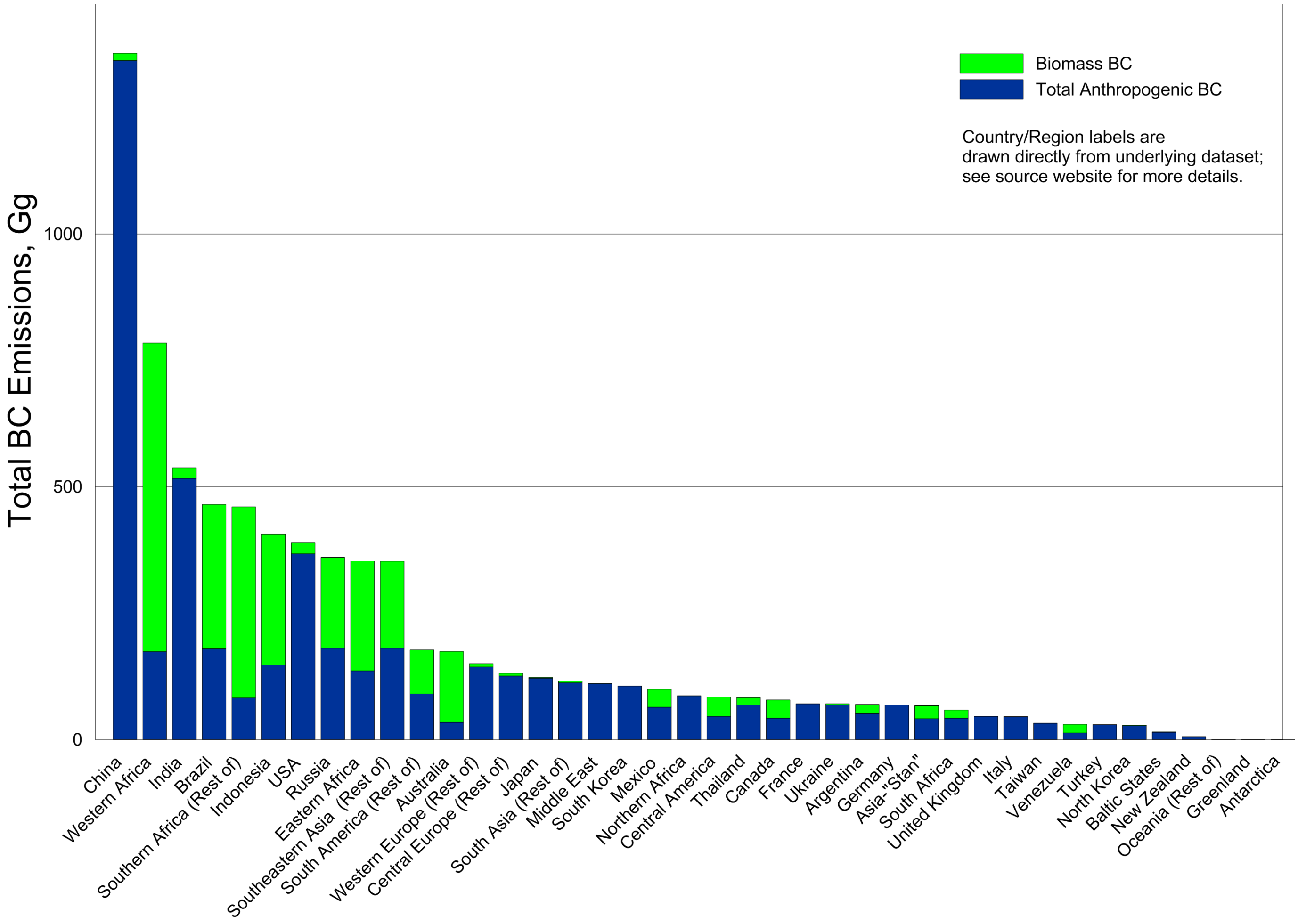
Figure 4-10. Global Distribution of BC and OC Emissions by Major Source Category. (Source: Lamarque et al., 2010)

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

Emissions estimates for BC and OC are generally more uncertain compared to estimates for CO_2, SO_2, or other pollutants primarily because BC is emitted by a large number of small, dispersed sources with irregular operating conditions, such as cookstoves, biomass burning, traffic, and construction equipment. Low technology-combustion (e.g.,

Table 4-6. OC/BC Ratios by Broad Source Categories. (Source: Lamarque et al., 2010)

Source Category	BC (Gg)	OC (Gg)	OC/BC
Energy/Power	54	368	7
Industry	1,497	2,249	2
Transport/Mobile	1,340	1,447	1
Domestic/Residential	1,947	7,746	4
Open Biomass Burning	2,755	23,868	9
Waste	35	47	1
Totals	**7,628**	**35,725**	**4.7**



Country/Region labels are drawn directly from underlying dataset; see source website for more details.

Figure 4-11. BC Emissions by World Region, 2000 (Gg). (Source: Lamarque et al., 2010, based on data from *http://acd.ucar.edu/~lamar/ipcc_ar5/bc.tar.gz*)

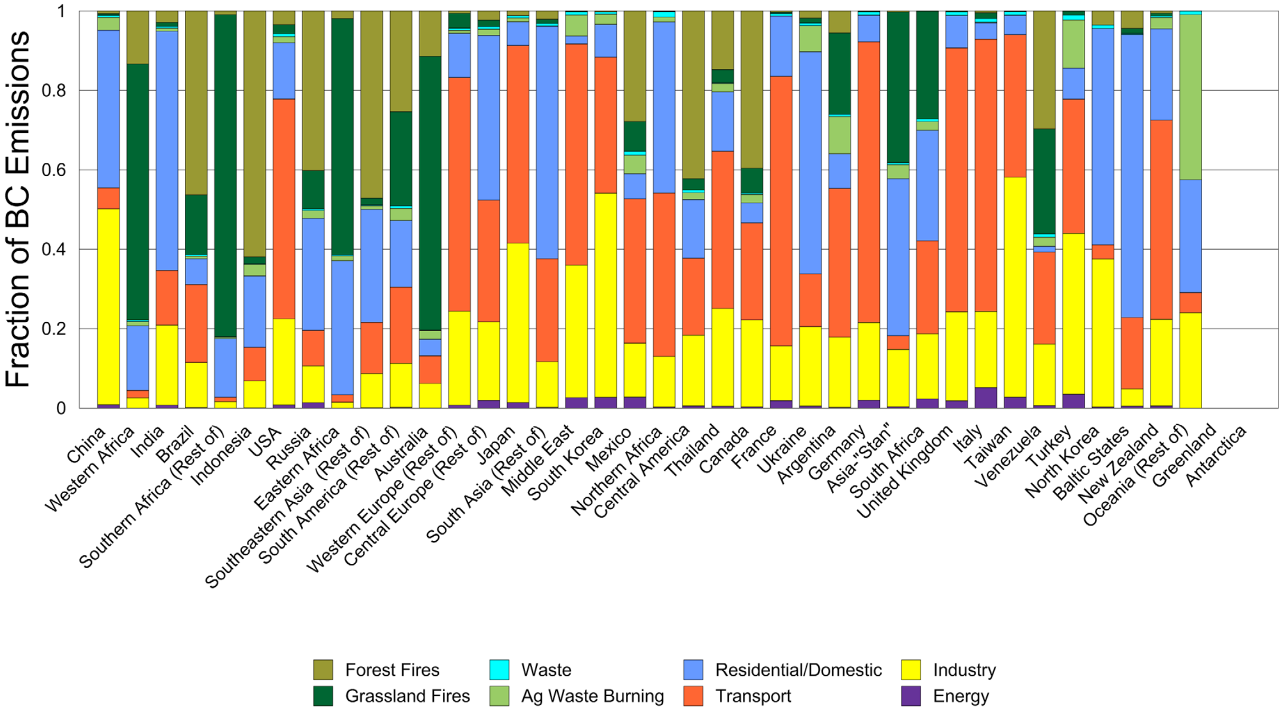


Figure 4-12. Global BC Emissions by Source Categories and Region. (Source: Lamarque et al., 2010)

"open burning") contributes greatly to both the emissions and uncertainties. There has not been a lot of work done on estimating uncertainties with BC emissions estimates. However, Bond et al. (2004) do present a bottom-up estimate of uncertainties in source strength by combining uncertainties in PM emission factors, emissions characterization, and fuel use patterns. They judge the precision of total BC emissions estimates to be within a factor of two. Advances in emissions characterizations for small residential, industrial, and mobile sources and top-down analysis combining field measurements and modeling with iterative inventory development will likely be required to reduce these uncertainties further. The general "factor of 2" in overall uncertainty estimated by Bond et al. (2004) is comparable to the range of estimates of climate forcing by BC given in the 4th IPCC assessment (IPCC, 2007).

4.4.2 Black Carbon Emissions North of the 40th Parallel

Emissions north of the 40th parallel are thought to be particularly important for BC's climate-related effects in the Arctic (Ramanathan and Carmichael, 2008; Shindell et al., 2008b). The 40th parallel north is a circle of latitude that is 40 degrees north of the Earth's equatorial plane. Globally, it crosses Europe, the Mediterranean Sea, Asia, the Pacific Ocean, North America, and the Atlantic Ocean. In the United States, the 40th parallel approximately bisects New York City in the East and San Francisco in the West, passing near Trenton, NJ, Philadelphia, PA, Columbus, OH, Indianapolis, IN, Springfield, IL Kansas City, MO, and Denver, CO. The importance of BC emissions, and especially marine shipping activities (which is a significant source contributor in the Arctic), affecting the Arctic region has been highlighted recently in several articles and reports (Arctic Council, 2009; Skeie et al., 2011). Arctic shipping emissions (which are not fully characterized in Figure 4-13) have been recently published in work on regional inventories by Corbett et al. (2010), Peters et al. (2011), and Paxian et al. (2010).

Global inventories indicate that most BC emissions, particularly from fossil fuels, occur in the Northern Hemisphere. Therefore, emissions north of 40°N latitude may be of particular concern in understanding the impacts of BC on climate. In addition, communities in proximity of the Arctic that are health receptors also stand to benefit from BC emissions reductions north of 40° latitude (these issues are dealt with in greater detail in Chapters 2, 6, and 12). Figure 4-13 presents the magnitude of global BC emissions and source contributions by latitude. Transportation is the largest source of global BC emissions north of the 40th parallel, though open burning, residential burning, and industrial sources all contribute emissions north of 40°N in the Bond et al. inventory. These patterns have implications for assessing the contribution of source regions to snow melt in the Arctic as well as total BC-related forcing in the Northern Hemisphere.

While Figure 4-13 details the global distribution of BC emissions by sector above the 40th parallel, BC emissions from U.S. sources north of the 40th parallel are displayed in Table 4-7. About 260,000 of the 637,000 tons (41%) are estimated to be emitted in areas north of the 40th parallel. In terms of the percentage of emissions from specific source categories occurring above the 40th parallel, most categories show BC emissions contributions north of the 40th parallel that are proportionate to the number of U.S. counties in that region (about 38%). The exceptions are the Fossil Fuel Combustion and Biomass Combustion categories. "North of 40" emissions from biomass burning are seen to be 51% of the total domestic BC emissions from this source category, which is attributable to the wildfire emissions from Alaska. However, as discussed earlier in this chapter, Alaskan wildfire activity is highly variable from year to year, so these emissions may vary. (Furthermore, mitigating wildfire emissions presents particular challenges, as discussed in Chapter 11.) BC emissions from fossil fuel combustion north of the 40th parallel represent only a small percentage (6%) of all emissions across the United States for this source category.

In terms of the contribution of specific source categories to total U.S. BC emissions (from all

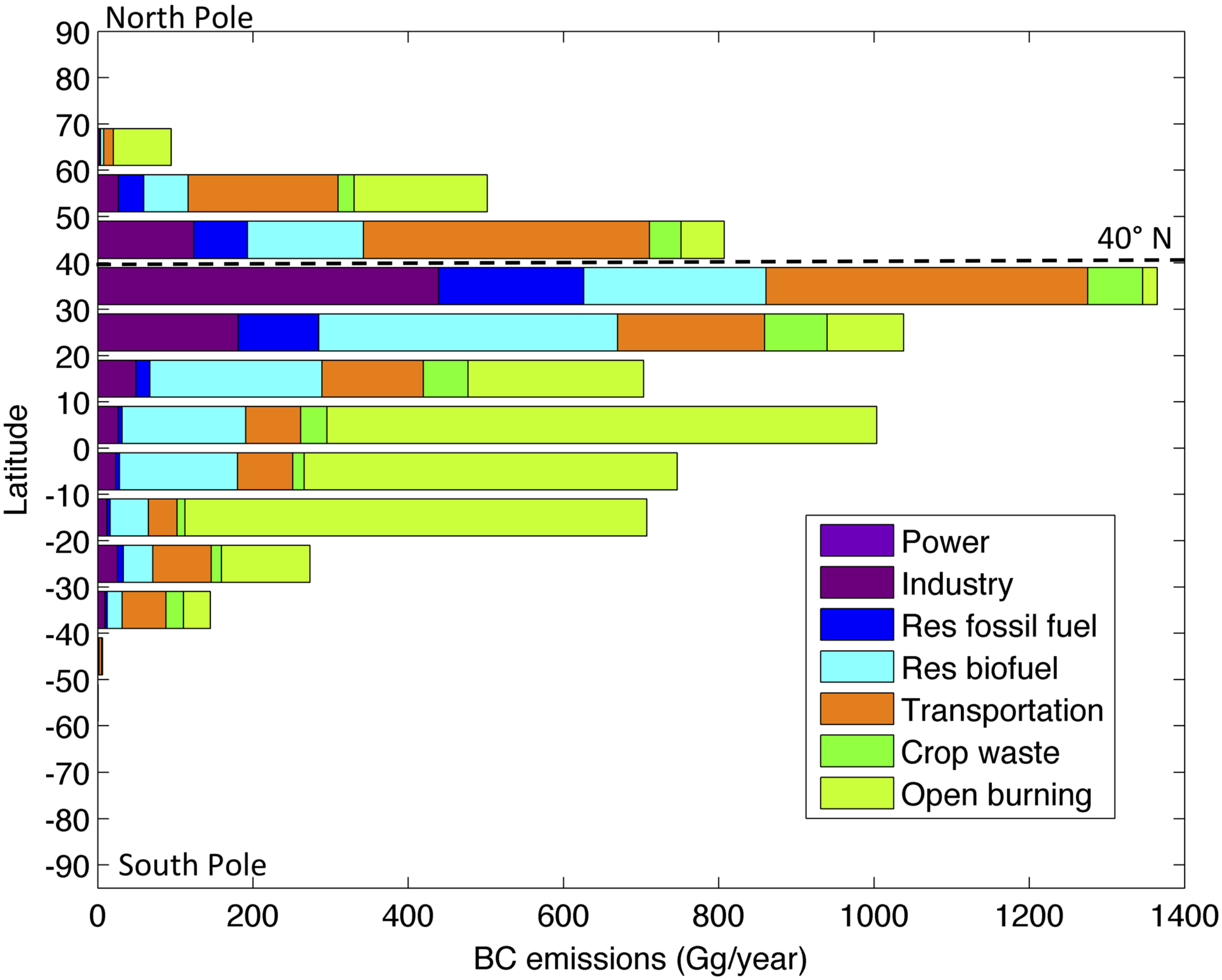


Figure 4-13. Geographical Distribution of Global BC Emissions by Latitude. (Source: Bond, 2008)

Table 4-7. A Comparison of BC Emissions Nationally to Those from Sources "North of 40th Parallel" in 2005 (short tons). (Source: U.S. EPA, 2005a)

Source	Total US BC Emissions	BC Emissions Estimated North of the 40th Parallel	Percent of BC Emissions From This Source Category Emitted North of the 40th Parallel	Percent of TOTAL BC Emitted North of the 40th Parallel
Biomass Combustion	250,499	128,501	51%	49%
Fossil Fuel Combustion	43,049	2,794	6%	1%
Fugitive Dust Sources	1,609	483	30%	0%
Industrial Sources	6,085	1,574	26%	1%
Mobile Sources	333,405	125,784	38%	48%
Other Minor Sources	2,525	755	30%	0%
Totals/averages	**637,172**	**259,891**	**41%**	

categories) occurring above the 40th parallel, it is important to note that biomass burning and mobile sources are by far the dominant contributors. Together, these sources make up approximately 97% of the U.S. BC emissions estimated to occur north of the 40th parallel. The contribution from mobile sources to total U.S. BC emissions north of the 40th parallel is similar (48%) to the contribution of mobile sources to total BC emissions nationally (52%), while biomass combustion contributes 49% of total U.S. BC emissions north of the 40th parallel (compared to 35% of total BC emissions nationally). Again, this reflects the heavy influence of Alaskan wildfire emissions.

4.4.3 Alternative Estimates of Global and Regional Emissions

In addition to the widely used Lamarque/Bond inventory discussed above, there are other global BC and OC emissions inventories compiled by other researchers. Seven other global BC and OC inventories are available in the published literature (Cooke and Wilson, 1996; Cofala et al., 2007; Penner et al., 1993; Junker and Liousse, 2008). The total BC emissions estimated in these inventories fall in the "factor of 2" error range estimated in Bond's BC inventory, which signals that these estimates are generally consistent with the estimates presented above. Most of these alternative emissions are developed using "bottom-up" approaches, similar to that used by Bond et al. (2004) and Streets et al. (2004). These are summarized and discussed further in Appendix 2. The alternative emissions inventories do not provide as much detail or as comprehensive an explanation of uncertainty in the estimates as the Bond inventories employed in this chapter.

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

Most of the regional BC inventory efforts to date have focused on the Asian sub-region (Zhang et al., 2007; Cao et al., 2006; Sahu et al., 2008; Streets et al., 2003a; 2003b; Ohara et al., 2007; Dickerson et al., 2002; Mayol-Bracero et al., 2002; Reddy and Venkataraman, 2002a; 2002b; Parashar et al., 2005) likely due to high emissions of BC and OC from diverse sources there. There are fewer regional BC inventories available for European countries. In general, global emissions inventories have to be used to estimate European BC emissions. Recent

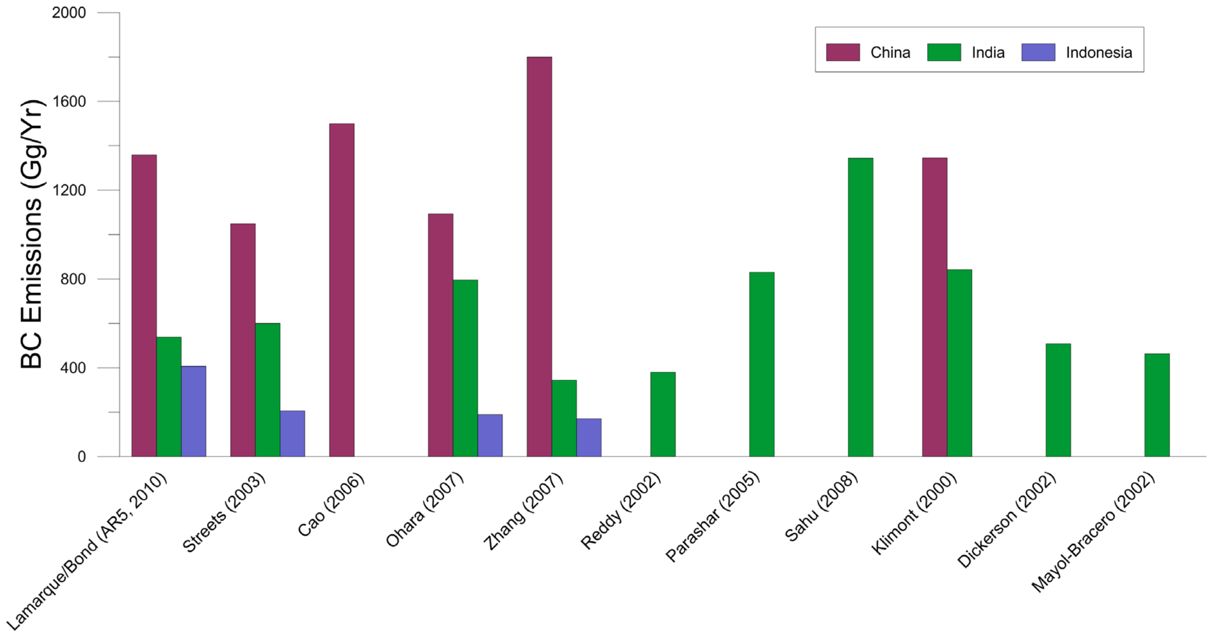


Figure 4-14. Comparison of Regional Inventories for China, India, and Indonesia with AR5 Estimates. (Source: U.S. EPA)

work by the Arctic Council to estimate BC and OC emissions for Arctic nations may provide useful information on regional inventories in those nations. A full list of available regional inventories, along with additional details about the methods used, is available in Appendix 2.

Figure 4-14 compares some of the different regional BC emissions estimates for China, India, and Indonesia to the estimates from AR5 inventories. In general, even though the publication year of the study (indicated on x-axis label in Figure 4-14) is different in most cases, these inventories are seen to be fairly consistent with one another, and also with the Bond global inventory. The range of emissions for a country from these various inventories also gives an indication of the amount of uncertainty in BC emissions estimates for a given region. All of the regional estimates are within the error bounds estimated by Bond et al. (2004) for BC emissions. Recently, EPA's Office of International and Tribal Affairs (OITA) commissioned a study (U.S. EPA, 2011b) to look at reducing BC emissions from various sectors in South Asia. Some of the studies highlighted in Figure 4-14 are discussed in more detail in Appendix A of that report, one of the interesting things the study points out is that BC emissions vary according to the season of the year for certain sectors in South Asia. As a result, emissions tend to peak during the dry season months preceding the monsoon in South Asia.

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

4.4.4 Inventory Comparisons for U.S. Black Carbon Emissions

Table 4-8 compares the U.S. portion of the 2000 global AR5-based BC and OC emissions estimates of Lamarque/Bond et al. (in green) to the EPA's BC estimates for 2002/2005 (in orange). U.S. emissions from the global inventory are aggregated to the highest level of source category detail possible to facilitate comparisons with EPA-based BC

Table 4-8. Comparison of BC and OC Emissions (in Gg) for the United States between AR5 Global Inventories and EPA Inventories. (Source: Lamarque et al., 2010)

BC Emissions (Gg) in AR5 and EPA Inventories

AR5 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 AR5 and EPA Inventories

AR5 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%**

estimates.[14] The degree of difference between the EPA inventory and the AR5 inventory for U.S. emissions is depicted as a percentage in the light blue column.

Total BC emissions for the United States are estimated to be about 390 Gg in the AR5 inventory, and about 572 Gg in the EPA inventories.[15] Most of this approximately 50% difference is driven by EPA estimates for open burning and (to a lesser extent) for mobile sources in the United States that are higher than those from the global inventories. As discussed previously, wildfire emissions can vary greatly from year to year and depend substantially on both fire reporting systems used and emissions calculation method (Larkin et al., 2009; 2010). This may explain some of the difference between the

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[14] In general, aggregating 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.

[15] EPA's estimate of the domestic BC emissions in Table 4-8 (572 Gg) is a bit smaller than the total BC emissions estimate shown earlier in this chapter (578 Gg). This difference stems from the fact that most of the sources in the "Other" mega source category from the U.S. inventory were not included here. In addition, note that 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. EPA derived inventory.

estimates for open burning as the AR5 estimates are based on the year 2000 and the EPA estimates on the year 2002 and different reporting systems and fuel loading/consumption characteristics are used. Also, EPA estimates include all nonroad and on-road emissions in the transportation source category, while global inventories group emissions from some of the smaller nonroad sources into the "Industry" category. This could account for global inventory estimates of U.S. emissions being lower for "transport" and higher for "industry" compared to the EPA estimates. In the case of OC emissions, Table 4-8 shows that the AR5 total is about 830 Gg while the EPA estimates are seen to be about 1,405 Gg, a difference of about 69%. As with BC, most of this discrepancy stems from fire emissions that EPA estimated to result in more OC than do the AR5 estimates for the United States. It is likely that fire emissions (both OC and BC) from many countries are under-estimated due to the methods used to estimate fire emissions in global inventories including an insufficient accounting for emissions from smaller fires.

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

4.5 Long-Range Transport of Emissions

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

The Task Force on Hemispheric Transport of Air Pollution (TF HTAP) organized under the Convention on Long-range Transboundary Air Pollution conducted a multi-model assessment of long-range transport of aerosols and other pollutants from four main source regions in the Northern Hemisphere approximating the populated regions of North America, Europe, South Asia, and East Asia (TF HTAP, 2010). The models included in the study produced widely varying estimates of the absolute amount of intercontinental transport of aerosols. Most of the diversity in model estimates appears to be due to differences in the representation of physical and chemical transformations that aerosols undergo in the atmosphere, which leads to differences in the estimated atmospheric lifetime of aerosols. Uncertainties in emissions estimates and atmospheric transport algorithms also contribute to the diversity of estimates. A systematic comparison between the TF HTAP ensemble estimates and observations in the mid-latitudes has not been conducted.

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

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

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

In recent work, Kopacz et al. (2011) estimated the contribution of BC emissions sources to BC concentrations and deposition in the Himalayas and Tibetan Plateau and the associated direct and snow-albedo radiative forcing. They conclude that emissions from northern India and central China and from western and central China contribute most of the BC in the Himalayas and Tibetan Plateau, respectively, although the contributions of different locations vary with season. However, they also show that the Himalayas and Tibetan Plateau region can receive significant contributions from very distant sources including biomass burning in Africa and fossil fuel combustion in the Middle East. They estimate that the snow-albedo effect of BC deposition on snow in the region results in a warming influence that is an order of magnitude larger than the direct radiative forcing influence.

Given the paucity of anthropogenic sources of BC in the Arctic, a large fraction of the climatic impact of BC in the Arctic can be attributed to long-range transport. Shindell et al. (2008b) examined the results of the TF HTAP multi-model experiments for insights about transport to the Arctic. Comparing to observations of BC at Barrow, Alaska, and Alert, Canada, all of the models appeared to underestimate the transport of BC to the Arctic. Consistent with the findings for the source-receptor relationships at mid-latitudes, they found that the models varied widely in terms of the absolute estimates of the contribution of different source regions, but were similar in their estimates of the relative contributions. The ensemble results suggest that European emissions are the largest contributors to surface BC in the Arctic (due to the high latitude, and therefore Arctic proximity, of many European sources), while East Asia is the largest contributor to BC in the upper troposphere (Figure 4-15) (Shindell et al., 2008b). Additional source apportionment analysis under the TF HTAP (2010) concluded that anthropogenic emissions from Europe and open biomass burning emissions from Eurasia both contributed about 35% of the surface BC in the Arctic. Anthropogenic emissions from the North

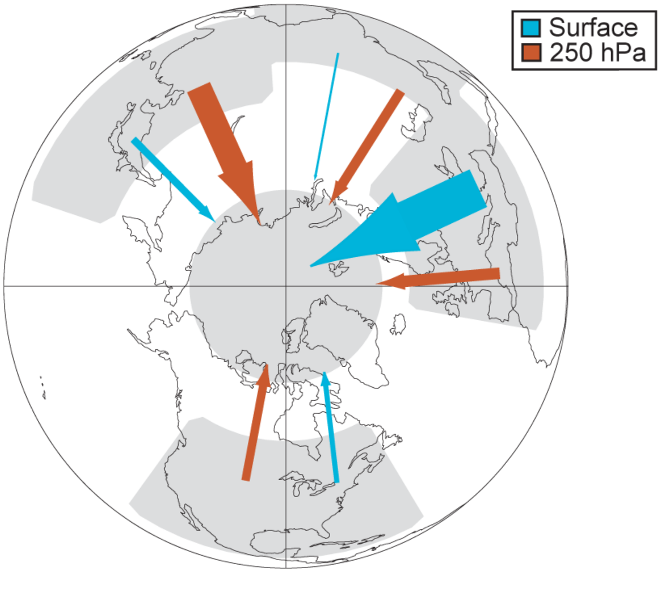

Figure 4-15. Relative Importance of Different Regions to Annual Mean Arctic BC Concentrations at the Surface and in the Upper Troposphere (250 hPa). Values are calculated from simulations of the response to 20% reduction in anthropogenic emissions of precursors from each region (using NO_x for ozone).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

American study region, not including open biomass burning, accounted for an average of 5% of surface BC in the Arctic region, with model estimates spanning the range from 2% to 10% (TF HTAP, 2010). However, unlike the rest of the Arctic, deposition of BC in Greenland, location of the second-largest ice sheet in the world, is most sensitive to North American emissions (Shindell et al., 2008b).

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

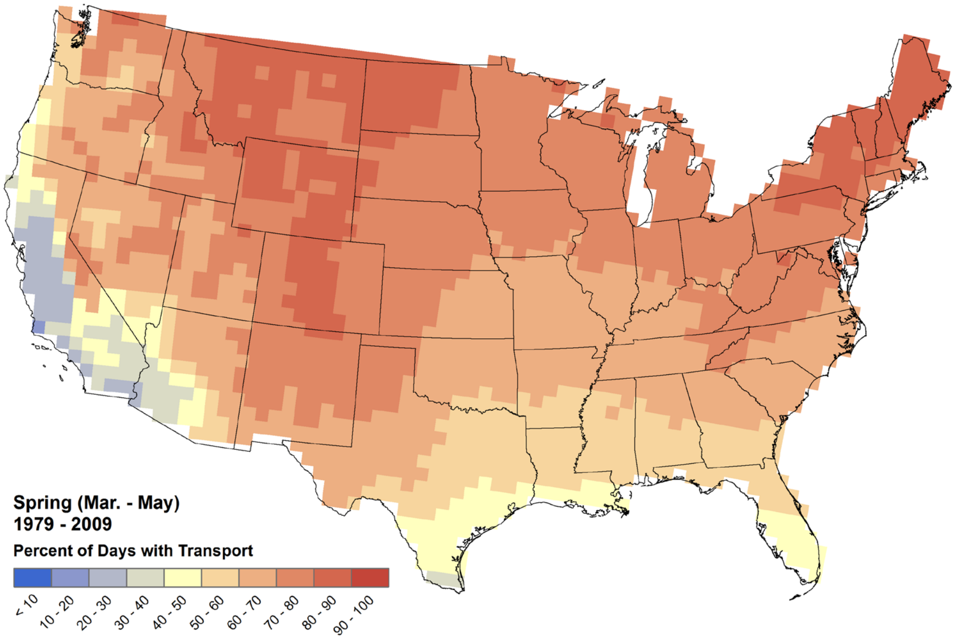

Figure 4-16. Potential for Transport of U.S. Emissions to the Arctic based on the percentage of spring days (March through May) in which seven-day forward trajectories initiated within the boundary layer (below 2000m) reached the Arctic Circle. These percentages are based on 31 years of trajectories using meteorological data from 1979-2009. (Source: USDA Joint Fire Science Program)

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

Within the United States, the potential for transport of domestic BC emissions to the Arctic is known to vary by location and season. Given its proximity to the Arctic, BC emissions sources in Alaska are likely to have an impact on the Arctic, depending on the synoptic weather conditions. For emissions sources in the contiguous United States, recent trajectory modeling work by the Joint Fire Science Program (DeWinter et al., 2011; Larkin et al., 2011) has shown that the probability of emissions impacting the Arctic is critically dependent on the specific injection height into the atmosphere and the specific synoptic weather patterns prevalent at the time. Seven day forward trajectories were computed for a 31 year period (1979-2009) using synoptic weather patterns starting at a number of injection heights for each location within the contiguous United States. Figure 4-16 displays the percentage of spring days (March-May) where any of the 7-day forward trajectory releases below 2000m reached the Arctic Circle. This analysis suggests that the potential for springtime transport of BC ground-level emissions from the contiguous United States to the Arctic can be significant. Over the southern portion of the United States, the potential for transport to the Arctic is frequent (40-60%), but a significant number of days without transport remain. For locations in the northern part of the United States and other higher-altitude locations, the analysis indicated that the potential for transport trajectories to the Arctic is very common (> 70%). However, even in areas which show a large seasonal and climatological potential for transport, it is possible to identify multi-day periods where transport to the Arctic is limited. The dependency on source location and synoptic weather conditions may have implications for understanding source apportionment and for implementing mitigation strategies. Tools are being developed to enable the use of daily predicted transport potential in mitigation strategies.

4.6 Historical Trends in Black Carbon Emissions

4.6.1 U.S. Black Carbon Emissions Trends

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

Since mobile source emissions are modeled, a time series of BC emissions can be generated more easily for this source category than for other U.S. source categories. Mobile sources have experienced a 32% reduction in BC, a 51% reduction in OC, and a 36% reduction in $PM_{2.5}$ emissions from 1990 - 2005. From 1990 to 2005, BC emissions decreased by 79%, 30%, and 25% for on-road gasoline, on-road diesel, and nonroad diesel sources, respectively. Continued reductions are expected for mobile sources in the next two decades, as discussed further in Chapter 8. BC emissions for nonroad gasoline sources, though extremely small, did not change from 1990 to 2005.

BC emissions trends for the other major source categories (open biomass burning, industry, and energy/power) are difficult to estimate due to lack of data and inconsistent measurements and methods over time. The methods used to estimate emissions from 1990 to 2008 have changed significantly, as has the way $PM_{2.5}$ estimates are used to derive BC emissions estimates. There are no BC estimates available for any non-mobile source categories for the year 1990. From 1990 to about 1998 there was about a 30% reduction in direct $PM_{2.5}$ emissions from EGUs and other power-generation sources due to controls on direct $PM_{2.5}$. It is expected that some of these reductions in direct $PM_{2.5}$ led to decreases in emissions of BC, but this is difficult to verify without consistent speciation data for the entire time period. In 1999, there was a major change in the methods used estimate $PM_{2.5}$ emissions. Based on these new methods, from 1999 to 2008 an additional 21% reduction in direct $PM_{2.5}$ is seen from this source category. In contrast, direct $PM_{2.5}$ emissions from industrial sources are estimated to have declined only 6% during the entire 1990-2008 period (U.S. EPA, 2010i).

Long-term trends in emissions from biomass burning categories (wildfires, prescribed burns, and agriculture burns) are not available due to significant year-to-year changes in the methods used to estimate emissions. For that reason, in the modeling assessments "average fires" are used to represent emissions from this source category. However, qualitative estimates of annual wildfire frequency/activity as well as future wildfire activity in the United States are available and suggest upward trends. Global climate change is expected to make the increased activity even greater with more fuel availability and drier, more combustion-friendly conditions.

4.6.2 Global Black Carbon Emissions Trends

There are a number of studies available which have looked explicitly at global BC emissions trends over time (e.g., Bond et al., 2007; Ito and Penner, 2005; Novakov et al., 2003). Figure 4-17 (Bond et al., 2007) shows the growth in global BC emissions from key source categories (excluding biomass burning) during the period between 1850 and 2000. The figure shows that emissions of BC have increased almost linearly, totaling about 1000 Gg (approximately 1.1 million tons) in 1850, 2200 Gg (approximately 2.4 million tons) in 1900, 3000 Gg (approximately 3.3 million tons) in 1950, and 4400 Gg (approximately 4.8 million tons) in 2000. The slower growth between 1900 and 1950 may be due to economic circumstances and also the introduction of cleaner technology, especially in developed countries. OC shows a similar pattern of linear growth that is slightly slower in the mid-1900s.

Figure 4-18 relates BC emissions trends from Bond et al. (2007) to earlier work done by Ito and Penner (2005), and by Novakov et al. (2003).[16] Ito and

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[16] Novakov et al. looked at BC from fossil fuel combustion only.

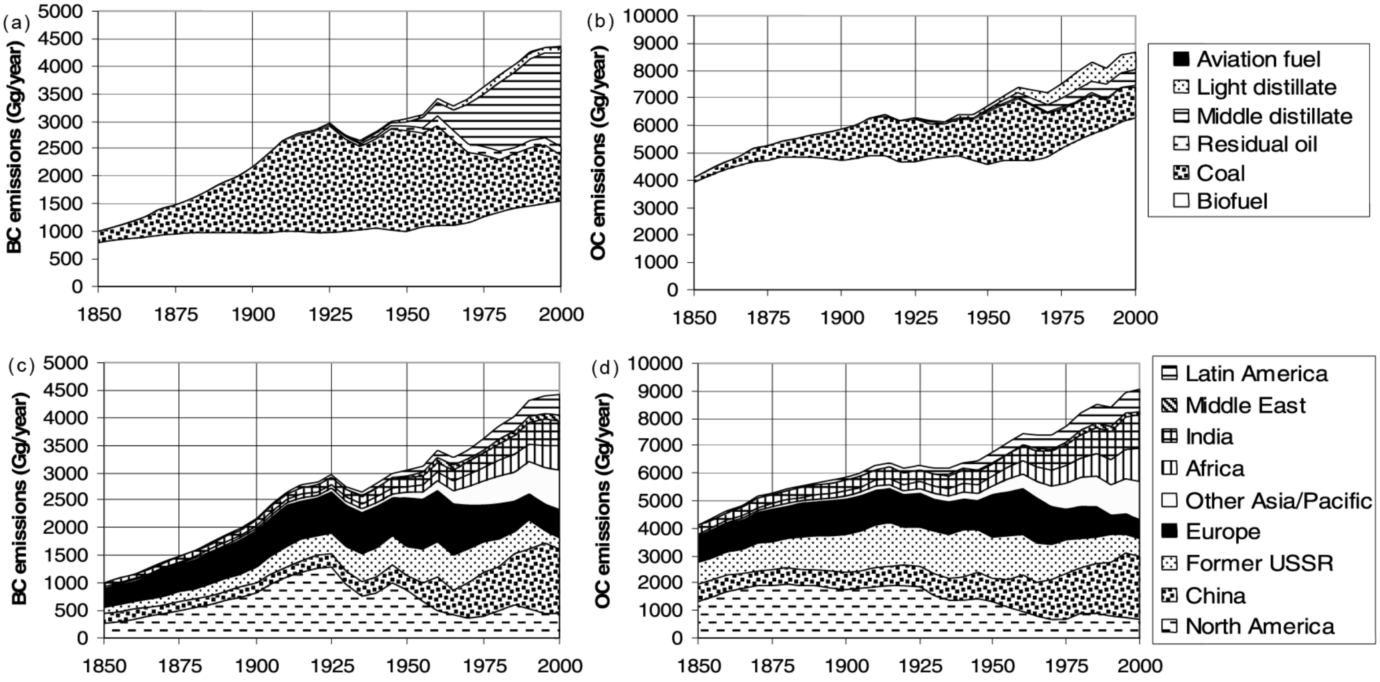

Figure 4-17. Historical Growth in Emissions of BC (Panels a, c) and OC (Panels b, d), Segregated by Fuel (Panels a, b) and World Region (Panels c, d). (Source: Bond et al., 2007)

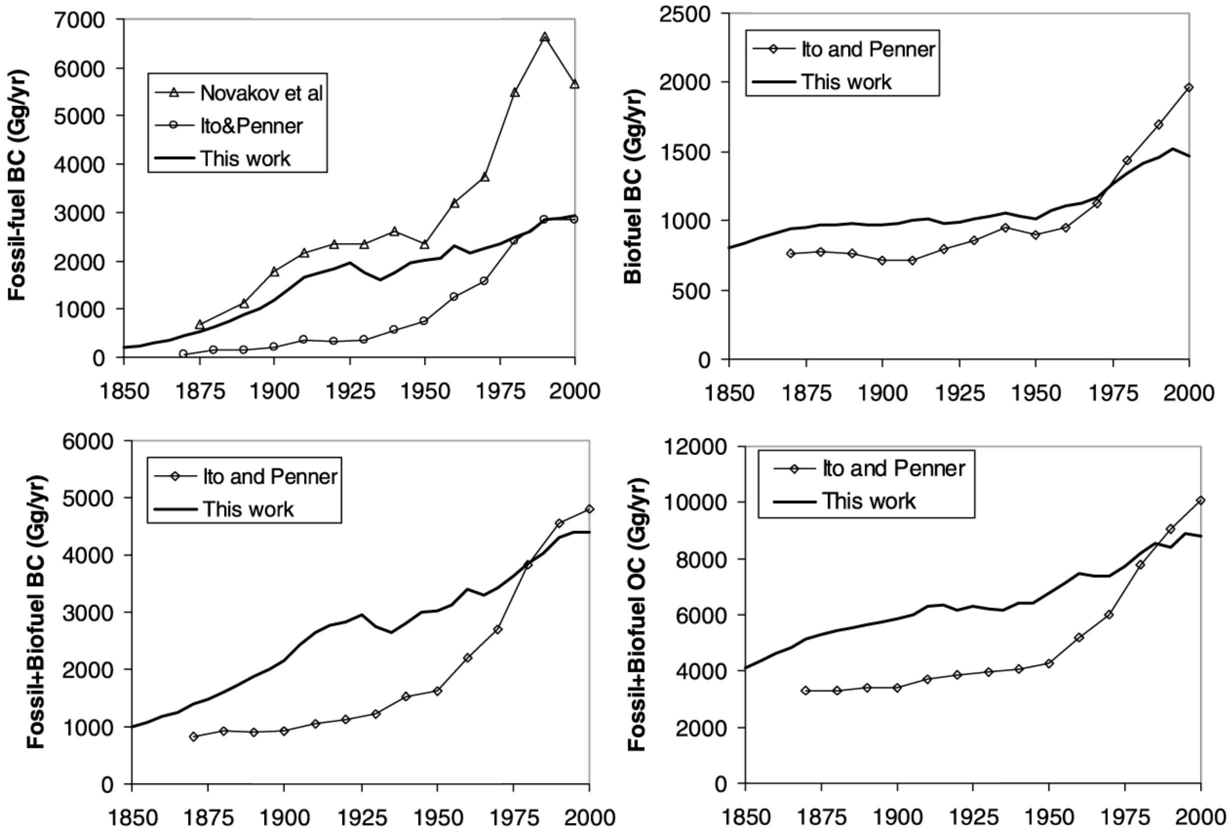

Figure 4-18. Historical Reconstruction of Global Emissions Trends. Comparison of Bond et al. (2007) ("this work") with Previous Studies.

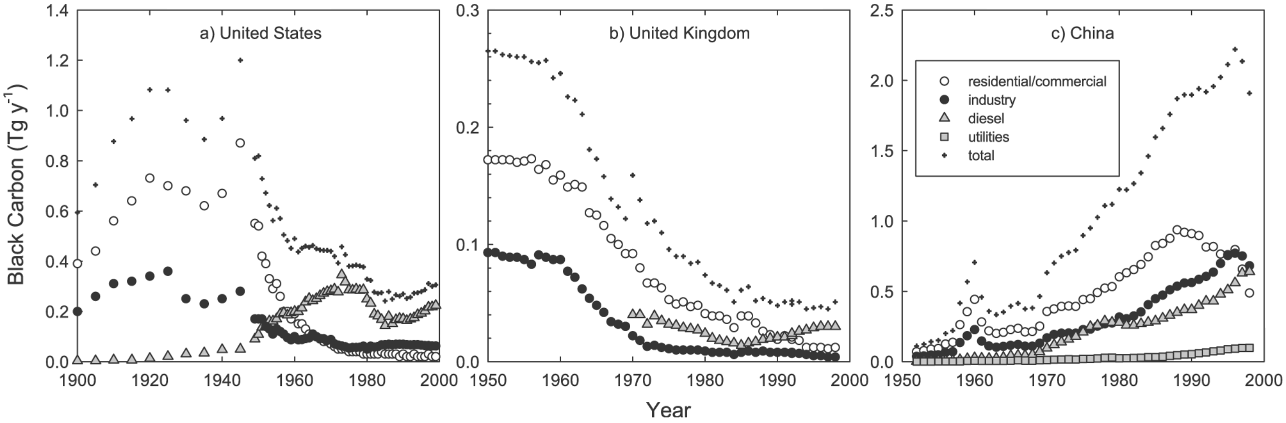

Figure 4-19. BC Emissions (Tg /y) in the United States, United Kingdom, and China (Novakov et al., 2003). BC emissions are estimated from annual consumption data for the principal BC producing fossil fuels and BC emission factors disaggregated by utilization sector. BC from biofuels and open biomass burning are not included.

Penner show a very similar trend and magnitude in BC emissions from biofuel, but the magnitude of fossil-fuel BC emissions is much lower. In the late 1900s, Bond et al.'s biofuel emissions increase less (about 30% between 1960 and 2000 vs. 100% for Ito and Penner). By contrast, Novakov et al. estimated higher fossil-fuel BC emissions than Bond et al., in the early 1900s. Novakov et al.'s work was based on total BC aerosol, while Bond et al. and Ito and Penner's work focused on fractions of PM less than 1 micron in diameter. Novakov et al. shows flat BC emissions between about 1910 and 1950, similar to Bond et al. (2007).

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

Figure 4-19 shows the estimated BC emissions trends for the U.K., United States, and China (Novakov et al., 2003). According to these data, emissions from the United States peaked in 1920, while Europe peaked in 1950 and has declined about 90%. Total global emissions of BC, however, have been steadily increasing since 1875 (Novakov et al., 2003). Presently, global BC emissions total approximately 8.4 million tons. Almost all of the increase in recent decades is from developing countries in Asia, Latin America, and Africa. China and India contribute nearly 25% of global BC emissions.

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

Chapter 5

Observational Data for Black Carbon

5.1 Summary of Key Messages

- Estimates of 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 (EC) and filter-based optical (BC) techniques. BC and EC values from these measurement methods are highly correlated, although the method-defined values may differ by as much as a factor of two; however, consistent measurements in long-term monitoring networks are sufficient to detect trends that correlate with emissions reductions. Published studies show that BC:EC ratios derived by commercial instrumentation are generally within 30%.

- The United States has recently standardized BC measurements for its major routine speciation monitoring networks. Additional research is needed to further standardize ambient and emissions measurement methods and to develop factors that harmonize existing measurements produced from different sampling and analytical techniques. It is also recommended that light absorption be reported in the original units of absorption along with any mass absorption coefficients or conversion factors used to convert absorption to BC mass concentration.

- Ground-level BC measurements across the globe indicate estimated concentrations ranging from < 0.1 μg/m^3 in remote locations to ~15 μg/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 North America or Europe. A comparison of urban concentrations to corresponding regional background levels reveals an urban increment of up to 2 μg/m^3 in North America and Europe compared to an urban increment of ~6-11 μg/m^3 in China.

- In the United States, BC comprises ~5-10% of average urban $PM_{2.5}$ mass.

- Long-term records of historical BC 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 the 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 is 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 two decades when U.S. ground-level ambient BC measurements are available, ambient BC concentrations have declined substantially, most likely due to reductions in mobile source emissions and other controls on direct $PM_{2.5}$ emissions. Since 2007, the decline may be due in part to recession-related decreases in vehicular travel and industrial output. The ambient concentration declines appear to be stronger in urban areas and may in fact be larger than the estimated average reductions in total BC and direct $PM_{2.5}$ emissions in the United States.

- Estimates of the total atmospheric column using remote sensing qualitatively show similar spatial variability in absorbing aerosol levels across the globe to ground level measurements. Remote sensing measurements that utilize multiple wavelengths also show that the absorbing particle mixture varies globally among areas dominated by urban-industrial sources, biomass burning, and wind-blown dust.

5.2 BC and Other Light-Absorbing Carbon: Measurement Methods

Current measurement techniques generally estimate BC on the basis of light absorption characteristics or by thermally isolating a specific carbon fraction. The techniques used currently to estimate BC mass concentrations are summarized in Table 5-1. These two general categories of BC measurement techniques can be viewed as different indicators of the chemical and physical properties of BC.[1] This is discussed further in the text box, "Measurement Approaches for BC." The two most common BC measurement techniques are thermal-optical and filter-based light absorption as denoted in the table below.

Table 5-1. Description of BC Measurement Techniques.

Method Type	Method Description	Prevalence of Use
Light absorption/optical	*Filter-based:* Light absorption by particles is measured through a filter loaded with particles; BC is quantified using factors that relate light absorption to a mass concentration.	High
	Photoacoustic: 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.	Low
	Incandescence: Incandescent (glowing) particle mass is measured; BC is quantified by calibrating the incandescent signal to laboratory-generated soot.	Low
Isolation of specific carbon fraction	*Thermal-Optical:* 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.	High
	Thermal: 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.	Low

Thermal-optical measurements involve exposing a particle-laden filter to a series of heating steps. These measurements involve a multi-step temperature program to evolve OC in pure helium and EC in a helium/oxygen atmosphere with an optical (transmittance or reflectance) correction for pyrolysis or the charring of organic material during analysis. Despite the seeming simplicity of thermally separating particulate carbon into two fractions, there is considerable uncertainty in assigning carbon mass to either OC or EC fractions. For example, charring of particles during the thermal analysis has produced erroneous OC and EC assignments (Cadle et al., 1980; Huntzicker et al., 1982; Yu et al., 2002b). In addition, there are several different commonly used temperature protocols that cause variation in the OC and EC assignments. Watson et al. (2005) and Chow et al. (2006) provide a detailed summary of the variety of methods used that demonstrates the wide range of thermal analysis protocols in use. The two thermal-optical methods that are predominantly used in monitoring networks in the United States are the thermal-optical transmittance (TOT) and thermal-optical reflectance (TOR) methods (Chow et al., 1993; Chow et al., 2007; Peterson and Richards, 2002). The TOT method differs from the TOR method in the thermal combustion program used and the method of correcting for char (transmittance versus reflectance). Long-standing reliance on the thermal optical methods has resulted in an extensive observational record based on OC/EC splits, and the frequent use of EC as an estimate of BC.

While EC is directly quantified as the mass of carbon atoms that evolve during a thermal or thermal-optical analysis, optical techniques observe the light-absorbing properties of the particles to estimate BC. Filter-based, optical instruments are relatively low cost, readily available, and simple to operate, and thus are frequently field deployed

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[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 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".

Measurement Approaches for BC

The chemical and physical properties of carbonaceous PM vary in terms of both refractivity (the inertness of the carbon at high temperatures) and light absorption. Each carbon measurement technique provides unique information about these properties. All current analysis methods are operationally defined, meaning that there is no universally accepted standard measurement approach. When developing methods and operational criteria to identify BC, some scientists use its optical properties or light-absorbing characteristics (optical or light absorption methods), some use its thermal and chemical stability (thermal-optical methods), while others use its morphology or microstructure or nanostructure (microscopy methods). One major class of methods, thermal or thermal-optical techniques, distinguishes refractory and non-refractory carbon as EC or apparent EC (ECa), and OC or apparent OC (OCa) respectively (see Figure 5-1). The second major class of methods, optical methods, quantifies the light absorbing component of particles as BC or apparent BC (BCa), which can be used to estimate BC concentrations and can also indicate the existence of components that absorb in the near-UV (i.e., brown carbon, BrC). Light absorbing carbon (LAC) is a term used for light-absorbing particles in the atmosphere, which includes BC and BrC.

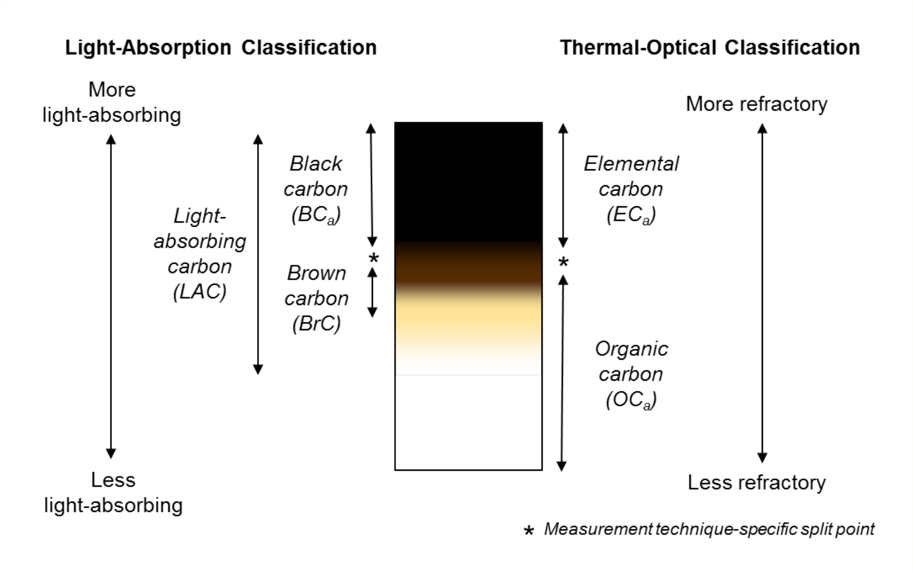

Figure 5-1. Measurement of the Carbonaceous Components of Particles. Black carbon and other types of light-absorbing materials can be characterized by measuring their specific light-absorbing properties, as seen on the left side of the figure (BCa/BrC/LAC). This contrasts with other approaches to characterizing particles based on measurements of the refractory nature of the material (inertness at high temperatures), as seen on the right side of the figure (ECa and OCa). (Source: U.S. EPA)

to measure BC. Filter-based instruments measure the quantity of light transmitted through a filter loaded with particles (Hansen et al., 1982; Lin et al., 1973; Rosen and Novakov, 1983). For filter-based optical instruments, the detected light absorption by particles can be converted to an estimated BC mass concentration. There are two main uncertainties associated with the quantification of filter based BC using optical methods: 1) a filter loading artifact and 2) the selection of an appropriate conversion factor. Several studies have shown that filter-based BC measurement can be affected by the amount and composition of particles loaded onto the filter (Arnott et al., 2005; Collaud Coen et al., 2010; Park et al., 2010; Schmidt et al., 2006; Virkkula et al., 2007; Weingartner et al., 2003). In addition, the selection of the conversion factor to relate light absorption to mass is a significant issue. No single factor is applicable to all methods, wavelengths, particle sizes, particle compositions, shapes and structures. There are a variety of conversion factors that have been published in scientific literature and are commonly applied to estimate BC (Gundel et al., 1984; Liousse et al., 1993; Petzold et al., 1997; Bond and Bergstrom, 2006; Novakov, 1982). Theoretical and empirical studies show that bounds can be placed on absorption efficiencies for different assumptions of BC aerosol origins and composition

(Alfaro et al., 2004; Andreae et al., 2008; Dillner et al., 2001; Cross et al., 2010; Chou et al., 2005; 2009; Favez et al., 2009; Fu and Sun, 2006; Fuller et al., 1999; Horvath, 1993; Jacobson, 2010; Knox et al., 2009; Liousse et al., 1993; McMeeking et al., 2005; Nordmann et al., 2009; Ogren et al., 2001; Ram and Sarin, 2009; Ramana et al., 2010; Rosen and Novakov, 1983; Schuster et al., 2005; Subramanian et al., 2010; Watson et al., 2005; Widmann et al., 2005; Chan et al., 2010). A suggested solution would be to quantify BC in the original light absorption units, which is a strength of the optical techniques. It is recommended that light absorption be reported in the original units of absorption along with any mass absorption coefficients or conversion factors used to convert absorption to BC mass concentration.

While the terms "BC" and "EC" are frequently associated with measurements from the two general categories of specific commercial instruments in the scientific literature, both of these measurement techniques provide estimates of BC concentrations (Wolff et al., 1982; Andreae and Gelencsér, 2006). Ambient monitoring studies that simultaneously utilized light absorption and thermal-optical methods show that the estimates of BC by the two techniques are on average near 1 and generally within 30% (70% of studies had ratios between 0.7 and 1.3); however there do exist studies reporting very low BC:EC ratios (~0.5) and very high BC:EC ratios (~2). Ambient inter-comparison studies have found that estimates of BC from thermal measurement methods are usually reliable predictors of ambient BC estimated via light absorption techniques and vice versa. The comparison of EC by thermal-optical methods and BC by light absorption is sensitive to the source of EC/BC and varies by location. While the estimates from the two techniques are highly correlated and display similar concentration values, they can vary by up to a factor of two among the limited number of studies available.[2] Further discussion of these comparisons can be found in Appendix 1.

5.3 Ambient Concentrations of BC

Currently, few countries have robust networks for ambient measurement of $PM_{2.5}$. Most available global ambient BC data are produced in the United States, Canada, Europe, and China, and the vast majority of these data are based on the more widely available thermal measurement techniques (see section 5.2). In the United States and Europe, limited light absorption measurements are available to supplement these thermal measurements. There is also a modest network of BC monitoring sites across the globe in remote areas to provide information about background levels.

5.3.1 Major Ambient Monitoring Networks

Figure 5-2 provides a map showing the extent of known BC monitoring networks around the globe. The existing networks in the United States, Canada, Europe (EUSAAR, EMEP), and Asia (CAWNET), as well as those with global coverage (GAW, ESRL/GMD) and ad hoc collections of special study data are shown.[3] The map separately shows locations using light absorption, thermal, or both measurement techniques. Most locations shown are in North America and these monitors mostly utilize thermal measurement techniques.

Ambient BC data in the United States are mostly available from $PM_{2.5}$ urban and rural speciation monitoring networks which use thermal measurements. The Interagency Monitoring of Protected Visual Environments (IMPROVE) network started collecting data in 1987, and the urban Chemical Speciation Network (CSN) started in 2001.[4] Urban BC is measured through the CSN network of approximately 200 monitors located in major urban areas.[5] In rural environments such as national parks and wilderness areas, the United States relies on the IMPROVE network to characterize air quality. This network consists of approximately 160 monitors. Like the CSN, the IMPROVE network utilizes thermal measurement technologies.[6] Other U.S. data include supplementary measurements from approximately 45 semi-continuous light absorption monitors (operational in 2007); 5 semi-continuous carbon measurements; and smaller networks of thermal-optical and light absorption monitors (SEARCH, Super-sites). See Appendix 1 for more details. Thermal-optical methods for measuring OC and EC are currently standardized and consistent among the IMPROVE (*http://vista.cira.colostate.edu/improve*),

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[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.

[3] These BC measurements are publicly available or have been included in peer-reviewed publications.

[4] The VIEWS web site (*http://vista.cira.colostate.edu/views/*) provides information on the start and end dates for each site.

[5] 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.

[6] 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.

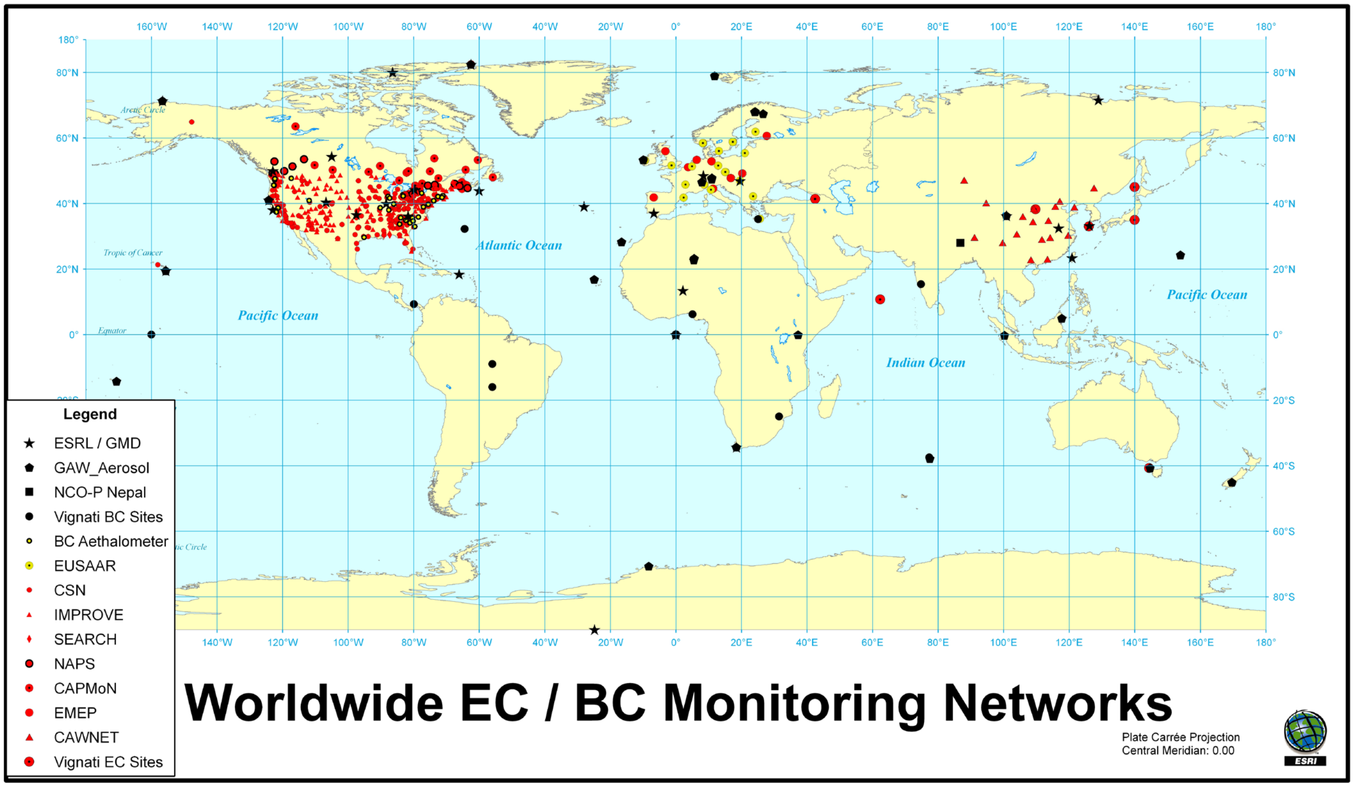

Figure 5-2. Ambient BC Measurement Locations Worldwide. Light absorption measurement locations are colored black. Thermal measurement locations are colored in red. A small subset of locations with both measurements is colored yellow. (Source: U.S. EPA)

CSN (*http://www.epa.gov/ttn/amtic/speciepg.html*), and SEARCH (*http://www.atmospheric-research.com)* networks.

5.3.2 Global Ambient Concentrations

Table 5-2 summarizes data from a number of studies and monitoring networks that help illustrate the range of BC concentrations across the globe. The table also indicates the BC measurement methods (thermal (T) and light absorption (LA)) for each study/monitoring network. While BC measurements for urban and rural areas are similar in North America and Europe, the reported concentrations for China are much higher. Both urban and rural BC concentrations in China are approximately 10 times higher than urban and rural concentrations in the United States, respectively.[7]

The United Kingdom shows higher BC concentrations at the upper range than the United States likely due to the influence of local sources on the individual monitoring sites. In general, roadside or near-source monitors yield higher values, as demonstrated by the curbside monitors in London which report considerably more BC than the urban-wide locations (Butterfield et al., 2010). The "Black Smoke" data for the UK that provide the basis for the five-decade trend discussed in section 4.6.2 are three to four times higher than co-located measurements of BC (Quincey, 2007).

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

5.3.3 Comparison of Urban and Rural Concentrations Globally

Available data suggest that BC concentrations vary substantially between urban and rural areas. Specifically, urban areas tend to have higher

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[7] As discussed in Chapter 4, the ratio between China and U.S. measured BC concentration is two to three times higher than the ratio of their estimated national BC emissions.

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

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

[a] CSN – Primarily urban network sites.

[b] IMPROVE – Rural network sites.

[c] BC data at State and Local Air Monitoring Stations from AQS, mostly with Magee Aethalometers.

[d] Personal communication with Tom Dann (Environment Canada).

[e] Monitoring was for the period 07/02 – 06/03 from Yttri et al. (2007).

[f] Data taken from *http://ebas.nilu.no/* or EUSAAR, the sites assigned to be urban are Ispra, IT (BC) and Melpitz, DE. Although not part of EUSAAR, the urban sites also include Ring A10, NL (EC). The northern EUSAAR remote location of Zeppelinfjell, NO, site is included with other Arctic sites listed separately.

[g] Urban network sites from Butterfield (2010); Curbside site at London Marylebone Road reported 10μg/m3.

[h] Data and urban/regional/remote classification was for the period 2006 from Zhang et al. (2008).

[i] Monitoring was for the period 03/06 – 02/08 from Marinoni et al. (2010).

[j] NOAA Global Monitoring Division Sites – For this table, we modified reported numbers in absorption units using a nominal mass extinction coefficient of 10m2g-1. One year from each site was eliminated as non-representative.

[k] Data taken from *http://www.airquality.co.uk/reports/cat05/1009031405_2009_BC_Annual_Report_Final.pdf*; curbside site at London Marylebone Road reported an average of ~40μg/m3 for each year.

concentrations. The global BC data (for 2005-2007 average or calendar year 2006) displayed in Figure 5-3 contrast the annual average rural and urban concentrations for North America, China, and Europe.[8, 9] The ambient rural concentrations provide an indicator of regional background concentrations resulting from regional emissions and transported aerosols. Levels in urban areas reflect the higher average concentrations resulting from the combination of local emissions and regional emissions. The portion of urban concentrations due to local emissions can also be described as the “urban increment” or “urban excess”.[10]

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[8] The data in Figure 5-3 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.

[9] The map in Figure 5-3 shows the 40th parallel, the importance of which is discussed further in Chapter 4.

[10] Because 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 (Abt Associates, 2005).

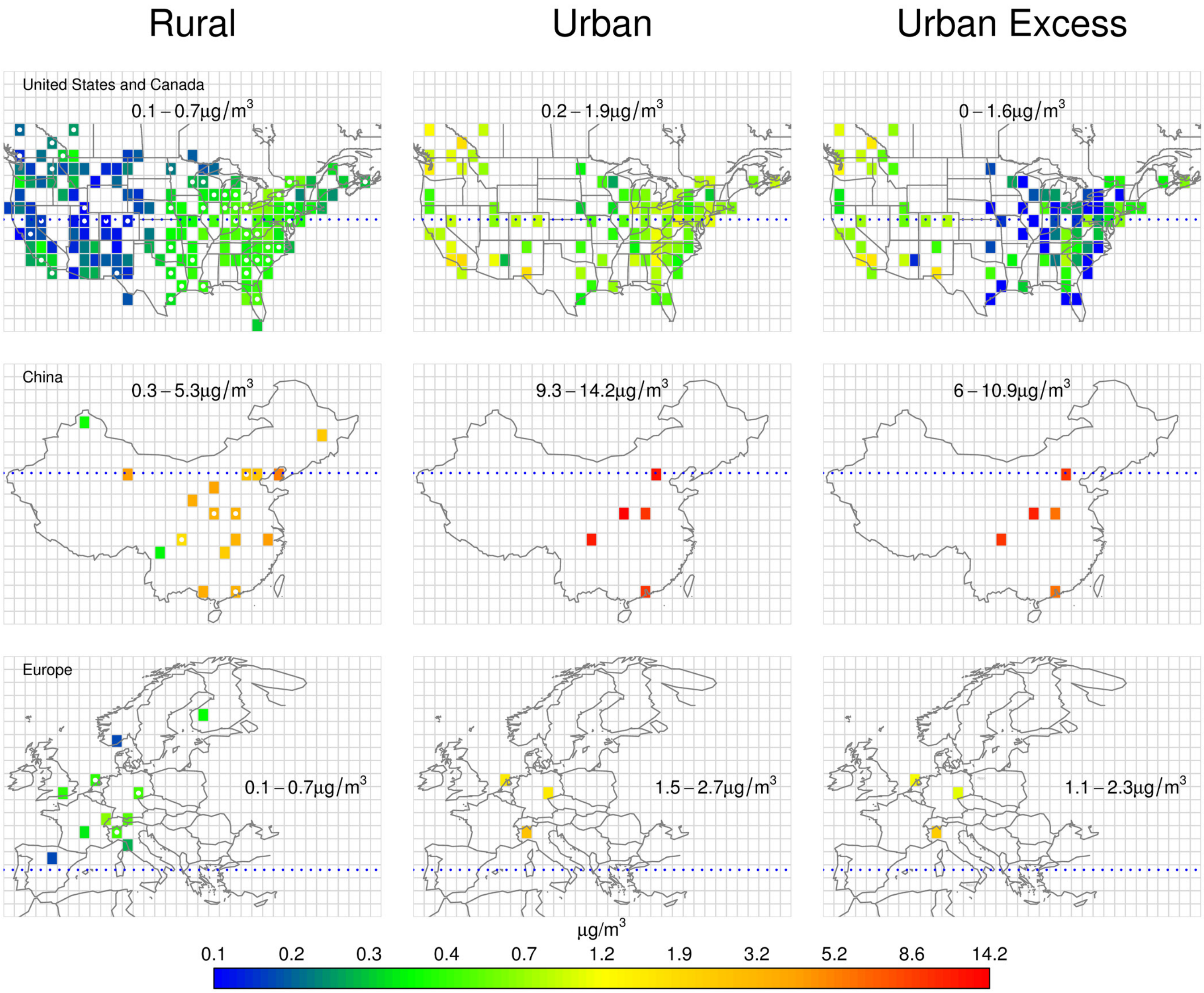

Figure 5-3. Spatial Distribution of Global BC Data. Rural, urban, and urban excess concentrations for 2005-2007. Grid squares with a white dot represent estimated rural concentrations from spatial interpolation of the nearest neighbors with measurement data. The 40th parallel is shown as a dotted line. (Source: U.S. EPA)

As demonstrated in Figure 5-3, urban BC measurements in North America are generally much higher than the nearby regional background levels. This suggests that there can be a substantial increment of local emissions in urban areas. For the period 2005 to 2007, the urban increment ranged from zero to 2.2 µg/m^3 (i.e., up to 92% of the total urban BC concentrations). In general, average urban concentrations are relatively consistent across North America, though the larger populated regions of the eastern United States, eastern Canada, and California contain most of the highest concentrations. However, the western United States and western Canada have lower regional background concentrations and therefore relatively larger urban increments, while higher rural concentrations in eastern North America result in smaller urban increments (more similar regional and urban average values). The higher regional background levels across eastern North America suggest higher and more consistent levels of BC emissions from sources across the region, and/or greater transport from clustered cities to surrounding rural areas.

Figure 5-3 also shows that Europe's measurement data are quite similar to those for North America. However, both China's regional and urban BC concentrations are much higher than those seen in

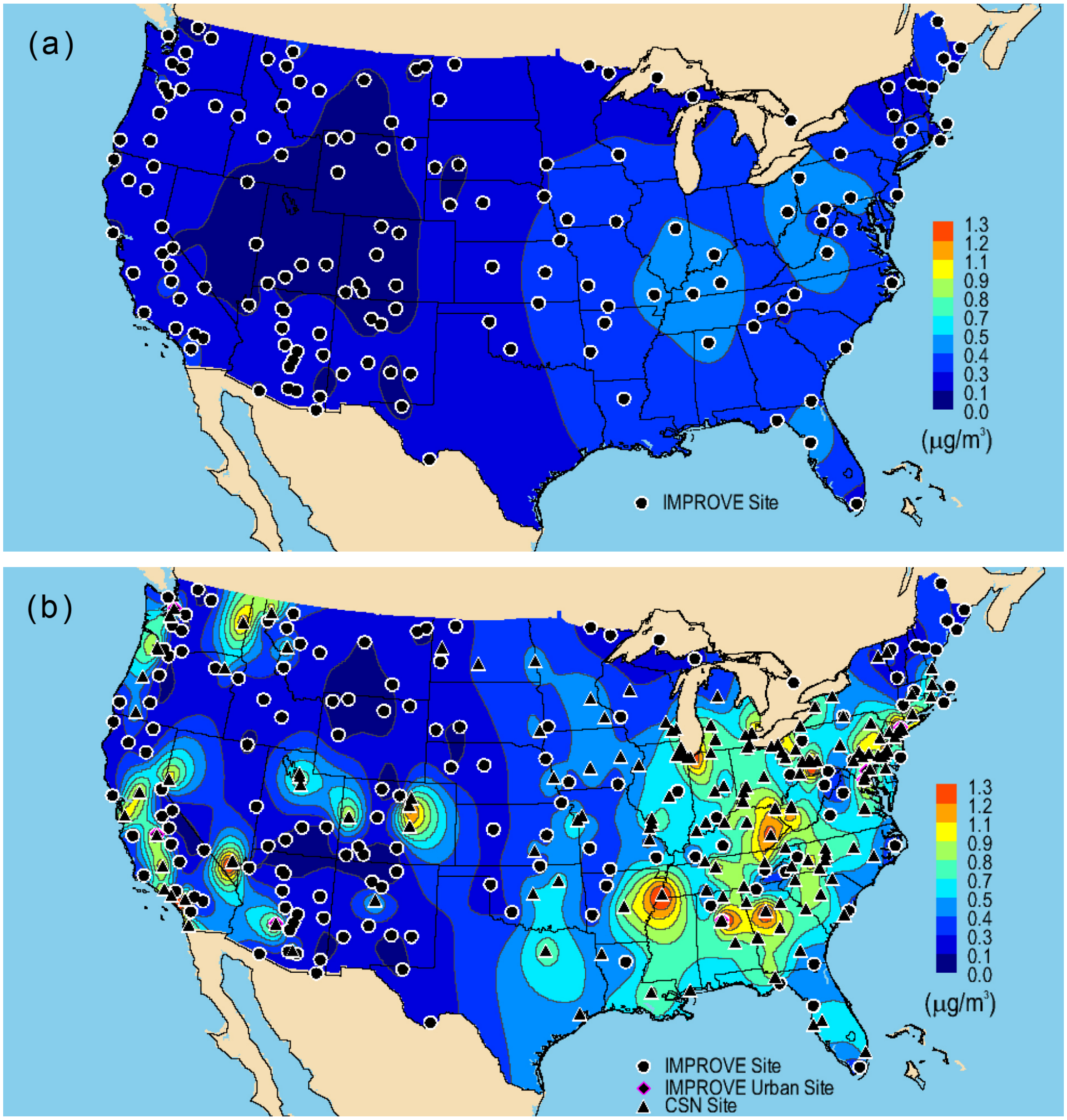

Figure 5-4. Annual Mean BC Concentrations (μg m^{-3}) for 2005–2008 in the United States. Panel (a) shows rural EC concentrations provided by IMPROVE network. Panel (b) shows the combined rural IMPROVE and urban CSN network EC data. IMPROVE site locations are shown as black circles, CSN sites are shown as black triangles, and urban IMPROVE sites are shown as magenta diamonds. (Adapted from Figures 7.5.1 and 7.5.2 of IMPROVE report V (2011) *http://vista.cira.colostate.edu/improve/*)

North America and Europe and its urban increments are approximately four times larger. This can be attributed in part to larger urban and regional emissions sources in China compared to North America and Europe.

To provide additional spatial detail to U.S. BC concentrations, Figure 5-4 shows the spatial distribution of background and combined urban plus background concentrations derived from EC measurements in the IMPROVE and CSN monitoring programs. While the background concentrations are relatively homogeneous over large sub-regions of the United States, Figure 5-4(a) indicates these background concentrations are generally higher in the Industrial Midwest. As shown in Figure 5-4(b), urban areas have much higher concentrations. High ratios of urban to rural BC concentrations

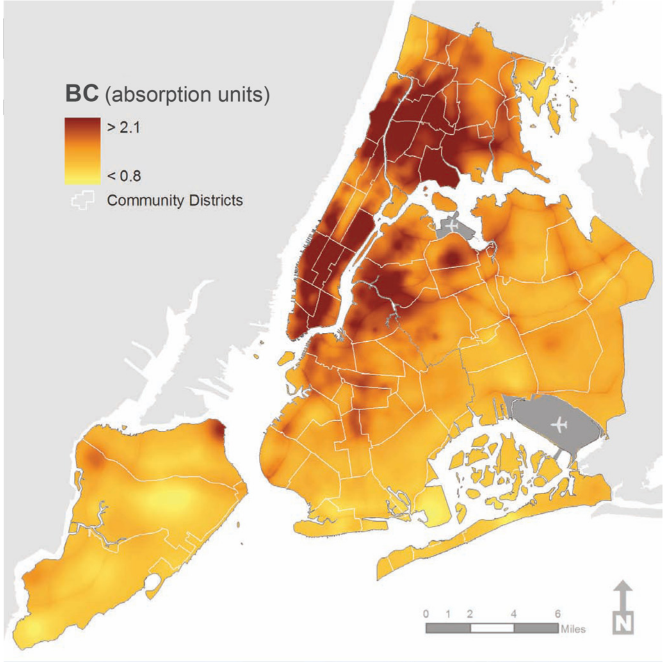

Figure 5-5. Urban BC Gradients for New York City. (Source: The New York City Community Air Survey, Results from Winter Monitoring 2008-2009, *http://www.nyc.gov/*)

demonstrate the localized impact of BC on surrounding rural regions and suggest urban BC emissions sources were significantly larger than rural sources. The mean urban to rural ratio was 3.3 ± 1.9 and was much larger than the mean ratio for sulfate, nitrate, or OM components of $PM_{2.5}$ (Hand et al., 2011).

In addition to the general contrast between measured urban and nearby rural BC concentrations, there can be substantial spatial variation in BC concentrations within a given city. Because global representations of BC concentrations are typically based on limited monitoring locations and are generally presented as average concentrations (often across monitors hundreds of miles apart), it is important to realize that ambient concentrations of BC in any urban area can vary widely from location to location within the city. BC concentrations can vary spatially within an urban area because the magnitude of monitored BC concentrations is dependent on the proximity of the monitor to roadways and other nearby sources. Therefore, concentrations measured may not be representative of other locations. Figure 5-5 illustrates the estimated spatial variability of BC in New York City.[11] This special study used 150 monitoring sites to reveal large gradients in apparent BC concentrations. Actual gradients may even be larger. Furthermore, the illustrative within-urban variability for NYC may also be representative of other urban areas with high population and emissions density. While most of the identified high concentration zones can be attributed to mobile source emissions density, this study also revealed significant BC emissions sources associated with residential oil combustion.

5.3.4 BC as a Percentage of Measured Ambient $PM_{2.5}$ Concentrations in the United States

Because total $PM_{2.5}$ mass is the basis for regulation of fine particles in the United States and also serves as the basis for BC emissions estimates, it is informative to estimate the contribution to total $PM_{2.5}$ mass from BC. However, given the limited BC data available on a global scale, this evaluation is based solely on data for urban areas in the United States that are regionally representative of large U.S. cities. Compared to U.S. rural locations, urban locations contain a higher percentage of BC and OC. While urban nitrate concentrations are also higher than surrounding rural areas, carbonaceous aerosols are responsible for most of the urban $PM_{2.5}$ increment. Other components, such as dust, are similar in both urban and rural environments (U.S. EPA, 2004b). Figure 5-6 shows the BC fraction of $PM_{2.5}$ mass for 15 selected U.S. urban areas. The values represent average concentrations among monitoring locations in the area. The average BC concentrations range from 0.6 $\mu g/m^3$ in St. Louis to 1.2 $\mu g/m^3$ in Atlanta. The percentage of $PM_{2.5}$ that is BC ranges from 4% in St. Louis to 11% in Seattle.[12] A more complete characterization of urban and rural $PM_{2.5}$ speciation components on an annual and seasonal basis can be found elsewhere (Hand et al., 2011; U.S. EPA, 2009b).

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

[12] Approximately 20-80% of the estimated ambient organic matter (OM) is directly emitted (Carlton et al., 2009). The other portion, termed secondary organic aerosol (SOA), is formed through chemical reactions of precursor emissions after being released from the sources (Saylor et al., 2006; Carlton et al., 2009; Chu, 2005). 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 (Bae et al., 2006; Turpin and Lim, 2001). 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 BrC, 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.

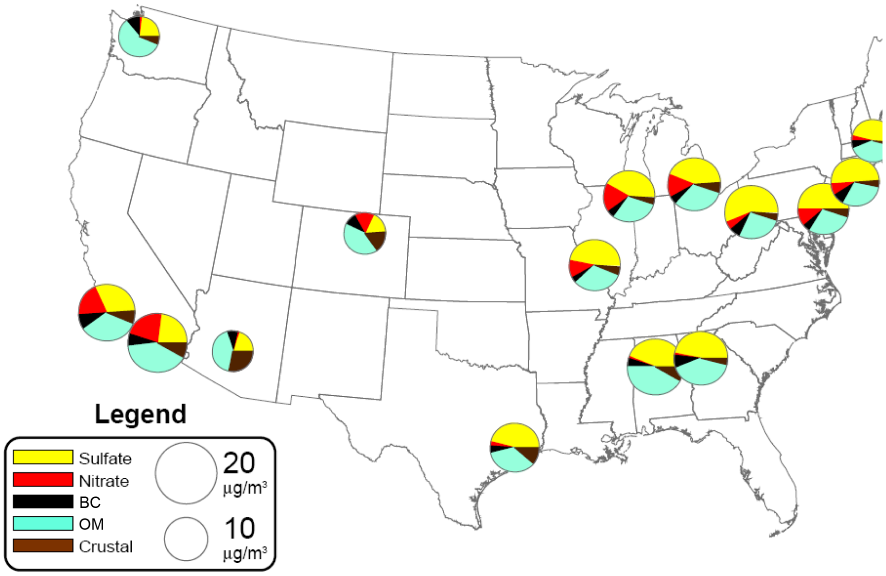

Figure 5-6. Composition of $PM_{2.5}$ for 15 Selected Urban Areas in the United States. Annual average $PM_{2.5}$ concentrations (µg/m^3) are presented where the circle size represents the magnitude of $PM_{2.5}$ mass. The BC and Organic Mass (OM) fractions are illustrated. OM represents OC together with its associated non-carbonaceous mass (e.g., hydrogen, oxygen and nitrogen), estimated by a material balance approach. Sulfates and Nitrates have been adjusted to represent their mass in measured $PM_{2.5}$. (Source: U.S. EPA, 2009b)

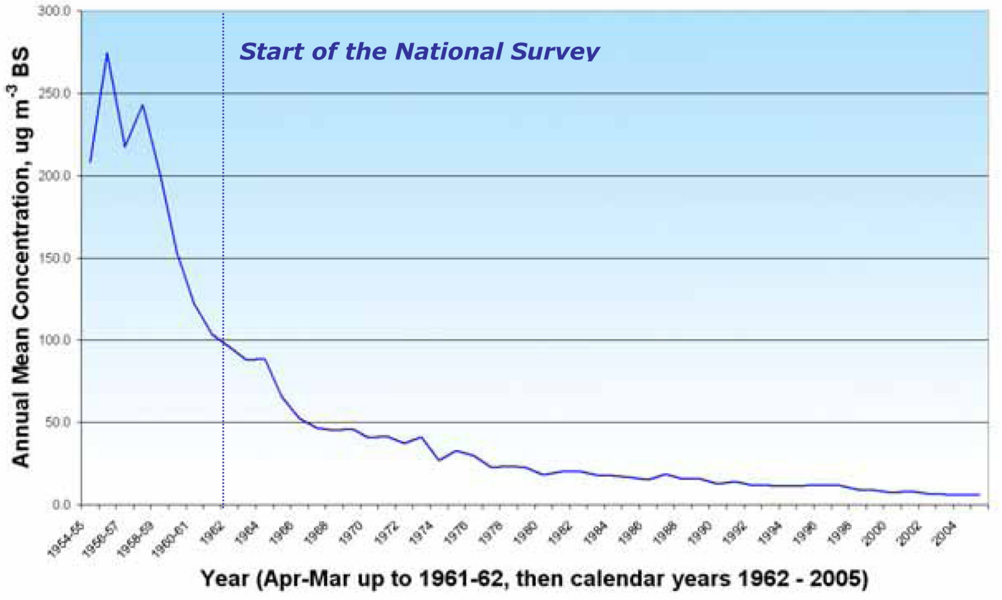

Figure 5-7a. Trends in Black Smoke Measurements (µg/m^3) in the United Kingdom, 1954-2005. The BS measurements are highly correlated with optical BC, although BS is 3 to 4 times higher than BC under current U.K. aerosol conditions. In 1961, the UK established a national air pollution monitoring network, called the National Survey, monitoring black smoke and sulphur dioxide at around 1,200 sites in the UK. (Bower et al., 2009)

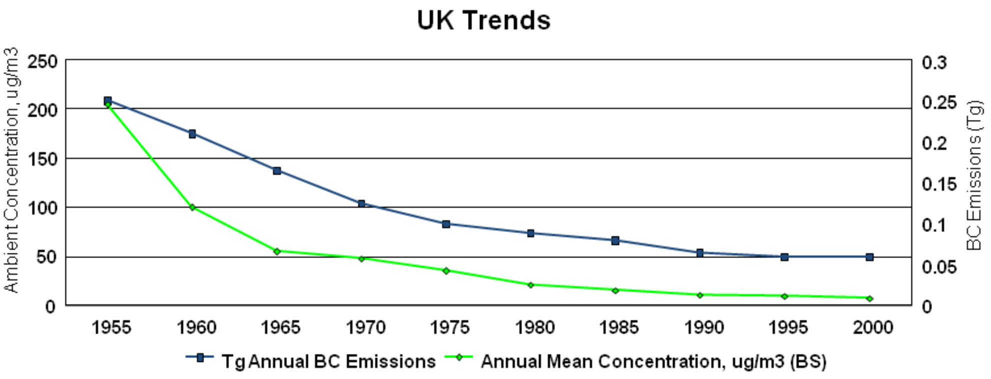

Figure 5-7b. Comparison of Ambient Black Smoke Measurements (µg/m^3, annual average) with Estimated BC emissions (Tg) in the United Kingdom, 1955-2000. (Source: U.S. EPA)

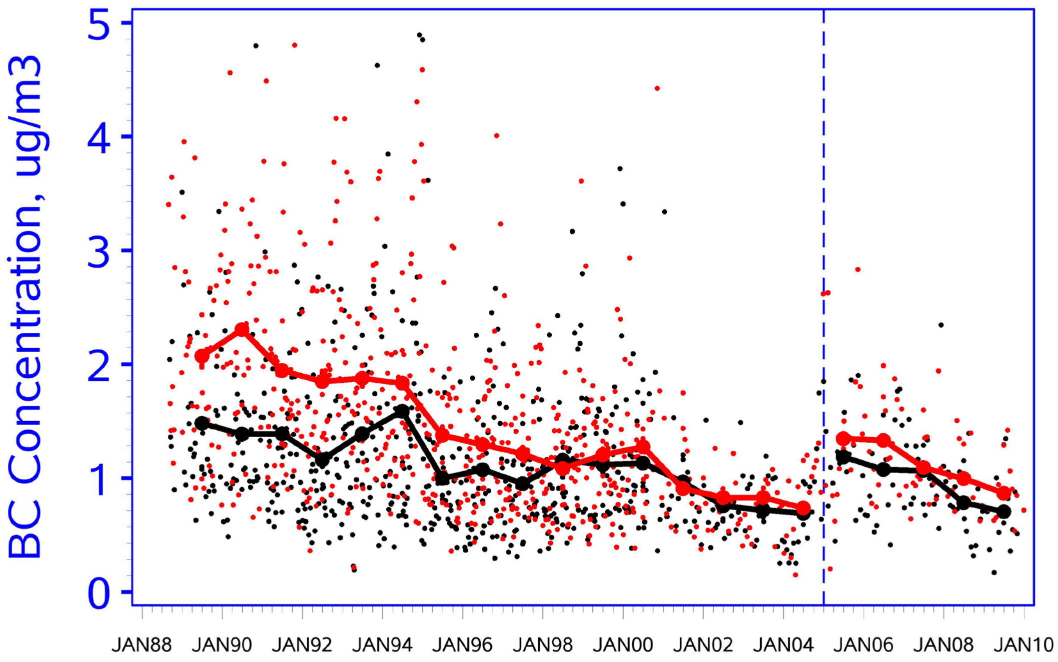

Figure 5-8. Ambient BC Trends in Washington, D.C. The red data points and line represent daily and annual average concentrations from Wednesdays (as a proxy for weekdays) and the black data points and line represent daily and annual average concentrations from Saturdays (as a proxy for weekends). The large dots represent the annual average concentration which is plotted at mid-year. This monitoring site changed its sampling protocol from twice per week (Wednesday and Saturday) to once every 3 days in August 2000. The apparent increase in BC concentration after January 2005 coincides with a carbon analyzer upgrade resulting in total carbon measurements with a higher proportion of BC. (Source: U.S. EPA, produced using data from VIEWS *http://vista.cira.colostate.edu/views*)

Note: The potential increase in reported EC measurements is described by White (2007), *http://vista.cira.colostate.edu/improve/Data/QA_QC/Advisory/da0016/da0016_TOR2005.pdf*.

5.4 Trends in Ambient BC Concentrations

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

Measurement data necessary for assessing long-term ambient trends in BC are limited even for the areas with currently robust monitoring programs such as the United States.[13] However, although limited, some information on changes in ambient concentrations over longer time periods is available and these data are useful in evaluating and corroborating emissions trends. Since most BC is directly emitted rather than the formed chemically from precursors in the atmosphere, ambient BC concentrations respond directly to emissions changes. Figure 5-7a shows the dramatic reduction in measured "Black Smoke" (BS) in the UK since the 1950s. This dramatic decline is attributable to a number of factors, including the introduction of cleaner fuels and technologies, and successful smoke control legislation (Bower et al., 2009). Figure 5-7b overlays these BS measurements and estimated BC emissions for the UK for the same time period, revealing large estimated emissions reductions corresponding to 80% of the reduction in black smoke.

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

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[13] 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.

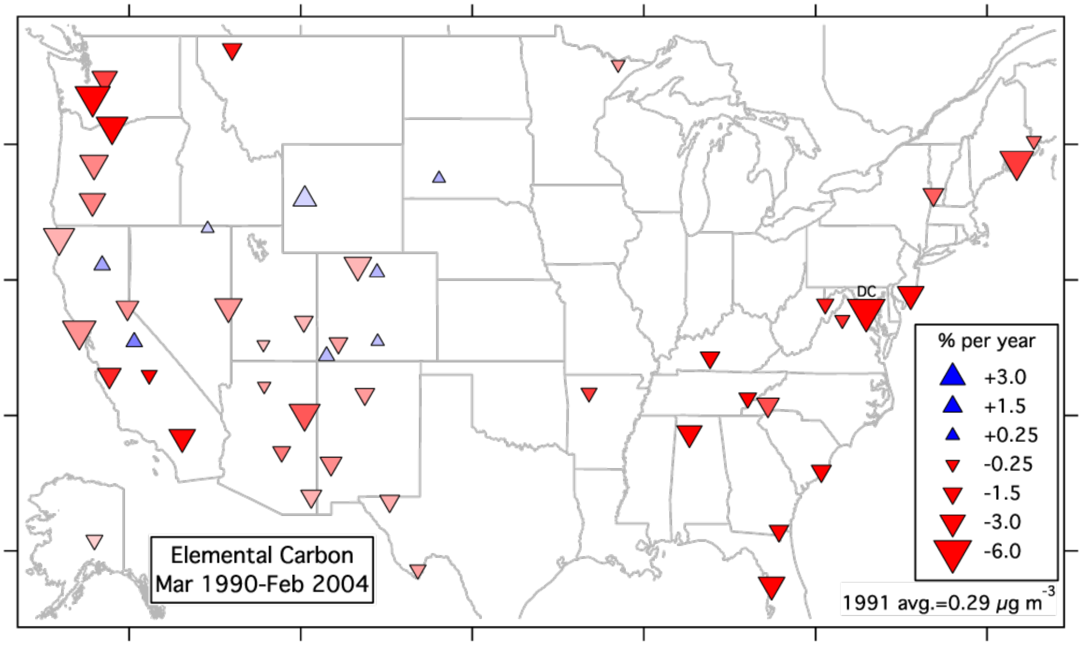

Figure 5-9. Trends in BC at All IMPROVE Network Stations with Sufficient Data between 1 March 1990 and 29 February 2004. Marker size indicates the magnitude of the trend. Triangle direction and blue or red color corresponds to the sign of the trend. Color saturation is proportional to the average concentration in 1991 with full saturation at twice the national median. The only urban site in Washington, D.C. is marked. Averages in the bottom right corner exclude Washington, D.C. (Adapted from Murphy et al., 2011)

concentrations domestically. Figure 5-8 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 are presented as separate time series due to a change in the frequency of sampling and with a separation at the end of 2004 due of an upgrade to newer analytical equipment. The trends show a substantial two decade decline in ambient BC concentrations. The percent change in Washington, D.C.'s BC on Wednesdays and Saturdays was 62% and 49% respectively, based on a comparison of average levels in 1989-1991 compared to 2002-2004, and 54% and 40% comparing 1989-1991 with 2007-2009. 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 5-9). BC concentrations in the rural 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). The large 22-year urban BC decline illustrated for Washington, D.C. may in fact be larger than the overall estimated nationwide reductions in BC and direct $PM_{2.5}$ emissions in the United States described in Chapter 4, and is the result of area-specific emissions dominated by certain emissions sectors.[14]

Figure 5-10 juxtaposes estimated annual average BC concentrations in the San Francisco Bay with annual consumption of diesel fuel in California (Kirchstetter et al., 2008).[15] Kirchstetter notes that the contrast in the trends in BC concentration and diesel fuel use is striking, especially beginning in the early 1990s when BC concentrations began markedly decreasing despite sharply rising diesel fuel consumption. This contrast suggests that control technologies to reduce BC emissions have been successful (see

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[14] As stated in Chapter 4, national BC emissions decreased by 79%, 30%, and 25% for on-road gasoline, on-road diesel, and nonroad diesel sources, respectively, from 1990 to 2005. Also, Chapter 8 (Table 8-1) shows that 45% of the on-road BC reductions are due to gasoline vehicles. Thus a combination of on-road gasoline and diesel are each potential contributors to the lower BC in our nation's capital and other urban areas.

[15] BC was estimated using Coefficient of Haze (COH) measurements, which are shown to be highly correlated with optical BC. See Appendix 1 for further details regarding COHs.

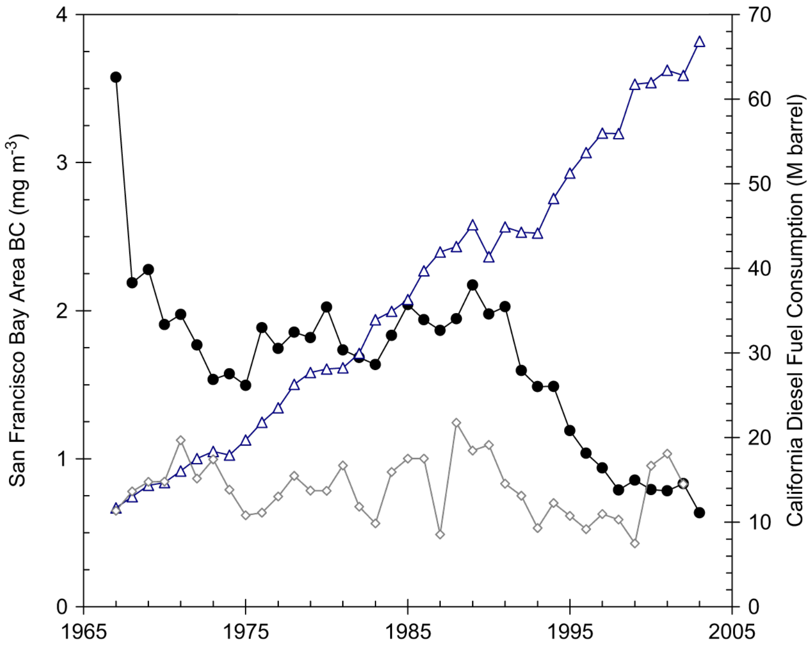

Figure 5-10. Estimated Annual Average Ambient BC Concentrations in the San Francisco Bay Area vs. Diesel Fuel Consumption. BC is shown as black dots. California on-road (blue) and nonroad (gray) diesel fuel consumption are shown as triangles and diamonds. (Kirchstetter et al., 2008)

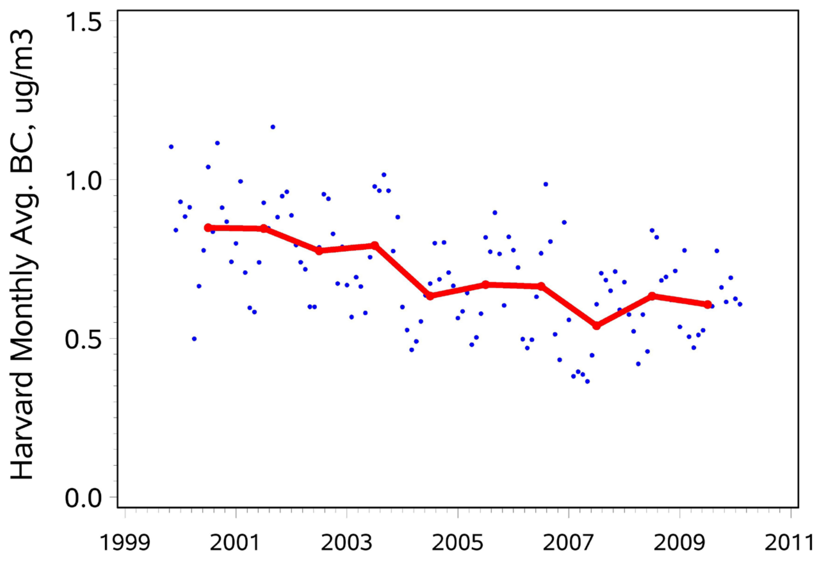

Figure 5-11. Ambient BC Trends in Boston (Harvard School of Public Health location). The blue squares are monthly averages and red line is the annual average of BC concentrations as directly reported by an Aethalometer. (These data were not adjusted using correction algorithms described in Appendix 1.) (Source: U.S. EPA, data courtesy of Harvard School of Public Health)

Chapter 8). Similarly, Figure 5-11 shows a data set from Boston, MA which displays a decline in BC concentrations during the period 2000-2009. These data are presented as monthly average concentrations to help reveal the underlying trend, but much higher temporal variability exists in the 5-minute or even hourly concentrations which respond to patterns in nearby mobile emissions and can even identify when a diesel truck passes by. The decline of BC concentration at this site has been attributed to diesel retrofits in Boston, but is no doubt also reflective of fleet wide changes in emissions especially due to diesel emissions standards (U.S. EPA, 2004c).

Figure 5-12 shows that BC concentrations have declined 32%, on average, for a 15-site subset of EPA's national urban CSN monitoring sites with the longest historical record. The figure shows the range of monthly average BC concentrations represented by CSN EC measurement data. Because EPA transitioned its urban EC monitoring to the IMPROVE protocol, CSN EC measurements have been produced by two different monitoring protocols. As discussed in section 5.2, EC may vary by a factor of two and in fact, that may also be true for these two data records. Therefore, the concentration data for these sites (which were all part of the first group which switched from the older CSN NIOSH-type monitoring protocol) are shown as reported after May 2007 while the earlier data are adjusted to be IMPROVE-like.[16] The details about this adjustment are described in Appendix 1. In general, the CSN sites and trend are representative of neighborhood, urban-wide, and regional-scale emissions influences and may not necessarily reflect local scale emissions changes. Nevertheless, the range of monthly average concentrations among the group of 15 urban locations illustrates

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[16] As discussed in Appendix 1, the adjustments are based on parallel measurements at 14 urban IMPROVE and 168 urban CSN locations between 2005 and 2011; the adjustment and trend prior to 2005, therefore, is more uncertain.

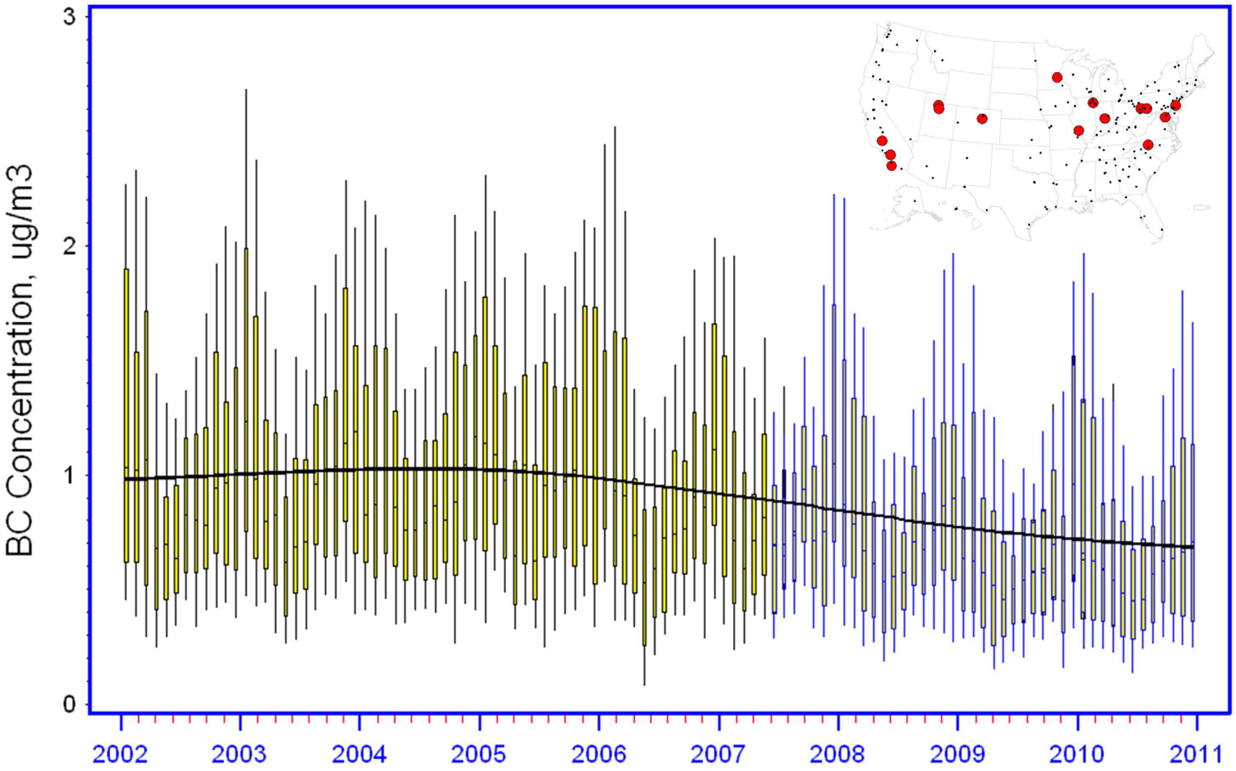

Figure 5-12. Ambient BC Trends (2002-2010), based on monthly distribution of average concentration among 15 CSN monitoring locations in the United States. The map shows the location of the 15 monitoring sites. Starting May 2007, the measured EC were produced with the IMPROVE monitoring protocol. The data prior to May 2007 were produced with the EPA NIOSH-type protocol and adjusted to represent the IMPROVE protocol. For a more detailed explanation of this graphic, see Appendix A, Figures A1-6 and A1-7 and surrounding text. (Source: U.S. EPA)

seasonal variability in BC. However, the higher winter BC concentrations, which may be due to local emissions patterns or climatology, were not evident at all sites and geographic regions.

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 substantially declined from the mid 1980s. The urban and regional BC declines can generally be attributed to emissions reductions associated with engine and fuel improvements (Bahadur et al., 2011; Kirchstetter et al., 2008; Minoura et al., 2006; Murphy et al., 2011) and reductions in residential biomass burning emissions (Burnet et al., 1988; Butler, 1988; Hough, 1988) to attain NAAQS (Bachmann, 2007) for PM and other pollutants.

5.4.2 Trends in Ambient BC Concentrations in the Arctic

Trends in BC concentrations at three Arctic locations (Alert Canada, Barrow Alaska, and Zeppelin Norway) are presented in Figure 5-13 (Hirdman et al., 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 emissions 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

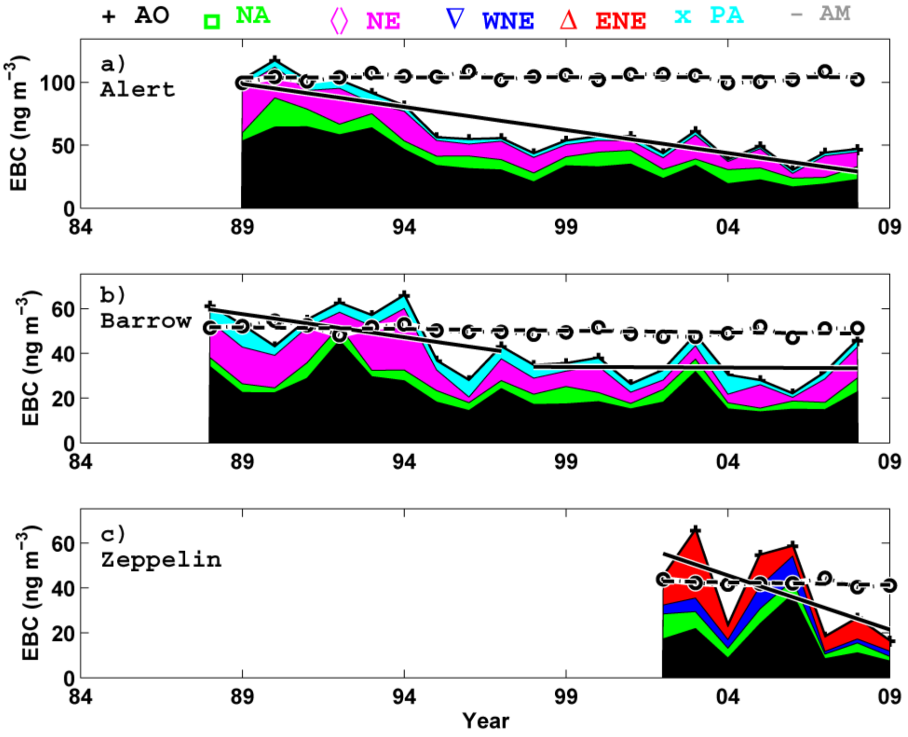

Figure 5-13. The Annual Mean BC Concentrations Measured at Alert (a), Barrow (b), and Zeppelin (c) and Split into Contributions from the Four Transport Clusters. The annual mean concentrations measured at Alert (a), Barrow (b), and Zeppelin (c) are split into contributions from four transport clusters. The solid line shows the linear trend through the measured concentrations. The circles show the annual mean BC concentrations when the cluster-mean concentrations are held constant over time (means over the first three years). This line is influenced only by changes in the frequencies of the four clusters. The dashed line shows the linear trend of these data. (Hirdman et al., 2010)

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.

5.5 Remote Sensing Observations

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

Unlike spatially discrete ambient BC monitors, remote sensing observations are global and thereby offer greater spatial surface coverage of BC levels and provide important estimates of BC where surface ambient measurements are not available. While remote sensing does not necessarily characterize surface concentrations, it provides important information on spatial variability of concentrations in BC and aerosols throughout the total atmospheric column. Combining these new data sources with traditional ground based (ambient) measurements has been used to derive the complete aerosol effect on the environment and climate (Falke et al., 2001; Husar, 2011). Integrated data sets of aerosol based extinction have relied heavily on AERONET sun photometer measurements in remote locations with low concentration and relatively homogeneous aerosol (Kaufman et al., 2001), while downwind of pollution or dust sources they have relied on MODIS characterization of the aerosol spatial distribution over the ocean and dark surfaces (Remer et al., 2002) and on TOMS over bright surfaces (Torres et al., 2002).

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

Koch compares estimated AAOD for 1996–2006 based on AERONET with OMI satellite retrievals for 2005–2007 (Koch et al., 2009; Torres et al., 2007). Koch notes that the two data sets broadly agree with one another. However, the OMI estimate is larger than the AERONET value for South America (with UV sensitive biomass combustion) and

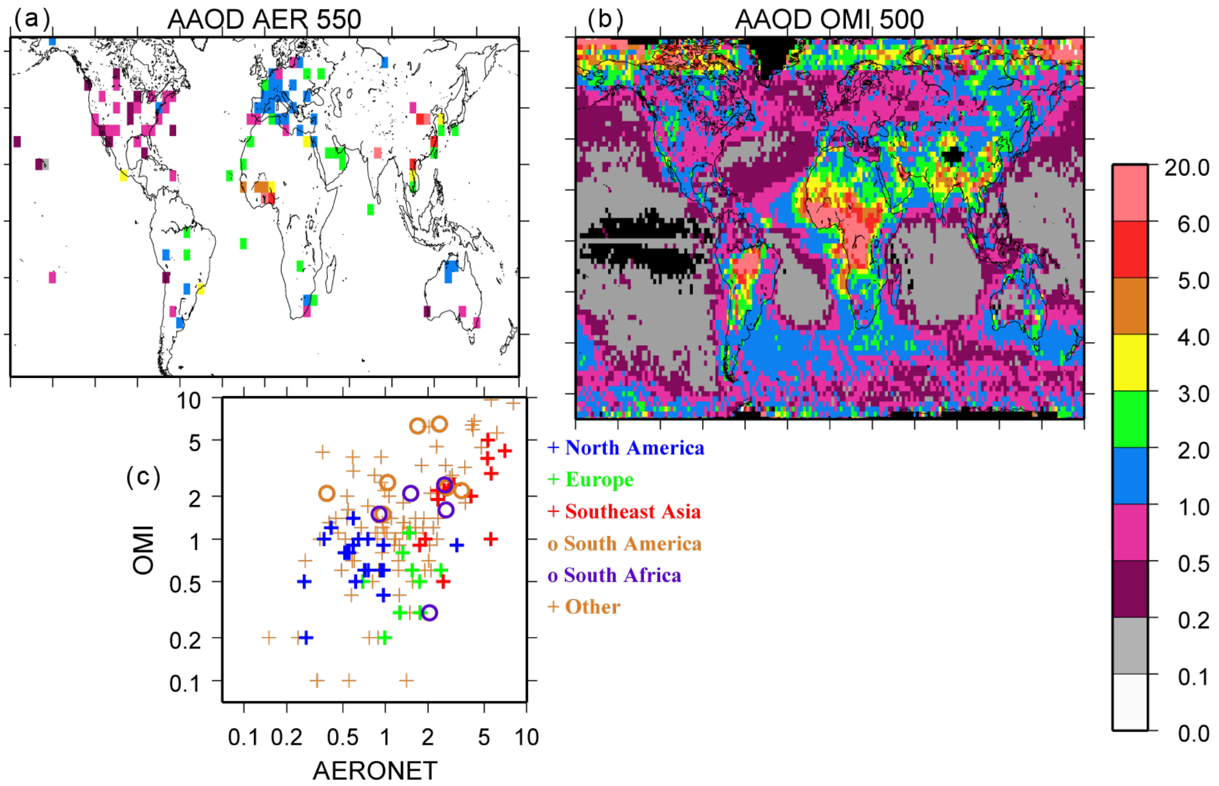

Figure 5-14. Aerosol Absorption Optical Depth (AAOD) from AERONET (1996-2006) and OMI (2005-2007). (a) Aerosol absorption optical depth, AAOD, (x100) from AERONET (at 550 nm), (b) OMI (at 500 nm) and (c) scatter plot comparing OMI and AERONET at AERONET sites. (Koch et al., 2009)

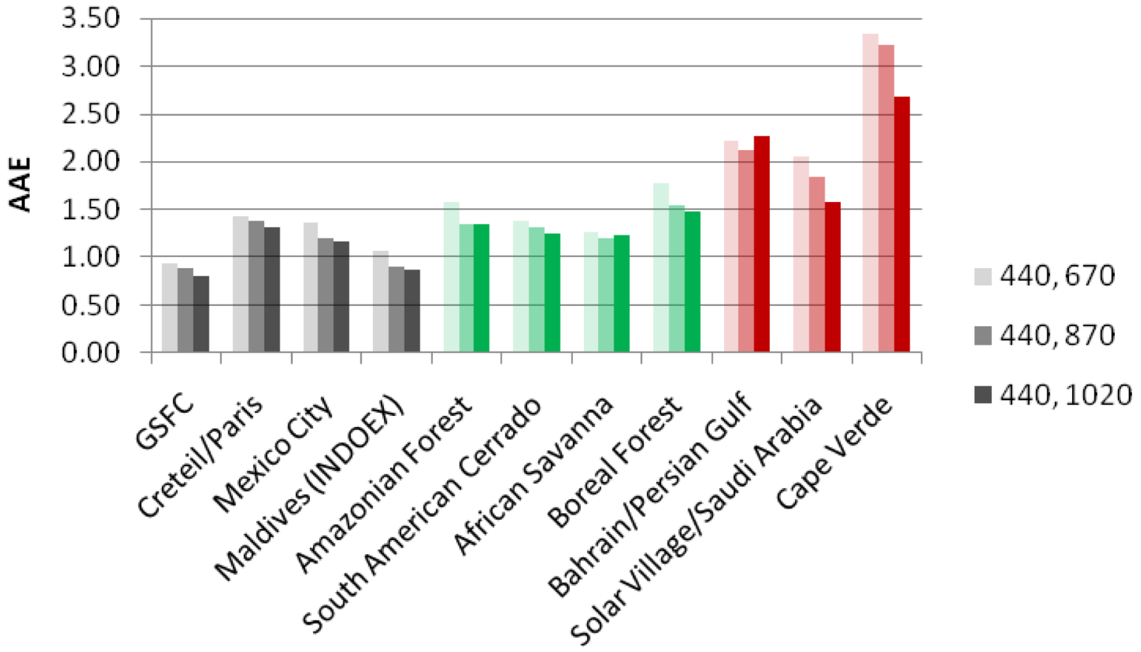

Figure 5-15. Absorption Angstrom Exponent (AAE) Values for AAOD Spectra Derived from AERONET Data. Black: Urban/Industrial or Mixed; Green: Biomass Burning; Red: Desert Dust. Shading for each location indicates wavelength pair (in nm) for AAE calculation. GSFC=Goddard Space Flight Center, Greenbelt, MD. (Russell et al., 2010)

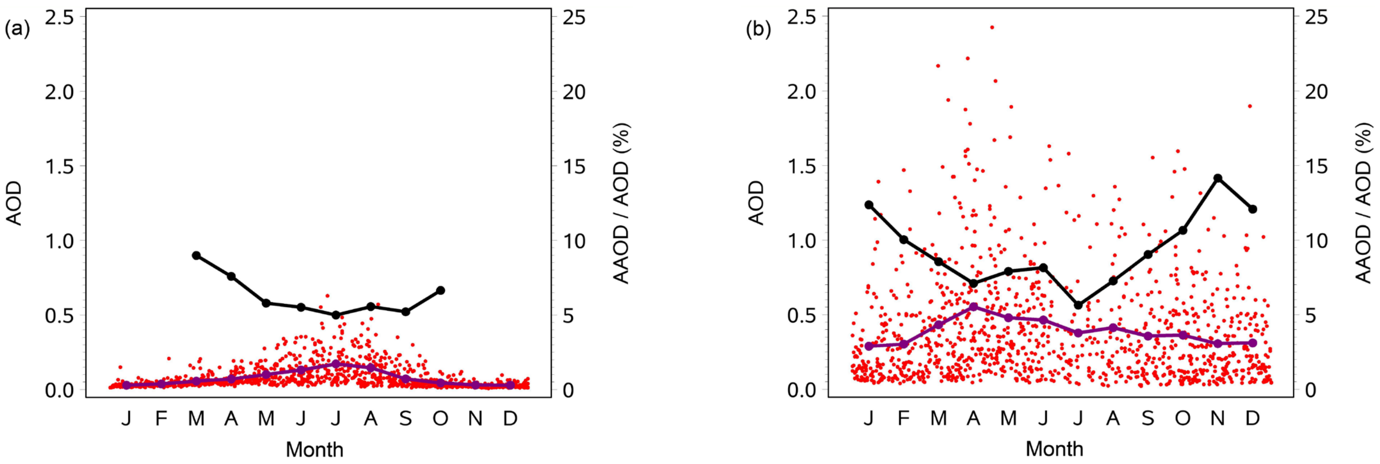

Figure 5-16. AERONET AOD and AAOD as a Percentage of AOD. (a) Goddard Space Flight Center (GSFC) in eastern United States and (b) Beijing, China from May 2005 to April 2009. Red dots represent AOD (shown on left vertical axis). Both AOD and AAOD are at 870nm. Horizontal axis shows the month of the year. The purple and black lines are the monthly average AOD and percent absorbing aerosol. Note that at GSFC, AOD is highest in the summer (associated with secondarily formed aerosols and winter-time AAOD observations are not available for GFSC because the calculation is not possible at low AOD values. (Source: AERONET data are based on Version 2, Level 2.0 inversion products with permission of Brent Holben, NASA and Hong-Bin Chen Chinese Academy of Sciences)

smaller for Europe and Southeast Asia, which are dominated by BC. The AERONET AAOD and OMI observations qualitatively agree with the ground level concentrations of BC for the United States, Europe, and Asia presented in Figure 5-14, and clearly increase the spatial characterization of aerosol absorption. As discussed below, aerosol absorption may not necessarily be associated with anthropogenic source emissions.

Multi-wavelength instruments, such as AERONET, can also characterize the wavelength dependence of absorption (often expressed as Absorption Angstrom Exponent, or AAE) to provide an indicator of the absorbing aerosol mixture. Using pairs of wavelength specific absorption measurements, Russell et al. (2010) find AAE values near 1 (the theoretical value for BC) for AERONET-measured aerosol columns dominated by urban-industrial aerosol, larger AAE values for biomass burning aerosols, and the largest AAE values for Sahara dust aerosols. Using these observations from multi-wavelength sensors can help distinguish the types of absorbing aerosols (Figure 5-14). It also demonstrates that the global AAOD observations presented in Figure 5-14 do not exclusively represent BC from anthropogenic sources.[17]

While a common limitation of remote sensing (which depends on solar light) is its general representativeness of daytime 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 5-15 illustrates the issue of incomplete data records and the contrast of AAOD levels across the globe.

Figure 5-16 presents AOD and AAOD as a percentage of AOD by date for two example locations from AERONET (Goddard Space Flight Center, GSFC in MD, and Beijing, China). The fraction of AOD that is estimated to be absorbing is lower at GSFC, but those data are not available during the winter months at GSFC due to insufficient AOD to calculate the absorbing portion. This is typical of the eastern United States and other U.S. locations where sufficient AOD only exists in the summer and which principally results from secondarily formed mostly scattering aerosols. Average percent absorbing aerosol for GSFC derived from AERONET is about 6-7%, but this does not represent winter-time conditions when BC may have its highest values.

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[17] 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), AOD, AAOD, and AAE, as well as shape and other aerosol properties (Mishchenko et al., 2007; Russell et al., 2010).

In contrast, AOD is sufficiently high year-round in Beijing and is several times the amount detected at GSFC; the Beijing absorbing portion (black line) ranges from about 8-9% in the summer to 15% in the winter. For the 7 months of the year where both locations provide data, the ratio of monthly average AAOD values between Beijing and GSFC ranges from 2 to 7. However, without year-round AAOD data for GSFC, the estimate of its annual average absorbing aerosol and its annual average ratio to Beijing is very uncertain. Thus, the lack of wintertime absorption measurements limits the value of remote sensing estimates to expand the spatial extent of ground level measurements for model evaluation and corroboration of emissions inventories.

5.6 BC Observations from Surface Snow, Ice Cores, and Sediments

Snow and ice cover approximately 7.5-15% of the Earth's surface, depending upon the time of year (Kukla and Kukla, 1974). The sunlight that reaches the snow surface typically penetrates about 10-20 cm into the snow, with the topmost 5 cm receiving the most sunlight and where light-absorbing impurities can significantly alter the amount of solar energy reflected by the snowpack (e.g., Galbavy et al., 2007). Black carbon measurements in snow, and related surface reflectivity measurements, are critical to accurately estimate climate forcing due to snow-bound BC. In addition, ice core measurements of BC provide an important record of natural and anthropogenic BC emissions transported to snow-covered regions. Lake and marine sediments also pose an opportunity to derive historical trends in BC emissions prior to the point of time when air monitoring data are available.

5.6.1 Measurement Approach

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

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

5.6.2 Surface Snow Data

Measurements of BC in the shallow surface layer of snow have been conducted since the 1980s by research teams at locations throughout the Northern Hemisphere and in Antarctica, although the measurements were sporadic (Figure 5-17). Two large field studies, Clarke and Noone (1985) and Doherty et al. (2010), significantly boosted the number of sampling locations during two windows of time (1983-1984, 2006-2009). However, even the highest number of measurement locations (55 sites in 2009) provides sparse geographic coverage of data, considering the high degree of spatial variability in BC concentrations. Recent model estimates by Flanner et al. (2007), seek to fill in the missing measurement gap with predictions of surface snow BC concentrations in the northern hemisphere, estimating values ranging five orders of magnitude (<1 to >1000 ng BC/g snow).

Recent surface snow results from Doherty et al., (2010) show that BC concentrations range over an order of magnitude in remote areas of the Northern Hemisphere (Figure 5-18). Even higher BC values in snow were reported for the Tibetan Plateau and throughout western China, up nearly another order of magnitude (Ming et al., 2009; Xu et al., 2006). BC removal from the atmosphere is primarily driven by precipitation (Ogren et al., 1984), thus BC concentrations in snow or ice are a function of the atmospheric concentration of BC above the surface and the frequency and amount of snowfall in a particular area. For example, Xu et al. (2009b) noted that BC concentrations on the Tibetan plateau were high during nonmonsoon periods with low precipitation, which they related to regional

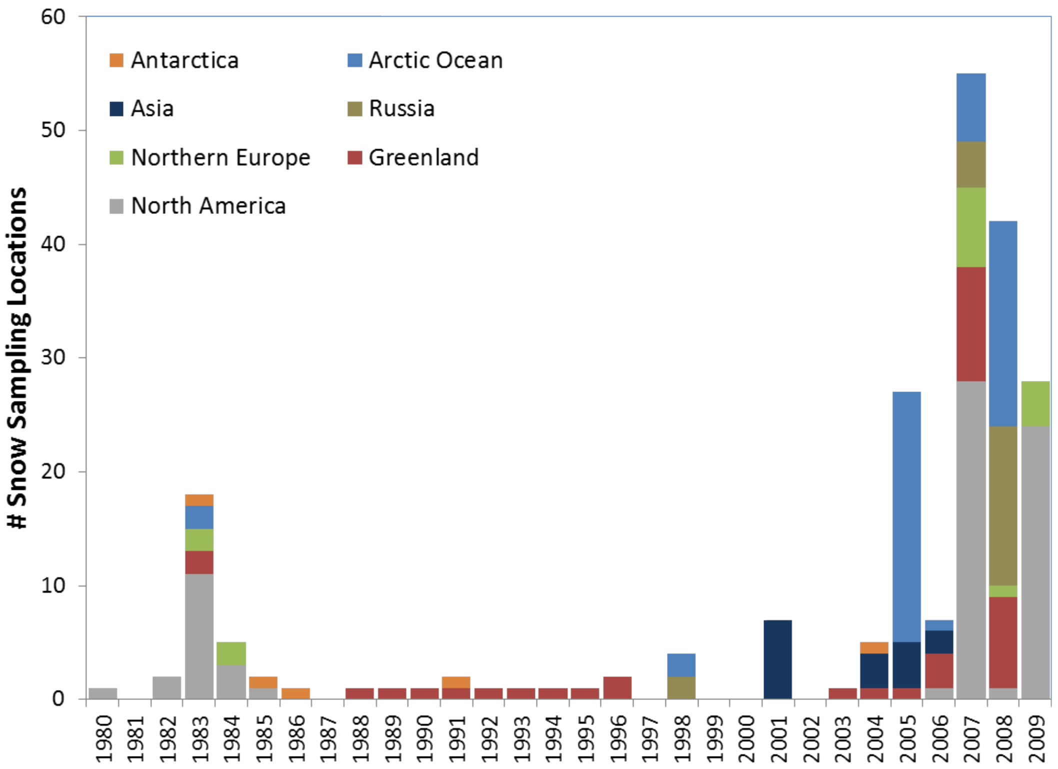

Figure 5-17. Locations of BC Measurements in Surface Snow and Shallow Snow Pits (snow pits are indicated for each year covered in the pit depth). (Sources: U.S. EPA, based on data reported in Cachier and Pertuisot (1994), Cachier (1997), Chylek et al. (1999; 1987), Clarke and Noone (1985), Doherty et al. (2010), Grenfell et al. (2002; 1981; 1994), Hagler et al. (2007a; 2007b), Hegg et al. (2009; 2010), Masclet (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))

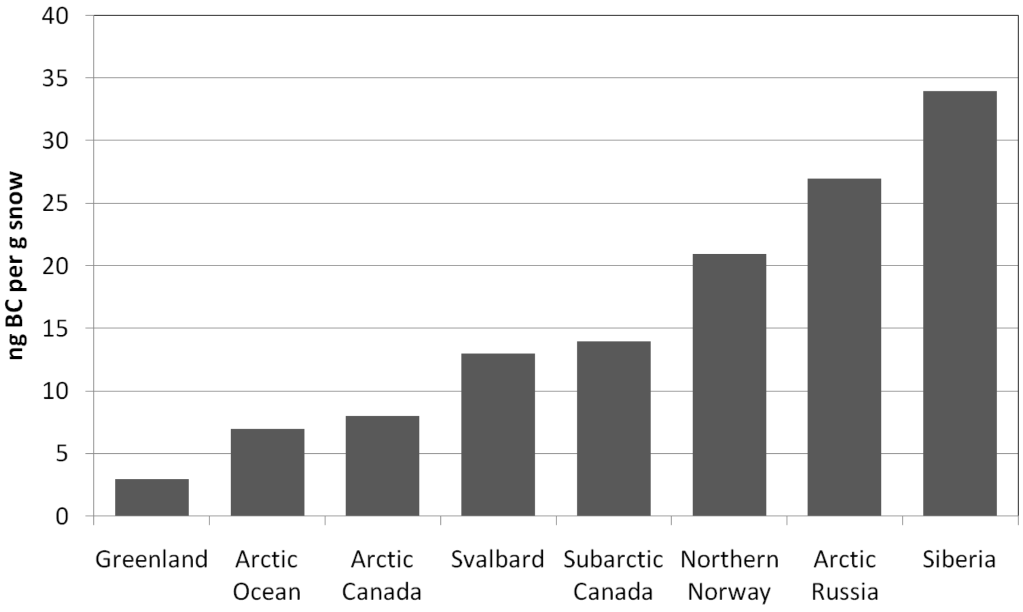

Figure 5-18. BC Concentrations in Surface Snow in Arctic and Subarctic Areas of the Northern Hemisphere. (Source: Derived from recent measurements reported in Doherty et al., 2010)

particulate pollution ("Asian Brown Cloud") elevating during the dry nonmonsoon period and then highly concentrating the infrequent precipitation with impurities. An additional important factor, discussed by several studies (Doherty et al., 2010; Flanner et al., 2007; Grenfell et al., 2002; Xu et al., 2006) is the potential increase in surface snow BC levels when melting snow leaves behind BC particles, further darkening the topmost layer of snow.

It is important to note that certain non-BC particulate species have been shown to absorb light when deposited to snow or ice. While dust is not as strong of a light absorber per unit mass as BC, dust can play a significant role in reducing snowpack reflectivity at high concentrations (Warren and Wiscombe, 1980). In addition, BrC in snow has been suggested to significantly absorb light (Doherty et al., 2010). Given that studies suggest that organic material in snow may undergo chemical transformation and loss from the snowpack due to sunlight-driven reactions (Hagler et al., 2007a; Grannas et al., 2004), BrC may absorb light to an even greater degree in fresh precipitation than what has been measured in aged snow samples. However, neither Grannas et al. nor Hagler et al. specifically measured BrC or the time evolution of light absorption.

5.6.3 Ice Core Data

Measurements of BC in ice cores are critical to understanding the longer-term trends of human influence on snow reflectivity. Ice cores, produced by drilling into permanent ice and carefully extracting a column of ice, have been collected and analyzed for BC at a number of locations in the Northern Hemisphere (Figure 5-19). In addition, an Antarctic BC ice core record spanning the past two and a half thousand years has just been completed as part the National Science Foundation WAIS Divide deep ice core project (Ross Edwards, personal communication). The ice cores with continuous BC data available primarily cover the past few hundred years, with the exception of the Dye 3 ice core in Greenland and the WAIS Divide core in Antarctica which extend back several thousand years. The layers of the ice core are dated using several strategies, including measuring certain chemical species with known seasonal variation, looking for certain known historical events that had unique

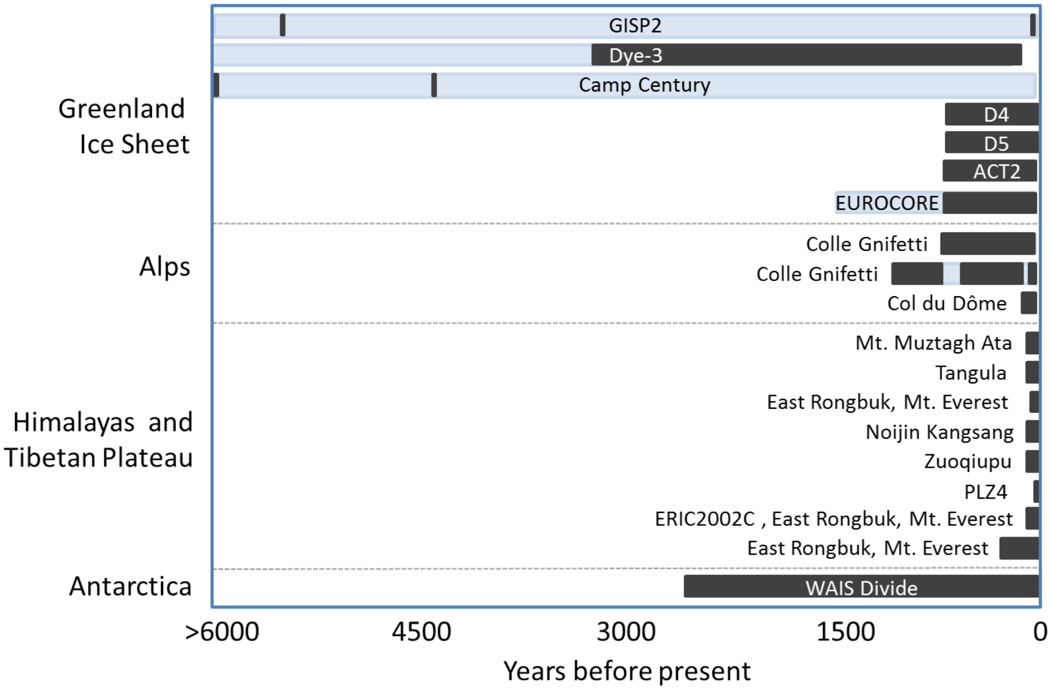

Figure 5-19. BC Ice Core Records Worldwide Labeled by Their Identifying Name. The extent of the bars (light blue and/or black) shows the time covered by the depth of the ice cores, with the black regions representing sections of the ice core that had BC concentrations reported. (Source: U.S. EPA, based on data 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. (2008b), McConnell et al. (2007), McConnell and Edwards (2008), Ming et al. (2008), Thevenon et al. (2009), and Xu et al. (2009a; 2009b))

chemical signatures (e.g., volcanic eruptions, nuclear explosions), and observing the visible layering of ice throughout the core (e.g., Hammer, 1978).

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 forward, while another ice core had continuously increasing levels until present day. Studies of ice core data collected to date find 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 associations with local emissions. In addition, deep ocean marine sediments reveal ancient BC trends related to natural emissions. Similar to ice cores, BC records in sediments initiate from the deposition of BC from the atmosphere, which relates to the atmospheric transport of BC emissions to a particular location. After depositing to the surface of a water body, the BC particles eventually transport downwards and, if the sediment is undisturbed, may form a permanent archive in the layers of sediment.

Lake sediment BC records have been quantified for several interior lakes in North America, including Lake Michigan, dating 1827-1978 (Griffin and Goldberg, 1983), and four lakes located in the Adirondacks of New York, dating 1835-2005 (Husain et al., 2008). Total carbon particles, associated with specific sources by particle shape, have also been measured in Lake Erie sediments, dating 1850-1998 (Kralovec et al., 2002). Historical BC records have also been obtained for a number of lakes in the Alps of northern Slovenia (Muri et al., 2002; 2003) and in ancient marine sediments, aged approximately 100 million to 5000 years before present, spanning southern to far northern latitudes of the Pacific Ocean and at several locations in the Atlantic Ocean (Smith et al., 1973).

The findings by Smith et al. (1973) reveal an approximate 10-fold increase in ancient BC deposited levels moving from the equator northward to 60°N (bisecting Canada), which they related to the increase in natural wildfire emissions moving from the equator northward. These trends lay the base pattern of deposited BC, to which anthropogenic emissions of BC would be added. Focusing on sediment findings that closely relate to U.S. emissions, Figure 5-20 presents estimates of atmospheric BC derived from sediment core measurements in the Adirondack region of New York State for deposition from approximately 1835 to 2005 (Husain et al., 2008) and overlays these estimates with long-term U.S. BC emissions data developed by Novakov (2003). The derived BC ambient estimates are well correlated with the historical BC emissions estimates for fossil fuel combustion in the United States, and Husain et al. (2008) attributed the decrease from 1920-2000 to reduction in BC emissions from U.S. fossil fuel combustion.

The ambient BC determined from Adirondack lake sediments by Husain et al. (2008), shown above, can be compared with records obtained from Lake Michigan sediments from Griffin and Goldberg

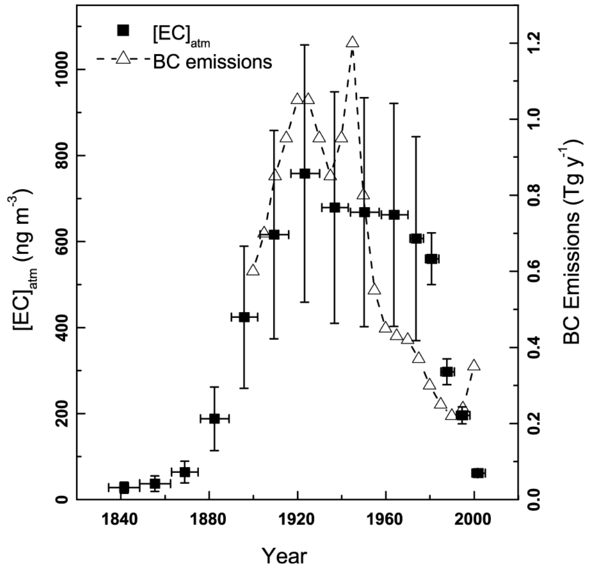

Figure 5-20. Atmospheric BC determined by Husain et al. (2008), for the Adirondack Region from 1835 to 2005. The measurements are compared with U.S. BC emissions. (Novakov et al., 2003)

(1983) and from Lake Erie sediments (Kralovec et al., 2002). While the multiple archives in multiple North American lake sediments reveal a significant increase in deposited BC levels after the late 1800s, the Lake Michigan time series shows an apparent peak in the record around 1940-1960, while the Adirondacks show a peak near 1910-1930, and the Lake Erie record shows a peak in 1970-1980. It is uncertain why the multiple sediment records in North America contrast in the timing of the peak BC concentration; possible explanations may include differences in local versus regional source contributions, dating methodology, carbon measurement approaches, and sediment deposition processes.

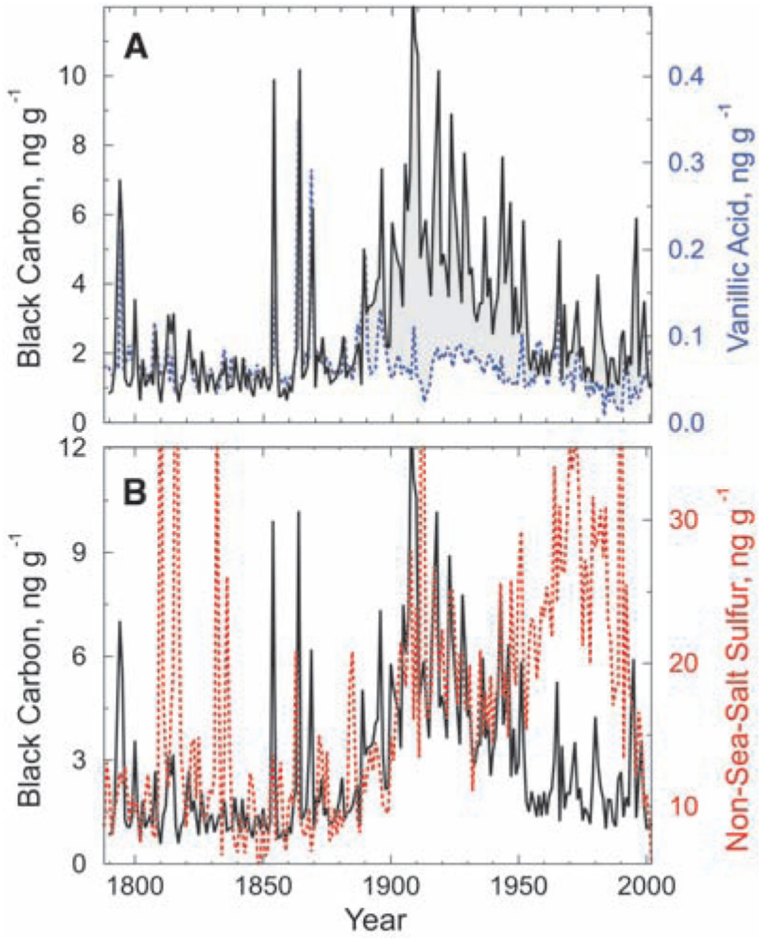

Figure 5-21. Annual Average Concentrations of (a) BC and VA and (b) BC and Non-Sea-Salt Sulfur (nss-S). The gray shaded region (between the black and blue dotted line) in the top figure represents the portion of BC attributed to industrial emissions, not boreal forest fires. (Source: Adapted from McConnell et al. (2007))

5.6.5 Arctic BC Snow and Ice Data – Source Identification

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

Historical trends in Arctic ice cores collected on the Greenland Ice Sheet improve our understanding of the historical impact of anthropogenic and natural emissions of BC on the Arctic. McConnell and Edwards (2008) and McConnell et al. (2007) provide monthly-resolution BC data in ice cores on the Greenland Ice Sheet. Similar to the lake sediment findings for the Adirondack Mountain region, the maximum BC concentrations in Arctic ice in the past hundred years occurred in the early 1900s corresponding to increases in a number of species associated with industrial emissions (e.g., cadmium, cesium, thallium, lead). McConnell et al. (2007) compare vanillic acid (VA), non-sea-salt sulfur (nss-S), and BC trends to apportion the BC due to industrial versus forest fire emissions (see Figure 5-21). VA is considered an indicator of forest fire emissions, while nss-S relates to industrial emissions and volcanic eruptions. In the postindustrial era, BC anthropogenic emissions contributed roughly 50-80% of the total BC loading in the ice during early 1900s and over past few decades the industrial input was on the order of 20-50% (estimated from Figure 5-21, originally published in McConnell et al., 2007). While nss-S correlated highly with the increasing BC during the late 1800s to mid-1900s, the trends did not match later; this may be related to changes in industrial emission factors. This study associates the high BC concentrations in the early 1900s with North American fossil fuel emissions and suggests that Asian emissions may play an important role past the mid-1900s.

Two recent studies attempted to attribute BC in snow to specific sources by collecting a large number of surface snow samples throughout the Arctic, which they measured for detailed chemical composition (Hegg et al., 2009; 2010). In Hegg et al. (2010), statistical analyses revealed that the measured species could be grouped into four unique factors with source-defining chemical characteristics (for example, sodium and chloride indicating a marine environment), which the authors labeled as marine, boreal biomass, crop and grass biomass, and pollution. It should be emphasized that the "marine" category represents air masses with an ocean-like chemical signature (i.e., sea salt), which may also include emissions from other sources (biomass or fossil fuel combustion) that transported over the ocean and mixed with sea spray. Depending on the location of the sample within the Arctic and time of year, the estimated contribution from these four sources varied considerably (Figure 5-22). In Siberia, emissions from biomass burning were significant drivers of BC and other absorbing species. However, on the Greenland Ice Sheet and at the North Pole, pollution and crop/grass biomass were found to be the primary sources.

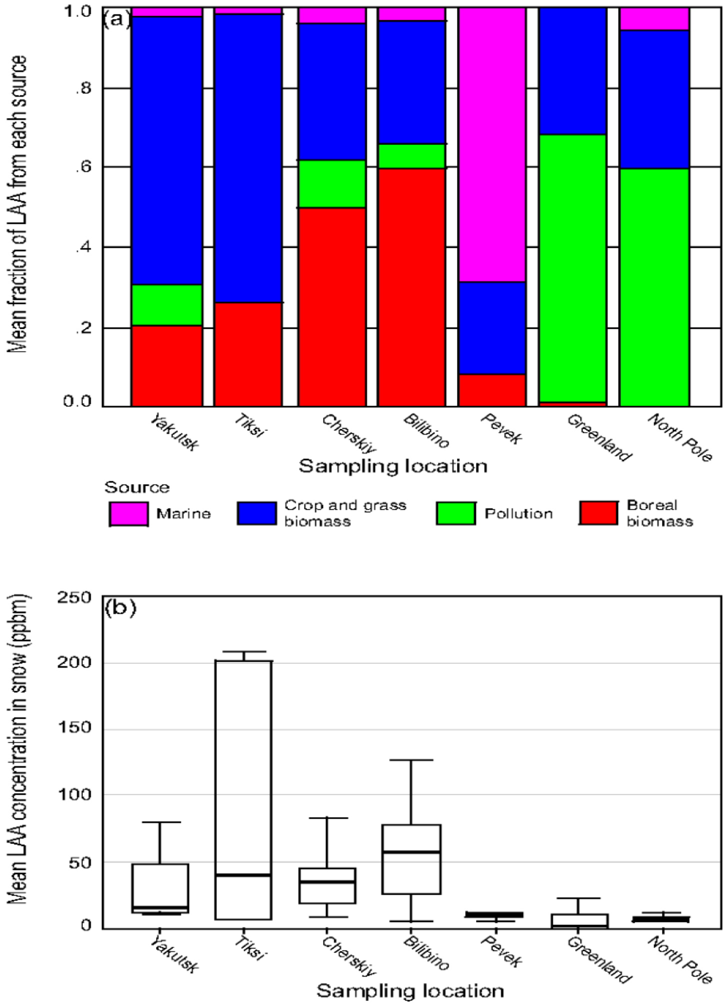

Figure 5-22. Sources of BC in Arctic Snow. (a) Fractional source contributions to Light Absorbing Aerosol (LAA) snow concentrations in Siberia (Pevek, Billbino, Cherskiy, Tiksi, Yakutsk), the Greenland Ice Sheet, and the North Pole. (b) The box and stem plots represent concentrations of LAA at each location, with error bars indicating the 95% confidence interval. LAA values are derived from a light absorption technique which converts to a mass estimate using calibration factors and are generally equivalent to BC values, although non-BC absorbing aerosols could bias the estimate high. (Source: Hegg et al., 2010)

5.7 Limitations and Gaps in Current Ambient Data and Monitoring Networks

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

Chapter 6

Benefits of Reducing Black Carbon Emissions

6.1 Summary of Key Messages

- Mitigation of BC offers a clear opportunity: continued reductions in BC emissions can provide significant near-term benefits for climate, public health, and the environment.

- It is generally impossible to reduce BC in isolation from other co-pollutants: most BC mitigation strategies involve reductions in total emissions of fine particles, including other $PM_{2.5}$ constituents. Fortunately, all control measures that reduce $PM_{2.5}$ pollution—including BC and other constituents—will achieve substantial health benefits. Therefore, mitigation strategies targeting BC that also reduce total direct $PM_{2.5}$ emissions could potentially result in hundreds of thousands of avoided premature deaths each year. The health benefits alone may be large enough to justify mitigation in many regions and sectors.

    - Because a variety of PM constituents are associated with adverse health impacts, significant health benefits are anticipated to result from reducing exposure to PM containing BC, regardless of the precise chemical composition of the emissions mixture.

    - Controls on direct $PM_{2.5}$ emissions (including BC) can be particularly beneficial since they are more co-located with population than other PM components. According to U.S. EPA estimates, the benefits from control of direct $PM_{2.5}$ emissions are 7 to 300 times greater than the benefits per ton estimated for reductions of PM precursors such as NO_x and SO_x.

    - The magnitude of human health benefits of emissions reductions depends both on how much exposure is reduced and the size of the affected population. The largest health benefits from $PM_{2.5}$ including BC control strategies will be achieved in areas near the emissions source and where exposure affects a large population.

    - Programs to reduce $PM_{2.5}$ in the United States and in other developed countries have greatly reduced the negative health impacts of $PM_{2.5}$, including BC. New programs introduced for mobile and stationary sources will continue to reduce $PM_{2.5}$-related health impacts in the United States and the countries that implement them over the next several decades.

    - The largest opportunities for achieving health benefits of BC mitigation measures are in lesser developed countries, due to high emissions co-located with large populations, particularly in South and East Asia.

- Estimating the climate benefits of mitigation strategies is more challenging than health benefits estimation because key scientific issues remain unresolved. Despite uncertainties in the magnitude of effects, the literature suggests climate benefits could be achieved through some mitigation measures.

    - BC reductions are expected to reduce the rate of warming soon after measures are implemented, though the full climate response may take several decades to fully manifest. In sensitive regions (e.g., the Arctic), or in regions of large emissions (e.g., South and East Asia), additional benefits may include slowing the rate of ice, snow, and glacier melt, and reversal of adverse precipitation changes.

    - Achieving climate benefits is most likely to result from the control of emissions from BC-rich sources such as diesel engines, cookstoves, brick kilns and coke ovens.

    - BC mitigation alone cannot change the long-term trajectory of global warming, which is driven by CO_2 emissions. However, as illustrated by a recent UNEP/WMO analysis, controlling current BC and long-lived GHGs in concert would greatly improve the chances of limiting global temperature rise below 2°C relative to pre-industrial levels.

  - BC mitigation may have particular benefits for the Arctic and the Himalayas, two regions that are particularly sensitive to BC deposition. In the Arctic, both global and high-latitude BC reductions can provide climate benefits. The most effective emissions reductions per unit of emissions will come from within- or near-Arctic sources, but overall reductions in global BC emissions will be critical to slowing the rate at which the Arctic is warming and melting. Slowing the rate of change may avoid reaching certain global tipping points as discussed in earlier chapters. In the Himalayan region, a range of local measures is likely to provide climate benefits, including sizable benefits for radiative forcing, ice/snow melt, precipitation and surface dimming.

- Limiting BC emissions also is expected to provide a number of environmental benefits, including improved visibility, reduced surface dimming, reduced impacts on ecosystems, and less damage to building materials.

- More research is needed on the benefits of individual control measures in specific locations to support policy decisions made at the national level. Research is also needed to design approaches to valuing the climate impacts of BC directly, and to incorporate those approaches into useful metrics for evaluating policy decisions, similar to the social cost of carbon (SCC).

6.2 Introduction

This chapter summarizes available information regarding the potential public health, climate and environmental benefits of reducing BC emissions, both in general and from particular economic sectors. The literature on the health benefits of reductions in fine particles (including BC) is well-developed, particularly in the United States where emissions control programs have been evaluated extensively. These analyses provide a high degree of confidence that BC mitigation strategies will produce significant public health benefits, not just from the BC reduction, but also from the reduction in co-emitted gaseous and particulate pollutants as well. As BC effects on climate are less certain than on health, there is also less certainty about the climate benefits of BC mitigation. Although the body of literature on climate benefits is limited, it provides important insights regarding which strategies are most likely to provide climate benefits. Non-climate environmental benefits of BC reductions are also less certain than health benefits, but the literature on $PM_{2.5}$ impacts provides important qualitative information regarding the likelihood of such benefits. It is important to note that a quantitative assessment of the benefits of specific mitigation measures was not conducted for this report due to time and resource constraints. Further analysis would be necessary to quantify the public health, climate and environmental benefits of specific BC reduction strategies, either individually or in combination.

6.3 Public Health Benefits of Reducing Black Carbon Emissions

All control measures that reduce $PM_{2.5}$ emissions, of which BC is a component, are virtually certain to achieve health benefits. The adverse health impacts of $PM_{2.5}$ are well documented in the scientific literature, previously discussed in Chapter 3 and in EPA's recent PM ISA (U.S. EPA, 2009a). The well established linkage between $PM_{2.5}$ and health effects is important, because mitigation strategies aimed at reducing BC almost always involve reductions in co-pollutants, including other PM constituents or precursors, as well as BC. BC is emitted in a mixture of other pollutants that are also associated with negative health effects, including primary particulate matter (including OC), precursors of secondary particulate matter (SO_2, NO_x, NH_3), and ozone precursors (NO_x, CO, VOCs). Reductions in all of these pollutants can provide human health benefits. Though the literature on differential toxicity of $PM_{2.5}$ components and mixtures is currently inconclusive, studies continue to provide evidence that many $PM_{2.5}$ constituents are associated with adverse health impacts, as discussed in Chapter 3. Because of the well documented health impacts of $PM_{2.5}$, there is high likelihood that BC mitigation strategies that reduce $PM_{2.5}$ will produce public health benefits. This section summarizes what is known about the health benefits of BC emissions reductions in the United States and globally.

6.3.1 Health Benefits in the United States

Historically, the United States has been quite successful in achieving significant $PM_{2.5}$ reductions through air quality protection programs such as attaining the $PM_{2.5}$ NAAQS and implementing a variety of mobile source rulemakings. These reductions yield large human health benefits. As discussed in Chapter 4, however, emissions of BC in the United States are still fairly substantial. Additional control programs, largely those affecting mobile diesel engines, are expected to achieve further BC reductions by 2030 with significant public health benefits (see Chapter 8). EPA's analyses of these benefits provide useful information regarding the potential public health improvements that can be achieved via strategies targeting direct emissions of $PM_{2.5}$, including BC. Below, we describe how the benefits of policies that reduce atmospheric pollutant loads are developed, and discuss in more detail specific estimates of the health benefits that $PM_{2.5}$ mitigation policies can help achieve.

6.3.1.1 Methods for Estimating the Health Benefits of Reducing $PM_{2.5}$

Peer-reviewed methods for estimating the public health benefits of $PM_{2.5}$ emissions reductions have been used in a variety of settings and the sophisticated models and techniques developed for application in the United States have been refined by EPA over the course of several decades. These methods rely upon atmospheric models to translate emissions changes into concentration changes, and epidemiologically derived concentration-response functions to calculate the change in a health endpoint attributable to the concentration change. Valuation techniques are then used to quantify the economic impact of the health benefits, with the total monetized benefit calculated as the sum of the values for all non-overlapping health endpoints. Health benefits are often monetized using the Value of a Statistical Life (VSL), which is determined by studies of individuals' willingness to pay (WTP) for reducing their risk of mortality. This approach is the standard method for assessing benefits of environmental quality programs and has been used widely in EPA regulatory documents (e.g., U.S. EPA, 2006c), as well as in the peer-reviewed literature (e.g., Levy et al., 2009; Hubbell et al., 2009; Tagaris et al., 2009).

Air pollution affects a variety of health endpoints, as discussed in Chapter 3. For some, currently available data are insufficient to enable quantification or monetization of effects. Table 6-1 summarizes the health endpoints that have been included in recent EPA $PM_{2.5}$ benefit assessments. The table indicates which effects have been quantified and monetized (left column), and which are discussed only qualitatively (right column).

Overall, the $PM_{2.5}$ control program in the United States has proven highly protective of public health. Multiple studies estimate significant reductions in $PM_{2.5}$-related mortality and morbidity since the implementation of the Clean Air Act (CAA). In a recent study, EPA also estimated that programs implemented as a result of the 1990 CAA Amendments have avoided about 160,000 annual premature adult deaths by 2010, and will avoid 230,000 by 2020 (Table 6-2). Direct $PM_{2.5}$ reductions specifically have been associated with 22,000 to 60,000 annual avoided premature deaths between 2000 and 2007 (Fann and Risley, 2011). Improvements in particulate air pollution, particularly in urban areas, have also been estimated to contribute 15% of a 2.72 year increase in average life expectancy among 211 counties between 1980 and 2000 (Pope et al., 2009).

6.3.1.2 The Potential Benefits of Further Mitigation of $PM_{2.5}$ (Including BC) Emissions

Despite significant improvements, the health burden of $PM_{2.5}$ in the United States is still substantial. Based on 2005 air quality data and population, Fann et al. (2011) estimated that about 130,000 annual premature deaths and 19% of all ischemic

Table 6-1. $PM_{2.5}$ Health Endpoints Included in EPA's Regulatory Impact Analyses. (Source: U.S. EPA, 2006, Table 5-2)

Quantified and Monetized $PM_{2.5}$ Health Endpoints	Un-Quantified and Non-Monetized $PM_{2.5}$ Health Endpoints
Premature mortality based on cohort study estimates and expert elicitation estimates	Low birth weight, pre-term birth and other reproductive outcomes
Hospital admissions: respiratory and cardiovascular	Pulmonary function
Emergency room visits for asthma	Chronic respiratory diseases other than chronic bronchitis
Nonfatal heart attacks (myocardial infarctions)	Non-asthma respiratory emergency room visits
Lower and upper respiratory illness	UVb exposure (+/-)
Minor restricted activity days	
Work loss days	
Asthma exacerbations (among asthmatic populations)	
Respiratory symptoms (among asthmatic populations)	
Infant mortality	

heart disease-related deaths nationwide were attributable to $PM_{2.5}$ exposure. Fortunately, a number of additional $PM_{2.5}$ control programs that will help to control $PM_{2.5}$ (including BC) over the next several decades have already been adopted and are expected to yield significant health benefits. In the illustrative regulatory impact analysis conducted by EPA in 2006 for the revised $PM_{2.5}$ NAAQS, benefits

Table 6-2. Changes in Key Health Effects Outcomes in the United States Associated with $PM_{2.5}$ Resulting from the 1990 CAA Amendments. (Source: U.S. EPA (2011a) Table 5-6, adjusted from 2006$ to 2010$)

Health Effect Reductions	Year 2010		Year 2020	
	Incidence Avoided	Valuation (millions 2010$)	Incidence Avoided	Valuation (millions 2010$)
Adult Mortality	160,000	$1,300,000	230,000	$1,800,000
Infant Mortality	230	$2,100	280	$2,700
Chronic Bronchitis	54,000	$26,000	75,000	$39,000
Acute Bronchitis	130,000	$66	180,000	$100
Acute Myocardial Infarction	130,000	$15,000	200,000	$23,000
Hospital Admissions, Cardiovascular	45,000	$1,400	69,000	$2,200
Lower Respiratory Symptoms	1,700,000	$32	2,300,000	$45
Upper Respiratory Symptoms	1,400,000	$45	2,000,000	$65
Asthma Exacerbation	1,700,000	$97	2,400,000	$140
Lost Work Days	13,000,000	$2,200	17,000,000	$2,900

Notes:

1. EPA's estimates of reductions in respiratory hospital admissions, respiratory emergency room visit, and minor restricted activity days are not included here, because the 2011 report presented these estimates only as combined totals for both $PM_{2.5}$ and ozone.
2. Estimates reflect annual incidence avoided and valuation for reductions in direct $PM_{2.5}$ and $PM_{2.5}$ precursors.

Table 6-3. List of Benefits, Costs, and Benefit to Cost Ratios for U.S. Rules with Direct PM Reductions (Billions 2010$).[a] (Source: U.S. EPA)

Rule (by Sector)	Annual Benefits[b]	Annual Costs	Benefit/Cost Ratio	Benefit Year
Transportation				
Light Duty Tier 2	$34	$7.2	4.7	2030
Heavy Duty 2007	$92	$5.5	16.7	2030
Non-road Diesel Tier 4	$105	$2.5	41.5	2030
Locomotive & Marine Diesel	$9.9–23.8	$0.8	13.1–31.4	2030
Ocean Vessel Strategy	$119–292	$3.4	35.5–87.1	2030
Stationary Sources				
2006 PM NAAQS[c]	$22	$7.1	3.1	2020
Cement NESHAP	$8.3–20.1	$1.01–1.04	8.2–19.4	2013
Stationary Spark Ignition RICE NESHAP	$0.52–1.2	$0.26	2–4.7	2013
Stationary Compression Ignition Engine NESHAP	$0.95–2.3	$0.37	2.5–6.2	2013

[a] Rules include a combination of direct $PM_{2.5}$ and PM precursor reductions. These estimates have been adjusted from the dollar years in the original analysis to 2010$.

[b] 3% discount rate used for benefit estimates.

[c] 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.

Table 6-4. Direct $PM_{2.5}$ National Average Benefits per Ton Estimates by Source Category for the United States (3% Discount Rate, Thousands of 2010$). (Source: *http://www.epa.gov/air/benmap/bpt.html*, based on method described in Fann et al., 2009)

Source Category	Monetized Benefit Per Ton in 2015	Monetized Benefit Per Ton in 2020	Monetized Benefit Per Ton in 2030
Area Source			
Pope et al. (2002)	$360	$400	$480
Laden et al. (2006)	$880	$970	$1,200
Mobile Source			
Pope et al. (2002)	$280	$300	$360
Laden et al. (2006)	$680	$740	$890
EGU and Non-EGU			
Pope et al. (2002)	$230	$250	$290
Laden et al. (2006)	$560	$610	$710

Notes:

1. These estimates have been adjusted from 2006$ to 2010$.
2. These are U.S. national average estimates, and these estimates may vary for different geographic locations in the country.

are estimated to be $22 billion per year in 2020 (2010$) (U.S. EPA, 2006c). Benefits for the non-road diesel rule have been estimated at $105 billion per year in 2030 (2010$) (U.S. EPA, 2004a). The benefits for these and other rules with direct $PM_{2.5}$ reductions are shown in Table 6-3, along with the costs and benefit-cost ratios for each rule.[1] While the relationship between benefits and costs for $PM_{2.5}$ reductions is discussed in more detail in Chapter 12, it is useful to note that quantified benefits exceed costs for each rule, often by a significant margin. For the nonroad diesel rule, quantified benefits are estimated to be over 41 times greater than costs in 2030.

As a shorthand approach for assessing potential health benefits resulting from different mitigation strategies when air quality modeling is unavailable, Fann et al. (2009) developed values of monetized health benefits per ton of emissions reduced for SO_2, NO_x, and direct $PM_{2.5}$ in the United States.[2] For directly emitted $PM_{2.5}$ (including BC) from all sources, these benefits (on average) range from $230,000 to $880,000 per ton of $PM_{2.5}$ reduced in 2015 (2010$).[3] While EPA has not separately estimated the benefits per ton for BC reductions specifically, Table 6-4 illustrates the results for reductions in total direct carbonaceous emissions (i.e., BC + OC) for 2015, 2020, and 2030. It is clear that controls on all sources of direct $PM_{2.5}$ can produce substantial public health benefits in the United States; furthermore, these benefits are 7 to 300 times greater than the benefits-per-ton estimated for reductions of other PM precursors such as NO_x and SO_x (Fann et al., 2009), indicating that controls on direct $PM_{2.5}$ may be particularly effective for protecting public health. The authors attribute this largely to the fact that carbonaceous particles tend to be emitted in close proximity to population centers.

These $PM_{2.5}$ monetized benefit-per-ton estimates are useful for evaluating the benefits associated with incremental $PM_{2.5}$ air quality improvements in the United States and represent the premature mortality and premature morbidity benefits associated with reducing one ton of $PM_{2.5}$ from a specific source. As discussed above, these estimates are based upon the methodology described in

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[1] The benefits, costs, and associated benefit-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.

[2] The benefit-per-ton estimates found in Fann et al. (2009) reflect a specific set of key assumptions and input data. As EPA updates these underlying assumptions to reflect the scientific literature, the benefit-per-ton estimates are re-estimated and are available at: *http://www.epa.gov/air/benmap/bpt.html*.

[3] According to Fann et al. (2009), the wide range in these benefit-per-ton estimates reflects several key sources of heterogeneity, including variability in source parameters which affect pollutant dispersion and human exposure, and variability in location-specific factors such as population density and baseline health status. In addition, the estimates vary depending on which morbidity and mortality effect estimate are utilized from the underlying epidemiological references.

Fann, et al. (2009) that used an innovative reduced-form air quality model to estimate changes in ambient $PM_{2.5}$ concentrations resulting from a variety of emissions control strategies applied to different classes of emissions sources. The estimates originally developed by Fann, et al. (2009) have been updated to incorporate revised VSL estimates (*http://www.epa.gov/air/benmap/bpt.html*). The monetized mortality and morbidity benefits of changes in ambient $PM_{2.5}$ were estimated using the Environmental Benefits Mapping and Analysis Program (Abt Associates, 2008) and developed for specific U.S. urban areas and the United States as a whole.[4]

While EPA strives to incorporate quantitative assessments of uncertainty in the health impacts estimates, there are aspects for which only qualitative assessments are possible. Key assumptions underlying the estimates are presented in detail in the regulatory impact assessments for each regulation. Typically, health impact assessments include uncertainty in the concentration-response function but are unable to include uncertainty in emissions, simulated concentrations, and projected population and mortality rates.

6.3.2 Global Health Benefits

Though the United States has already made great strides toward reducing BC through its efforts to reduce $PM_{2.5}$ emissions, BC emissions remain very high in some parts of the world due to industrial production, residential burning of solid fuel, and transportation (see Chapter 7). Furthermore, unlike in the United States and Europe, where additional controls are already planned for key source categories such as mobile diesel engines, emissions from many other international sources are not yet subject to plans for control. As a result, the largest remaining achievable increment of public health benefits from controls on BC is international, particularly in South and East Asia, where large populations are exposed to high concentrations. While a growing body of literature examines the climate benefits of controlling BC emissions globally (see section 6.4), only a few studies have examined the associated health benefits. For these few studies, the estimated public health benefits are very large, and for many control measures the benefits greatly exceed the costs of controls, suggesting that these reduction measures will be advantageous for society independent of the level of climate benefits achieved. This section (1) describes studies that have estimated the potential health benefits that can be achieved through mitigating BC emission and (2) discusses approaches that have been used to value the health benefits that could be achieved on a global scale.

6.3.2.1 Estimating the Benefits of Global BC Mitigation

In a study focused specifically on the health impacts of BC reductions, Anenberg et al. (2011) estimated that halving anthropogenic BC emissions (but not any co-emissions) globally would avoid 157,000 (95% confidence interval, 120,000-194,000) annual premature deaths worldwide. Over 80% of these health benefits occurred in East Asia (China; 54%) and South Asia (India; 31%), where large populations are exposed to high concentrations (see blue bars in Figure 6-1). Halving all anthropogenic BC emissions in each major world region individually demonstrated that the vast majority of avoided deaths from halving BC emissions occur within the source region, with very little impact from extra-regional emissions. This is because BC impacts on health are driven by surface concentrations where humans live. BC emissions that are transported to other regions are usually conveyed at high altitudes, where they may have more widespread impacts on climate, but impact human health less. Per unit of emissions, the mortality impact of BC emissions was estimated to be 50% larger for South Asia than for East Asia (see red diamonds in Figure 6-1). This is likely because emissions changes in East Asia have smaller impacts on concentrations and because mortality rates are higher in South Asia.

Anenberg et al. (2011) found that halving global residential, industrial, and transportation emissions contributed 47%, 35%, and 15% of the avoided deaths, respectively, from halving all anthropogenic BC emissions. Residential and industrial sector contributions to global BC-related mortality are each 1.2 times greater than their contributions to global BC emissions, owing to their co-location with dense populations, mainly in developing regions. In contrast, the contribution of transportation emissions to mortality is 40% lower than the contribution of that sector to global BC emissions, since transportation emissions are more evenly distributed among developing and less populated developed regions. Avoided deaths were likely underestimated for the residential sector since indoor exposure was excluded from the study. Figure 6-2 shows that while the industrial and residential sectors in East Asia have the greatest BC emissions ("mitigation potential"), all three sectors in South Asia have the greatest estimated

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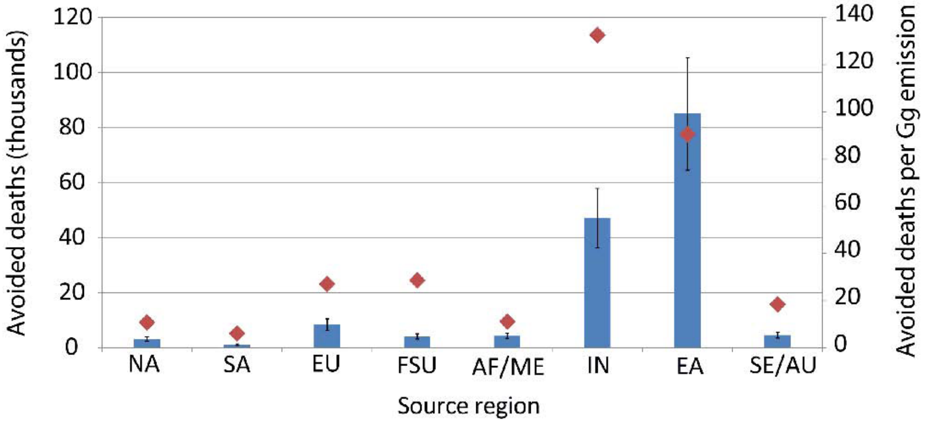
[4] For further information about the underlying methodologies and analytical assumptions used to develop these estimates, as well as, relevant uncertainties involved in the estimates see Fann, et al. (2009) and EPA regulatory impact analyses including the SO_2 NAAQS RIA (U.S. EPA, 2010h) and the Portland Cement NESHAP RIA (U.S. EPA, 2010b) available at *http://www.epa.gov/ttn/ecas/ria.html*.



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

Figure 6-1. 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., 2011)

Note:
Confidence intervals (95%) reflect uncertainty in the Concentration Response Function only.

mortality impacts per unit of emissions ("mitigation efficiency"). Outside of South Asia and East Asia, estimated mitigation efficiency is greatest for the Former Soviet Union, Southeast Asia/Australia, and Europe, while mitigation potential is likely greatest for the residential sector in Africa/Middle East and for the transportation sector in Europe and North America.

It is important to note that these estimates understate the full public health benefits that would be achieved by reductions in global BC emissions. Since controls to reduce BC will generally reduce other directly emitted particles as well, halving global BC emissions would likely result in far larger changes in overall $PM_{2.5}$ emissions. In fact, Anenberg et al. (2011) estimated that halving global anthropogenic OC emissions along with BC resulted in eight times more avoided premature deaths annually than halving BC alone. Nevertheless, this study demonstrates that BC mitigation efforts are likely to be more effective at reducing mortality in some regions than others, largely driven by population exposure. Although the coarse grid resolution (~170 km on a side) used by Anenberg et al. (2011) was unable to capture fine-scale spatial gradients in population and concentration, emissions from different sectors result in different exposure patterns. Therefore, the health response to controlling emissions from different regions and from different source sectors is likely to vary. Finer scale models can be used to investigate how different mitigation strategies impact health within individual world regions.

While Anenberg et al. (2011) examined broad percentage decreases in BC emissions from individual source regions and sectors, actual mitigation measures will affect the full mixture of emissions from individual sources. Currently, the most comprehensive assessment of more realistic emissions control measures is the *Integrated Assessment of Black Carbon and Tropospheric Ozone* sponsored by the United Nations Environment Programme (UNEP) and the World Meteorological Organization (WMO). Using an integrated modeling approach addressing a range of co-emitted pollutants, the UNEP/WMO Integrated Assessment identified a small number of emissions reduction measures that would achieve major benefits for near-term climate change. This suite of measures included both BC reduction measures and methane reduction measures. For BC, the assessment modeled the impact of both "technical measures," such as improving coke ovens and brick kilns and increasing use of diesel particulate filters, and "non-technical measures," such as eliminating high-emitting vehicles, banning open burning of agricultural waste, and eliminating biomass cookstoves in developing countries. Specifically, the Assessment evaluated the health benefits of the following BC measures:

- Use of diesel particle filters as part of a Euro VI package for on-road and off-road diesel vehicles

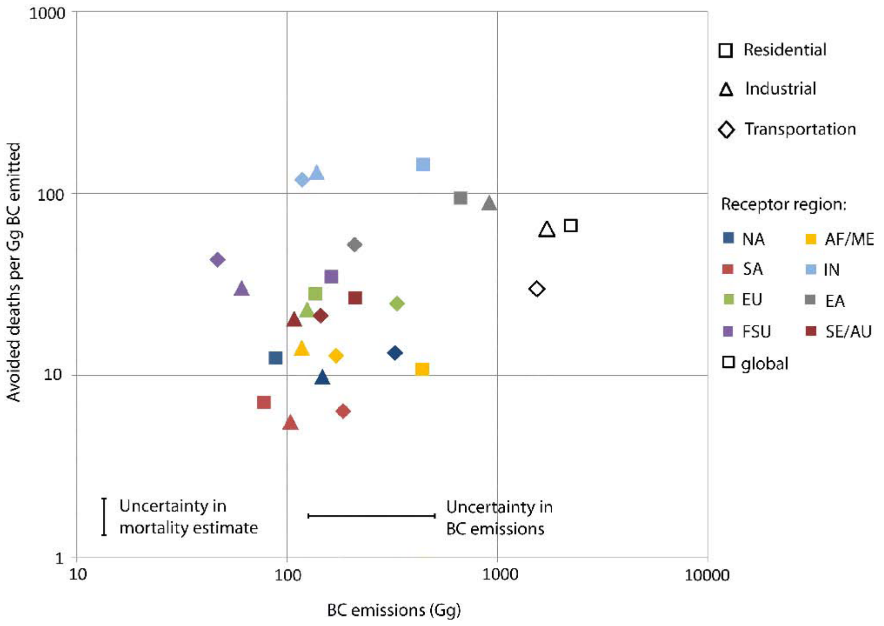


Figure 6-2. Annual Avoided Premature Cardiopulmonary and Lung Cancer Deaths Per Unit BC Emissions Reduced ("mitigation efficiency") versus Total BC Emissions (Gg; "mitigation potential") for Particular Source Sectors within Each Region. (Anenberg et al., 2011)

Notes:

1. Avoided deaths are estimated in the three simulations where global emissions in each sector are halved, and shown for each receptor region; these deaths are compared with emissions from each region, assuming that deaths from inter-regional transport are negligible.
2. Uncertainty in the mortality estimates is calculated from the uncertainty in the CRF only (~22% and 56% from mean for cardiopulmonary and lung cancer mortality).
3. Uncertainty in BC emissions is assumed to be a factor of 2 from the central estimate (Bond et al., 2004; 2007).
4. Since these uncertainties are factor differences from the central estimate, they are identical for each data point.

- Introduction of clean-burning cook stoves for cooking and heating in developing countries
- Replacement of traditional brick kilns with vertical shaft kilns and Hoffman kilns
- Elimination of high-emitting vehicles in on-road and off-road transport (excluding shipping)
- Ban of open field burning of agricultural waste
- Substitution of clean-burning cook stoves using modern fuels for traditional biomass cook stoves in developing countries

Together, these BC measures were estimated to reduce global anthropogenic BC emissions by 75%, along with substantial reductions in co-emitted OC, NO$_x$, and CO.

The UNEP/WMO Assessment estimated that fully implementing these measures by 2030 would avoid 0.6-4.4 million PM$_{2.5}$ related premature deaths and 0.04-0.52 million ozone-related premature deaths annually around the world, based on 2030 population projections (Shindell et al., 2012; UNEP and WMO, 2011b). Consistent with the results of Anenberg et al. (2011), over 80% of the health benefits occur in Asia. Figure 6-3 shows that implementing the BC and methane measures would reverse the trend of increasing air pollution-related deaths in Africa and South, West, and Central Asia (although methane and BC measures are shown together here, BC measures contribute ~98% of the total health benefits). Figure 6-3 also shows the additional benefits achievable in areas already making progress. The study also found that the substantial health benefits of the joint air quality/climate mitigation measures examined occur regardless of whether measures to reduce long-lived GHG have been implemented. A follow-on study to the UNEP/WMO Assessment found that in Africa, Asia, and Latin America and the Caribbean, improved biomass cookstoves would generate the greatest health benefit of all the measures examined, with substantial additional benefits from mitigation measures for the transportation sector (UNEP, 2011). In Europe and North America, switching to pellet stoves from current domestic wood-burning technologies was estimated to deliver the greatest health benefit. The study also found that banning the burning of agricultural crop residues would produce a small benefit in all regions.

One additional study examined the potential health benefits of global reductions in vehicle emissions specifically, accounting for the full

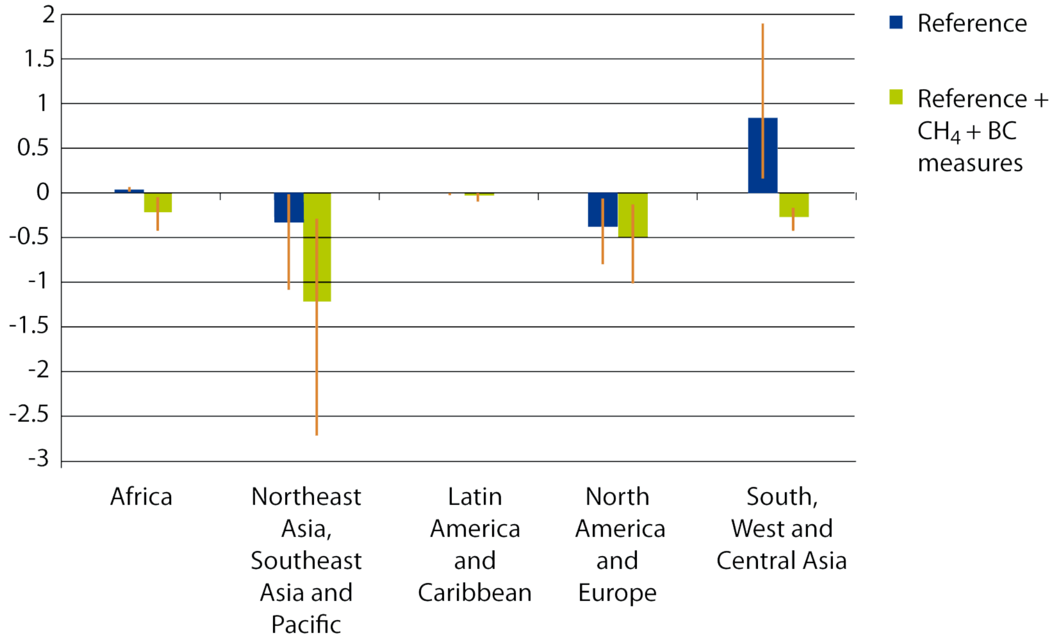


Figure 6-3. Comparison of Premature Mortality by Region (millions of premature deaths annually), showing the change in 2030 in comparison with 2005 for the reference scenario emission trends and the reference plus methane and BC measures. The lines on each bar show the range of estimates. Over 98% of the benefits were attributed to the BC measures. (Source: UNEP and WMO, 2011b)

mixture of emissions affected by control measures (Shindell et al., 2011). Emissions from on-road motor vehicles, including cars, trucks, and motorcycles, are growing rapidly in many countries, due to rising personal vehicle ownership and usage. Shindell et al. (2011) assessed the potential health benefits of imposing existing gasoline and diesel European vehicle emissions standards in developing regions. Specifically, the study examined the adoption of Euro 6 vehicle standards in China and India, Euro 4 vehicle and Euro 3 motorcycle standards in Africa and the Middle East, Euro 6 vehicle and Euro 3 motorcycle standards in Brazil, and Euro 3 motorcycle standards in the rest of Latin America, based on the authors' judgment of local financial, technical, and institutional capacity. The European standards are more stringent than the standards currently planned in these regions. Imposing these standards was estimated to avoid 120,000-280,000 premature $PM_{2.5}$ and ozone-related deaths worldwide, based on 2030 population, largely resulting from local emissions controls. Small benefits were also seen in areas with no additional local emission controls due to long-range transport of ozone and $PM_{2.5}$ in the atmosphere. The study concluded that tighter vehicle emissions standards are likely to lead to significant health benefits, in addition to climate benefits in most cases.

In addition to the BC and sector-specific studies described above, there is a small but emerging body of literature assessing the global health benefits of $PM_{2.5}$ emissions reductions. The results from additional global and international studies are summarized in Appendix 3. Many of these studies estimate the avoided premature deaths associated with reductions in BC and other $PM_{2.5}$ constituents, while other studies attempt to compare the costs and benefits of potential mitigation strategies. These studies indicate that a large number of premature deaths can be avoided annually by undertaking strategies to reduce BC emissions (Shindell et al., 2011; 2012; Anenberg et al., 2011; Wilkinson et al., 2009; Saikawa et al., 2009; Jacobson, 2010). The studies that include a benefit-cost comparison show that estimated human health benefits significantly exceed the estimated costs for certain BC mitigation strategies (Smith and Haigler, 2008; Kandlikar et al., 2009; Baron et al., 2009). Thus, BC reductions appear advantageous to society independent of the level of climate benefits achieved. This is particularly true of sources associated with high human health exposures, such as cookstoves (which are often used indoors in confined spaces) and vehicles located in densely populated areas.

Valuing the global health benefits of emissions controls is difficult due to limited data on willingness to pay for reducing mortality risks. In general, studies valuing reduced mortality around the world have used two methodologies. In the first, a uniform VSL, generally from U.S. or European studies, is applied to avoided premature deaths in all countries, regardless of income or other economic disparities. While ethically appealing, willingness-to-pay is a function of income and thus would reasonably be expected to vary around the world. The second approach is to adjust VSLs from the developed world by measures of income or other economic welfare in other countries, for example income elasticity. While this method is often criticized because it seems to imply that lives in different countries are valued at different levels, it accounts for differences in income levels and is therefore often the preferred approach among economists. This approach still has limitations, including the inability to account for differences in social values and culture.

Acknowledging the advantages and disadvantages of these valuation approaches, the UNEP/WMO Assessment and Shindell et al. (2011) reported the monetized health benefits of mitigation measures using both methods (UNEP and WMO, 2011a). The UNEP Assessment estimated the monetary value of the ozone and $PM_{2.5}$-related premature deaths avoided as a result of the methane and BC measures was $1.7-10.9 trillion in 2010 $US. Consistent with the mortality results, the vast majority of the monetized health benefits resulted from the BC measures. Shindell et al. (2011) estimated that the global health benefits of imposing tighter vehicle emissions standards in the developing world are valued at $0.7-2.6 trillion in 2010 $US. Since the majority of the health benefits resulting from these emissions control measures occur in developing countries, the income-adjusted VSL approach leads to lower valuation estimates compared with the uniform VSL approach.[5] Shindell et al. (2012) conclude that since about half of the benefits of all BC mitigation measures are attributable to improved efficiencies for implementing improved brick kilns and cleaner burning stoves, which lead to net cost savings, and another 25% to regulatory measures on high-emitting vehicles and banning agricultural waste burning, which require primarily political rather than economic investment, the majority of the BC measures could be implemented with substantially greater benefits than costs.

6.3.3 Conclusions Regarding Potential Health Benefits

All control measures that reduce $PM_{2.5}$ pollution are virtually certain to achieve health benefits. Programs aimed at reducing $PM_{2.5}$ in the United States, such as rules targeting light and heavy duty vehicles, diesel emissions, and marine vessels, as well as industrial stationary sources, have greatly reduced $PM_{2.5}$ concentrations (including BC) and $PM_{2.5}$-related mortality. These programs have very favorable benefit-cost ratios, particularly for the mobile source sector. While progress has been made, the $PM_{2.5}$ health burden in the United States remains significant. EPA has introduced a number of programs for both mobile and stationary sources that are estimated to have a substantial impact on air quality and, as a result, $PM_{2.5}$-related health impacts, over the next several decades. However, additional controls for transportation and stationary sources, as well as for residential wood burning, can further reduce the remaining BC emissions (Chapters 8-11).

The largest opportunities for achieving the health benefits of BC mitigation measures are in lesser developed countries due to high emissions located in densely populated areas, particularly in South and East Asia. Although the body of literature is limited, available studies demonstrate that mitigating BC emissions would have substantial benefits for global public health, potentially avoiding millions of premature deaths each year valued in the trillions of $US. Although valuing health benefits around the world is complicated by data limitations, several studies undertaking such analyses have found that the mortality benefits alone are quite substantial and may alone justify mitigation efforts. Reducing BC emissions from transportation and residential sources, in addition to some BC-rich industrial sources, would likely achieve the greatest combined health and climate benefits. More information on the benefits and costs of individual measures in each country is needed to support policy decisions made at the national level.

6.4 Climate Benefits of Reducing Black Carbon Emissions

A number of recent studies and assessments have pointed to the possibility that reducing BC could provide climate benefits within the next several decades. Some of these studies have focused exclusively on BC, without adequately treating co-emitted pollutants, and/or have estimated direct forcing effects only, without accounting for the potential off-setting cloud interaction effects. As the treatment of BC's atmospheric chemistry and co-pollutants in climate models has advanced, however, studies have begun to focus on certain key sectors and regions as potentially fruitful mitigation options for climate. The recent UNEP/WMO (2011a) assessment, for example, indicates that a small group of carefully targeted BC measures could help improve chances of keeping the Earth's temperature increase to less than 2°C relative to pre-industrial levels (see section 6.4.1). However, the climate benefits of reducing BC emissions are less well understood and less certain than the public health benefits. Because BC concentrations and

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[5] The UNEP/WMO Assessment (2011) did not evaluate the full costs of implementing the modeled measures, but did report cost information where available for key demonstration projects in different countries and sectors.

their climate impacts vary spatially and temporally (as discussed in Chapter 2), the location and timing of emissions reductions is critically important for estimating climate benefits of mitigation. In addition, emissions control measures for BC also reduce co-emitted pollutants that lead to cooling (e.g., SO_2, OC). Because many of these co-emitted pollutants lead to climate cooling, the climate benefits of BC emissions control measures may be offset. Therefore, the full mixture of emissions must be considered in estimating the climate benefits of potential mitigation measures. Since these factors are complex and often not well understood, quantitative analysis of the climate benefits of BC mitigation strategies is difficult and the number of related studies is limited.

Of the studies currently available, some have focused on the physical climate benefits of BC mitigation—estimating changes in temperature, ice melt, or radiative forcing. A number of these studies explicitly compare the climate benefits of BC reductions to the climate benefits of reductions in other GHGs. These studies often use metrics such as GWP or GTP (introduced in Chapter 2) as the basis for comparing alternative climate mitigation strategies. Other studies have extended the analysis of climate benefits by attempting to place an economic value on avoided impacts. In a few cases the economic benefits of particular BC mitigation strategies were compared to those of alternative strategies targeting either BC or long-lived GHGs. The next several sections describe (1) the potential physical benefits that can be achieved through BC mitigation, (2) how those benefits compare to benefits that could be achieved through CO_2 mitigation, and (3) the potential value of the climate benefits of BC mitigation.

6.4.1 Studies Estimating Physical Climate Benefits

As discussed in Chapter 2, the nature and distribution of BC and its mechanisms of action mean it can have important direct and indirect effects on climate that differ from those of GHGs. Unlike with GHGs, these effects are not limited to those derived from radiative forcing on a global scale. Rather, the effects associated with BC include alteration of cloud properties, which affects cloud reflectivity, precipitation, and surface dimming. In addition, deposited BC can result in disproportionate warming in areas covered by snow and ice, which is greatest near source regions (e.g. the Himalayas) but still significant in the Arctic.

Most studies on the climate benefits of BC mitigation have focused on estimating the impacts of broad-scale global or regional emissions reductions. Several investigators have used global climate models to examine reductions of BC, OC and in some cases associated GHGs from the fossil, biofuel, and biomass sector sources. Most of these have focused on the effect of global reductions on radiative forcing or temperature. As discussed below, the results generally suggest that the largest climate benefits are likely to accrue from strategies that reduce emissions from BC-rich sectors such as mobile diesel engines and other fossil fuel combustion sources, as opposed to sectors where the quantity of BC compared to co-emitted pollutants is smaller (e.g., biomass burning). Below, we discuss (1) the global climate benefits of BC emissions reductions, (2) benefits of BC reductions specific to ice-covered regions, and (3) key uncertainties in these estimates that are due to insufficient scientific understanding of how aerosols affect climate.

6.4.1.1 Global Climate Benefits of BC Reductions

As discussed previously, some co-emitted pollutants lead to cooling that can counteract the warming by BC. Changes to the entire emissions mixture must therefore be considered when estimating the climate impacts of BC mitigation measures. Several studies have examined the climate impacts of eliminating all emissions from individual sources and found the largest and most consistent benefits in terms of negative forcing (cooling) result from reductions in emissions from fossil fuel sources. For example, Jacobson (2010) found that eliminating all fossil fuel soot reduced surface air temperature by 0.3-0.5 K (13-16% of total net global warming). Another study that evaluated multiple models found that global reductions in open biomass burning (where the emissions mixture typically includes a higher concentration of cooling compounds) produced small but positive climate benefits (Kopp and Mauzerall, 2010).

Several studies have examined the climate benefits of emissions reductions from individual emissions source sectors, again finding that sectors with higher BC ratios generally have larger positive forcing. These studies are generally consistent in finding that transportation and household biofuel combustion contribute more than any other sector to positive forcing. For example, Unger et al. (2010) examined the warming impacts of each major sector's emissions, taking into account the full mixture of aerosols and gases (short-lived and long-lived) from each sector. On-road motor vehicles and household biofuels, major sources of global anthropogenic BC emissions, were found to contribute more than any other sector to globally averaged near-term warming (by 2020) (Figure 6-4). Koch et al. (2007a)

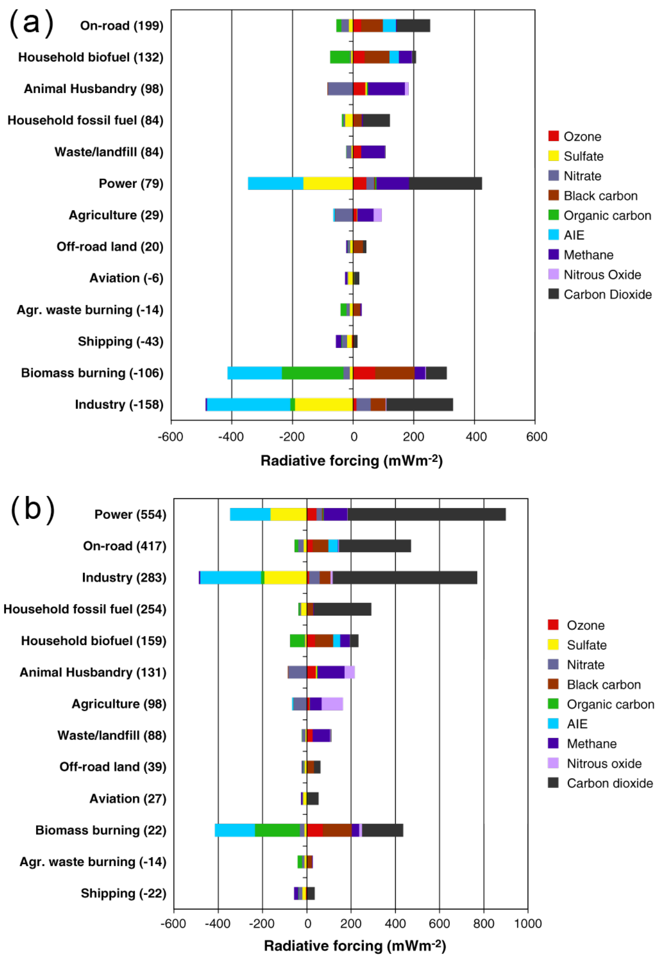

Figure 6-4. Global Radiative Forcing Due to Perpetual Constant Year 2000 Emissions, Grouped by Sector, in 2020 (Top) and 2100 (Bottom) showing contribution from each species. The sum is shown on the title of each bar, with a positive radiative forcing indicating that removal of this emissions source will result in cooling. From Unger et al. (2010) (also shown in Chapter 2).

examined the direct radiative forcing of aerosols only, finding that the sectors that are responsible for the largest BC radiative forcing are residential (0.09 W m^{-2}) and transport (0.06 W m^{-2}), but that co-emitted scattering (i.e., cooling) components reduce these impacts to 0.04 W m^{-2} and 0.03 W m^{-2}, respectively. Shindell et al. (2008a) found that across-the-board emission reductions in household fuel burning in Asia and in transportation in North America are likely to offer the greatest potential for near-term climate benefits. Although these studies were limited to direct radiative effects, Bauer et al. (2010) also found that reducing diesel emissions would reduce positive forcing (i.e., warming) even when accounting for cloud changes, which can result in cooling. In another study examining the effects of all aerosols, Bauer and Menon (2012) reached similar conclusions. This study focused on regional differences in the impact of emissions from different source categories, and concluded that the largest opportunities to reduce positive forcing due to all aerosols included transportation in all regions, agricultural burning in Europe and Asia, and residential cooking and heating ("domestic sector") in Asia.

This body of literature suggests that both transportation and residential BC sources may be attractive targets for BC mitigation measures. Although several of these studies found that reductions in industrial and power generation emissions may accelerate near-term warming (Unger et al., 2009; Shindell et al., 2008a), this broad categorization includes a variety of different types of sources, some of which are major emitters of SO_2, a precursor of sulfate (a "cooling" aerosol). Within the industrial sector, however, are some sources (e.g., brick kilns, coke ovens) that are major emitters of BC in the developing world. Controlling emissions from these specific BC-rich sources will likely also lead to climate benefits, as discussed below. As with any strategy development, determining the specific optimal measures to implement depends on a number of factors in addition to the climate and public health benefits. These factors are discussed in more detail in Chapter 7.

Several recent studies have examined how specific emission control measures are expected to reduce emissions. To date, the most comprehensive assessment of this type is the UNEP/WMO Assessment described in Section 6.3.2 (UNEP and WMO, 2011a; Shindell et al., 2012). This study found that implementing an illustrative set of BC and methane (CH_4) emission control measures together would reduce future global warming by 0.5°C (0.2°C - 0.7°C), with about half the reduction specifically from the BC measures. Implementing the BC and CH_4 emission control measures by 2030 was estimated to halve the expected increase in temperatures for 2050 compared with the reference scenario (based on current policies and energy and fuel projections), as shown in Figure 6-5. This study used a range of values from the literature reflecting the indirect and direct radiative forcing effects of BC and OC to provide a range of expected outcomes that account for uncertainty. A follow-on study to the UNEP/WMO Assessment found that the measure likely to produce the greatest near-term global climate benefit is switching from traditional biomass cookstoves to cleaner burning stoves, followed by reducing emissions from the transportation sector (UNEP, 2011).

Because transportation emissions are expected to contribute the most to BC-related warming in the future (Koch et al., 2007b), Shindell et al. (2011) examined the climate benefits of imposing tighter vehicle emission standards in China, India, Africa, the Middle East, and Latin America. Relative to no additional controls, imposing these standards led to significant reduction in warming in the Northern Hemisphere extra-tropical region (reduction of 0.22°C, with a potential range from 0.04°C to 0.38°C) and in the Arctic (reduction of 0.28°C, with a potential range from 0.02°C to 0.47°C) over the next 50 years, although the total reduction in global warming after 2040 overall was small. Controlling emissions from heavy-duty diesel trucks in India and Brazil was found to have the greatest climate benefits, followed by controls on medium-duty diesel vehicles in India and light-duty petrol vehicles in North Africa and the Middle East. Controlling emissions from light-duty gasoline vehicles everywhere and from motorcycles and medium-duty trucks in some regions also provides climate benefits that are more limited. These BC reductions were also associated with precipitation changes, but such results are highly uncertain and warrant further study. Despite large uncertainties, this study demonstrates the substantial climate benefits of controlling emissions from the motor vehicle sector around the world.

Few studies have examined the climate benefits of specific particle control programs on smaller, more localized scales. A recent study of particular relevance examined the results from California's laws to reduce particle pollution, in particular those regulating diesel emissions. The study found that these rules reduced atmospheric concentrations of BC with a measurable impact on regional radiative forcing. Modeled results indicate that the decrease in BC emissions in California has led to a cooling

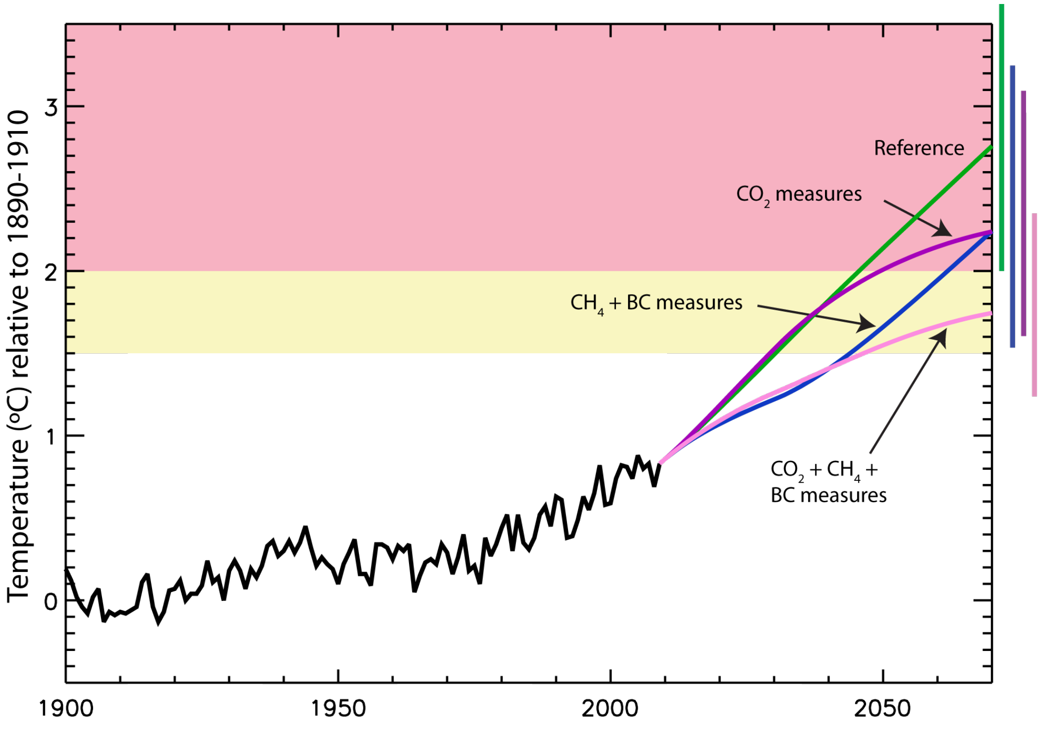

Figure 6-5. Observed Deviation of Temperature to 2009 and Projections under Various Scenarios. Immediate implementation of the identified BC and CH$_4$ measures, together with measures to reduce CO$_2$ emissions, would greatly improve the chances of keeping the Earth's temperature increase to less than 2°C relative to pre-industrial levels. The bulk of the benefits of CH$_4$ and BC measures are realized by 2040. (UNEP and WMO, 2011a)

Note: Actual mean temperature observations through 2009, and projected under various scenarios thereafter, are shown relative to the 1890–1910 mean temperature. Estimated ranges for 2070 are shown in the bars on the right. A portion of the uncertainty is common to all scenarios, so that overlapping ranges do not mean there is no difference. For example, if climate sensitivity is large, it is large regardless of the scenario, so temperatures in all scenarios would be towards the high end of their ranges.

of 1.4 W m^{-2} (±60%) (Bahadur et al., 2011). So, while uncertainties remain, as outlined in previous chapters, emerging research suggests that targeting emissions reductions from key sectors can have measurable benefits for climate.

6.4.1.2 Benefits of BC Reductions on Snow- and Ice-Covered Regions

As noted above, BCs effects are highly regionalized and include impacts on precipitation, atmospheric stability, and snow/ice melting that differ in important ways from those driven by GHGs. Global modeling provides useful insights into potential responses in regions identified in Chapter 2 as being particularly affected by BC emissions. For example, Jacobson (2010) found the extreme strategy of eliminating all anthropogenic emissions from sources of fossil fuel and biofuel BC would reduce global temperatures by 0.4 to 0.7° C; with a reduction of about 1.7° C in the Arctic. This is consistent with other modeling and analysis discussed in Chapter 2 that suggest a larger impact of BC and other pollutants on the Arctic, and thus greater potential benefits from emissions control measures.

Recent findings of the Arctic Council Task Force on Short-Lived Climate Forcers suggest that mitigating sources of BC emissions in or near the Arctic will have greater climate benefits in that region, with important seasonal and spatial variations. Impacts in the Arctic are greatest during the spring and summer months when the solar radiation is the strongest. Specifically, in its 2011 *Progress Report and Recommendations for Ministers* (Arctic Council, 2011), the Task Force noted that:

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

The Task Force highlighted the importance of reducing emissions from in-Arctic sectors such as land-based transportation, open biomass burning, residential heating, and marine shipping in the Arctic, but also noted that emissions from outside the Arctic, especially those in close proximity to the Arctic, are important for Arctic climate change, partly because of the volume of these emissions and (as noted above) because of the relatively small cooling effect of co-emitted pollutants in the region.

The 2011 Arctic Monitoring and Assessment Programme (AMAP) report, "The Impact of Black Carbon on the Arctic" examined the influence of BC emitted from different sources and world regions on radiative forcing and temperature in the Arctic (Quinn et al., 2011). This study found that in general, BC *deposition* on snow and ice in the Arctic (which contributes to the snow/ice albedo effect described in section 2.6) exerts a greater warming effect than within-Arctic direct radiative forcing from BC in the atmosphere. Furthermore, both direct forcing and snow/ice albedo forcing in the Arctic *per unit of emissions* were shown to be highest for emissions originating near to or within the Arctic region, with

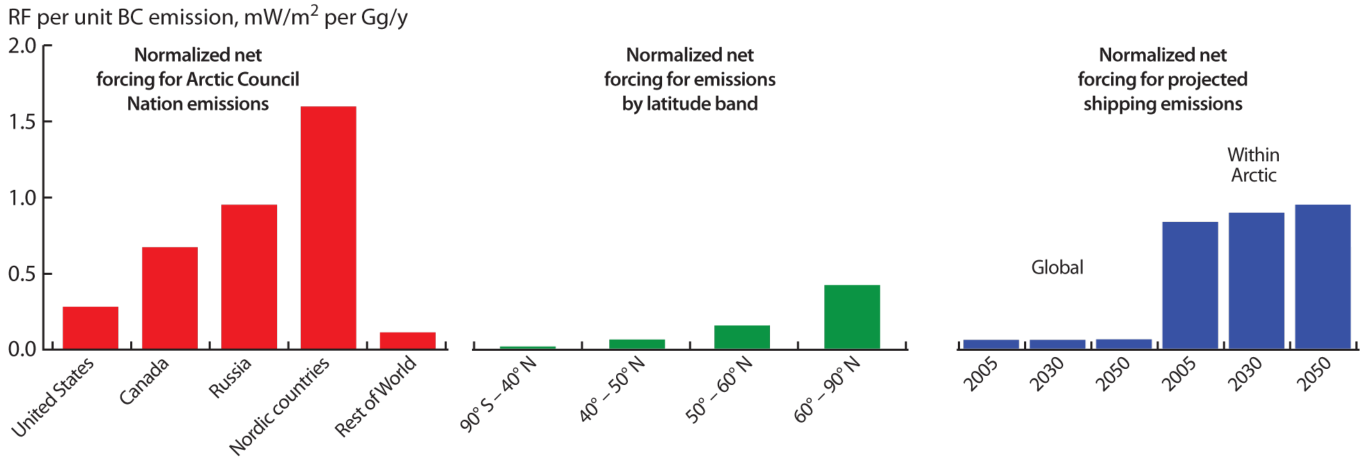

Figure 6-6. Summary of Normalized Net Forcing per Unit of Emissions (includes atmospheric direct forcing by BC and BC-snow/ice forcing) due to emissions from Arctic Council nations and the rest of the world, the indicated latitude bands, and global and within-Arctic shipping. (Source: Quinn et al., 2011)

the Nordic countries (i.e., highest latitudes) having the highest forcings per unit of emissions, followed by Russia, Canada, and the United States (Quinn et al., 2011, p. 95). These impacts are illustrated in Figure 6-6. This is because within-Arctic BC sources are more likely to cause warming near the surface and to lead to BC deposition on snow and ice surfaces. Furthermore, warming at high altitudes reduces atmospheric energy transport into the Arctic; as a result, AMAP (Quinn et al., 2011) found that direct atmospheric forcing by BC that has been transported into the Arctic at high altitudes may have a relatively small impact on Arctic surface temperatures.

However, the Arctic climate is coupled with that of the Northern Hemisphere and is thus sensitive to changes in radiative forcings in other nearby regions. Since the bulk of global BC emissions occur outside the Arctic, and since global BC forcing results in poleward transfer of heat energy, Arctic temperatures are significantly affected by BC direct forcing occurring outside the Arctic region. Overall, when total global emissions are considered, BC emissions in the rest of the world are the dominant influence on radiative forcing in the Arctic (Figure 6-7). The AMAP results confirm both that latitude and the total magnitude of emissions matter:

> *It has been suggested that emissions north of 40°N have a large impact on the Arctic particularly in the winter and spring when the polar dome extends to the mid-latitudes over Europe and Asia …. To test this assumption and to compare the potential impact of sources on Arctic climate as a function of latitude between 40°N and 90°N, a set of experiments was performed with emissions gridded by latitude band. The latitude bands included in the analysis were 90°S to 40°N, 40°N to 50°N, 50°N to 60°N, and 60°N to 90°N. … [E]missions in the most southerly latitude band (90°S to 40°N) result in the largest direct RF due to the magnitude of the emissions of BC in the northern hemisphere tropics and mid-latitudes. Atmospheric direct RF also is relatively large for the 40°N to 60°N latitude band because of the magnitude of emissions and likelihood of transport to the Arctic. Emissions within the Arctic (60°N to 90°N) result in a smaller absolute direct RF because of their lower magnitude compared to emissions in more southerly latitude bands. Absolute BC-snow/ice RF increases with latitude between 40°N and 60°N. This result confirms that emissions from lower latitudes are less effectively deposited in the Arctic since they reach the Arctic at higher altitudes …. Normalized BC-snow/ice RF increases dramatically with increasing latitude band … confirming the efficiency with which sources close to the Arctic are transported to and deposited within the Arctic. This result also indicates that per unit emission, sources within the Arctic yield the largest RF. (Quinn et al., 2011, p. 57)*

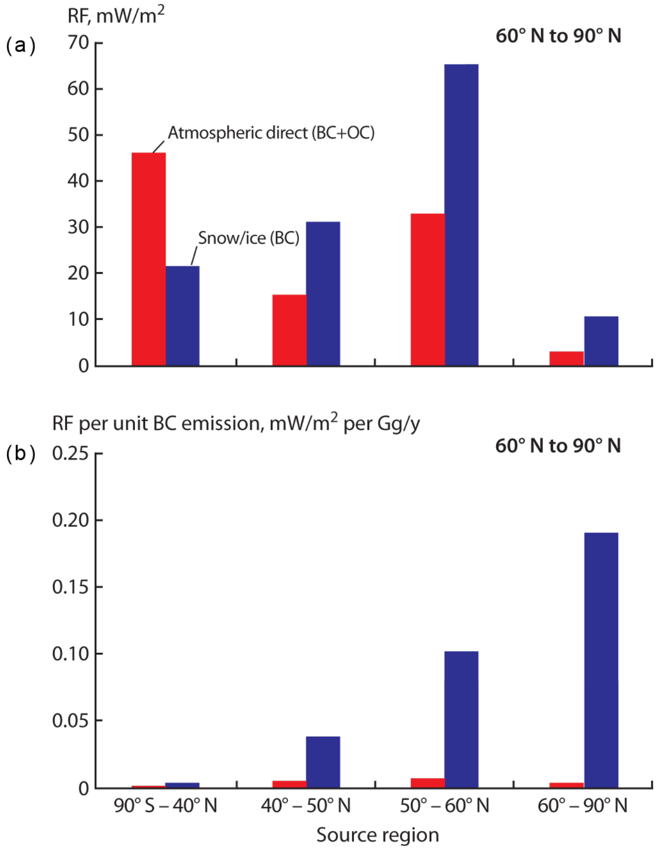

Figure 6-7. Contribution to Radiative Forcing of Carbonaceous Aerosol Emissions within Different Latitude Bands. (a) Absolute and (b) Normalized per unit emission atmospheric direct radiative forcing due to BC+OC and BC-snow/ice radiative forcing as a function of latitude band. (Quinn et al., 2011, Figure 8-9)

Importantly, AMAP (Quinn et al., 2011) indicates that reductions in OC emissions will provide benefits in the Arctic too, since OC emissions appear to exert positive radiative forcing over snow- and ice-covered surfaces. Note that the radiative forcings reported in Figure 6-7 are for the combination of OC+BC emissions. Similarly, sulfate aerosols, which normally exert a cooling influence, appear to have a much weaker cooling effect over snow and ice (Quinn et al., 2011). While aerosol indirect forcing effects in the Arctic are highly uncertain, current evidence suggests that indirect and semi-direct effects are less negative in the Arctic than for the global average. This all means that a wide array of measures aimed at reducing BC and OC in or near the Arctic will provide benefits to the Arctic region, as will BC-focused strategies elsewhere. AMAP (Quinn et al., 2011) concluded that global strategies to manage BC emissions are necessary to mitigate Arctic climate change.

There is relatively less information regarding specific benefits of BC reductions for other snow- and ice-covered regions, including the Himalayas. As discussed in section 2.6.5, Menon et al. (2010) modeled the impacts of estimated increases in BC between 1990 and 2000 and found that in particular BC from fossil fuel/biofuel use in India may be responsible for some of the observed patterns and trends in snow and ice melting and precipitation in the Himalayan region. Such changes may have significant implications for water supply in the region. While a number of studies have suggested BC and associated emissions may play a role in reduced monsoon rains, current modeling capabilities do not provide a basis for reliable quantitative assessments of the extent to which emissions reductions might reverse observed changes in precipitation.

6.4.1.3 Key Uncertainties in Estimating Climate Benefits of BC Reductions

As discussed in detail in Chapter 2, the results from the studies described in the previous subsections have some level of uncertainty. The primary sources of this uncertainty include lack of understanding about:

- The climate effects of reductions in co-emitted pollutants, especially brown carbon (BrC) emissions and the extent to which they contribute to radiation absorption (Magi, 2009).
- The effect of model representation of aerosol mixing state and aerosol-cloud interactions, including radiative effects and precipitation effects, on estimated climate impacts. These processes can have a major influence on the overall warming or cooling effect of emissions changes.
- Effects of non-BC aerosols in the Arctic.
- Effects of other atmospheric processes (such as atmospheric transport and deposition) on climate outcomes.

In addition, errors in the emissions inventories of BC and OC and other reflecting agents from each sector, particularly for residential solid fuel combustion, may lead to over or under estimation of the magnitude of the climate impact.

6.4.2 Comparing Climate Benefits of Reductions in BC vs. CO_2

While studies performed to date do not include the full set of aerosol interaction effects, co-emissions, or other uncertainties, taken as a whole they suggest that reductions of BC may be among the most effective strategies for reducing near-term warming, and can *complement* GHG reductions as part of an overall climate strategy (Grieshop et al., 2009; Kopp and Mauzerall, 2010). As described in Chapter 2, BC reductions can reduce the rate of climate change and provide climate benefits in the near term. However, BC reductions today have much smaller effects on temperatures in 100 years. Therefore, BC emissions reductions cannot *substitute* for CO_2 reductions for purposes of alleviating long-term warming. Studies indicate that BC emissions reductions that come at the expense of reductions in CO_2 emissions would result in short-term cooling but add an additional commitment to long-term radiative forcing due to the life time of CO_2 in the atmosphere (Lack et al., 2009).

The UNEP/WMO BC and Tropospheric Ozone Assessment (UNEP and WMO, 2011a) described in sections 6.3.2 and 6.4.1 compared the climate benefits of groups of BC and CH_4 mitigation measures to a scenario developed by the International Energy Agency in which long-lived GHG concentrations were reduced to a level of 450 ppm CO_2eq (International Energy Agency, 2009). As illustrated by Figure 6-5, the reductions in CH_4 and BC combine to produce a noticeable impact on near-term warming as compared to the reference case or CO_2 measures by themselves. The analysis showed that even aggressive CO_2 reductions may not keep climate change from approaching 2°C by mid-century.[6] At the same time, it is important to note that the benefits of reducing BC and CH_4 are insufficient to avert warming over the long term. Reducing short-lived climate forcers now, while neglecting to achieve aggressive CO_2 reductions, may not keep temperatures from reaching the 2°C mark in 2070 and beyond. These results, and those from other studies on the temporal aspects of reducing BC and other short-lived forcers, underscore the scientific rationale for reducing long-lived GHGs and BC simultaneously as two distinct, complementary strategies that act on different time scales to address global warming and other effects of climate change.

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[6] An increase in global mean temperatures of 2^{o}C since preindustrial times was adopted as an international target under the UN's Copenhagen Accord in December 2009.

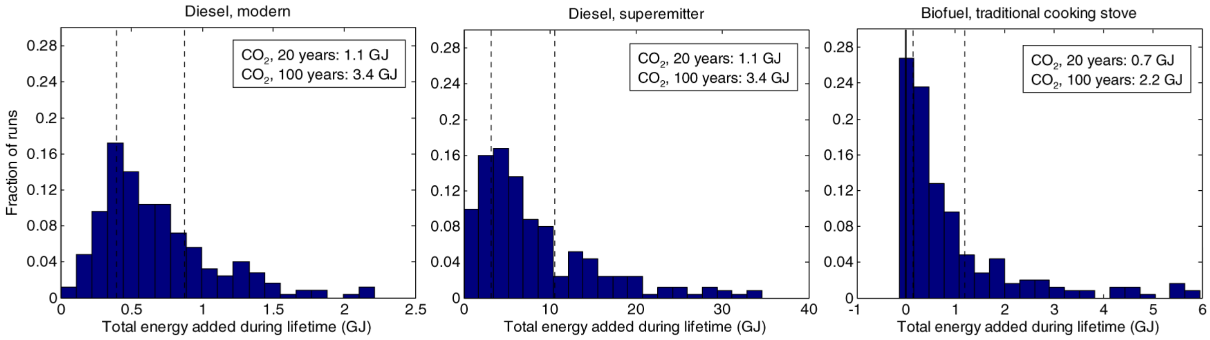

Figure 6-8. Integrated Forcing by Aerosols Emitted from Burning 1 Kg of Fuel from Different Sources, based on results of 250 Monte Carlo simulations. (Note scale differences.) (Bond, 2007, Figure 6)

Among the other studies comparing BC and CO$_2$ strategies, Grieshop et al. (2009) used a valuation that "one ton of black carbon causes about 600 times the warming of one ton of carbon dioxide over a period of 100 years" to state that eliminating present-day emissions of BC over the next 50 years would have "an approximately equivalent climate mitigation effect to removing 25 Gt C from the atmosphere over the same period."[7]

Some studies also account for changes in co-emissions from BC and GHGs that may affect climate outcomes. For BC, the co-emissions (e.g. SO$_2$, OC) are often reduced by the same control measures that affect BC and the climate impacts are generally realized on the same timescale as (i.e., short atmospheric lifetimes lead to near-term climate effects). For CO$_2$, there is a gap between climate effects due to CO$_2$ and effects due to co-emitted species, since the latter generally have a shorter lifetime. Furthermore, with CO$_2$, it is often possible to reduce co-emissions separately with end-of-pipe technologies, which makes it possible to make independent decisions about how much to reduce CO$_2$ vs. co-emissions from a given source. All of this illustrates the importance of accounting for co-emissions in climate models, but also the complexity of assessing the impact of specific control measures.

Other studies have indicated that implementing aerosol mitigation measures for BC-rich sources can yield more cooling over the short term (10-20 years) than eliminating CO$_2$ emissions from those sources (Bond, 2007; Jacobson, 2005; Sarofim, 2010). Sarofim (2010), for example, addressed one specific mitigation option (retrofits of some U.S. diesel vehicles) and showed that the CO$_2$ equivalent reductions calculated by using a GWP would lead to radiative forcing reduction from black carbon mitigation peaking in the year that the vehicles are retrofitted and dropping to almost zero change in global radiative forcing after 20 years as the retrofitted vehicles are retired. In contrast, the radiative forcing reduction from the CO$_2$ equivalent mitigation calculated using GWPs peaks about a decade after the start of the mitigation period at only a tenth of the BC peak, but at the end of the century the radiative forcing reduction is still more than half of what it was at that peak.

Bond (2007) examined emissions from multiple source types and compared the total radiative (integrated) forcing from those sources over 20 years for carbonaceous aerosols (both OC and BC) to the integrated forcing from CO$_2$ (an approach similar conceptually to using GWP weightings). The study showed that the aerosol emissions resulting from burning 1 kg of fuel in a super-emitting diesel vehicle has more than a 90% chance of contributing more total forcing than CO$_2$ from that source over a 20 year timeframe, and even for a normal (pre-2007) diesel, the aerosol emissions resulting from burning 1 kg of fuel are likely to contribute more than half as much warming as the CO$_2$ emissions over 20 years (see Figure 6-8). This study did not account for the indirect effects of aerosols or snow albedo effects. Jacobson (2005) did include co-emissions and more cloud interactions, and still found that diesel vehicles warmed climate more than gasoline vehicles for 13-54 years, because the higher BC emissions from diesel vehicles outweighed the lower CO$_2$ emissions over that timeframe.

A different approach avoids the limitations of choosing a single metric to compare emissions of

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[7] In this study, 25 Gt was chosen because it equals one "wedge" from the Pacala and Socolow (2004) study that identified large-scale mitigation options over the next 50 years. However, Grieshop et al. (2009) did not involve any calculations to compare the short-term and long-term effects of implementing a BC wedge rather than an additional greenhouse gas wedge, did not examine co-emissions, and did not take into account cloud interaction effects.

BC and CO_2, and instead investigates how reductions of BC over the entire century would change the difficulty of meeting radiative forcing targets. Kopp and Mauzerall (2010) calculated the optimal CO_2 emissions pathways in order to meet a 2.21 W m^{-2} target in 2100. Rather than assessing the benefits of BC reductions in the near future like the previous studies, this study assessed the radiative benefits of BC reductions at the end of the century and then translated those benefits into near term CO_2 emissions targets. This study included both co-emissions and an estimate of indirect effects. They found that meeting this target required 50% reductions of CO_2 by about 2050. However, if this target were tightened to accommodate the positive radiative forcing from carbonaceous aerosols (both OC and BC) from contained combustion source (fossil fuels and biofuels), then the 50% reduction of CO_2 would need to occur 1 to 15 years earlier, depending on the assumptions about carbonaceous aerosol emissions pathways and forcing strength.

6.4.3 Valuing the Climate Benefits of BC Mitigation

Another way to evaluate the benefits of BC mitigation strategies and to compare them with the benefits of other climate mitigation strategies is to use valuation techniques to create monetary estimates of avoided damages. This would be equivalent to the approach adopted to compare the health benefits of different regulatory approaches discussed above in section 6.3. However, methods for establishing the economic value of the climate damages associated with BC are still being developed. Two metrics, the Global Damage Potential (GDP) and the social cost of a pollutant, involve monetization of the damages of climate change. Assessing the value of damages through a single metric (i.e., dollars) provides useful information that can help inform policymakers regarding the scale and scope of the climate impacts of BC and the benefits that can be gained from BC mitigation. However, no study to date has fully monetized the climate impacts of BC. An analysis of this type would need to include the benefit of avoiding risks and impacts associated with warming (especially near term warming and rate of change),as well as the value of avoiding impacts such as accelerated ice and snow melt and changes in precipitation induced by BC.

Currently, efforts to develop valuation methods for climate impacts have focused on CO_2. In computing the value to society of avoided climate damages, EPA assigns a benefits dollar value to CO_2 emission reductions using estimates of a "social cost of carbon" (SCC) developed by a U.S. federal government interagency working group in 2010. The SCC is an estimate of the monetized damages resulting from an incremental increase in CO_2 emissions in a given year; likewise, it can be thought of as the monetized benefit to society of reducing one ton of CO_2. The SCC estimates are intended to include an array of human-induced climate change impacts, such as changes in net agricultural productivity, human health, property damages from increased flood risk, and the value of ecosystem services due to climate change. Current SCC values, such as those utilized by EPA to analyze the benefits of the 2010 *Light-Duty Vehicle Greenhouse Gas Emission Standards and Corporate Average Fuel Economy Standards* (U.S. EPA, 2010c), are subject to a number of limitations, including the incomplete way in which the underlying climate models capture catastrophic and non-catastrophic impacts, the incomplete treatment of adaptation and technological change, uncertainty in the extrapolation of damages to high temperatures, and assumptions regarding risk aversion. The SCC estimates developed for CO_2 have been controversial due to the difficulty of estimating economic impacts across nearly every sector of the economy as well as valuation issues regarding impacts on natural ecosystems. Furthermore, these estimates were developed exclusively for CO_2 and are not directly transferrable to other GHGs or BC.[8]

It might be possible to use a similar approach to develop a social cost specific to BC using integrated assessment models (IAMs) that combine economic growth, climate processes, and feedbacks between the global economy and climate into a single framework to translate BC emissions into economic

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[8] One approach that might appear tempting is to use existing estimates for the SCC for CO_2, 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 CO_2 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 2.) 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 CO_2-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-CO_2 forcers (Interagency SCC Group, 2010), though it is unclear how large such an error would be.

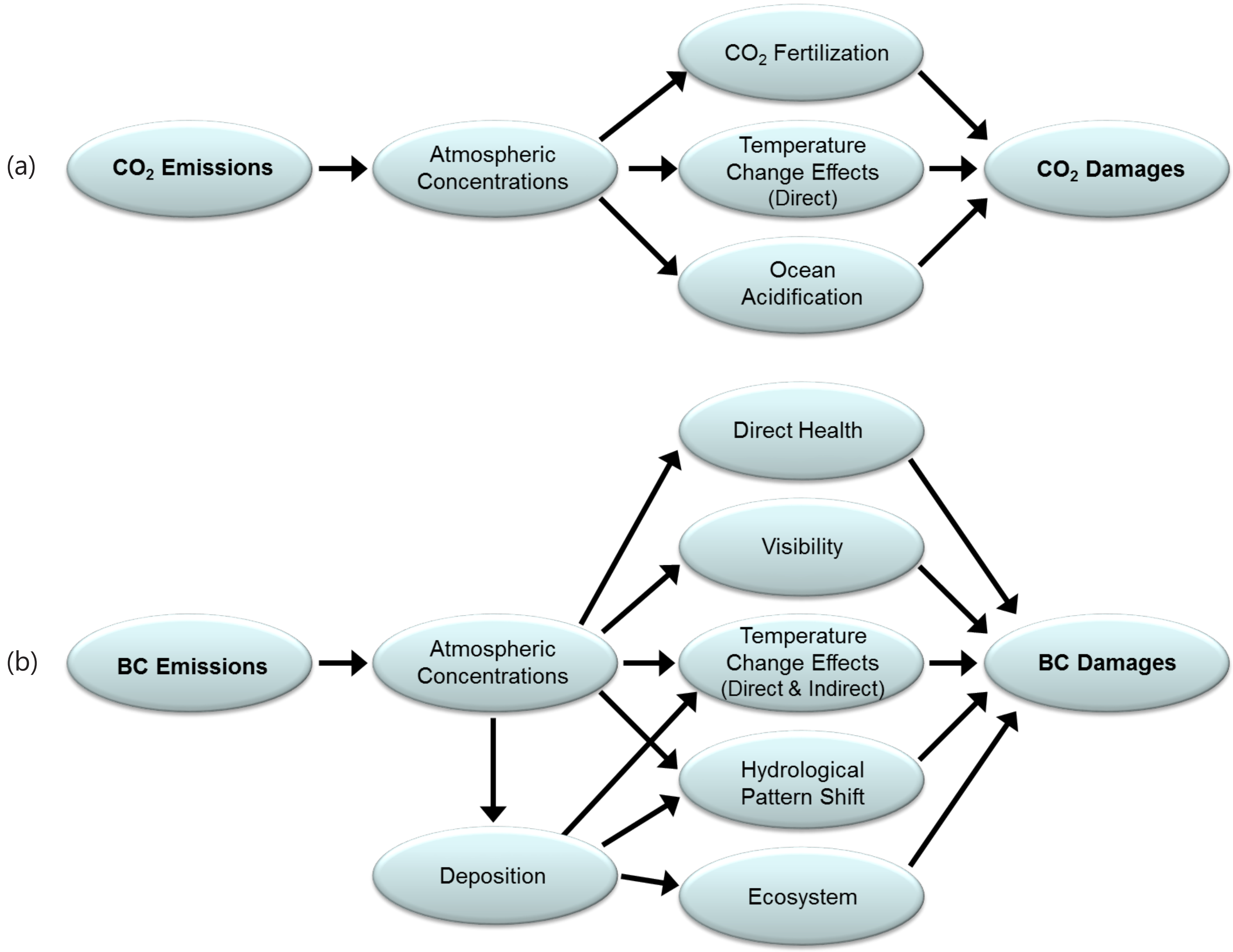

Figure 6-9. Cause and Effect Chains for (a) CO_2 and (b) BC from Emissions to Damages. (Source: U.S. EPA)

damages. However, there are a number of factors that would complicate these calculations. These difficulties stem partly from limitations in the capabilities of IAMs, and partly from the complexity of the cause-effect chain needed to measure the physical links between emissions and climate change impacts, and to calculate damages (see Figure 6-9).

Few IAMs are designed to demonstrate regional impacts, and currently these models do not adequately consider the impact of BC and other short-lived climate forcers on the *rate* of climate change. In addition, the feasibility of considering indirect climate effects such as the impact of BC on snow and glacier melt and changes in precipitation patterns in the IAMs must be evaluated. In one aspect, at least, calculating a social cost for BC might actually be easier than calculating a social cost for CO_2: the short lifetime of BC and the relatively immediate nature of the climate impacts reduce the extent to which social cost calculations would depend on the social discount rate selected. Due to the complexities involved with valuing the climate benefits of BC reductions, a top-down approach using IAMs may not be preferred. Rather, a bottom-up approach that considers location, emission profiles of sources, and ambient concentrations and deposition of BC similar to the approaches used to quantify health effects may be needed.

The cause-effect chain from emissions to impacts and damages is also complex for BC. The regional nature of many BC impacts, the importance of location of emissions, and BC's impacts on precipitation, snow/ice, and surface dimming add additional complexities to any such approach that are not present for CO_2 SCC calculations. In addition, the peer reviewed literature lacks impact functions and valuation methods necessary to assess many of these BC effects. Finally, because BC is emitted as part of a mixture, incorporation of the climate impacts of reducing other co-emitted aerosols into a social cost approach would reflect the net impact more accurately.

The Global Damage Potential (GDP) compares the relative damage resulting from an equal mass of emissions of two climate forcers (IPCC, 2009) – effectively, the ratio of the social cost of a climate forcer to the social cost of carbon. The GDP is meant to be a parallel approach to the GWP, and therefore might potentially be calculated using some of the same assumptions that go into the GWP calculation. One key assumption is that the background concentration is a constant, modified only by the initial pulse of emissions. This simplifies the calculation of the GWP or GDP because it requires no assumptions about a reference emissions pathway. However, because the impact of increased concentrations depends on the starting concentration, this simplification means that the metric may not accurately reflect actual damages.

As this discussion of GDP and SCC illustrates, calculating economic damages associated with specific climate forcers is extremely complicated. Even where risks and impacts can be identified and/or quantified with physical metrics, it may be difficult to monetize these risks and impacts (e.g., such as ecosystem damage or the potential to increase the probability of an extreme weather event) such that an accurate cost-benefit comparison could be undertaken. Both the GDP and the social cost calculation depend on the physical aspects of the climate system as well as the economic linkages between climate change impacts and the economy (IPCC, 2009). Therefore, the GDP and the social cost require calculations of the entire cause and effect chain, but as a result contain a large amount of uncertainty. Additional work is needed to design approaches to valuing climate impacts of BC directly, and to incorporate those approaches into metrics comparable to the SCC.

Some authors have attempted to incorporate economic valuation approaches into a comparative framework that enables direct comparisons between the benefits of BC mitigation and the benefits of CO_2 mitigation. If fully developed, such approaches could be utilized to help policymakers choose among an array of mitigation choices involving different pollutants, different sources, and different timeframes. Using a computer model that included economic considerations, Manne and Richels (2001) examined relative tradeoffs between different gases that vary over time and are calculated to optimally achieve a given target. The paper demonstrated that if a long-term temperature stabilization target is the only policy goal, then reductions of short-lived gases have little value compared to long-lived gases as long as the target will not be reached for several decades, but that the value of these short-lived gases rises rapidly as the temperature approaches the target. Manne and Richels also examined a case in which the rate of change of temperature was a goal along with the long-term temperature change, finding that in that case the relative prices of the different gases stay more constant over time. This kind of approach, including economic considerations for cost of control but without looking at the benefits of those controls, is known as a cost-effectiveness analysis. The relative tradeoff between a given gas and CO_2 is also known as the Global Cost Potential (GCP).

6.4.4 Conclusions Regarding Climate Benefits

The climate benefits of BC mitigation are less well understood and less certain than the health benefits. Studies examining across-the-board emissions reductions from individual sectors find that the warming impact of source sectors generally corresponds with the OC/BC ratio of that sector, with the benefits from source sectors that have low OC/BC ratios (e.g., fossil fuel-based sectors) being higher.[9] Most studies are consistent in finding that the transportation sector contributes the most to positive radiative forcing, followed by household biofuels. Industrial sources are often found to have a net cooling impact; however, this broad categorization neglects to highlight several sub-sector sources that are major BC emitters, including brick kilns and coke ovens in developing countries. The magnitude of the results from studies estimating the climate benefits of emissions reductions are often uncertain due to uncertainty regarding co-emitted species, indirect effects, and effects on precipitation. However, the studies described above demonstrate that several specific and presently available emissions control measures are likely to have substantial climate benefits. These include emissions control measures for vehicles, residential burning of solid fuel, and major industrial sources of BC including brick kilns and coke ovens. The available literature also strongly suggests that BC mitigation can provide particular benefits to sensitive regions, including the Arctic and the Himalayas. These regions stand to benefit disproportionately from reductions in BC, especially if reductions can be achieved from sources within the regions themselves.

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[9] Bond et al. (2011) found that for direct forcing only, a ratio of about 15:1 for OC to BC is close to climate neutral.

Comparing the climate benefits of BC and CO_2 mitigation is complicated by the many differences in lifetime and mechanisms of impact between these climate forcers. BC reductions can reduce the rate of climate change and provide climate benefits in the near term, but cannot *substitute* for CO_2 reductions for purposes of alleviating long-term warming. Thus, controlling both short-lived forcers and long-lived greenhouse gases is necessary to achieve the target of constraining temperature rise to no more than 2°C as agreed upon by the international community.

Several metrics have been suggested to value the climate benefits of BC mitigation in economic terms. Assessing the value of damages through a single metric (i.e., dollars) provides useful information that can help inform policymakers regarding the scale and scope of the climate impacts of BC and the benefits that can be gained from BC mitigation. However, no study to date has fully monetized the climate impacts of BC, and a great deal of additional work is needed to design approaches for doing so.

6.5 Environmental Benefits of BC Reductions

In addition to health and climate benefits, there are additional environmental benefits related to reductions in $PM_{2.5}$ including BC. While EPA has had some success in quantifying and valuing benefits from improved visibility, other important impacts such as ecosystem effects and damage to building materials are not easily quantified. In general, however, the environmental benefits of reducing $PM_{2.5}$ have been shown to be quite large in the United States. Importantly, the majority of environmental benefits globally are likely to accrue to other countries, i.e., those investing in emission reduction programs. This is particularly true for those areas where ambient $PM_{2.5}$ is interfering with rainfall patterns (discussed above, in the section on climate impacts) or causing surface dimming on a broad scale. The next sections describe what is known about how $PM_{2.5}$ reductions can (1) improve visibility, (2) reduce pollutant impacts on ecosystems, and (3) enhance the longevity of building materials.

6.5.1 Visibility Impacts

Visibility impairment is caused by the scattering and absorption of light by suspended particles and gases in the atmosphere. A number of other factors can influence visibility, such as the relative atmospheric humidity, intensity of sunlight, presence of cloud cover, distance from the object being viewed, physical characteristics of the object being viewed, and physical capabilities (i.e., eyesight) of the viewer (Malm, 1999). However, when $PM_{2.5}$ is present in the air, its contribution to visibility impairment typically greatly exceeds that of naturally occurring atmospheric gases (U.S. EPA, 2011d). As a result, in otherwise constant conditions, visibility impairment is greater when PM is present. Reductions in air pollution from implementation of various programs associated with the Clean Air Act (CAA) Amendments of 1990 provisions have resulted in substantial improvements in visibility, and will continue to do so in the future.

Visibility directly affects people's enjoyment in a variety of daily activities and their overall sense of well-being. Individuals value visibility both in the places they live and work, in the places they travel to for recreational purposes, and at sites of unique public value, such as national parks. Visibility economic benefits consist of the aesthetic benefits of better visibility, improved road and air safety, and enhanced recreational activities like hunting and bird watching. As with health benefits, visibility improvements are valued using WTP studies. EPA estimates that in 2010, improvements in visibility related to the 1990 CAA Amendments were valued at approximately $36 billion annually (2010$) (U.S. EPA, 2011c). Almost three-quarters of these benefits ($27 million) result from residential visibility improvements. It is important to note that residential benefits in this EPA study reflect benefits from all metropolitan statistical areas in the country, whereas recreational benefits are limited to visibility improvements in Class I areas managed by the National Park Service in California, the Southeast and Southwest.

6.5.2 Ecosystem Impacts

Ecosystems perform a number of functions that contribute to human welfare, including the provision of food and raw materials, filtering of air and water, and protection from natural hazards such as floods (U.S. EPA, 2011d). Additionally, people may seek out certain ecosystems for their aesthetic value. Atmospheric $PM_{2.5}$ negatively affects the ability of ecosystems to perform these and other valuable welfare functions. $PM_{2.5}$ can impact ecosystems through direct deposition on plants, animals, or bodies of water. In areas with high emissions, $PM_{2.5}$ deposition on leaves interferes with a plant's ability to perform basic metabolic functions. Increases in trace metal and organic matter in bodies of water as the result of $PM_{2.5}$ deposition are toxic for aquatic life forms (U.S. EPA, 2009b). Indirect impacts are caused when plants and animals take in pollutants through affected soil or water. As with direct impacts, indirect impacts can alter normal biological processes and be toxic to living organisms.

The relationship between $PM_{2.5}$ and ecosystem effects is difficult to quantify because of the variability in $PM_{2.5}$ emissions and composition. Pollutants accumulate over time in affected organisms, making it difficult to link impacts to ambient $PM_{2.5}$ concentrations. Additionally, the impact of negative welfare effects may vary based with geographic location. The impact of damage to a national park would likely be valued differently than damage to commercial farm land, which would in turn be valued differently than damage to private non-commercial property (U.S. EPA, 2011d).

6.5.3 Materials Co-benefits

$PM_{2.5}$ deposition on materials such as stone, metal, and painted surfaces leads to damage by accelerating the natural weathering process. Chemical reactions with acidic gases worsen the impact of $PM_{2.5}$ -related damage. Additionally, accumulation of $PM_{2.5}$ on surfaces, referred to as soiling, affects the aesthetic properties of materials and necessitates more frequent cleaning or repainting of affected surfaces. Research has not established any quantitative relationship between the ambient concentrations of $PM_{2.5}$ and the rate of damage or soiling caused by $PM_{2.5}$ deposition (U.S. EPA, 2011d).

6.5.4 Conclusions Regarding Environmental Benefits

The environmental benefits of reducing BC are likely to be substantial, both in terms of the range of impacts avoided and the value to society, although it is difficult to quantify these impacts currently. Due to the difficulties involved in quantifying and valuing environmental benefits, EPA often addresses these benefits qualitatively with the exception of visibility benefits as previously discussed.

6.6 Conclusions

All control measures that reduce $PM_{2.5}$ pollution are virtually certain to achieve health benefits, and several studies examining costs and benefits of BC mitigation suggest that the health benefits alone may justify mitigation. Programs to reduce fine particles in the United States and in other developed countries have greatly reduced the negative health impacts of $PM_{2.5}$, including BC. EPA has determined that there is insufficient information at present to differentiate the health effects of the various constituents of $PM_{2.5}$; thus, EPA assumes that many constituents are associated with adverse health impacts. New programs introduced by EPA for mobile and stationary sources will continue to reduce $PM_{2.5}$-related health impacts in the United States over the next several decades. The largest potential benefits of BC mitigation measures are achievable internationally, due to high emissions co-located with large populations, particularly in South and East Asia. More information on the benefits and costs of individual measure in each country is needed to support policy decisions made at the national level.

Estimating the climate benefits of BC mitigation is less well understood and less certain compared with estimating health benefits. However, several conclusions can be drawn from the literature examining the climate impacts of BC reductions. Current studies indicate that BC reductions can reduce the rate of climate change in the near-term. Controlling emissions from motor vehicles and residential burning of solid fuels is likely to benefit climate, though residential emissions are particularly difficult to estimate and errors in current understanding of the composition of emissions may affect this conclusion. Major industrial sources of BC, such as brick kilns and coke ovens, will likely also lead to climate benefits. There are several key uncertainties which further research is needed to address, including the extent to which different emissions mixtures result in equivalent climate effects, how the indirect effects of those mixtures influence the climate outcomes, and the potential benefit of various mitigation strategies for precipitation and meteorology. Additional work is needed to design approaches to valuing climate impacts of BC directly, and to incorporate those approaches into useful metrics for evaluating policy decisions, similar to the social cost of carbon (SCC). It is also important to note that BC mitigation cannot substitute for CO_2 reductions for the purposes of alleviating long-term warming. The literature suggests that mitigating both short-lived climate forcers and long-lived greenhouse gases is necessary to achieve internationally agreed upon goals of temperature rise.

Overall, the literature points to substantial health and climate benefits of BC mitigation from some sources, particularly for control measures targeting emissions from motor vehicles, residential combustion of solid fuels, and some high BC-emitting industrial sources such as brick kilns and coke ovens. Mitigation measures for each of these sectors exist and have been proven to be effective in different parts of the world, including in the United States, as detailed in Chapters 8-11. Chapter 7 describes how the information presented in this chapter can be used to evaluate policy options.

Chapter 7

Mitigation Overview: Designing Strategies for Public Health and Near-Term Climate Protection

7.1 Summary of Key Messages

- Existing particle control programs have been effective in reducing BC in many regions, particularly control programs affecting emissions from mobile and stationary sources.
  - While BC is not the *direct* target of existing programs, it has been reduced through controls aimed at reducing ambient $PM_{2.5}$ concentrations and/or direct particle emissions.
  - Past experience suggests that available control technologies and approaches can reduce BC emissions from many key source categories at reasonable cost. However, information is currently limited regarding the effectiveness of control strategies for reducing BC in a targeted fashion and the associated costs of those strategies.
- While global BC emissions are likely to decrease in the future, 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 particularly in the transportation sector.
  - 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 low-cost reductions in BC emissions from a number of key source categories. BC emissions reductions are generally achieved by applying technologies and strategies to improve combustion and/or control direct $PM_{2.5}$ emissions from sources.
  - Some of the strategies utilized by developed countries have also been undertaken in developing countries or could be adopted on a broader scale internationally. In other cases, developing countries have a different mix of sources and constraints that may require different types of control strategies.
- In selecting BC mitigation measures, policymakers must consider three overlapping goals: climate benefits, health benefits, and environmental benefits. In most cases, policymakers will seek to achieve multiple goals simultaneously; this requires taking into account a suite of impacts and attempting to maximize co-benefits and minimize tradeoffs across all objectives (health, climate, and environment).
- With a defined set of goals, policymakers can evaluate the "mitigation potential" within each country or region. The mitigation potential depends on total BC emissions and key emitting sectors, and also depends on the availability of control technologies or alternative mitigation strategies.
- Selection of ideal emissions reduction strategies will depend on a range of constraining factors, including:
  - Timing
  - Location
  - Atmospheric Transport
  - Co-emitted Pollutants
  - Cost
  - Existing Regulatory Programs
  - Implementation Barriers
  - Uncertainty
- 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.

  - While all PM mitigation strategies that reduce human exposures will benefit health, 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 also maximize climate benefits.

  - The location and timing of the reductions are also very important. The largest climate benefits of BC-focused control strategies may come from reducing emissions affecting the Arctic, Himalayas and other ice and snow-covered regions. This would include BC emitted directly in those areas as well as BC transported into those areas from other areas.

- Cost is a prime consideration in both developed and developing countries, as is feasibility of implementation. Some physical and political constraints may hinder full implementation of even those strategies for which there is high confidence of large health and climate benefits.

- Optimizing climate, public health and environmental benefits requires a broad, multi-pollutant approach to BC mitigation that includes looking at the entire suite of options and evaluating them carefully to understand the full range of costs and benefits.

7.2 Introduction

As outlined in the previous chapter, reducing BC emissions has tremendous potential for improving global public health while achieving climate benefits. The optimal path forward will vary as decision-makers in each country weigh desired health and environmental outcomes, costs and benefits, and mitigation potential. This is a complex calculus that depends on a large number of considerations. This chapter presents a decision framework to help guide policymakers who want to develop BC mitigation strategies.

First, the chapter examines what is known about the overall impact of existing or planned control programs on emissions of BC and how current BC emissions are projected to change over the next several decades in response to these control programs and/or economic growth and development. Next, the chapter describes a decision framework, which includes both key factors to consider when developing a BC mitigation strategy and how different approaches for reducing BC emissions have potential to provide climate and public health benefits. The chapter concludes with several examples of how the decision framework could help guide mitigation choices. The chapter is followed by more detailed mitigation chapters covering four major emissions sectors—mobile sources; stationary sources (including both power generation and industry); residential heating and cooking; and open biomass burning. Chapters 8-11 describe projected changes in emissions in these sectors in the United States and globally, available control technologies and strategies and their associated costs, and implementation challenges. Chapter 12 provides a summary of how the mitigation options in these various sectors translate into near-term mitigation opportunities for BC to benefit climate and public health.

7.3 Effect of Existing Control Programs

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

In recent years, the overarching $PM_{2.5}$ control program for stationary sources in the United States and Europe has focused mainly on secondarily formed particles such as sulfates and nitrates, rather than on direct $PM_{2.5}$ emissions. This is because PM controls motivated by public health and environmental goals are focused on reducing total PM mass (a large portion of which is sulfates and nitrates formed in the ambient air from SO_x and NO_x) at least cost. PM controls oriented toward climate would have to consider the light absorbing and scattering properties of the various PM constituents.

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

In general, available estimates of BC emissions reductions are calculated from analyses of $PM_{2.5}$ controls. As discussed in Chapter 4, EPA's trends report (2010i) shows that U.S. emissions of direct $PM_{2.5}$ have declined by 58% since 1990, a reduction of over 1.3 million tons. Over half of this reduction has come from controls on stationary fossil fuel combustion, with substantial reductions also occurring in emissions from industrial processes and mobile sources. Using speciation factors, it is possible to calculate BC reductions in these sectors, but these estimates are generally rough. Information on BC reductions is strongest for the mobile source sector, where the BC fraction of emissions and the impact of specific controls are well understood. As discussed in more detail in Chapter 8, mobile source BC emissions declined by approximately 32% between 1990 and 2005. For other sectors, precise, measured data about the effectiveness of specific controls for reducing BC emissions is often not available. As described in Chapter 5, however, recent ambient BC measurements do appear to indicate a decline in neighborhood/urban and regional scale concentrations of BC in the United States between the mid 1980s and the present (see section 5.4.1).

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

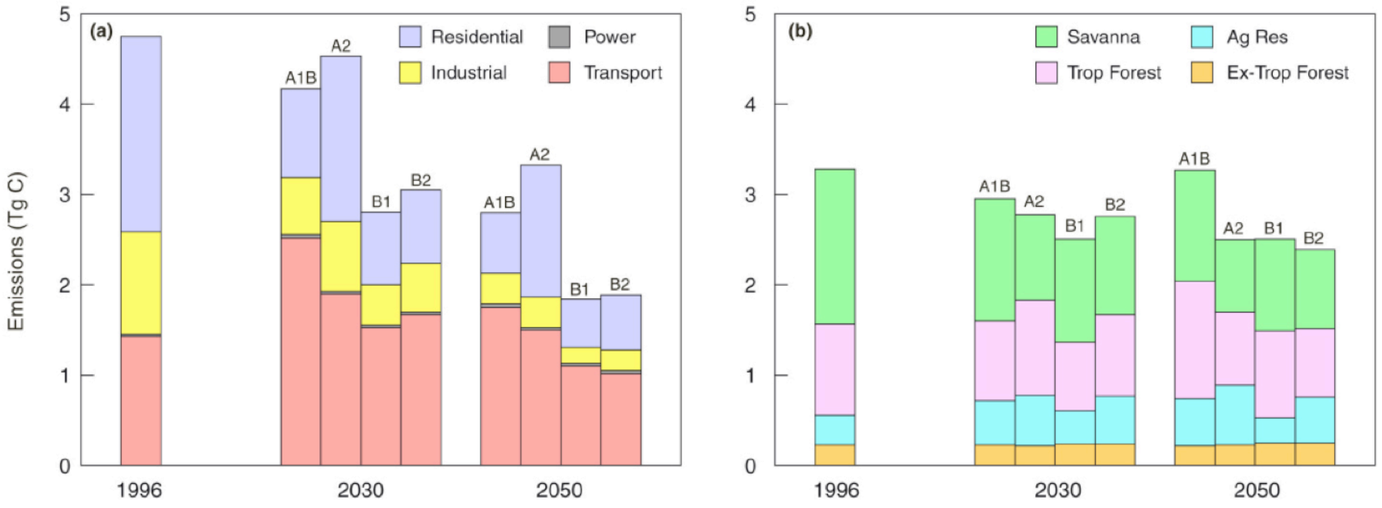
Despite what is known from analysis conducted in the United States, many of the strategies that have been applied domestically differ in important ways from control strategies that have been adopted internationally. Some of the strategies utilized by developed countries have also been undertaken in developing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

7.4 Future Black Carbon Emissions

The influence of BC on climate and public health in the future, and the need to more precisely determine the effectiveness of various mitigation strategies for reducing BC, depends in large part on the magnitude of future emissions. This section describes what is known regarding these future emissions, but available estimates are variable and uncertainty about future emissions trajectories remains high.

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

Recent studies (Streets et al., 2004; Cofala et al., 2007; Jacobson and Streets, 2009; Rypdal et al., 2009) provide a snapshot of potential future BC emissions trends. These studies have produced a range of estimates for future BC emissions depending on assumptions about economic growth, population levels, and development pathways. In an analysis of future BC emissions trends based



Energy-related combustion

Open biomass burning

Figure 7-1. Global BC Emissions Forecasts for Various Sectors under Alternative IPCC SRES Scenarios (in teragrams (Tg) of carbon). Scenarios generally show a modest decrease in BC emissions from all sectors as compared to 1996 baseline emissions. (Streets et al., 2004)

on the IPCC SRES scenarios, Streets et al. (2004) projected BC emissions to decrease globally by 9% to 34% by 2030 relative to 1996 levels depending on assumptions about economic growth and development. However, there was considerable variation among projections for the different sectors depending on the SRES scenario examined (Figure 7-1). Thus, while aggregate emissions were generally projected to decline under alternative growth scenarios, emissions growth was projected for certain sectors or regions. The sectors where Streets et al. (2004) indicate a potential for future emissions growth include residential emissions in Africa, open biomass burning emissions in South America, and transportation emissions in the developing world (for example, where fuel sulfur levels are still too high for implementation of DPFs—see Chapter 8 and Appendix 3). In general, industrial emissions and transport emissions were projected to decline in developed countries.

An analysis by Jacobson and Streets (2009) found that under the assumptions embedded in the A1B scenario for IPCC, total global BC emissions may increase substantially. Again however, this analysis indicates that projected emissions growth or decline varies significantly among regions and sectors, as Figure 7-2 illustrates. In general, BC emissions in developed countries are projected to decline, while emissions in developing countries may grow. Transportation (mobile source) emissions in particular are projected to grow in several world regions but decline in others, as illustrated by growth factors greater than or less than one, respectively (Jacobson and Streets, 2009). In developed countries, the majority of the emissions reductions in the transportation sector are projected to result from implementation of 2007 U.S. on-highway diesel engine standards and similar standards, such as Euro V, that lead to the use of diesel particulate filters (DPFs) in the diesel fleet. Also, in the United States, other standards for diesels (nonroad diesels, locomotives, and commercial marine) contribute to these reductions. However, other studies have indicated that emissions from shipping in the Arctic region may increase due to the retreat of Arctic sea ice, opening up new shipping routes and increased economic activity in that region (Corbett et al., 2010).

In its most recent work, the IPCC has also developed four "Representative Concentration Pathways" for use as a consistent set of emissions inputs for projecting future climate change. These four pathways (Figure 7-3) are defined by the total radiative forcing resulting from each pathway in 2100, including GHGs and other forcing agents, which ranges from 2.6 to 8.5 W m^{-2}. Global BC emissions in all four pathways peak in 2005 or 2010, are 8 to 20% below 2010 levels by 2030, and continue decreasing for the rest of the century to about half of 2010 levels. Emissions for the

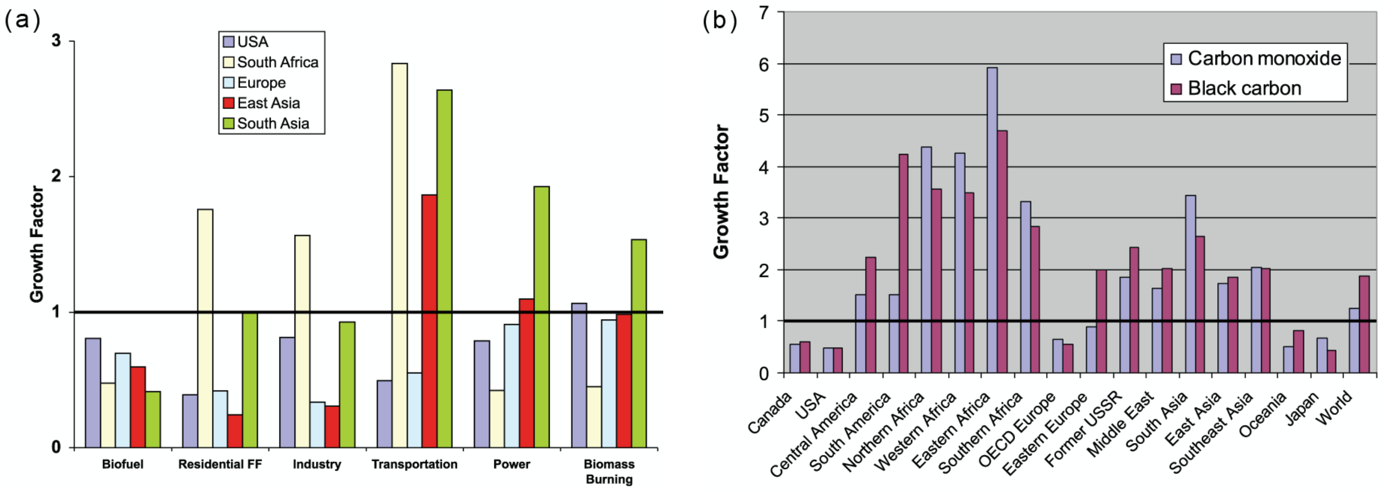


Figure 7-2. Black Carbon Emissions Growth, 2000-2030 under IPCC A1B Scenario. Top: 2000-2030 Black Carbon Emissions Growth Factors by Sector for Selected World Regions (from IPCC A1B scenario). Bottom: 2000-2030 Black Carbon and Carbon Monoxide Emissions Growth Factors for Transportation (Mobile) Sector in Specific Regions. Emissions in sectors with a growth factor less than one (see dark line, added) will decline. (Source: Jacobson and Streets, 2009)

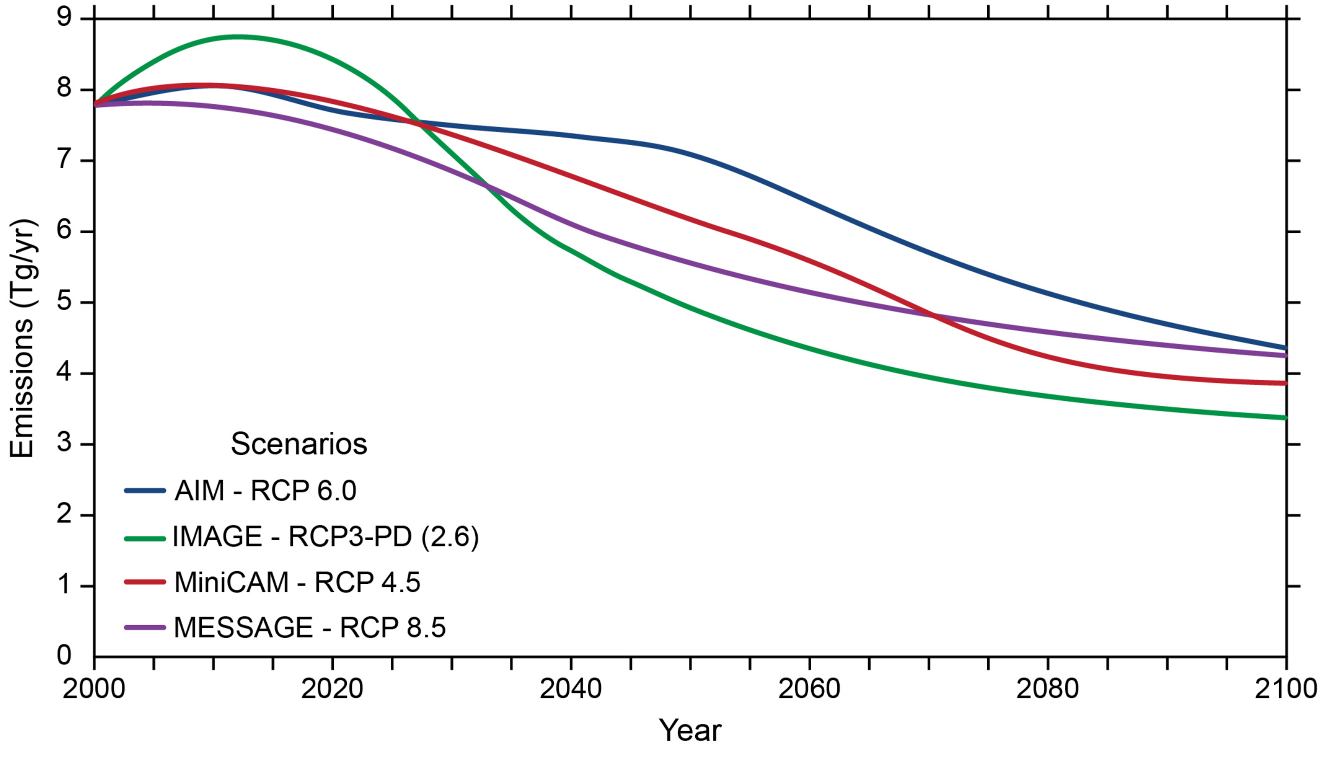


Figure 7-3. Future Emissions of BC under IPCC Representative Concentration Pathways, 2000-2050 (Gg/year).

Notes:

1. RCP2.6 (RCP 3-PD) – van Vuuren et al., (2007)
2. RCP 4.5 – Clarke et al. (2007); Smith and Wigley (2006); and Wise et al. (2009)
3. RCP 6.0 – Fujino et al. (2006); and Hijioka et al. (2008)
4. RCP 8.5 – Riahi et al. (2007)

RCP pathways are reported in combinations of five regions and four sectors on the website[1] (though the underlying, gridded dataset is further disaggregated).

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

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

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

Some countries have already begun looking at these possibilities. For example, the Arctic Council countries (Canada, Denmark, Finland, Iceland, Norway, Russia, Sweden, and the United States) formed a special Task Force on Short-Lived Climate Forcers in 2009 to consider whether additional or accelerated mitigation strategies may be needed to address warming in the Arctic region. Noting that emissions in the Arctic region from sources other than land-based transport—particularly residential heating, agricultural and forest burning, and marine shipping—will likely remain the same or increase without new measures, the Task Force has recommended that "Arctic Council nations individually and collectively work to implement some early actions to reduce black carbon" (Arctic Council Task Force on Short-Lived Climate Forcers, 2011, p. 5). A new Project Steering Group (PSG) under the Arctic Council's Arctic Contaminants Action Programme (ACAP) is currently investigating which measures would be implemented, and the impact of such measures on total BC emissions from Arctic Council nations. The PSG has identified a number of key sources affecting the Arctic and is developing pilot programs to advance BC emissions reduction efforts, particularly in the Russian far north.

7.5 Key Factors to Consider in Pursuing BC Emissions Reductions

Significant reductions in BC emissions are expected to occur in certain regions in the coming decades; however, these reductions will be gradual, and even after they are fully realized, substantial BC emissions will remain in some sectors and regions. There is a core set of factors that policymakers can consider to improve the likelihood that selected mitigation strategies will achieve substantial public health and environmental benefits and reduce the rate of near-term warming. Policymakers can examine the challenge of BC mitigation from the perspective of three different (but overlapping) goals: climate benefits, health benefits, and environmental benefits. With a defined set of goals, policymakers can evaluate the "mitigation potential" within each country or region. The mitigation potential depends on total BC emissions and key emitting sectors, and also depends on the availability of control technologies or alternative mitigation strategies, such as fuel switching, improvements in energy efficiency, or changes in land-use patterns. The ideal emissions reduction strategies will also depend on a

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[1] *http://www.iiasa.ac.at/web-apps/tnt/RcpDb/dsd?Action=htmlpage&page=welcome#*

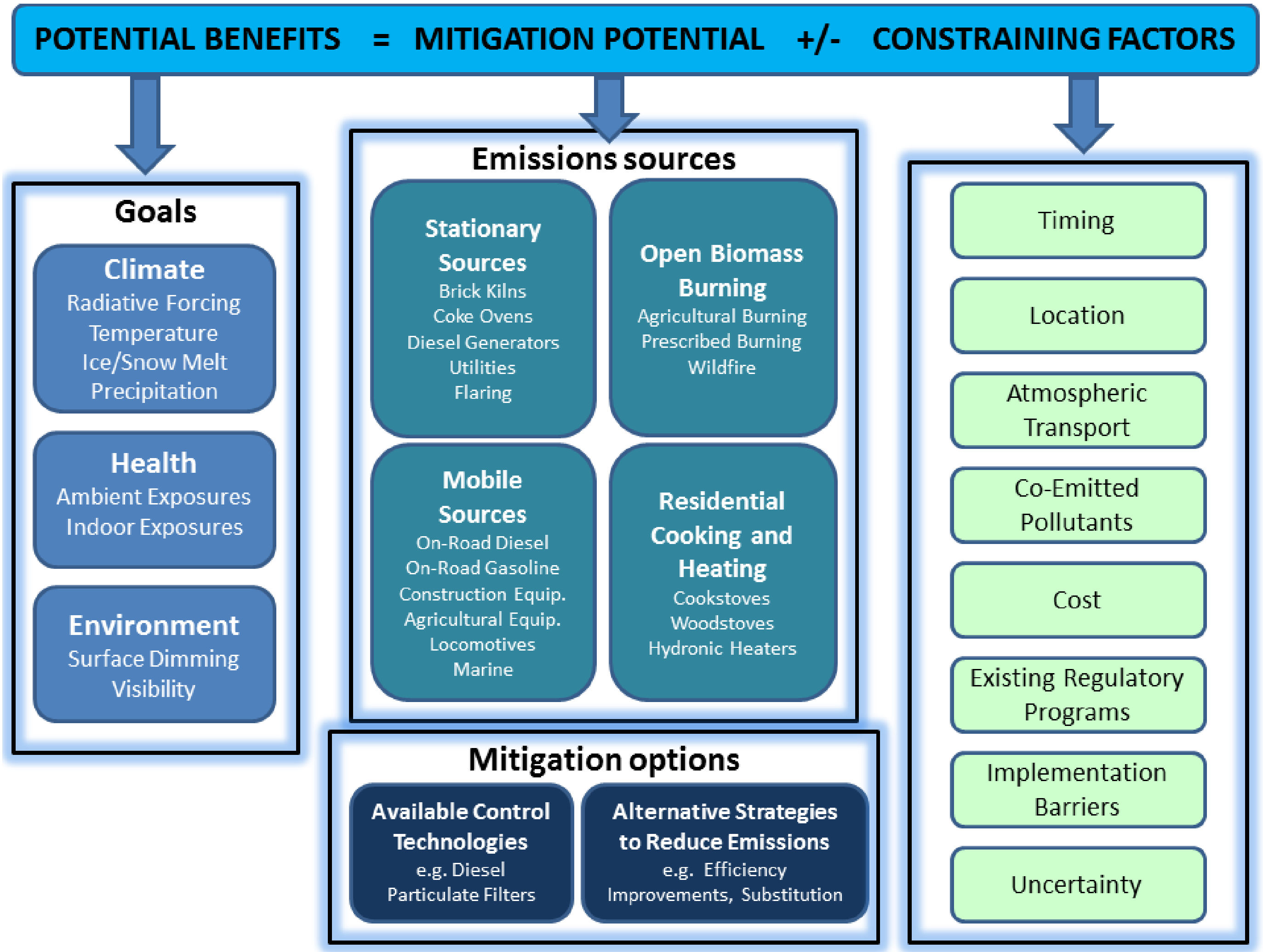

Figure 7-4. Policy Framework for Black Carbon Mitigation Decisions. (Source: U.S. EPA)

range of constraining factors, such as cost, location, and co-emitted pollutants. This decision-making framework is illustrated in Figure 7-4. Each of the key factors policymakers must consider is discussed further below.

7.5.1 Defining Goals: Climate, Health and Environmental Outcomes

Reducing BC emissions offers a win-win opportunity with the potential to achieve benefits for climate, public health and the environment simultaneously. The preferred mitigation strategies could differ depending on the main policy goal. Policymakers focused primarily on climate objectives might choose different mitigation approaches than policymakers who are interested in maximizing public health benefits or protecting visibility. BC plays an important role in all of these effects, but achieving different goals might require different mitigation strategies oriented toward different sources in different locations. In most cases, policymakers will be seeking to achieve multiple goals at the same time; thus, preferred strategies will likely take into account a suite of impacts and decision-makers will attempt to maximize climate, health and environmental benefits and minimize tradeoffs across all objectives. However, it is important for policymakers to be clear about what the goals are, and to prioritize among them when tradeoffs are involved.

Often, policymakers will be considering BC reductions as merely one part of a broader PM mitigation agenda, and decisions will be driven first by health considerations. This is consistent with the mitigation pathway in developed countries, where the primary goal has been improving public health, and the secondary goal has been reducing non-climate environmental effects of PM (such as acid deposition and visibility impairment). PM mitigation programs in most countries have not given direct consideration to climate benefits. Policies have generally focused on reducing emissions of secondary PM precursors (SO_2 and NO_x) as the most cost-effective strategy for those health benefits. While enormously successful for public health, strategies that reduce SO_2 and NO_x emissions to control secondarily formed PM are not expected to result in substantial decreases in BC emissions. Moving forward, as climate becomes an additional consideration, SO_2 and NO_x reductions should be accompanied by corresponding reductions in BC. The practical result would be additional policies that more actively target sources of direct PM emissions. This would ensure that ongoing reductions in SO_2 and NO_x emissions, which are critical to achieving public health and non-climate environmental goals, are complemented and enhanced by BC emissions reductions for climate.

Even within a specific category (climate, health, environment), there are multiple sub-goals to be considered. While related, these different objectives may require different strategies and approaches. For example, BC reductions could provide climate benefits in the form of reduced radiative forcing and temperature, but could also be aimed at reducing precipitation impacts and the melting of ice and snow. The optimal control strategy will depend on the impact(s) of concern. Scientific evidence discussed in Chapter 2 suggests that direct radiative forcing and temperature increase are driven largely by BC emissions, offset by emissions of other aerosols. Reducing these impacts, therefore, may require strategies that preferentially reduce BC relative to cooling species. On the other hand, a broader variety of aerosols (including nitrates, sulfates, OC) are contributing to snow/ice albedo effects, ABCs, and precipitation effects, so to reduce these impacts, a wider variety of $PM_{2.5}$ reduction strategies may be beneficial. For health, policymakers are interested in both indoor and outdoor exposures and risks. However, reducing indoor exposures to $PM_{2.5}$ (including BC) often requires different strategies than reducing ambient $PM_{2.5}$ concentrations. Identifying clearly which set of impacts the mitigation action is designed to reduce is critical for selecting appropriate measures.

Because of the nested nature of the climate, public health and environmental goals driving policy decisions about BC mitigation, cost-benefit analysis conducted to support mitigation decisions should incorporate public health and welfare benefits as well as climate benefits. Such analysis can fully inform decision makers regarding available choices, but can also be complicated by uncertainties in valuing the climate impacts. This is particularly true for other environmental outcomes, such as for visibility and impacts on agriculture. The impact of BC on these outcomes is less well understood, and therefore there is less known quantitatively about expected benefits of BC reductions. Nevertheless, as discussed in Chapter 6, public health benefits of many BC reduction strategies may be large enough to justify the costs, regardless of the climate impacts.

7.5.2 Identifying Opportunities for Emissions Reductions

Designing BC mitigation programs requires policymakers to carefully evaluate both the total BC emissions inventory in a specific region or country, and the contribution of specific sectors to that total. As discussed in detail in Chapter 4 and mentioned earlier in this chapter, BC emissions—and therefore BC mitigation potential—vary widely by region. In some countries, aggressive programs to reduce PM emissions mean that the remaining emissions are limited, making further reductions more difficult and expensive to achieve. Nevertheless, virtually every country, including the United States, still has substantial BC emissions remaining across a number of source categories, meaning that further BC reductions are possible. For example, while U.S. BC emissions are expected to drop considerably by 2030 due to controls on new mobile diesel engines (see Chapter 8), significant emissions remain from the existing fleet of diesel vehicles. Achieving additional reductions from mobile diesel engines is currently possible through EPA's National Clean Diesel Campaign and SmartWay Transport Partnership Program. However, funding would be required in order to address a majority of the legacy fleet of existing vehicles. Furthermore, as discussed in Section 7.4, several recent studies identify regions and emissions source categories in which BC emissions are likely to increase over time. These sectors represent important potential mitigation opportunities, even though total emissions from some of them may not be large at present.

Careful investments in emissions inventories and emissions measurements can greatly improve policymakers' ability to identify key emitting sectors and sources. This includes evaluating current emissions and anticipated future BC emissions in terms of both specific facilities (how many? where?) and control technologies already in place. Having a clear understanding of exactly where potential emissions reductions could be achieved based on current sources and technologies opens the door for practical conversations about specific avenues and means for achieving those reductions. This includes considering both:

- **Availability of control technologies:** As individual countries examine the options for reducing BC to improve air quality and benefit the climate, they must consider the availability of control technologies that are effective for BC mitigation. Technologies to reduce PM and co-pollutant BC emissions are in widespread use globally, as are technologies that reduce PM and co-pollutants more broadly.

- **Alternative mitigation strategies:** Even when conventional control technologies are ineffective or costly, other strategies may be available to reduce BC emissions. In some cases, BC reductions may be achieved not through end-of-pipe controls (such as particle filters) but through substitution of new, cleaner technologies (such as improved cookstoves and brick kilns) or more efficient combustion practices that substantially alter the emissions stream. Changes in land-use patterns or greater reliance on alternative forms of energy can also result in BC reductions.

Previous chapters have covered in detail what is known regarding emissions sources from the United States and across the globe. The following chapters will go into greater detail regarding specific technologies and strategies available to control BC emissions from the main contributing sectors. Appropriate mitigation choices for individual countries will vary based on scientific variables and policy drivers.

7.5.3 Key Considerations

Even with clearly defined goals and carefully constructed emissions inventories, there are a number of other factors for policymakers to consider that are critically important for mitigation decisions. These include:

- **Timing:** Seasonal timing of emissions reductions is important. As discussed in Chapter 2, the effects of BC in snow- and ice-covered regions are accentuated during times of increased sun exposure. Policies designed to reduce BC impacts on the Arctic, therefore, might need to consider the seasonality of emissions. For example, impacts of open biomass burning on the Arctic spring melt could be influenced by adjusting the timing of agricultural burning. Similarly, the benefits of emissions controls for the Indian Monsoon will depend on the timing of controls relatively to seasonal weather patterns.

- **Location:** Considering the location of emissions is also important when seeking mitigation strategies that will be beneficial for both climate and public health. Because BC is a regional rather than global pollutant, it is important to evaluate the benefits of emissions reductions in terms of specific regional impacts, factoring in source region, emissions transport, and receptor region conditions. For climate, impacts on snow- and ice-covered regions such as the Arctic are of particular concern, as are impacts in regions where the precipitation patterns are heavily impacted by all aerosol emissions, such as India. Location matters for human health benefits too, with proximity of emissions reductions to areas of large population being a prime consideration. In general, the largest human health benefits from BC-focused control strategies occur near emissions sources, where exposure affects a large population. Despite the historical focus on secondary PM for public health, many of the BC sources ripe for mitigation are in large urban areas. These sources of direct PM, from the transportation, residential and industrial sectors, are tied to everyday human activities. This means that the benefit of reducing BC emissions for the health of the people living nearby is expected to be very high.

- **Emissions Transport:** The net impacts of emissions from any specific source will depend partly on atmospheric fate and transport. Meteorological conditions, plume height and other factors will affect the extent to which emissions from particular sources in particular locations affect climate, health and environmental endpoints of interest. Identifying how reductions in emissions from specific sources translate into climate, health and environmental responses in receptor regions requires sophisticated models that can account for myriad chemical and physical processes, as well as climate responses.

- **Co-emitted pollutants:** Maximizing benefits across different goals requires explicitly accounting for co-emitted pollutants. Depending on the goal, reductions in some species may be more valuable than reductions in others. The largest benefits in terms of direct forcing would come from two types of strategies: first, strategies that reduce BC more than emissions of other (cooling) PM constituents like SO_2, NO_x and/or OC; and second, strategies that reduce BC in conjunction with reductions in other GHGs like CO_2 and CH_4. The first approach would include strategies such as controls on mobile diesel engines (see Chapter 8), while the second approach would include strategies

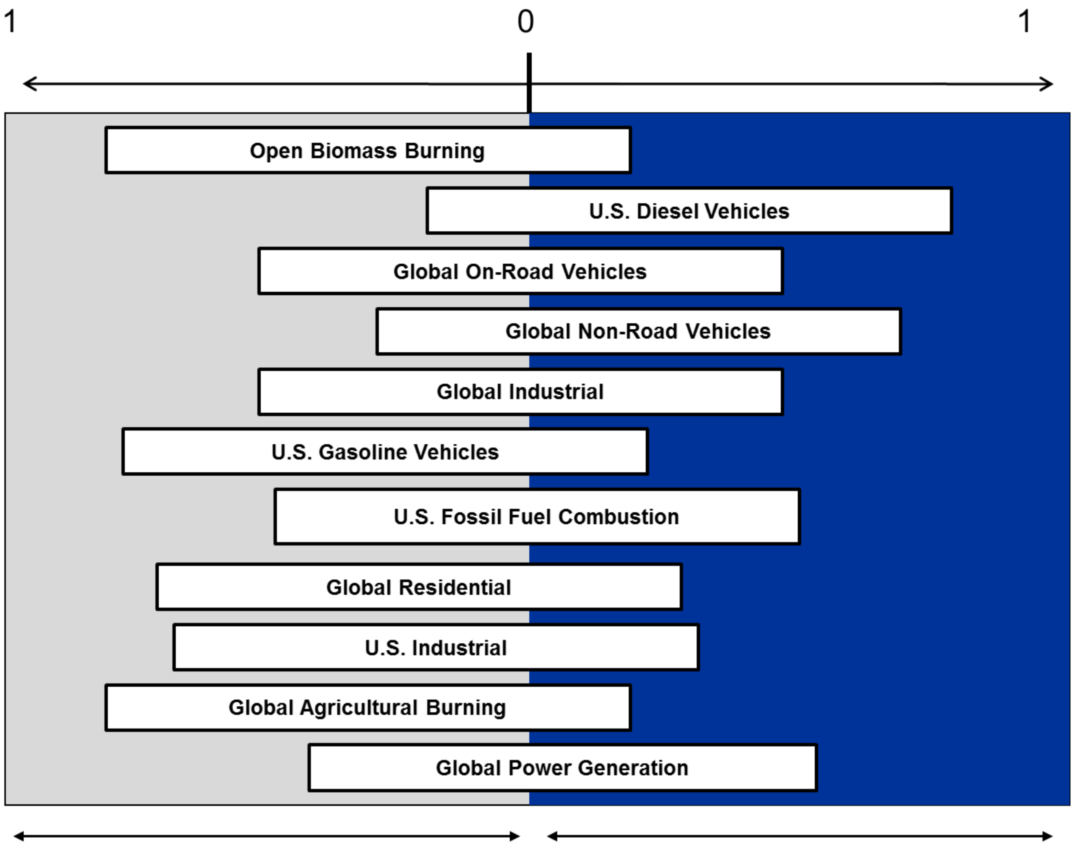

Figure 7-5. OC (left) and BC (right) Emissions from Key U.S. and Global Emissions Source Categories, Expressed as a Fraction of Total Carbon (OC + BC) Emissions from that Category. (Source: U.S. EPA)

> such as improvements in combustion and fuel efficiency in residential cookstoves, use of prepared fuels (briquettes or pellets), or a shift to clean cooking fuels (see Chapter 10). For both types of strategies, optimizing control opportunities requires careful analysis of the full emissions stream. When all climate effects (i.e., effects beyond direct forcing) are considered, and particularly when health and environmental benefits are added to the equation, the relative importance of various co-pollutants changes. For reducing direct forcing, mitigation strategies might focus on reducing BC preferentially. If the focus shifts to health impacts or precipitation effects, other aerosol species become equally important and total $PM_{2.5}$ reductions are likely to be preferred over BC-focused strategies.

It is very difficult to establish short-hand approaches for estimating the impacts of specific sources based on their chemical emissions profile. In a number of studies, the OC/BC ratio[2] has been used to rank the net warming potential of individual source categories, based on an assumption that OC will primarily reflect light and thereby induce a negative radiative forcing (cooling) effect in the atmosphere, and that BC will primarily absorb light and induce a positive forcing (warming) effect. Figure 7-5 illustrates the variation in emissions profiles among sources by comparing relative OC/BC ratios. Emission sources with low OC/BC ratios are generally thought to have the largest potential to warm the climate, though there is no agreement within the scientific literature about how to interpret specific ratios. Bond et al. (2011) found that for direct forcing only, a ratio of about 15:1 for OC to BC is close to climate neutral; however, this does not include cloud indirect effects or co-emissions of substances other than OC. Bond et al. also find that the neutral ratio varies substantially depending on where the emissions occur.

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[2] The most commonly used ratio is OC:BC or OC/BC, but other ratios include OC/EC, and OM/BC, where OM represents the total mass of organic matter. Chapter 4 of this report provides OC/BC and BC/$PM_{2.5}$ ratios for a number of source categories.

Several factors limit the value of the OC/BC ratio as a short-hand for estimating the climate forcing impact of a combustion source. Assuming that all particle OC (including BrC) is scattering, regardless of location, may underestimate the positive forcing (warming) impact of a given source. Also, the OC/BC ratio ignores other climate-relevant pollutants. Sulfates, nitrates and secondary organic aerosols (SOA) or their precursors, sulfur dioxide (SO_2), NO_x, and VOCs, form additional light-scattering material within the plume. Ramana et al. (2010) note that the extent of BC-induced warming depends on the concentration of both sulfate and OC. Further, the aging process described in Chapter 2 induces optical changes in an emitted particle mixture, including coating of BC particles, leading to enhanced light absorption (Lack and Cappa, 2010). These effects are not captured by an OC/BC ratio. Finally, many analyses that employ OC/BC thresholds for "net warming effects" do not take into account other effects, such as effects on precipitation and all the indirect effects related to particle-induced changes in clouds.

Despite the many limitations of OC/BC ratios, they provide a simple way to evaluate potential climate benefits and therefore continue to be used to prioritize mitigation options. For the reasons stated above, these ratios should serve only as a approximate indicator of potential radiative effects of categories of emissions sources. The specific circumstances or policy goals should override generic OC/BC rankings in cases where, for example, emissions are affecting the Arctic, since even mixtures that contain more reflective aerosols can lead to warming over such light-colored surfaces. In addition, OC/BC ratios are irrelevant to effects that are shared among BC and other aerosols. This includes precipitation or dimming effects, and impacts on public health. For these types of effects, mitigation strategies that reduce direct $PM_{2.5}$ emissions or overall ambient $PM_{2.5}$ concentrations will provide the largest benefits, and the ratio of BC to other constituents is far less important.

- **Cost:** Cost is a prime consideration in both developed and developing countries. Regardless of whether control technologies or alternative strategies to reduce pollution are available, policymakers are only likely to pursue such approaches if the costs are limited and/or clearly outweighed by the benefits of mitigation. For some sectors and locations, BC control costs are low in comparison to the public health benefits that can be achieved (the most easily quantified stream of benefits). However, control costs can vary significantly depending on the technologies being considered, and in many cases, particularly in developing countries, there is a lack of information about costs of BC control.

- **Existing Regulatory Programs:** Countries with existing regulatory programs and statutory mandates may face legal constraints in terms of the types of emissions reductions they can pursue. There may also be technological barriers and phase-in schedules that need to be considered. For example, regulations that require application of specific control technologies or approaches may dictate the extent of emissions reductions achieved. Many regulations are phased in or become effective over an extended time period. Controls on new mobile diesel engines, for example, only reduce BC emissions as new vehicles and engines are purchased and deployed to replace older models. Policymakers must work within existing regulatory constraints in selecting mitigation initiatives.

- **Implementation Barriers:** As policymakers look across the range of available emissions reductions, the other key factor to consider is feasibility of implementation. This is perhaps the most important consideration, because there are some constraints that simply cannot be overcome. Many examples exist for how physical and political constraints hinder full implementation of even those strategies for which there is high confidence of large benefits. Replacement of the 500-800 million traditional solid-fuel burning cookstoves in developing countries has been the subject of numerous public and private campaigns, with one notable large-scale success (China) and many cases of limited success (Sinton et al., 2004). Only recently has progress begun to accelerate, as illustrated by the rapid growth in stove sales tracked by the Partnership for Clean Indoor Air (see Chapter 10) and the emergence of the Global Alliance for Clean Cookstoves. The historically slow progress on this issue is attributable, in part, to the scale of the problem and the difficulty of implementing replacement programs in diverse local environments.

  Some studies have suggested altering the timing of agricultural burning in the United States to avoid transport to the Arctic during the spring melt season (CATF, 2009a). However, the seasonally dependent cycles of planting, harvesting and pest-control (among other considerations) make this a difficult strategy to implement. Also, for stringent PM standards for new vehicles, nonroad diesels, locomotives, and commercial marine (other than ocean going vessels), ultra low sulfur diesel fuel is required to enable use of diesel particulate filters. Many countries already have regulations requiring such fuels. As another example, there is near universal agreement that retrofitting the existing diesel fleet across the globe would reap tremendous public health and climate benefits; however, this kind of program would be very expensive. Cultural practices also come into play. Residential heating in the Arctic region has recently been identified as a key contributor to BC impacts in the Arctic. Nevertheless, both the rising cost of fossil fuel in relation to wood and the cultural connection to in-home wood heating makes a widespread transition to pellet stoves (a recommendation from the UNEP/WMO report) problematic.

- **Uncertainty:** In cases where there is a high degree of uncertainty, policymakers may hesitate to take mitigation actions. For example, if the emissions mixture from particular sources is not well characterized, or if the effectiveness of control strategies is not well understood, the benefits of pursuing reductions may be questioned by decision-makers. This is particularly true if the health and environmental co-benefits of a strategy are limited, or if costs are high. Under these circumstances, policymakers may choose to postpone BC mitigation actions, or limit those actions to a narrower set of controls with more clearly defined costs and benefits.

All of these factors affect which mitigation options are most desirable under different circumstances. Clearly, the complexity of the decision-making calculus requires the balancing of multiple considerations simultaneously. In some cases, these considerations may involve tradeoffs, and policymakers will have to evaluate which options are best given competing goals and considerations. The following section illustrates how the different factors in the decision-making framework can affect which options are preferable under different circumstances.

7.6 Applying the Mitigation Framework

To craft policy that addresses remaining BC emissions, as well as anticipated increases in BC emissions in some sectors and regions, policymakers are faced with a complex set of considerations and potentially competing goals. Optimizing climate and public health benefits requires a broad, multi-pollutant approach to BC mitigation that includes looking at the entire suite of options and evaluating them carefully to understand the full range of possible benefits. Currently, most countries implement a pollutant-by-pollutant approach to air quality management, looking at each pollutant in isolation and developing strategies targeting each one. Though successful in many respects, this approach may not result in the most effective or efficient strategies for achieving multiple objectives. As a result, many countries are moving toward a multi-pollutant approach, where strategies incorporate many pollutants by sector or by region. However, most have yet to incorporate climate pollutants and benefits into those strategies.

A recent case study of Detroit, Michigan conducted by EPA explored how costs and benefits would change if such a multi-pollutant framework was designed (Wesson et al., 2010). This case study compared a traditional pollutant-by-pollutant strategy (status quo) with one that sought to maximize emissions reductions and risk reduction for ozone, PM and selected air toxics (multi-pollutant, risk-based). The study found that the multi-pollutant, risk-based strategy produced over twice the monetized benefits of the status quo. Though the multi-pollutant, risk-based strategy cost slightly more, it resulted in a much more favorable benefit/cost ratio. This result was due, in part, to a shift from reductions in secondary PM to a strategy that achieved greater reductions in primary PM, particularly in emissions from sources affecting vulnerable and susceptible populations. This is exactly the kind of shift in strategy that could achieve BC benefits. This suggests that multi-pollutant assessment can produce more cost-effective strategies, and that targeting primary PM emissions (with BC benefits) can have large health benefits, especially in urban areas.

Policymakers facing choices among different BC mitigation options can apply the 3-step framework described above (defining goals, identifying emissions reduction potential, and weighing key factors) to evaluate the pros and cons of different options and to identify the options that best maximize co-benefits. In many cases, the most desirable mitigation approaches will be determined by the primary goal (or goals) combined with constraints on resources, technologies, or anticipated impacts. Several examples can help illustrate how the factors in the framework affect the attractiveness of different mitigation options.

In the United States, BC is managed as part of the larger PM mitigation program designed to achieve a suite of health and environmental goals. While this generally means focusing on reducing total PM mass, a greater emphasis on climate protection as a goal in the PM program might require more focus on reducing BC emissions within the overall PM mixture. The largest domestic categories of BC emissions are mobile diesel engines and wildfires; emissions from both sectors are large, so both present some mitigation potential. However, there are substantially different mitigation challenges for each sector. Mobile diesel emissions are well characterized and dominated by BC (as opposed to other co-pollutants such as NO_x, SO_x, or OC); they come from a defined set of controllable sources; they can be controlled with known technologies with clearly identified costs; they tend to be concentrated in urban areas where they affect large populations; and they can be regulated through centralized federal programs. On the other hand, mobile diesel emissions from U.S. sources are spread out geographically and while they contribute to overall atmospheric warming and have a large health impact in urban areas, they are not as readily transported to the Arctic. Controls on mobile diesel engines can be expected to provide widespread health and environmental benefits, and climate benefits in the form of overall reductions in radiative forcing. These global climate benefits will accrue in part to the Arctic, helping to alleviate impacts (such as rapid temperature increase) in that region. However, since emissions within or near the Arctic appear to be most closely linked to Arctic impacts (Quinn et al., 2011), diesel reductions in the United States may not be the most effective emissions reductions for reducing impacts in the Arctic. They will have even less impact on the Himalayas.

Wildfire emissions in the United States are less well characterized in terms of volume and radiative forcing potential, in part because they are dominated by OC rather than BC. They are unpredictable and extremely variable across time and space. They are difficult to control with known technologies or approaches, and often occur in locations far removed from people. Implementation barriers are much higher, and anticipated impacts of emissions reduction strategies are much more uncertain. On the other hand, wildfire emissions have been shown to be one of the prime contributors to BC deposition and atmospheric loading in the Arctic, and even lighter-colored OC has been linked to warming in the region (Quinn et al., 2011). Because the volume of emissions from a single event is large and emissions plume height enables long-range transport, wildfire emissions can travel long distances and have impacts far from the location of the fire. In the United States in 2002, nearly 50% of wildfire emissions occurred in Alaska, making them extremely important for impacts on the Arctic.[3] Therefore, if policymakers are seeking mitigation options that provide climate benefits to the Arctic specifically, they might be interested in attempting to control wildfire emissions.

Outside of the United States, similar tradeoffs and challenges exist. Agricultural burning in Africa is one of the world's largest sources of BC. Despite high OC co-emissions, this agricultural burning may have substantial climate impacts because emissions are lofted above deserts, which, like snow and ice, are light-colored surfaces. However, because much of the burning takes place in rural areas, controlling these emissions would provide fewer benefits for health than reducing emissions from the transportation sector in larger African urban areas, or reducing emissions from solid-fuel cookstoves. Again, policymakers must consider a whole range of factors in determining which options are preferable.

Some of the largest climate benefits of BC-focused control strategies may come from reducing emissions that affect the Hindu Kush-Himalayan-Tibetan Plateau (HKHT), as well as other ice- and snow-covered regions. The HKHT and Indian subcontinent is also experiencing some of the most dramatic impacts on precipitation from particle pollution; these multiple climate effects mean there is a tremendous opportunity in that region to maximize climate benefits (including reduced interference with natural precipitation patterns in the region). An aggressive mitigation strategy could reap substantial benefits directly to that region's population, one of the world's largest. Here, many of the variables in the framework align: a variety of sources (cookstoves, diesel trucks, brick kilns) are affecting both climate and health, and low-cost technological solutions are available to help mitigate these emissions. These technologies are discussed further in the next several chapters, and some of the key options for the region are highlighted in Chapter 12.

7.7 Conclusions

Mitigating BC emissions depends on a clear prioritization of goals and emissions reduction opportunities. The challenge for policymakers is clear: identifying feasible and cost-effective mitigation strategies requires carefully weighing a large number of factors, in many cases with incomplete information. The challenge is rarely a purely technological one. As the following chapters will illustrate, many effective control technologies are available to reduce BC emissions. These control technologies can provide cost-effective BC emissions reductions from key source categories,

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[3] Alaskan wildfire emissions exhibit significant interannual variability. Emissions in 2002 were particularly high. Factors such as the timing, extent, and location of the fire, the total volume (and type) of fuel consumed, and the burning conditions all affect the fire emissions and the net impact on climate.

and current studies (discussed in Chapter 6) suggest that applying available technologies and strategies will produce near-term climate benefits, especially at the regional level. Available control technologies can also provide substantial health and environmental benefits. The challenge is identifying the strategies that maximize benefits across all these categories. Despite what is known and achievable, the path forward is neither straightforward nor easy, and will vary greatly by country and region of the world. Nevertheless, as demonstrated in the previous chapter, the potential benefits of action are large.

The next four chapters provide an overview of BC mitigation options, including costs where that information is available, in each of four major source categories: mobile sources, stationary sources, residential cooking and heating, and open biomass burning. Policymakers can view these options through the lenses discussed above and determine which, if any, strategies are appropriate to reduce BC emissions in their specific context. In Chapter 12, some of the clearest mitigation opportunities based on current emissions, control technologies, and expected benefits are discussed. These are options that satisfy many of the key criteria that decision-makers care about, as presented in the mitigation framework above. They are options that are most clearly linked to beneficial outcomes, because of their high BC emissions reductions potential, their importance for particular regions, or the lack of constraints on implementation. However, policymakers will need to adapt even these important mitigation options to their particular local and national circumstances.

Chapter 8

Mitigation Approaches for Mobile Sources

8.1 Summary of Key Messages

- In the United States, mobile sources accounted for 52% of total BC emissions in 2005, approximately 93% of which came from diesel vehicles or engines. On a global basis, mobile sources are responsible for approximately 19% of BC emissions, with total mobile source emissions and the percentage attributable to mobile sources both significantly lower in developing countries.

- In the United States, new engine requirements have resulted in a 32% 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 86% reduction in BC emissions from mobile sources is projected from 2005 to 2030, leading to a total decline of 90% in BC emissions between 1990 and 2030. 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.

    - 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 $14,000/ton (2010$) based on prior EPA rulemakings.

- Mobile source BC emissions in other developed countries have been declining rapidly since the 1990s due to regulations on PM emissions from new engines, mainly diesel trucks, and substantial further emissions reductions are expected by 2030 and beyond. Internationally, other developed countries have and are continuing to adopt emission standards (including those for diesel engines with ultra low sulfur fuel) similar to EPA emission standards, which also results in harmonization of standards. However, standards for new engines lag behind in some regions.

- Of the on-highway and nonroad diesel engines currently in operation in the United States, many of which will remain in operation for the next 20 to 30 years, there are approximately 11 million legacy fleet engines that are emitting PM at elevated levels compared to new engines.

- For policymakers seeking additional BC emissions reductions beyond those that will be achieved as a result of the new engine regulations already in place, there are currently available, cost-effective diesel retrofit strategies that can reduce harmful emissions from in-use engines substantially.

    - DPFs in a retrofit program for in-use vehicles can reduce PM emissions by up to 99%, at a cost of $8,000 to $15,000 for passive DPFs, and $20,000 to $50,000 for active DPF systems. However, not all engines are good candidates for DPFs because of old age or poor maintenance. Other cleaner engine strategies include engine repowers, engine upgrades, and replacement of the engine (sometimes including the vehicle or piece of equipment). EPA's National Clean Diesel Campaign has provided grant funds to support diesel engine retrofits, repowers, and replacements.

    - Other strategies to reduce emissions from existing engines include improved fleet maintenance practices, idle reduction programs, advanced aerodynamics, more fuel efficient tires and more efficient supply chain management strategies, including shifts in mode of transportation. EPA's SmartWay Transport Partnership is designed to encourage industry to adopt these best practices for reducing emissions and improving fuel economy.

    - Internationally, retrofit programs present significant financial and logistical challenges. This is particularly true in developing countries, where infrastructure is lacking to assist with vehicle registration, inspection and maintenance programs, technology certification/verification programs, and application of readily available technologies. Vehicles in these regions tend to be older and less well-maintained than in developed countries, and the availability of low-sulfur diesel fuel is limited. In addition, the costs of DPFs may be prohibitive for some countries.

8.2 Introduction

A number of $PM_{2.5}$ control strategies have proven successful in reducing BC emissions from mobile sources, which represent one of the most important categories of BC[1] emissions globally, especially within developed countries (see Chapter 4). The two principal strategies include: (1) emissions standards for new vehicles and engines, with emissions reductions occurring as the vehicle and engine fleet turns over, and (2) controls or strategies that reduce emissions from existing in-use engines, such as diesel retrofits. In this chapter, these two major strategies are explored, with emphasis on describing the anticipated impact of these approaches on emissions by 2030. It is important to note that these strategies are complementary, and can be employed simultaneously. The joint application of new engine standards and controls on in-use engines has been very successful in both the United States and Europe in reducing direct PM emissions—including BC—from mobile sources.[2]

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

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

8.3 Emissions Trajectories for Mobile Sources

As discussed in Chapter 4, mobile sources remain the dominant emitters of BC in developed countries. In the United States, for example, mobile sources were responsible for about 52% of BC emissions in 2005, almost all of which (93%) came from diesel vehicles or engines. If wildfire emissions are excluded, then mobile sources account for 69% of the 2005 domestic inventory. On a global basis, mobile sources are responsible for approximately 19% of the BC (Bond et al., 2004) with total emissions and percentage attributable to mobile sources both significantly lower in developing countries. A number of studies have projected that these emissions are likely to increase globally in the future, largely due to growth in the transportation sector in developing countries (Streets et al., 2004; Jacobson and Streets, 2009) (see Chapter 7). However, mobile source BC emissions in developed countries have been declining rapidly since the 1990s. Regulations on (PM) emissions from new engines, particularly in the United States and Europe, have been effective in reducing emissions of BC from on-road vehicles (mainly diesel trucks), and nonroad diesel engines, locomotives, and commercial marine vessels, although Europe has not currently adopted stringent locomotive and commercial marine standards as the United States has. Substantial emissions reductions are expected over the next two decades and beyond.

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[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.

[2] Roughly 98% of the exhaust PM emitted from mobile sources is 2.5 microns or smaller in size. This is true for both diesel and gasoline vehicles/engines. All exhaust particulate from mobile diesel sources is commonly referred to as "diesel PM" and this convention is used in this chapter. These emissions do not include secondary PM (SOA, nitrates, sulfates) formed from mobile source emissions in the atmosphere or tire and brake wear emissions.

In the United States, new engine requirements have resulted in a 32% 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 86% reduction in BC emissions from mobile sources is projected from 2005 to 2030, leading to a total decline of 90% in BC emissions between 1990 and 2030 as shown in Table 8-1. Most of these reductions are concentrated in the diesel fleet. For example, from 1990-2005, there was a 30% decline in BC emissions from diesel trucks. Due to new regulations, a further 95% decline is projected in diesel truck BC emissions by 2030 (97% total decline since 1990). Other categories of diesel engines, such as nonroad diesels (e.g., agricultural, construction equipment), commercial marine diesels (excluding ocean going vessels), and locomotives are also projected to have major declines (75-92%) in BC emissions from 2005 to 2030 in the United States. BC emissions from gasoline vehicles and nonroad gasoline engines, which are much smaller sources of BC, are projected to decline by 80% during 1990-2030 time period, with a 23% reduction occurring from 2005-2030. Most of that reduction will come from on-road gasoline vehicles due to the use of catalysts that decrease PM.[3,4]

Considering only the emissions from U.S. mobile sources occurring north of the 40th parallel in 2005, EPA estimates there will be a substantial decline of approximately 85% in these emissions by 2030 as well. As discussed in Chapter 4, emissions from sources in northern latitudes are of particular interest, due to the proximity of these emissions to the Arctic and the greater likelihood of transport to that sensitive region. However, the projected decline in mobile source emissions north of the 40th parallel does not reflect potential future increases in emissions from marine freight transport that may occur under future climate scenarios. The total or seasonal loss of Arctic sea ice may result in new marine trade routes through the Arctic. Such developments could potentially result in greater emissions in the Arctic, with greater potential for deposition on remaining ice. U.S. emissions inventories currently contain no projections of these potential future emissions in the Arctic area. However, some studies have been done of emissions from shipping and aircraft in the Arctic area (Corbett et al., 2010; Wilkerson et al., 2010).

Table 8-1 shows the emissions reductions in BC (as well as $PM_{2.5}$ and OC) going from 1990 through 2030 for various mobile source sectors which are discussed in the following sections. The basis for the emissions inventories here is discussed in the mobile source section of Appendix 2. The numbers are based largely on the MOVES and NONROAD models, which represent EPA's projections for emissions reductions that will occur as a result of the engine and tailpipe emissions regulations already promulgated by EPA, but do not include any additional emissions reductions that would occur as a result of engine retrofits or replacements. Also, Figure 8-1 shows the reductions in BC graphically from 1990 through 2030.

8.4 New Engine Standards in the United States

In the United States, PM emissions standards for new mobile source engines are being phased in across different sectors between 2007 and 2020, mostly for diesel engines. These standards will lead to the large reductions in mobile source emissions of BC illustrated in Table 8-1.[5] The realized reductions depend on the rate of fleet turnover—i.e., the rate at which older vehicles and engines are replaced with new vehicles that comply with the latest emissions standards. The rate of fleet turnover depends heavily on the type of vehicle or engine, with on-road engines such as passenger cars and light-duty trucks being replaced more frequently than some other types of mobile sources, such as nonroad equipment. The state of California has its own diesel PM standards as promulgated by the California Air Resources Board (CARB). These standards are, in general, similar if not identical to the Federal standards. CARB also has its own gasoline PM standards. A detailed list of the mobile source PM standards is contained in Appendix 5.

The emission standards and/or control technology cited below to reduce PM (and thus BC) emissions do not include programs such as increased use of

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[3] 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. The use of catalysts on these vehicles to decrease HC, CO, and NO_x also results in substantial PM and BC reductions. In general, BC emissions from gasoline vehicles and engines have been less studied than those from diesel engines.

[4] 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.

[5] 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.

Table 8-1. Mobile Source BC, OC, and $PM_{2.5}$ Emissions 1990-2030 (short tons). (Source: U.S. EPA)

Source Category	Year				% Change	
BLACK (ELEMENTAL) CARBON	**1990**	**2005**	**2020**	**2030**	**1990→2005**	**2005→2030**
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	31,254	9,356	-25%	-92%
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[a]	283	410	457	553	45%	35%
Total BC Emissions (Mobile)	**487,628**	**333,400**	**99,940**	**47,557**	**-32%**	**-86%**
ORGANIC CARBON						
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[a]	1,372	1,988	2,217	2,682	45%	35%
Total OC Emissions (Mobile)	**418,088**	**205,172**	**132,252**	**134,025**	**-51%**	**-35%**
DIRECT $PM_{2.5}$						
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[a]	2,178	3,156	3,519	4,257	45%	35%
Total $PM_{2.5}$ Emissions (Mobile)	**985,671**	**626,859**	**270,559**	**228,318**	**-36%**	**-64%**

[a] Non landing and take-off (LTO) emissions not included; also, planned technology and operations improvements that require funding for implementation are not included in the forecast.

electrification (either for light-duty vehicles using hybrids or electric vehicles or, more importantly, truck stop electrification which reduces idling of the diesel truck engine and use of auxiliary power units on heavy-duty trucks which are typically small diesel engines). They do not include benefits from reduced idle programs or other transportation control measures (such as reduced commuting, increased

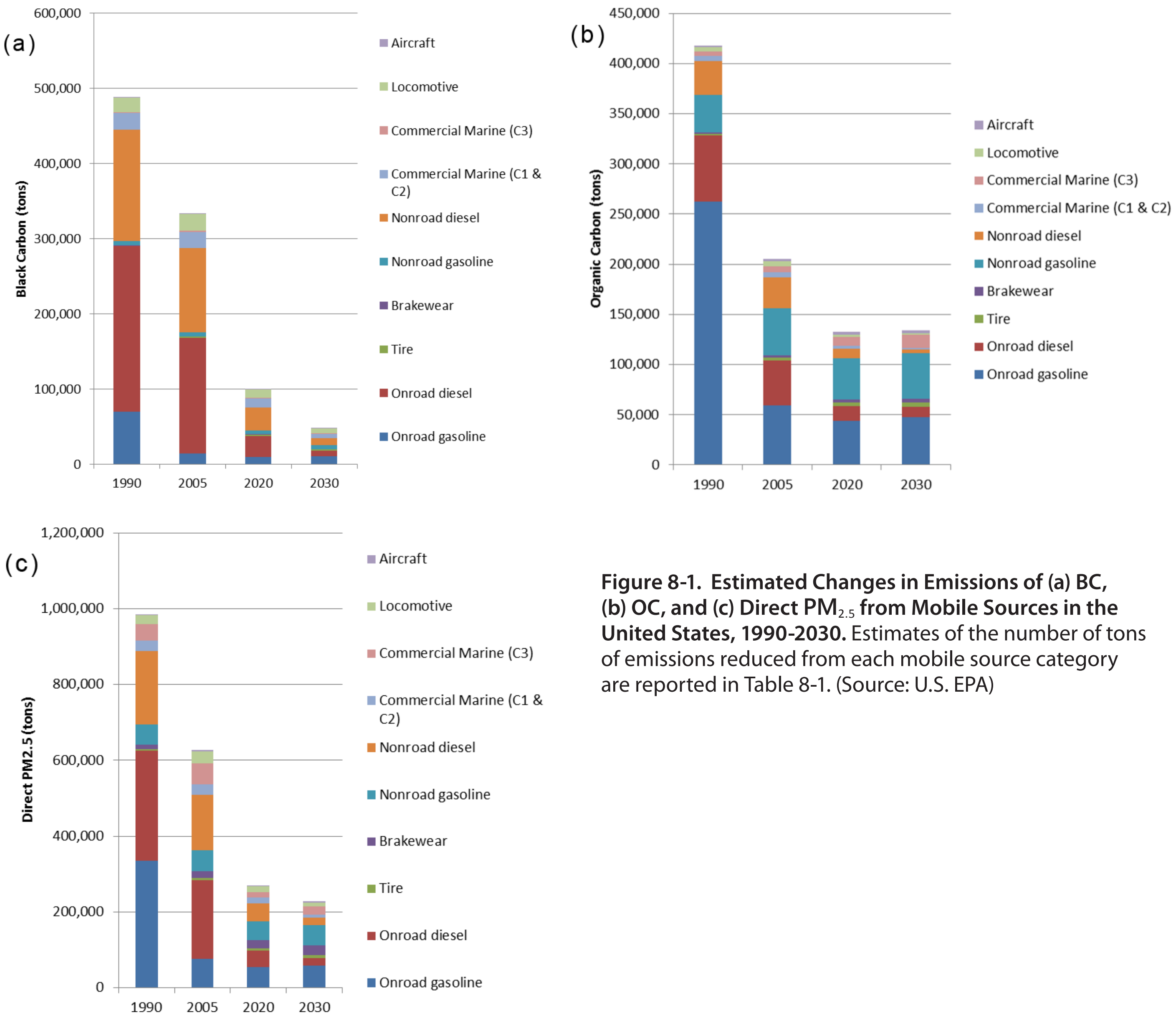

Figure 8-1. Estimated Changes in Emissions of (a) BC, (b) OC, and (c) Direct $PM_{2.5}$ from Mobile Sources in the United States, 1990-2030. Estimates of the number of tons of emissions reduced from each mobile source category are reported in Table 8-1. (Source: U.S. EPA)

use of mass transportation, increased bicycling/walking). These types of programs are discussed more generally in a later section of this chapter.

The reductions in BC also do not consider how BC would be affected by future fuel economy standards such as those for light-duty vehicles (which are mostly gasoline-powered and thus a smaller source of BC emissions) and diesel vehicles (which are mostly heavy-duty trucks and a larger source of BC emissions). EPA has issued light-duty vehicle fuel economy standards effective for the 2012-2016 model years. EPA also just issued final rulemaking for heavy-duty vehicle fuel economy standards for the 2014-2018 model years. Additional fuel economy improvements for light-duty vehicles for model years 2017-2025 have recently been proposed.[6] Basically, these standards will not increase BC emissions.

These rulemakings and other forces in general will result in changes in vehicle technology. The introduction of and increased use of electric vehicles is certainly already occurring. There have been several studies (Jacobson and Delucchi, 2011; Delucchi and Jacobson, 2011) examining alternative energy sources including one on providing worldwide energy (for electric power, transportation, heat/cooling) by wind, water, and sunlight on a widespread basis in the 2030-2050 time frame. These alternative power sources could

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[6] See *http://www.epa.gov/otaq/climate/regulations.htm*.

greatly reduce emissions of PM and BC. Also, it will be important to determine the effect from increased use of biofuels on BC emissions, which is currently an area of significant uncertainty.

8.4.1 On-road and Nonroad Diesel Engines

Diesel PM, as it exits the engine, is 70-80% BC for the pre-2007 model year diesel trucks and current diesel nonroad engines (excluding commercial marine oceangoing vessels which are discussed separately). The main source of diesel PM has traditionally been heavy-duty diesel trucks with gross vehicle weights from 8,501 to 80,000 lbs. The first standards controlling diesel PM for on-road engines were standards for visible smoke (which has some correlation with PM) effective with the 1970 model year followed by increasingly stringent PM mass standards starting with the 1988 model year. For the 2007 vehicle (engine) model year, stringent emission standards of 0.01 g/BHP-hr (grams per brakehorsepower/hour – a standard unit for emissions from heavy-duty mobile source engines) became effective for heavy-duty diesel engines, which represents over 99% control from a pre-control diesel engine in the 1970 time frame.[7]

As a result of these standards, BC emissions have been dramatically or even preferentially reduced as the major PM constituent.[8] To meet these stringent PM standards, virtually all new on-highway diesel trucks in the United States, beginning with the 2007 model year, have been equipped with DPFs. DPFs typically eliminate more than 90% of diesel PM and can reduce BC by as much as 99%. The type of DPFs typically used on new model year vehicles are called "wall flow" filters with a catalyst coated on a ceramic monolith with the exhaust flowing through the filter walls trapping the PM and allowing the exhaust gases to flow through. The trapped PM is then oxidized by reaction with compounds such as oxygen and nitrogen dioxide on the catalyst surface. This technology preferentially removes solid particles such as BC. BC emissions from the heavy-duty diesel truck fleet have been reduced by 30% from 1990-2005, and EPA projects that the application of DPFs will result in a further 95% reduction by 2030, from 153,477 tons to 7,615 tons. EPA's earlier rulemakings concluded that use of DPFs separate from the overall emission control system could result in a minimal fuel economy penalty (~1%) due to additional pumping work to force the exhaust gases through the DPF at high engine loads, but that the overall fuel economy impact would be neutral due to optimization of the complete emission control system. This was one of the primary reasons the Agency took such a systems approach. Now that the heavy-duty on-highway program is fully phased-in, some manufacturers are claiming a 5-6 percent fuel economy improvement through the use of integrated emission control systems. Additionally EPA and NHTSA projected that these overall optimized emission control systems could be further improved as part of the technology packages engine manufacturers are projected to use to comply with the Agencies' recently finalized Heavy-Duty Fuel Efficiency and Greenhouse gas rulemaking.

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

Nonroad diesel engines also emit a significant amount of BC. EPA's first emission standards for PM for these engines began in 1996. Recent rules issued in 2004, to be effective with the 2012 calendar year for newly manufactured engines, will result in widespread use of DPFs with dramatic reductions (~ 99% from a pre-control engine) in PM and BC. These standards will be fully phased in around 2015 for new model year nonroad diesel engines but will be phased into the fleet some years later with fleet turnover. EPA's latest version of the NONROAD model calculates the effect of all of these regulations, including those resulting in use of DPFs. EPA calculates a 92% decrease in emissions between 2005 and 2030, from 112,058 tons of BC in 2005 to 9,356 tons in 2030, despite substantial expected growth in use of these engines over this time period. Cumulatively, this will be a 94% decrease from 1990 to 2030.

A general note is that the recent down turn in the economy (not accounted for in these projections) can result in lower fleet turn-over than seen historically for on-road light-duty vehicles and

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[7] 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.

[8] Ultrafine particles (generally those smaller than about 0.10 microns in size) from pre-2007 diesel engines generally comprise primarily 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).

trucks. This can also be an issue with nonroad engines. These changes by themselves would increase emissions since increased numbers of older vehicles or engine are being used. Also, a shift in travel patterns and freight movement can occur, such as altered use of intermodal freight facilities. Economic downturns may also reduce total usage for both on-road and nonroad vehicles, which would reduce total emissions. Similarly, increases in fuel prices and land-use patterns will affect transportation patterns. Also, any shift in travel patterns and freight movement such as altered use of intermodal freight facilities would affect BC emissions. Finally, it is important to note that the total emissions reductions achieved will depend on the extent to which older vehicles/engines officially retired from service are still utilized for limited purposes in the United States or are exported to other countries (especially in Central and South America) for continued use.

As mentioned briefly in the introduction to this chapter, an important prerequisite for the application of DPFs is a switch to low-sulfur fuel. Low-sulfur fuel is needed, and has been required in the United States by regulation, to preserve catalytic activity of the emission control system, which is poisoned by sulfur. In issuing diesel PM regulations for on-road heavy-duty vehicles, nonroad diesels, and commercial marine (categories 1 and 2)/locomotives, EPA determined that the emission standards being required could be met only with use of ultra low sulfur diesel fuel. Specifically, sulfur interferes with the ability of the DPF to passively regenerate. For NO_x control with urea selective catalytic reduction (SCR), sulfur compromises low-exhaust temperature NO_x reduction performance. Fuel sulfur also results in sulfate PM due to catalytic oxidation of sulfur oxides over the DPF, which increases PM. Noncatalytic diesel particulate filters that would be compatible with higher sulfur diesel fuels are harder to regenerate (i.e., removal of accumulated diesel particulate in the filter) and are not as effective for PM control. They also do not control the organic fraction of PM as effectively and, thus, do not meet stringent PM standards (U.S. EPA, 2001). Such filters though could be among possible control technologies for larger commercial marine diesels (category 3) which use heavy fuel oil instead of conventional diesel fuel; these engines are discussed later.

EPA first regulated sulfur content in on-road diesel fuel to 500 ppm in 1993, resulting in typical fuel sulfur levels of about 300 ppm. Prior to that, the sulfur level in on-road diesel fuel was about 2,000 ppm. In 2006, the sulfur level was limited to 15 ppm for on-road diesel fuel and has been reduced gradually in nonroad diesel fuel, first to 500 ppm in 2007 for all categories except ocean-going vessels, and, starting in 2010, to 15 ppm for most categories. In the case of locomotive and marine diesel fuel (for categories 1 and 2 marine diesel), this second step will occur in 2012. Thus, all highway diesel vehicles and nonroad engines in the United States must now or will soon operate on "ultra-low sulfur diesel" (ULSD). Typical in-use fuel sulfur levels are about 7 ppm. Of course, as discussed later, fuel for the larger C3 marine (such as heavy fuel oil, HFO) diesel (ocean-going) engines has significantly higher sulfur levels and would not be suitable for diesel particulate filters.

It is important to note that the net climate impact of the application of DPFs will be offset somewhat by the necessary co-emissions reductions in sulfate, which is reflecting (cooling).[9] Also, while diesel PM from pre-2007 engines has a high level of BC in PM, it also has some OC (about 22%), which is also greatly reduced by the DPF in later model years. The net climate impact of the application of DPFs will be affected by these reductions in OC emissions. Still, given the predominance of BC in diesel exhaust (70-80%), emissions reductions from this source category have a strong likelihood of providing climate benefits.

The EPA nonroad diesel rule[10] issued in 2004 provides an aggregate cost estimate for controlling PM emissions using DPFs on new engines of about $14,000 per ton ($2010). This cost figure includes the additional cost of ULSD fuel, engine costs, any changes in maintenance costs, and equipment costs. As shown in Table 8-2, similar cost estimates were developed in 2001 for the Heavy-Duty Diesel Rule

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[9] 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 SO_2 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 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 levels.

[10] *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*).

Table 8-2. Cost Estimates for Particulate Matter Controls on New Diesel Engines (2010$), based on Recent U.S. EPA rulemakings. These costs include the additional cost of requiring Ultra Low Sulfur Diesel Fuel. (Source: U.S. EPA)

Rule	Estimated Cost (2010$) Per Ton $PM_{2.5}$ Reduced	
	NPV, 3% rate	NPV, 7% rate
Heavy-Duty Diesel Rule (2001)	$16,652	$19,216
Nonroad Diesel Rule (2004)	$13,762	$14,461
Locomotive/Marine Rule (2008)	$8,579	$9,778

for on-road[11] and the 2008 rule controlling emissions from locomotive and marine diesel engines.[12] It is important to note that the values reported in Table 8-2 are adjusted from the original values developed by EPA to 2010$ as a function of GDP to ensure consistency with other costs presented in this Report. A large fraction of the cost is due to the requirements for ultra low sulfur diesel fuel.

It is important to note that the controls applied under these regulations affect multiple pollutants, not just BC. At this time, there is no methodology to allocate these costs specifically to BC or other PM components but it is useful to note that for these diesels the BC is the largest PM component. Furthermore, the analyses conducted during the 2001-2008 time frame utilized the best cost information available at that time, as well as emissions reductions (total tons reduced) based on EPA's then-current emissions models. Since then, the emissions models have changed so that the reductions estimated in the earlier rulemaking analyses would be somewhat different today. The magnitude of the reductions was determined doing emissions inventory estimates for given years both with and without the standard being considered in effect. One cannot obtain the tons reduced by given standard just from emissions inventory data for a given year compared to another year since the total reduction reflected in the inventory from one year to another is the result of all the standards in place (and vehicle/engine turn over) for all mobile sources rather than just a single standard for a particular category. Also, the inventory and cost numbers used in these calculations have not been updated since they were obtained. In the absence of new analysis, the $14,000 cost/ton (the average costs in Table 8-2) is the best available EPA information for control of diesel PM from newly manufactured on-road vehicles and nonroad engines meeting EPA emission standards. The total costs and benefits of these regulations are discussed separately in Chapter 6. As an aside, EPA cost estimates made in rulemakings tend to be higher than the actual cost due to improvements in technology to meet the standard that were not considered when the rule was issued (Anderson and Sherwood, 2002).

8.4.2 On-road and Nonroad Gasoline Engines

On-road gasoline PM emissions have decreased dramatically over the years, especially with the use of catalysts and unleaded gasoline starting with the 1975 model year vehicles. For example, PM emissions for a typical car using leaded gasoline in 1970 were about 0.3 g/mile compared to emissions from current vehicles with unleaded fuel of about 0.001 g/mile, a reduction of over 99% (Coordinating Research Council, 2008). While BC emissions were not usually measured in the PM from cars in the 1970s, some limited measurements suggest that BC made up about 10-20% of the PM at that time, compared to about 20% of PM mass in 2005. Thus, the per-vehicle PM reductions since 1970 have resulted in a substantial reduction in BC emissions. Most of this BC comes under "rich" operating conditions (where there is insufficient air for full combustion, such as during cold-start or high load conditions). EPA's most recent modeling indicates that BC emissions from on-road gasoline engines have declined 79% since 1990, from 69,629 in 1990 tons to 14,510 tons BC in 2005, and will decline a further 31% by 2030 (to 10,027 tons).

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[11] *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).

[12] *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*).

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

It should be noted that most current technology vehicles now emit below the Tier 2 PM standard by a significant margin. However, a relatively new technology, gasoline direct injection (GDI), is being utilized for a number of reasons such as improved fuel economy and performance. GDI engines differ from conventional fuel injected engines in that the fuel is injected directly into the cylinder (like in a diesel engine) rather than at the intake port. GDI vehicles are expected to constitute a major part of the new vehicle fleet in the coming years and may be 90% of new vehicle sales in model year 2016. The specific technology for injecting and guiding the gasoline spray into the engine coupled with the catalyst may have an impact on the magnitude of the PM emissions. Recent studies performed by EPA determined that some “wall guided” GDI engines perform slightly worse than currently produced “port fuel injection” (PFI) engines with respect to PM but that “spray guided” GDI engines perform on par with PFI engines. Indications are that most manufacturers utilizing GDI technology will be migrating to “spray guided” designs, but regardless EPA anticipates future GDI designs will perform on par with or better than current technology.

CARB has issued a preliminary discussion paper discussing the option of tightening the PM mass standard effective for the 2015 model year (California Air Resources Board, 2010). The present CARB PM standard (LEV II) is 0.01 g/mile, which is also the EPA emission standard. The possible standard presented in the discussion paper is 0.003 g/mile starting in 2017. A 0.001 g/mile standard is being considered starting for the 2025 model year. CARB had also considered a standard specifically for BC, but announced at its November 2010 LEV III (Low Emission Vehicle) workshop that it would not set such a standard.

Nonroad gasoline engines are either 2-stroke engines (where lubricating oil is mixed into and burned with the gasoline) or 4-stroke engines. The 2-stroke engines are smaller engines and tend to be used more in lawn and garden equipment, such as handheld string trimmers; they have also been used in lawn mowers and snow blowers. They can also be used in recreational marine, although most engines there are now 4-stroke engines. The 4 stroke engine is used in equipment such as lawn mowers, small generator sets, industrial equipment, and recreational equipment such as marine engines. These engines emit significant PM mass, especially the 2-stroke engines, where the PM has a large contribution from the lubricating oil. They can also be used in larger equipment such as farm and construction equipment although, here, the dominant engine type is diesel. EPA estimates that BC emissions from nonroad gasoline engines will decline approximately 5% (from 5,444 tons to 5,174 tons) between 2005 and 2030, largely due to changes needed to meet standards for volatile organic compounds (VOC), CO, and NO_x emissions standards being applied to several categories of nonroad gasoline engines, which will also reduce PM. Current information, which needs to be updated, used in EPA air quality modeling suggests that BC is approximately 10% of PM mass with the same number being used for both 2-stroke and 4-stroke engines even though 2-stroke engines have oil added to gasoline. PM emissions from nonroad gasoline engines, particularly the 2-stroke engines, have been characterized far less thoroughly than emissions from on-road gasoline vehicles, and EPA’s estimates of BC emissions are highly uncertain. EPA places a high priority on obtaining better BC emissions data from both 2-stroke and 4-stroke nonroad gasoline engines.

8.4.3 Other Mobile Sources – Commercial Marine Vessels, Locomotives, and Aircraft

Locomotives have used diesel (diesel electric) engines (both 2-stroke and 4-stroke engines) predominantly since the 1950s. EPA has implemented several tiers of emission standards for PM for these engines with the most recent set of standards to be effective in 2015. These newest standards will result in the use of DPFs on new locomotives which preferentially reduce BC. In addition, national emission standards require that older locomotives that are remanufactured must be certified to more stringent emission standards than their prior certification level.

Commercial marine vessels are classified as C1, C2, and C3 based on engine size. C1 marine engines are similar in size (less than 5 l/cylinder or for some categories less than 7 l/cylinder) to those used in construction/farm equipment. C2 marine engines (between 5 or 7 and 30 l/cylinder) are similar to locomotive diesels. The C3 engines (greater than 30 l/cylinder vessels) are similar to those used in some power plants and are used in ocean-going vessels. The most recent set of emission standards for these engines will result in most new C1 and C2 commercial marine engines having DPFs starting in 2014. Ultra low sulfur diesel fuel is being required for these engines. For these engines, there will be a dramatic drop in PM emissions and an even more dramatic drop in BC emissions. Like locomotives, older marine diesel engines must be certified to more stringent emission standards upon remanufacturing, compared to their previous certification level. The level of the standards to which these remanufactured engines must be certified varies depending on engine type and year of manufacture for the original engine.

PM emissions from C3 engines comprise mainly sulfate (about 75%) and relatively little BC (can be less than 1% although as discussed in Appendix 2 this percentage can vary significantly). Due to recent work with the International Maritime Organization (IMO), there will be large reductions in the higher sulfur level of the fuel (largely bunker diesel fuel composed of especially high molecular weight, even solid, hydrocarbon compounds) used in these engines (see Appendix 4). As this sulfur level is reduced, PM will be greatly reduced but BC levels are expected to stay roughly the same on a per-vessel basis and will constitute a larger percentage of the PM emissions. There is some increase in BC emissions from 2005 due to an increase in usage of these vessels. Though C3 marine is responsible for less than 1,000 tons of BC emissions for the entire United States, there is some concern that emissions from these vessels could have disproportionate impact on the Arctic, especially if Arctic marine traffic increases as shipping lanes open due to ice melt in the region. Additional BC emissions data and modeling/deposition studies are needed to clarify the impact of C3 marine vessels.

C3 marine usually uses heavy fuel oil (HFO), which can be solid at room temperature (and is heated before going into the engine), rather than the conventional distillate diesel fuel used by C1/C2 commercial marine and other nonroad diesels. HFO contains higher molecular weight hydrocarbon compounds than conventional distillate diesel fuel. This affects the characteristics of the PM emissions. As is also discussed in Appendix 4, HFO contains higher fuel sulfur levels and cannot be used with DPFs.

There has been limited research into the BC emissions from aircraft. Additional characterization of aircraft emissions would help to improve understanding of BC emissions from aircraft, although there is sufficient information to develop a PM inventory and an initial BC and OC inventory.

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

8.5 New Engine Standards Internationally

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

8.5.1 International Regulations of Diesel Fuel Sulfur Levels

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

The United Nations Environment Programme's (UNEP) Partnership for Clean Fuels and Vehicles (PCFV), founded at the World Summit on Sustainable Development in 2002, promotes low sulfur fuels and cleaner vehicle standards and technologies. This partnership has over 100 members from the oil and gas industry, engine and retrofit manufacturers, government agencies, and environmental NGOs. Currently, the PCFV is conducting a low sulfur campaign with a call for global adoption of 50 ppm sulfur gasoline and diesel. The implementation of 50 ppm sulfur programs would allow countries to begin to deploy DPFs, which would produce significant reductions in $PM_{2.5}$ and BC. However, the U.S. EPA believes a further reduction to sulfur levels at or below 15 ppm is needed for DPFs to function for their intended lifetime. Further detail on the diesel sulfur reduction activities of countries outside the United States, Canada, Japan, and the European Union is provided in Appendix 4. Most of the actions underway in other countries focus on fuels for on-road vehicles. Sulfur limits for nonroad diesel fuel are also needed on an international basis to facilitate BC control. It is important to note that the cost to provide the ULSD fuel will vary from one country to another depending on fuel supplies and refinery capabilities. Thus, while the benefits of low sulfur fuels and advanced emission control technologies far outweigh the costs, the often substantial upfront costs of upgrading existing refineries presents a challenge for many governments.

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

8.5.2 Standards for New Engines Outside the United States

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

In Europe, DPFs were first applied to light-duty diesels; these requirements are relatively recent, with the latest standards, known as Euro 5, becoming effective in 2009. Nonroad diesels will start to phase in DPFs starting with what are termed Stage IIIB standards in 2011. The nonroad reductions will be followed by Euro 6 on-road heavy-duty diesel standards which will require DPFs on all new trucks starting in 2013. Likewise some locomotive engines will have DPFs by 2011 although commercial marine diesels are not regulated.[13]

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. China is following the European emission standard progression with some time lag; however, China has not yet implemented low sulfur fuel nationwide to enable widespread use of DPFs. 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. In addressing the future impact of possible standards, it is important to account for both the vehicle/engine standards and growth in the number of vehicles/engines as well as increases in usage (such as vehicle miles traveled).

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[13] The European standards use the PMP (particle measurement program) methodology with thermal treatment (catalytic oxidation) to remove volatile particles 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 U.S. test procedure. With the PMP it is important to distinguish between particle mass and particle number where organics, which tend to be small in size, make a contribution. The treatment of organics is an important distinction for PM control and may affect the control technology used, which could affect BC reductions.

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

For control of BC from C3 marine internationally, EPA is working with the IMO, the Arctic Council, and others to recommend what can be done to better define and reduce BC from C3 engines in international waters. Such work would include developing a definition for BC emissions from international shipping. It would also include considering measurement methods for BC and identifying the most appropriate method for measuring BC emissions from international shipping. It would also include investigating appropriate control measures to reduce the impact of BC emissions from international shipping in the Arctic. Control measures that can be evaluated include speed reductions, improved routing/logistics, vessel, propeller and engine modifications, DPFs (such as non-catalytic ones that could be used with higher sulfur fuel), water-in-fuel emulsification, use of slide-valves, and possibly alternative fuels (MEPC, 2010; UNEP and WMO, 2011a). Some of these measures have been discussed in a recent research article (Corbett et al., 2010) and an earlier Arctic Council report (Arctic Council, 2009). Finally, the effect of using a distillate diesel fuel (similar to what is used for diesel trucks and nonroad diesels) versus HFO on BC emissions should be investigated. Use of a distillate fuel is expected to result in less organic emissions and could increase the BC/PM ratio although the total mass of BC emitted might decrease.

8.6 Mitigation Approaches for In-use Mobile Sources in the United States

Though emissions standards for new engines will reduce emissions over time, existing engines can remain in use for a long time (20 to 30 years) (U.S. Census Bureau, 2004). Opportunities to control BC emissions from in-use vehicles center almost exclusively on diesel engines. Despite EPA's diesel engine and fuel standards taking effect over the next decade for new engines, in-use diesel engines will continue to emit large amounts of PM and BC, as well as other pollutants such as NO_x, before they are replaced. For this reason, strategies to reduce emissions from in-use engines have received a great deal of attention, particularly because communities near freight corridors and other large concentrations of diesel-powered engines are disproportionately affected by the pollution. EPA's NCDC estimates that Diesel Emission Reduction Act (DERA) funding could be used to apply in-use mitigation strategies to 11 million of the on-highway and nonroad engines now in the U.S. diesel fleet.

A variety of strategies are available to reduce substantially harmful emissions from in-use vehicles, and many of these strategies are cost-effective given the health benefits associated with reducing PM emissions. As used by EPA, the term diesel retrofit includes any technology or system that achieves emissions reductions beyond that required by the EPA regulations at the time of new engine certification. Diesel retrofit projects include the replacement of high-emitting vehicles/equipment with cleaner vehicles/equipment, repowering or engine replacement, rebuilding the engine to a cleaner standard, installation of advanced emissions control after-treatment technologies such as DPFs, or the use of a cleaner fuel (U.S. EPA, 2006a).

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

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

8.6.1 Available Retrofit Technologies and Strategies for In-use Engines

8.6.1.1 Diesel Exhaust After-treatment Devices

Typically, after-treatment diesel retrofit involves the installation of an emission control device to remove emissions from the engine exhaust. This type of retrofit can be very effective at reducing PM emissions, eliminating up to 99% of BC in some cases. Of the diesel retrofit devices currently available, DPFs most effectively reduce BC. For the sake of completeness, various diesel retrofits are covered below.[14] Further information is available from NCDC, including a table of emissions reductions and typical costs for various diesel retrofits.[15] EPA and CARB adhere to rigorous verification processes to evaluate the performance and reliability of available retrofit technologies. These processes evaluate the emission reduction performance of retrofit technologies, including their durability, and identify engine operating criteria and conditions that must exist for these technologies to achieve those reductions. Federal funding under the NCDC requires recipients to use EPA or CARB-verified diesel retrofit technologies for clean diesel projects.

As previously mentioned, DPFs are wall-flow exhaust after-treatment devices that are effective at significantly reducing diesel PM emissions by 85% to 90% and BC emissions by up to 99%. Because BC exits the engine in solid particle form, DPFs can reduce BC up to 99%. The small amount of PM that does penetrate a DPF is composed of mainly sulfate and OC. DPFs typically use a porous ceramic, cordierite substrate, or metallic filter to physically trap PM and remove it from the exhaust stream. The collected PM is oxidized primarily to CO_2 and water vapor during filter regeneration. Regeneration can be passive (via a catalyst) or active (via a heat source) and is necessary to keep the filter from plugging and rendering the engine inoperative. Regular engine maintenance is essential to DPF performance.

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

For large, on-highway trucks, retrofitting passive DPFs generally costs between $8,000 to $15,000, including installation, depending on engine size, filter technology and installation requirements. Active DPF systems can cost $20,000 for a heavy duty diesel truck and up to $50,000 for a large piece of nonroad equipment. Vehicle inspection, data logging, and backpressure monitoring systems are required with each installation; these costs along with installation of the device, are typically included in the cost of the DPF (U.S. EPA, 2010a). However, operating costs incurred due to application of DPFs are not included in the estimates above. Operating costs could include the differential cost for using ULSD, fuel economy impacts related to increased exhaust backpressure, or changes to maintenance practices related to the use of retrofit technologies. There is no increased cost for use of ULSD in the United States because ULSD is now the predominant diesel fuel used in both highway and nonroad applications. In addition, data from existing retrofits show no significant difference in fuel economy for fleets with and without these retrofit technologies.[16,17]

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

- Partial Diesel Particulate Filters (PDPFs) provide moderate (around 30% to 50%) reduction of PM from diesel exhaust. However, while limited test data exists on the effectiveness of PDPFs to reduce BC, it is likely that these devices result in minimal BC reductions (UNEP and WMO, 2011). PDPFs typically employ structures to briefly retain particles for oxidation, structures to promote air turbulence and particle impaction, and catalysts to oxidize diesel particles. Partial flow filters are capable of oxidizing the soluble organic

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[14] See *http://www.meca.org/cs/root/diesel_retrofit_subsite/what_is_retrofit/what_is_retrofit*.

[15] See *http://www.epa.gov/cleandiesel/technologies/retrofits.htm*.

[16] These cost estimates are from NCDC's Cost-Effectiveness Paper 2006, updated to 2010 dollars.

[17] NREL Ralph's Grocery study at: *http://www.nrel.gov/docs/fy03osti/31363.pdf* and/or Clean Air Task Force (2009b). The carbon dioxide-equivalent benefits of reducing black carbon emissions from U.S. Class 8 trucks using diesel particulate filters: a preliminary analysis. Available on the Internet at *http://www.catf.us/resources/publications/view/100*.

fraction of diesel exhaust and likely some BC. As of October 2010, only three PDPF technologies were verified by CARB (none by EPA), and these were only verified for transport refrigeration units (TRU). These devices cost about $4,000-$8,000 per unit.

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

8.6.1.2 Cleaner Engine Strategies

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

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

Vehicle/Equipment Replacement: When no diesel retrofit solutions can be cost-effectively implemented for a particular vehicle or piece of equipment, the option exists to retire the vehicle/equipment from service before the end of its useful life and replace it with a newer model. While this is typically the most expensive method of reducing emissions, this can be the most feasible strategy for a particular vehicle or piece of equipment. For example, significant emissions reductions can be achieved by scrapping older model drayage trucks at ports and replacing them with newer, clean diesel trucks. One benefit to replacing an entire vehicle or piece of equipment is that newer models often have improved non-engine systems and parts that are preferred by operators.

8.6.1.3 Other Emissions Reduction Strategies

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

Improved Fleet Maintenance Practices: Since a small percentage of vehicles in a given fleet may be responsible for a majority of the fleet's emissions,

Local Retrofit Projects in the United States

Agricultural Vehicle Repowers

The Air Pollution Control District in San Joaquin Valley received $2 million to repower 33 pieces of agricultural equipment 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 over 160 tons and PM by nearly 6 tons.

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 equipment 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.

one of the first steps for reducing emissions is to take an inventory and inspect vehicles and equipment. This information may be used to identify vehicles in need of repair and find candidates for other mitigation options. Repair of poorly operating engines typically decreases emissions and improves fuel economy. Furthermore, regularly performed maintenance will extend the life of vehicles and equipment (Partnership for Clean Fuels and Vehicles, 2009). For example, many manufacturers prescribe that engines be rebuilt after accumulating a set number of hours of use. An engine rebuild involves replacing some old parts and restoring durable parts to original factory specifications. In some cases, an after-treatment technology could be installed at the time of engine rebuild. This would save time since the vehicle or equipment would not need to be removed from service any longer than prescribed for normal maintenance.

Cleaner fuels can lead to BC reductions via multiple pathways. As previously stated, ULSD fuel is necessary for diesel particulate filters and other after-treatment 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 or LNG. 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. As previously stated, it will also be important to determine the effect from increased use of biofuels such as biodiesel on BC emissions.

Another form of fuel switching is electrification. As previously stated in this report, power plant supplied electricity has extremely low emissions of BC in the United States. 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 allows marine vessels to shut down their engines and run normal operations by plugging into electrical connections at docks. When a vessel is at berth, it typically runs its auxiliary diesel engines to provide power for normal operations (referred to as hotelling). For example, CARB has estimated that 1.8 tons per day of diesel

PM was emitted by approximately 2,000 hotelling ocean-going vessels in California in 2006 (Regional Planning Organization, 2006). Hotelling emissions can be dramatically reduced if the vessel uses "shore power" electricity while at port. It should be noted that emissions of pollutants from other sources should be considered when pursuing this and other alternative fuels/energy sources. For example, electrification shifts the emissions from the mobile source to the power plant.

Fuel economy improvements may yield reductions in BC. Some fuel savings devices, such as low-rolling-resistance tires and aerodynamic technologies (e.g., trailer gap reducers, trailer boat tails, and trailer side skirts) reduce fuel use with little change to engine operation. These fuel saving devices likely result in PM reductions; however, additional research is needed to quantify the emission reductions. Hybrid vehicles are potential technologies for CO_2 reductions, but further research is necessary to determine the extent of PM or BC reductions.

Idle reduction: Long-duration idling of truck and locomotive engines consumes an estimated 1 billion gallons of diesel fuel annually, resulting in thousands of tons of PM, a significant fraction of which is BC (i.e., 15-40%) (Gaines et al., 2006; Lim, 2002). It is important to consider that while BC is a significant fraction of overall diesel PM, BC/PM ratios differ during idling. The reduction in PM due to idling has definite health benefits, and the reduction in fuel use results in reduced CO_2 emissions and, in turn, climate benefits. However, the net climate benefit due to reduction in idling PM is less understood. Furthermore, idling increases fuel and engine maintenance costs, shortens engine life, increases driver exposure to air pollution, and creates elevated noise levels. Idle reduction programs and technologies are already prevalent in the US. They serve as one of the simplest and lowest cost methods to reduce emissions from engines. Because reducing idling reduces engine operation, emissions of all pollutants are lower. Strategies for reducing idling include both operational and technological methods. Examples of on-board truck technologies include:

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

Off-board technologies include truck stop electrification, which provides conditioned air and electricity to truck cabs for accessory loads while at a truck stop. These systems also may provide telephone, cable TV, and internet access. A majority of U.S. states and many municipalities have anti-idling regulation in place to limit idling of vehicles (American Transportation Research Institute, 2011).

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

8.6.2 Cost-Effectiveness of Retrofits

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

In 2009, EPA published a Report to Congress, *Highlights of the Diesel Emission Reduction Program*, which provides information on the overall cost-effectiveness of various diesel emissions reduction strategies funded under the Diesel Emissions

Reduction Act. The Report estimates that the average cost-effectiveness of the DERA projects funded in 2008 ranged from $9,000 to $27,700 per ton of $PM_{2.5}$. According to this analysis, which is currently being updated, diesel retrofit strategies compare favorably with other emissions reduction strategies used to attain national ambient air quality standards that range from $1,000 to $20,000 and as high as $100,000 per ton of $PM_{2.5}$ on an annualized basis. However, most diesel retrofit strategies are less cost-effective than regulatory programs designed to set PM emissions standards for new diesel engines, such as the emissions standards for 2007 and later model year heavy-duty highway engines.

8.6.3 Applicability of Diesel Retrofits

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

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

Currently, PM mitigation strategies for marine and locomotive engines are limited. No DPFs are verified or certified by federal or state agencies for these engines. Therefore, upgrading/replacing engines and fuel switching are currently the two most viable mitigation strategies for these engines. Fuel switching could also include the use of shore power for larger marine vessels, which eliminates local PM emissions while ships are at port. New emissions reduction technologies are being developed to reduce locomotive and marine emissions. For example, marine engine upgrade kits have been implemented with funding support from the EPA Emerging Technologies Program.[18]

8.6.4 Experience with Diesel Emissions Reduction Programs in the United States

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

8.6.4.1 National Clean Diesel Campaign (NCDC)

The National Clean Diesel Campaign (NCDC) is a partnership that aims to accelerate the implementation of emissions control strategies in the existing fleet through approaches such as retrofitting, repairing, replacing, repowering, and scrappage of diesel vehicles and equipment; reducing idling; and switching to cleaner fuels. This ten-year effort by EPA to bring together industry, environmental groups, local and State governments and Federal programs has resulted in significant experience with various fleet types and technologies and reduced emissions from thousands of engines. Several initiatives through the Campaign have targeted specific sectors, such as Clean School Bus USA and the clean ports program, demonstrating a variety of technologies and strategies on those fleets.

In 2005, a dedicated source of funding was authorized by Congress for implementation of NCDC projects on a wider scale. The Energy Policy Act of 2005[19] provided EPA with grant and loan authority to promote diesel emissions reductions from the existing in-use fleet in the United States and authorized appropriations of up to $200 million per year to the Agency under the DERA provisions for FY2007 through FY2011. The DERA Program may serve as one of the best avenues and foundations for reducing BC emissions in the United States (U.S.

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[18] See *http://www.epa.gov/cleandiesel/projects/projects.htm*.

[19] *http://www.gpo.gov/fdsys/pkg/PLAW-109publ58/pdf/PLAW-109publ58.pdf*

EPA, 2009a). Congress appropriated $169.2 million in funding under this statute in FY 2008 through FY 2010. In addition, the American Recovery and Reinvestment Act of 2009 allotted the NCDC $300 million. The Diesel Emissions Reduction Act of 2010 was signed into law in January 2011. This law authorizes DERA for $100 million per year from FY2012 through FY2016.

DERA offers a funding vehicle for immediate BC reductions within the in-use fleet. The first year of DERA funding reduced emissions from more than 14,000 diesel-powered highway vehicles and pieces of nonroad equipment. DERA funding supported a wide range of verified technologies, cleaner fuels, and certified engine configurations, such as repowers, replacements, idle-reduction technologies, biodiesel, and retrofit devices such as DPFs. DERA funding also supported diesel programs in state governments.

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

From 2008-2010, EPA received applications requesting more than $665 million, which equates to $7 for every $1 available in clean diesel funding. Thus, there remains strong interest in utilizing DERA to reduce diesel emissions. Additionally, a large number of high emitting engines remain currently in use. In moving forward with the program, a few challenges remain. For example, there are too few verified technologies for nonroad and marine engines and older diesel trucks, limiting the extent of achievable emissions reductions. The nonroad sector offers another challenge because of the number and diversity of nonroad engine types, the range of horsepower and the varying usage and duty cycles of the equipment.

Because BC is a regional pollutant, EPA, through the DERA Program, provides assistance to state and local governments in developing their own clean diesel programs. This includes targeting current nonattainment areas where clean diesel strategies can assist in meeting local emissions reduction goals and providing high quality data to states that depend on the performance of diesel emissions reduction strategies in their air quality plans. In addition, EPA conducts in-use testing—confirming the performance of verified technologies in the field—while working cooperatively with industry groups, engine manufacturers, and state agencies to expand the list of clean diesel technology options.

8.6.4.2 SmartWay

In 2004 EPA launched its SmartWay Transport Partnership. SmartWay is an innovative, voluntary partnership between EPA and private industry to reduce fuel use and emissions from goods movement. SmartWay promotes fuel-saving technologies and emission control technologies; some technologies—like idle reduction or newer truck replacements—do both. Since most cargo-hauling large trucks, locomotives, barges, and other freight vehicles use diesel fuel, and these vehicles remain in the legacy fleet for decades, reducing fuel use and emissions from goods movement and the legacy fleet can have a major impact on diesel emissions, including emissions of BC.

SmartWay provides shippers as well as truck carriers with standardized tools and approaches to assess, benchmark, track and reduce fuel use and emissions from goods movement. SmartWay offers technical assistance to enable partners to improve performance. The program offers incentives (SmartWay logo eligibility, SmartWay partner ranking, recognition of top performers) to encourage continual improvement. To enable this improvement, SmartWay helps its shipper and carrier partners identify fuel-saving operational and technical solutions through its technology program. This technology program researches and evaluates fuel-saving technologies, develops standardized protocols for the measurement of technology improvements (e.g., fuel consumption, aerodynamic impacts, long-duration idle reduction), and officially verifies the benefits of certain technology types (i.e., long-duration idle reduction technologies, low rolling resistance tires, and aerodynamic components).

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

Nearly 3,000 companies, from small firms to Fortune 500 companies, belong to SmartWay. To date, these SmartWay partners have saved $6.1 billion dollars by cutting their fuel use by 50 million barrels of oil. This is equivalent to taking 3 million cars off the road for an entire year. Improving supply chain efficiency helps these companies grow the economy, protect and generate jobs, cut imports of foreign oil, contribute to energy security, and be good environmental stewards.

In developing new national standards to bring cleaner, more efficient trucks to market, EPA and the Department of Transportation's National Highway Traffic Safety Administration (NHTSA) drew from the SmartWay experience. This experience includes developing test procedures to evaluate trucks and truck components and determining how these features and components perform. While focused on American freight-efficiency, SmartWay has responded to industry demand to recognize the importance of the global supply chain by expanding its tools and building the capacity for SmartWay-based programs in other countries.

8.6.4.3 Congestion Mitigation and Air Quality Improvement Program (U.S. DOT)

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

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

8.6.4.4 State Programs

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

Several states have passed legislation similar to California's. New Jersey has instituted a mandatory retrofit program requiring owners of diesel vehicles to retrofit with best available retrofit technology (BART). The state reimburses vehicle owners/operators for all expenses. New York has also instituted a mandatory retrofit program that applies to all heavy-duty state-owned and contractor vehicles.

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

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

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[20] *http://www.tceq.texas.gov/airquality/terp.*

8.7 Mitigation Approaches for In-use Mobile Engines Internationally

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

EPA has often advised other nations and supported international demonstration projects in an effort to transfer information and technologies to those that seek to reduce emissions from mobile sources. Additionally, EPA's diesel retrofit experts have advised and participated in several pilot retrofit projects where diesel trucks and buses were fitted with various exhaust after-treatment devices. Low-sulfur diesel was obtained for the projects in most cases. The projects have shown generally that, if appropriate fuel is provided and engine maintenance is addressed, DPFs are viable options to reduce PM (and thus BC) on some vehicles. Following a relatively small EPA supported pilot project in Beijing in 2006, city authorities went on to retrofit more than 6,000 vehicles with active DPFs prior to the Beijing Olympics. That number is now above 8,000 and growing. EPA has also assisted in retrofit projects in Mexico City, Bangkok, Santiago, and Pune (India).

As noted earlier, the SmartWay program recognizes the importance of the global supply chain and has shared its program and technology expertise with other countries. EPA hosted a SmartWay International Summit in December of 2008 to offer guidance to numerous countries which are also developing freight sustainability programs. As a result of that Summit and other capacity building and information sharing, multiple countries and regions have gone on to implement SmartWay-like programs. Mexico, Canada, France and Australia have each developed and launched freight sustainability programs using SmartWay templates for tools and program design, partnership structures and best practices. Additionally, a consortium of SmartWay Partners and other businesses in the European Union have beta-tested SmartWay tools with the intent of developing a SmartWay platform for implementation throughout the region.

More recently, EPA has collaborated to help China develop multiple freight sustainability projects utilizing SmartWay technologies and program design elements. EPA first provided technical expertise for the Green Truck Pilot in Guangzhou in 2009. The World Bank funded the retrofitting of SmartWay technology on local trucks and demonstrated notable fuel savings. Based on those results, the World Bank secured funding from the Global Environmental Facility for the Guangdong Green Freight Demonstration Program. This $18 million project will implement truck retrofits and upgrades using SmartWay technologies and financing methods, as well as logistical improvements, driver training and capacity building for governmental officials. The Ministry of Transport and the China Sustainable Energy Program are developing a nationwide freight efficiency program, built in part on these pilot projects.

Chapter 9

Mitigation Approaches for Stationary Sources

9.1 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 20% 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 \$48/ton $PM_{2.5}$ to \$685/ton $PM_{2.5}$ (2010\$) 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.

9.2 Introduction

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

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

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[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.

9.3 Emissions from Key Stationary Source Categories

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

Stationary sources account for a slightly higher percentage (20%) of total worldwide BC emissions, and more than 30% of BC emissions from contained combustion (i.e., sources other than open biomass burning) (see Table 4-6 and Bond et al., 2004). In certain developing world regions, such as China and India, stationary sources represent a very significant percentage of the BC inventory. The regions with the highest percentage of "contained" BC emitted from industry and power generation are China, the former USSR, India, and central/South America (Zhang et al., 2007). Key source categories include brick kilns, coke production/iron and steel production, and industrial boilers. As discussed in Chapter 7, however, BC emissions from industrial sources are expected to decline worldwide under most scenarios. This decline is anticipated to occur in developing countries as well as developed countries.

In the United States, direct emissions of PM and BC from stationary sources have been reduced significantly due to improved combustion efficiencies in industrial operations and implementation of federal and state clean air regulations over the past several decades. This declining emissions trend is expected to continue as further reductions will be needed to meet revised air quality standards and mitigate adverse effects on public health and the environment. EPA's modeled emissions inventory projections indicate that direct PM emissions from stationary sources are expected to decline by about 20% between 2005 and 2020. For example, sources in nonattainment areas will be required to implement emissions reduction strategies to help areas attain the 1997, 2006, and any future revisions to the $PM_{2.5}$ NAAQS. Certain facilities will also be required to comply with revisions to maximum achievable control technology (MACT) and new source performance standards (NSPS) for specific source categories. These standards will lead to control of some sources that currently do not have any PM controls; they will also lead to improved levels of control for certain sources that already have PM controls. Older power generation sources may be retired as well. However, in an overall sense, near-term BC emissions reductions from domestic stationary sources are expected to be modest when compared to expected reductions in other sectors, such as the mobile source category.

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

ICI boilers are a wide-ranging category of combustion units that collectively can burn a wide variety of combustible fuels, including coal, oil, natural gas, and biomass. There are thousands of ICI boilers across the country, varying in size from a few million Btu/hr for small commercial or industrial units to over 10 million Btu/hr for large boilers. Their operations range from intermittent to near-steady state. Most large units are covered under new regulations issued in April 2011 that include stringent standards for PM, mercury, and certain hazardous air pollutants, although EPA is currently reconsidering these standards.[2] Under the proposed reconsideration of this rule issued in December 2011, EPA has projected that the new emissions limits applicable to major source boilers and process heaters would reduce $PM_{2.5}$ emissions from these sources by 41,200 tons by 2014.

Stationary engines burning diesel fuels also account for substantial BC emissions. These engines are similar to mobile diesel engines and typically use the same fuels, but they can also operate using natural gas or heavier fuel oil grades than mobile diesel engines. They are used to perform a range of different tasks, such as pumping water or oil

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[2] The April 2011 final rule is available at *http://www.epa.gov/airquality/combustion/actions.html#feb11*. EPA issued a proposed reconsideration of this rule on December 2, 2011 (see *http://www.epa.gov/airquality/combustion/actions.html*), and expects to issue a revised final rule in 2012.

through pipelines, operating equipment in remote locations, or providing backup power generation.

Many other categories of industrial sources emit relatively low amounts of BC. In the current U.S. inventory, the "natural gas combustion" sector appears to have substantial BC emissions, but this is likely due to severe constraints on the data used to generate these estimates.[3] Given current knowledge of the utility and major source boiler inventory and the mechanisms of BC formation, EPA does not believe that there are significant BC emissions from natural gas combustion sources with good combustion practices. It is recommended that additional source testing and research be conducted to improve current emission factors associated with natural gas combustion. It is also recommended that additional source testing and research be conducted on the related category of oil and gas flaring (see additional discussion in section 9.7.4 below).

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

9.4 Available Control Technologies for Stationary Sources

This section provides an overview of the main technologies for reducing $PM_{2.5}$ emissions from stationary sources. Several post-combustion PM control technologies have been in operation for many years and have been demonstrated to be quite effective in reducing $PM_{2.5}$. These technologies are also considered to be relatively effective at controlling BC because BC is a filterable component of $PM_{2.5}$. Many studies to date have assumed that $PM_{2.5}$ control technologies will reduce similar fractions of $PM_{2.5}$ and BC mass. However, it has also been recognized that reduction efficiency declines to some extent as particle size decreases (and BC particles are commonly smaller than 1 micrometer in diameter). For this reason, it is recommended that additional source testing and research be conducted on stationary sources to better understand control efficiencies for BC and to develop improved emission factors for specific source categories.

The two most effective control technologies for $PM_{2.5}$ (and therefore for BC) are fabric filters (sometimes called baghouses) and ESPs. Although there are other technologies used to reduce emissions of PM (such as cyclones and Venturi scrubbers), they are often designed to control larger particles (PM_{10} and larger), and therefore are considered to be less effective in terms of BC mitigation. EPA provides a thorough overview of the principles of operation, design variations, applicability, performance, and associated costs of fabric filters and ESPs in the 2002 EPA Air Pollution Control Cost Manual (see U.S. EPA, 2002b).

9.4.1 Fabric Filters

A fabric filter unit consists of one or more isolated compartments containing rows of fabric bags in the form of round, flat, or shaped tubes, or pleated cartridges. Particle-laden gas passes up (usually) along the surface of the bags then radially through the fabric. Particles are retained on the upstream face of the bags, and the cleaned gas stream is vented to the atmosphere. The filter is operated cyclically, alternating between relatively long periods of filtering and short periods of cleaning. During cleaning, dust that has accumulated on the bags is removed from the fabric surface and deposited in a hopper for subsequent disposal. Fabric filters are not recommended for boilers burning oil because particles from oil combustion are sticky and tend to clog the filter.

A properly designed and well run baghouse will generally have extremely high particle collection efficiencies (i.e., greater than 99.9%). Baghouses are particularly effective for collecting fine particles from power generation and a range of industrial facilities. For example, tests of bag houses on two utility boilers (Broadway and Cass, 1975; Cass and Broadway, 1976) showed efficiencies of 99.8 % for particles 10 μm in diameter and larger and 99.6% to 99.9% for particles 2.5 μm in diameter and smaller. Studies have shown that collection efficiencies greater than 99% can be achieved for particles less than 1 μm in diameter (NESCAUM, 2005; Buonicore and W.T. Davis (eds.), 1992). A recent report for the U.S. Forest Service on the applicability of different PM emissions control technologies to small wood-fired boilers found that mechanical collectors, such as multicyclones, were only modestly effective in reducing PM emissions, with an average of about

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[3] The current AP-42 emissions factor for BC from natural gas combustion is considered to be highly questionable. Bond et al. (2006b) 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.

15% control efficiency. In the Forest Service study, fabric filters achieved 74% reduction of $PM_{2.5}$, even with some of the uncontrolled flue gas circumventing the baghouse (Hinckley and Doshi, 2010).

9.4.2 Electrostatic Precipitators

An ESP is a particle control device that uses an electrical charge to move the particles out of the flowing gas stream and onto collector plates. Appropriately designed ESPs are effective at removing particles from sources operating at high temperatures and having large volumes of gas. They operate on many types of facilities that emit PM and BC. ESPs typically achieve greater than 99% PM removal efficiency (depending upon the design parameters of the specific unit), although the removal efficiency is generally lower for submicrometer size particles such as BC. Smaller particles are more easily carried by the gas stream, and therefore the ESP collection efficiency for very fine particles like BC is typically lower than the efficiency for removing larger particles (i.e. greater than 1 micrometer in diameter). In general, fabric filters are more effective than ESPs at removing BC.

To address the lower ESP collection efficiency on submicrometer particles, a hybrid PM collection system can be employed. Some designs place the baghouse downstream of an existing ESP to improve overall collection efficiency. Others integrate the ESP and baghouse components. This type of system can achieve 99.99% control of all particle sizes from 0.01 to 50 micrometers (Zhu, 2003).

Sources such as biomass combustors that also generate significant levels of condensable PM may benefit from wet ESP designs, in which the collector surfaces are washed with water (either continuously or intermittently) to clean the particles from the collectors. Tubular ESPs are most commonly used for operations where the PM is either wet or sticky (U.S. EPA, 2002b, Chapter 1). Typical applications include sulfuric acid plants, coke oven by-product gas cleaning (tar removal), and recently in iron and steel sinter plants. Because wood combustion systems in particular can produce PM that is sticky, tubular ESPs may be appropriate for use in small systems for reduction of PM and BC.

A relatively new technology known as an agglomerator can also be used in conjunction with a control device (such as an ESP) in utility or industrial applications. This technology is installed in the high velocity ductwork leading to the control device. It pre-treats the dust particles prior to entering the device, agglomerating small and large particles together, thereby making it easier for the control device to collect the larger particles. It has been shown to improve the ESP collection efficiency of very fine particles (less than 1 micrometer in size) by 75-90% (Truce and Wilkison, 2008).[4] There are a number of commercial installations of the Indigo Agglomerator technology in place; most installations are upstream of an ESP, but this technology has also been successfully operated in conjunction with a fabric filter or wet scrubber.

Wet particulate scrubbers are generally not appropriate for control of BC. Collection efficiencies for wet scrubbers vary with the particle size distribution of the waste gas stream. In general, collection efficiency for submicrometer particles is much lower than for ESPs or fabric filter systems. Submicrometer particle collection efficiencies for wet scrubbers typically are on the order of 50% or less, although cyclonic wet scrubbers may be able to remove as much as 75% of submicrometer particle mass (U.S. EPA, 2002b).

9.4.3 Diesel Particulate Filters and Oxidation Catalysts

There are more than a million stationary diesel engines in use today and together these sources have substantial emissions of PM and NO_x. For most diesel engines, BC is a significant component of untreated exhaust; these emissions can be reduced through DPF technology.

DPFs were originally developed for mobile engine applications (see Chapter 8). They include variations such as diesel particle traps and catalytic and noncatalytic soot oxidation systems. These units typically involve mechanical filtering of soot particles and a mechanism for oxidation of the soot to CO_2. This second step is sometimes referred to as regeneration, and eliminates the need for collecting and disposing of the captured particles. Catalysts are used to enhance the oxidation process. Depending upon the design and operation of the DPFs, removal efficiencies of between 40% and 99% can be achieved (van Setten et al., 2001).

To ensure optimal performance of DPFs and to avoid poisoning of the catalyst, the diesel engine should burn fuel with low sulfur content. DPFs have been identified by the California Air Resources Board (CARB) as a verified technology for stationary engines serving prime and emergency standby generators and pumps.[5] In some situations, such as where loads are not transient and exhaust

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[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*.

temperatures are high enough, non-catalytic DPFs can be used with fuels having a higher sulfur content.

EPA issued new source performance standards in July 2006 (71 FR 39153) for new compression ignition (CI) stationary internal combustion engines. These standards implemented new restrictions on emissions of PM, NO_x, VOC, and CO as well as new limits on the level of sulfur permitted in diesel fuel. In June 2011, EPA further strengthened these standards, establishing stringent new performance standards for stationary CI engines with a displacement of 10 to 30 liters per cylinder (76 FR 37954), consistent with recent revisions to standards for similar mobile source marine engines.[6]

9.5 Cost-Effectiveness of PM Control Technologies

The cost-effectiveness of an emissions control device for a stationary source is often expressed in terms of dollars per ton of pollutant reduced. Factored into this amount are capital costs (amortized over several years) for design and installation of the control equipment, and annual costs for operating and maintaining the equipment. Because many emission standards for stationary sources to date have included emission limits for total filterable PM (as opposed to $PM_{2.5}$ or BC), many of the cost-effectiveness values found in published reports are expressed in terms of the cost per ton of reducing total PM. It has been noted earlier that we assume that BC emissions will be reduced with the operation of a fabric filter or ESP on a stationary source. However, it is acknowledged that actual control efficiencies for capturing submicrometer BC particles are uncertain, and they are likely to be somewhat lower than the assumed control efficiencies for total PM or $PM_{2.5}$. For this reason, additional research and source testing is needed to develop improved measurement techniques and development of robust emission factors for specific source categories.

The effectiveness of a given control technology used for a specific source category will depend not only upon the performance of the particular technology, but also upon the level of control that is already in place. For instance, most large coal combustion sources, such as EGUs, are likely to be well controlled to comply with prior PM emission standards. In contrast, smaller and older coal combustion units that have not been subject to similar emission standards may in some cases be able to reduce PM emissions for a lower dollar cost per ton because installation of the same technology will remove a greater mass of PM (including $PM_{2.5}$ and BC) compared to a well-controlled EGU. Therefore, some sources that have been completely exempt from PM control because of their age, small size, or limited operation (such as certain distillate oil or coal combustion systems) may present favorable mitigation opportunities. Thus, a reasonable and cost-effective mitigation strategy requires detailed knowledge of the sources and their emissions, on both a per-source basis and across the full population of those sources.

The 2002 EPA Air Pollution Control Cost Manual and several related 2003 control technology fact sheets provide typical cost effectiveness ranges for PM reduction by fabric filters and ESPs. The cost-effectiveness range identified for a fabric filter was $51 to $462 per ton (2010$); for an ESP it was $48 to $685 per ton (2010$).

Table 9-1 presents information on PM control cost ranges as adapted from multiple sources referenced in a 2009 NESCAUM report for ICI boilers (NESCAUM, 2009). Capital costs can vary significantly depending on the source specific characteristics. For some large utility boilers (500 megawatts), a fabric filter can require an investment on the order of $70 to 105 million (2010$) (NESCAUM, 2010). Cost-effectiveness values per ton are commonly higher for oil combustion units than for those burning coal or wood. In general, however, PM control technologies are well-established.[7]

The reduction of BC and $PM_{2.5}$ from industrial categories can provide significant public health benefits, particularly for communities located close to emissions sources. For example, recent regulations on different sizes of ICI boilers in the U.S. were estimated to provide $73,000 - $294,000 in health benefits per ton $PM_{2.5}$ reduced (2010$) (U.S. EPA, 2011a).[8] This suggests that controlling BC

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[6] In addition, the action revises the requirements for engines with displacement at or above 30 liters per cylinder to align more closely with recent standards for similar mobile source marine engines, and for engines in remote portions of Alaska that are not accessible by the Federal Aid Highway System. See *http://www.epa.gov/ttn/atw/nsps/sinsps/fr28jn11.pdf*.

[7] It should also be noted that control of PM and BC results in millions of pounds of particulates being captured and disposed of as solid waste; and in some cases it is discharged as part of wastewater discharges. The problems associated with disposal of coal combustion residues are well recognized, and it will be important to manage such wastes effectively in the future.

[8] For major boilers: $73,000 to $182,000 per ton of directly emitted fine particles (range from Pope to Laden, 3% discount rate, 2014 analysis year, 2010$). For area boilers: $122,000 to $294,000 per ton of directly emitted fine particles (range from Pope to Laden, 3% discount rate, 2014 analysis year, 2010$). See *http://www.epa.gov/ttn/ecas/regdata/RIAs/boilersriafinal110221_psg.pdf*, Tables 7-2 and 7-3.

Table 9-1. PM Control Costs for ICI Boilers. (Source: U.S. EPA, based on data from (NESCAUM, 2009). Costs have been updated to 2010$.)

Fuel Type	Technology	PM Reduction Potential	Size of ICI Boiler (mmBTU/hr)	Capital Costs (Million, 2010$)	Cost Effectiveness ($/ton removed, 2010$)	Reference
Coal	Dry ESP	90–99%	250	$3.5 – 47	$184 – 1,529	MACTEC (2005)
Coal	Fabric Filter	90–99%	250	$2.4 – 27	$498 – 1,183	MACTEC (2005)
Oil	Dry ESP	90–99%	250	$2.4 – 26	$2,739 – 24,715	MACTEC (2005)
Wood	ESP	99.5%	Medium	—	$239 – 344	STAPPA (2006)
Wood	Fabric Filter	99.5%	Medium	—	$173 – 293	STAPPA (2006)

emissions from industrial sources may be highly cost-effective and should remain a part of any overall BC reduction strategy.

9.6 Mitigation Approaches Other than PM Control Technologies

9.6.1 Process Modification/Optimization

As a product of the combustion process, BC can be reduced by approaches other than direct reduction using PM control devices. Process modification and/or optimization can be an effective means of reducing PM emissions. Some general examples of process optimization include reducing the frequency of mass transfer operations, improving operational efficiency, and the proper use of dust collection devices at the point of generation.

Cost values for these approaches are difficult to estimate. Often, steps to improve operational efficiency require only investments in instrumentation or operator training to yield ongoing reductions in fuel consumption and emissions of all pollutants, including BC. Changes in fuel can be more expensive and are incurred over the entire period in which they are used, but are dependent upon fluctuating and often highly localized market conditions. However, for many situations, particularly smaller boilers for which the costs of control technology investment and operation would make up a significant fraction of the system's initial capital and operating cost, conversion to fuels (such as natural gas) that generate lower PM and BC can be less expensive than use of post-combustion control equipment.

One specific example technique to reduce PM emissions for existing boilers is a boiler tune-up. Fuel usage can be reduced by improving the combustion efficiency of the boiler. At best, boilers may be 85% efficient and untuned boilers may have combustion efficiencies of 60% or lower. As combustion efficiency decreases, fuel usage increases to maintain energy output resulting in increased emissions. Lower combustion efficiency also results in formation of PM constituents like BC that are formed from incomplete combustion of the fuel. The objective of good combustion is to release all the energy in the fuel while minimizing losses from combustion imperfections and excess air. A tune-up can make a significant difference in energy consumption and emissions levels.

9.6.2 Fuel Substitution and Source Reduction Approaches for PM

The type of fuel and process has a great impact on PM emissions from combustion. Coal, oil, and natural gas are the most common fuels used. Of these fuels, coal combustion generally results in the highest PM emissions. As noted earlier, increased use of biomass fuels may also lead to higher BC emissions unless suitable control techniques are applied.

Fuel substitution can be an effective means of reducing PM emissions for many industrial fuel combustion processes that generate process heat or electricity. Switching to fuels that generate lower levels of PM and BC per Btu can be a viable alternative. In addition, fuel switching can lead to cleaner and safer unit operation. However, there are several factors to consider when evaluating whether fuel switching would be an appropriate option.

Fuel-switching can often be implemented for a lower capital investment than add-on control technologies. In some situations, the age of a boiler or space constraints may make fuel-switching more cost-effective. It is important to consider, however, that the lower capital cost must be weighed against a change in fuel prices, such as would be incurred by switching from coal to natural gas. The actual cost of converting to a different fuel that reduces BC emissions must account for the cost of installing the necessary fuel feed systems, fuel price differential, and changes in non-fuel maintenance and operational costs. Other considerations include the cost of extraction and associated environmental impacts. Even switching between types of coal can lead to additional capital costs in the event new coal pulverizers or a larger ESP is needed due to differences in coal hardness and fly ash resistivity.

Fuel substitution for the purpose of reducing BC emissions can also reduce emissions of SO_2, NO_x, and CO, depending upon the characteristics of the original and replacement fuels. A common conversion is from coal to natural gas, with coal to distillate oil an appropriate alternative. Switching from distillate oil to natural gas is also a possible approach for reducing BC emissions, but the reductions in BC for such a change will be less than when switching from coal to natural gas. Switching to natural gas from either coal or oil will likely result in significant SO_2 emissions reductions.

When considering fuel switching to reduce BC effects on climate, the impact on CO_2 emissions is obviously a consideration. Conversion from coal to natural gas will reduce CO_2 as well as BC. A further alternative may be to switch to a biomass-based fuel oil. The bulk of liquid biofuels appropriate for use in boilers is in the form of biodiesel, although there have been some evaluations of other biomass-based fuel oils developed specifically for use in boilers (Partanen and Allen, 2005; Adams et al., 2002).

9.7 Mitigation Approaches for Stationary Sources Internationally

As discussed earlier in this section, stationary source emissions of PM are generally considered to be well-controlled in most developed countries due to the operation of common control technologies such as fabric filters and ESPs. The picture is different in developing countries, where a number of specific industrial source categories have been identified as important contributors to BC emissions. The source categories of concern vary by country and region of the world. Mitigation opportunities exist in these countries and regions because known control technologies exist and have been demonstrated to be effective. This section will address the source categories that have been identified in the emissions inventories as being major contributors to BC emissions and for which known control technologies exist. The source categories are brick kilns, coke production/iron and steel production, power generation and industrial boilers, and oil and gas flaring.

9.7.1 Brick Kilns

Brick and masonry production in many developing countries (such as China, India, Bangladesh, Vietnam, Nepal, and Pakistan) has increased in recent decades in response to growing urbanization and increasing demand for construction materials. Currently, brick production is estimated to be growing at a rate of 4% per year.[9]

Conventional brick kilns (such as bull's trench, clamp, and intermittent downdraught kilns) generally are operated by small-scale ventures in rural areas, often with poor conditions for workers (French, 2007; Gupta, 2003). Low-quality coal and firewood are common fuels used in brick-making; in some cases, even waste fuels such as used tires are employed. These kiln designs have inefficient combustion, leading to high emissions of both greenhouse gases and PM (and associated local air pollution health effects). The inefficient operation of these kilns also leads to high fuel costs, and kiln operations have been found to contribute to localized deforestation when cheap firewood is harvested in lieu of purchasing more expensive coal to use as fuel.

The most basic BC mitigation technique is the replacement of inefficient kilns with kilns having improved, energy efficient designs, such as the vertical shaft brick kiln (VSBK), the tunnel kiln, or the hybrid Hoffman kiln (HHK). These kilns generally require less than 50% of the fuel needed for a conventional kiln (UNDP, 2007) and have been estimated to reduce PM emissions proportionally. Bond and Sun estimated that reducing emissions by switching to a more efficient kiln design can be cost-effective, in the range of \$6.5 to \$13 per ton (2010\$) of CO_2-equivalent (based on 20-year GWP) (Bond and Sun, 2005). China has taken steps over the past decade to promote the transition to the more efficient HHK in many areas. In Bangladesh, the United Nations Development Program initiated a \$25 million project in 2010 to implement 15 energy-efficient kiln demonstration projects over the next five years (UNDP, 2010).

Under a "business as usual" scenario, global BC emissions from brick kilns are expected to decline by about 11% (428 to 381 Gg) over the 2005-2030 time

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[9] See *http://www.resourceefficientbricks.org/background.php*.

period, reflecting a gradual introduction of more efficient kilns. However, the technical mitigation potential for this sector exceeds this projected reduction. Given the rapid rate of urbanization projected for coming decades in many countries, and the high fuel cost and significant health and climate impacts associated with uncontrolled brick kilns, appropriate policy options, technical assistance, and financial incentives could be considered to accelerate the transition to more efficient brick kilns.

9.7.2 Coke Production/Iron and Steel Production

Coke is a key input used in the production of iron and steel. In the coking process, coal is heated to very high temperatures for up to 36 hours in an airless furnace, and volatile carbonaceous gases are driven off. Modern plants minimize emissions by capturing the coke oven gas and using it in a separate chemical recovery process where it is refined into by-products and usually burned for heat production (RTI International, 2008). However, some small-scale plants located in developing regions or countries with economies in transition still do not capture the carbonaceous emissions from coke production. These plants in particular represent potential BC mitigation opportunities.

The global demand for coke and steel has increased significantly in the past two decades and is expected to increase significantly in the future. Bond et al. (2004) developed global emissions estimates for BC from coke production of 380,000 Gg, based on 1996 data (about 8% of estimated global “contained” emissions). It is acknowledged that this estimate was highly uncertain due to the lack of information regarding the number of polluting “beehive” or “indigenous” plants currently in operation globally. In the late 1990s, China was considered to have the largest coke production capacity of any country by far; and it continues to be responsible for more than 60% of global coke production (based on 2008 data) and more than a third of global steel production. Most coke production in China is conducted by state-owned enterprises. However, in 2004 it was estimated that smaller township and village enterprises operating less capital intensive “indigenous” plants were responsible for about 15% of the coke production in China; it is assumed that these smaller but uncontrolled plants are responsible for a majority of BC emissions from the industry (Dukan, 2010; Polenske and McMichael, 2002).

One mitigation option to reduce BC from coke plants is simply to phase out smaller uncontrolled operations. China has initiated policies to phase out certain plants with uncontrolled emissions, but the portion of the industry that has shut down or consolidated, and the extent to which emissions have declined to date, is not well characterized. Other standard mitigation options may be feasible depending on the size of the operation. Emissions generated during the “pushing” of coke from the oven to the quenching operation can be captured by a moveable hood system and then sent to a baghouse for control (with up to 98% PM reduction). Fugitive PM emissions can be minimized during other stages of the coking process with implementation of good combustion practices, frequent maintenance, and proper work practices. The use of a continuous opacity monitor on the combustion stack can help identify ovens that are in need of repair or maintenance (RTI International, 2006). BC emissions associated with larger coking operations could also be reduced via implementation of an energy recycling program to recover waste heat from the very high temperature coking process. The recovered heat would be converted to steam and used to power a generator which in turn would help provide electricity needed for plant operations, reducing the total amount of coal needed run the plant and the associated BC emissions (Polenske and McMichael, 2002).

Coke production and the iron and steel industry are important contributors to BC emissions in other regions as well. Bluestein et al. noted that emissions from uncontrolled blast furnaces in the former Soviet republics may have the potential to contribute to BC levels in the Arctic (Bluestein et al., 2008). However, additional information is needed to improve global inventories of BC from coke production. To the extent that existing sources in China and other coke producing nations are not recapturing exhaust gases, advanced technologies are readily available to reduce emissions significantly.

9.7.3 Power Generation and Industrial Boilers

The worldwide demand for energy is projected to increase substantially over the coming decades. The 2011 *International Energy Outlook* projects global energy demand to increase by 53% between 2008 and 2035. In countries outside the OECD, the increase in demand over this period is estimated to be 85%, while in OECD countries it is estimated to be 18% (USEIA, 2011). With this increased demand for power generation, increasing emissions of PM and BC can be expected unless actions are taken to implement energy conservation programs, to ensure high combustion efficiencies in power plants and industrial boilers, and to implement effective control technologies on new and existing facilities to the maximum extent possible.

Internationally, many sources of power generation (especially smaller power plants, industrial boilers, and stationary diesel engines) continue to operate without effective controls on PM and represent current opportunities for mitigation. In 2001, it was estimated that 20% of the power plants in China were operating without effective PM controls yet were responsible for 62% of the total $PM_{2.5}$ emissions from power plants. In addition, many industrial boilers in China are known to operate only wet scrubbers and cyclones, which are effective in capturing larger particles but have low fine-particle removal efficiencies (Zhang et al., 2007). In regions of the world where electricity from the grid is unreliable or not available, there is a substantial reliance on stationary diesel generators for power. For example, it is estimated that diesel generators in India account for as much as 17% of total power generation (USAID, 2010a). Diesel generators are also widely used in the Arctic region and contribute to BC deposition locally (Quinn et al., 2008). Since well-established control technologies are available to control emissions from these power generation sources effectively, additional emissions reductions can be achieved. However, further investigation is needed to determine the cost-effectiveness of control options in specific locations.

9.7.4 Oil and Gas Flaring

Natural gas is a byproduct of the oil extraction process and it is often treated as a waste gas and disposed of rather than captured for economic use. When not captured, it is either directly vented to the atmosphere or it is burned through a process called flaring. The combustion process during flaring can be inefficient and characterized by a distinct dark-colored, sooty plume. Oil and gas flaring and venting leads to significant emissions of greenhouse gases (especially methane) and a variety of other air pollutants, including BC, hydrocarbons and toxic air pollutants. Flaring can lead to significant health impacts on nearby communities. BC emissions from flaring are of particular concern if they can impact areas of snow and ice in the Arctic region.

Global estimates of pollutant emissions from flaring and venting are still quite uncertain. It has been estimated that globally the natural gas wasted due to flaring is about 5% of the total annual natural gas consumption. In 2002, the World Bank started the Global Gas Flaring Reduction initiative. Many countries are now self-reporting flaring and venting data. NOAA has also developed methodologies to estimate flaring activity through the use of satellite remote sensing data. Based on this information, the countries with the highest estimated levels of flaring are Nigeria, Russia, Iran, Iraq, and Angola (Buzcu-Guven et al., 2010). More work is needed to improve estimates of BC emissions from flaring. Mitigation of venting and flaring activities will be on ongoing challenge for the future. Reducing BC (and methane) emissions from flaring and venting activities would require expanded efforts to make use of the natural gas for power generation on site or to capture the gas so that it can be distributed and marketed. There are clear economic incentives for this. EPA is working with a number of governmental and private partners to address these issues through the Global Methane Initiative.

9.8 Technical and Research Needs

Emissions of BC from industrial sources, both domestic and international, currently represent a modest percentage of total BC emissions. In some regions, such as Asia, industrial emissions are more significant (almost a quarter of "contained" emissions) (USAID, 2010a). It is expected that over the next two decades, global emissions from the industrial category will become a greater percentage of "contained" global BC emissions as reductions occur in other sectors. The reduction of BC and $PM_{2.5}$ from industrial categories can be very cost-effective when considering the substantial health benefits they provide to local populations, in addition to broader climate benefits. For these reasons, controlling BC emissions from industrial sources should remain a part of any overall BC reduction strategy.

While this is the case, the emission factors and emissions inventories for key sectors are recognized by many experts to be uncertain, and there is a need to improve $PM_{2.5}$ and BC emission factors for industrial sectors. In some cases, only a few source tests may provide the basis for many emission factors. It is difficult to measure BC emissions that remain after control devices have treated the exhaust emissions because it is difficult to measure emission rates directly at the sources. BC emissions are extremely difficult to measure under real-world and field conditions. Our prior experience with PM control clearly indicates that some ultra-fine particles (BC and OC) are being captured in control devices for larger particles, but reduction efficiencies of control devices are generally considered to be lower for sub-micrometer BC particles than for total PM. To what extent is not well documented (Streets et al., 2001).

To help develop improved PM and BC emission factors, global inventories, and future year projections, additional source test information needs to be collected and evaluated for priority categories, both in the United States and abroad. This research would quantify the emissions of BC that pass through existing control devices into the ambient air, establish improved emission factors for different source categories, and assess the engineering modifications that can be made to these control techniques to enhance their BC capture capability. This could be facilitated by additional funding for technical and research programs, and greater collaboration between EPA, state governments, industry groups, academic institutions, and governments from other countries. Bilateral and multilateral assistance programs can also play an important role in evaluating the cost-effectiveness of BC control measures in priority world regions.

Chapter 10

Mitigation Approaches for Residential Heating and Cooking

10.1 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 (particularly gas-burning) appears to be the most effective strategy to reduce BC and OC from residential wood combustion (RWC).
  - U.S. RWC is approximately 3% of the domestic BC emissions inventory. Residential wood smoke contains $PM_{2.5}$, air toxic pollutants (e.g., benzene), CH_4, CO_2, OC, BC, and BrC.
  - EPA is currently working to establish new or revised new source performance standards (NSPS) 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 \$3,600/ton $PM_{2.5}$ reduced, while gas fireplace inserts average \$1,800/ton $PM_{2.5}$ reduced (2010\$).
  - 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.
  - According to the WHO, exposure to cookstove emissions leads to an estimated 2 million deaths each year; indoor smoke from solid fuels ranks as the six largest mortality risk factor and the fifth largest disease risk factor in poor developing countries. Reductions in exposure to these emissions likely represent the largest public health opportunity among all the sectors considered in this report.
  - The BC climate impacts from cookstoves are likely to be strong in a regional scope, and additional source testing and modeling is needed to clarify the composition of emissions from these sources and their net climate impact.
  - Cookstove mitigation activity today is difficult to quantify definitively: while the EPA-led Partnership for Clean Indoor Air (PCIA) reported that PCIA Partners sold about 2.5 million stoves in 2010, it is likely that 5-10 million "improved" stoves are sold each year by commercial entities. In addition, there are no reliable data on the quality or performance of many of these stoves and thus considerable uncertainty regarding the benefits of their use. The full market of stoves is on the order of 500-800 million homes (3 billion people); thus, significant expansion of current clean cookstove programs would be necessary to achieve large-scale climate and health benefits.

- Many improved cooking solutions exist, but all face important supply, cost, performance, usability, marketability and/or other barriers to large-scale progress. The potential climate and health benefits vary substantially by technology and fuel.
    - The performance hierarchy for improved cooking solutions appears to be as follows, in generally decreasing order for both costs and emissions performance: 1) electricity; 2) clean fuels such as LPG or ethanol; 3) advanced biomass stoves (e.g., forced air fan or gasifier stoves); 4) rocket stoves; and 5) other improved stoves. For all solid fuel stoves (3 and 4), processing the fuel into pellets or briquettes allows for greatly improved combustion with significant reductions to harmful emissions.
    - Well-designed biogas may be the cleanest, most climate-neutral (renewable) cooking solution suitable for large-scale use; solar stoves are ultimately the cleanest solution, but have not yet demonstrated an ability to reach large scales of sales or adoption.
- A number of recent developments—including the growth of a variety of promising businesses and business models; innovations in stove design, testing, and monitoring; carbon financing; research quantifying the health benefits of improved stoves; and new country-based and global efforts to address health risks—have created a real opportunity to achieve clean cooking solutions at a global scale.
    - Over the past nine years, the EPA-led PCIA has built a network of more than 540 Partners working in 117 countries to increase the use of affordable, reliable, clean, efficient, and safe home cooking and heating technologies. PCIA Partners sold approximately 2.5 million stoves in 2010, which may result in reduced indoor air pollution exposures for more than 12 million people—primarily women and children.
    - Launched in September 2010, the rapidly growing Global Alliance for Clean Cookstoves (the Alliance) is led by the United Nations Foundation and currently has over 275 partners, including 28 countries and significant U.S. government participation. The Alliance represents an enormous opportunity to rapidly increase the use of clean cooking solutions by building on the past experiences and successes of PCIA and other leaders in this field (e.g., Shell Foundation, GIZ, SNV, United Nations agencies, World Bank).
    - The Alliance's interim goal is for 100 million homes to adopt clean cooking solutions by 2020, with sales accelerating through this period. Achieving this scale of progress will not be easy—it will require significant investments, demand a coordinated global approach, and need to be based primarily on sustainable commercial businesses that produce high-quality stoves and fuels that meet local users' needs.
    - Developing globally recognized performance standards for stoves that are widely accepted by the cookstove community and adopted by country governments could spur wider development of clean cookstoves.

10.2 Introduction

Household energy use represents an extremely important source of BC emissions worldwide, accounting for 25% of the total global BC inventory. In developed countries, most of these emissions are associated with residential wood combustion (RWC), generally for heating. Total emissions from RWC in developed countries are estimated at about 4% of the total global inventory (311 Gg) and 16% of total residential emissions worldwide. In developing countries, emissions from residential combustion are more often linked to widespread use of small stoves for cooking and/or heating. These cookstoves utilize a wider range of fuels, including coal, natural gas, and dung as well as wood, charcoal, and other biomass-related fuels. The emissions from residential cookstoves represent a much larger fraction of the global inventory, accounting for 21% of total global BC emissions (1635 Gg) and 84% of emissions from residential sources worldwide. The variety of sources and fuels within the residential category, and the significant differences between developed and developing countries make this sector among the most challenging from a mitigation perspective. However, given the vast number of people dependent upon residential sources for everyday needs, such as heating and cooking, this sector likely represents the biggest opportunity for public health improvements through reductions of BC and overall $PM_{2.5}$.

This chapter is divided into two parts. First, it presents information regarding available mitigation approaches for residential wood combustion in the United States and other developed countries. There are a number of cost-effective, advanced mitigation technologies that are well known and easily deployed; the biggest challenge remains one of implementation and outreach. The chapter then examines the technologies and approaches available for reducing emissions from the residential sector in developing countries, where the scale of the problem is much broader, the range of sources and fuels more complicated, and the challenges to effective implementation much larger. It describes the advanced cookstove technologies that are currently available and their costs, and considers the emissions reduction potential if these technologies were adopted on a large scale.

10.3 Residential Wood Combustion in Developed Countries

There are an estimated 29 million wood-burning fireplaces, over 12 million wood stoves and hundreds of thousands of hydronic heaters (also known as outdoor wood boilers) throughout the United States. Emissions from these appliances contain $PM_{2.5}$, toxic air pollutants, and other pollutants that can adversely impact health and climate. The majority of these emissions come from old, inefficient wood stoves built before 1990. Wintertime wood smoke emissions contribute to $PM_{2.5}$ nonattainment and localized problems in many areas in the United States. For this reason alone, replacing inefficient wood stoves and educating wood burners on proper burn practices and stove operation are important strategies for reaching domestic air quality goals. In fact, there is far greater certainty about the public health benefits of reducing residential wood smoke emissions, both indoors and outdoors, than about the net climate impacts, especially in light of the high level of OC emissions from these sources.

10.3.1 Emissions from Residential Wood Combustion

Incomplete combustion of wood results in emissions of fine and ultrafine particles, including BC, BrC and other non-light absorbing OC particles. Inorganic materials, such as potassium, are also present in lesser quantities as part of the mix of emitted particles. In the United States, RWC contributes over 350,000 tons of $PM_{2.5}$ nationwide—mostly during the winter months. Of this, approximately 21,000 tons is BC, which is about 3% of total U.S. BC emissions. The key emitting source categories that comprise RWC are wood stoves, manufactured and masonry fireplaces, hydronic heaters, and indoor furnaces. The 2005 $PM_{2.5}$ inventory shows that cord wood stoves contribute about 52%, fireplaces 16%, hydronic heaters 16%, indoor furnaces 11% and pellet stoves and chimineas (free-standing outdoor fireplaces) the remaining 5%. Since 2005, the popularity and use of outdoor hydronic heaters has grown. As a result the emissions from these units are growing and are of particular concern to many areas, like the Northeast and Midwest.

In addition to $PM_{2.5}$ and BC, wood smoke contains toxic air pollutants such as benzene and formaldehyde, as well as CH_4, CO, and CO_2. Nationally, RWC accounts for 44% of polycyclic organic matter (POM) emissions and 62% of the 7-polycyclic aromatic hydrocarbons (PAHs), which are classified as probable human carcinogens.[1] All of these pollutants are products of incomplete combustion (PIC). These emissions are the direct consequence of poor appliance design and improper owner operation (e.g., using unseasoned wood) leading to incomplete combustion of the fuel.

OC emissions from RWC generally far exceed the BC emissions, making the OC/BC ratio relatively large. However, different wood burning appliances combust wood in varying ways, resulting in different OC/BC ratios. In general, wood stoves have lower OC/BC ratios than fireplaces (see Figure 10-1), and also represent a significantly larger percentage of the $PM_{2.5}$ emissions inventory. The type of wood burned also affects the amount of BC and OC emissions.

Despite the relatively high OC/BC ratio from RWC in general, it is important to consider the location of these emissions. While OC emissions are generally considered to have a cooling effect, OC emissions over areas with snow/ice may be less reflective than OC over dark surfaces, and may even have a slight warming effect (see Flanner et al., 2007). Significantly, the vast majority of residential wood smoke emissions occur during the winter months; the highest percentage of wood stove use is in the upper Midwest (e.g., Michigan), the Northeast (e.g., Maine), and the mountainous areas of the Pacific Northwest (e.g., Washington), where snow is present a good portion of the winter months.

10.3.2 Approaches for Controlling Emissions from RWC

Mitigation of RWC $PM_{2.5}$ emissions generally involves increasing the combustion efficiency of the source.

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[1] See EPA's 2005 National-Scale Air Toxics Assessment at *http://www.epa.gov/nata2005*.

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.

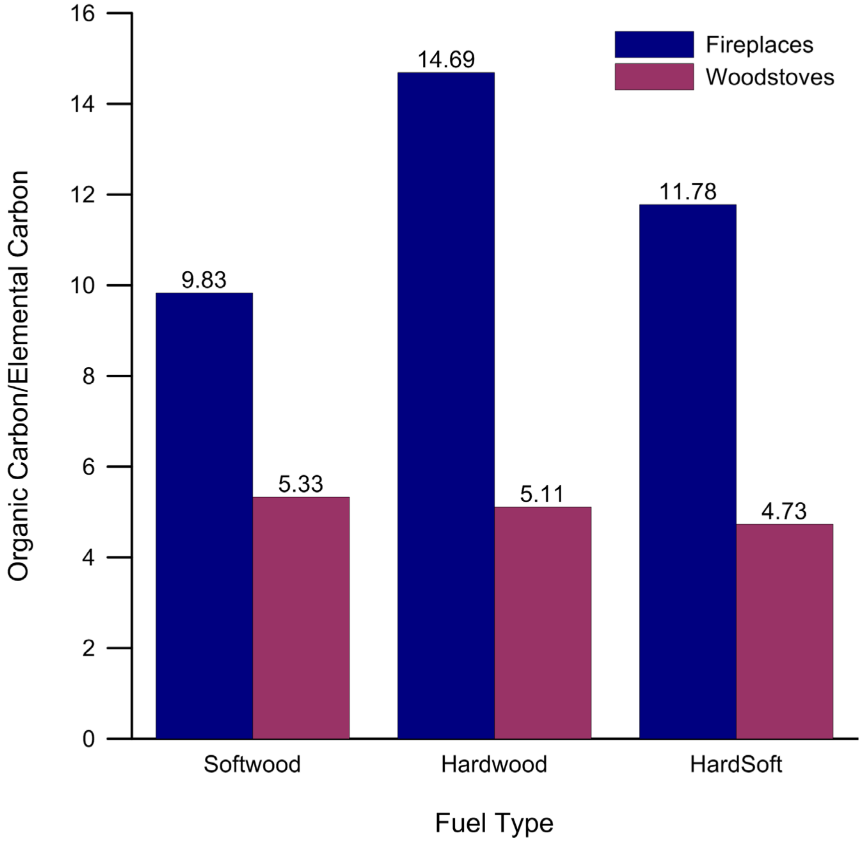

Figure 10-1. OC/BC Emission Ratios by Source Category and Fuel Type. (Source: U.S. EPA)

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 heaters[2] are typically designed to increase temperature in the firebox and to 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 OC/BC 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 an 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, wood species, moisture content, and so forth. A more efficient appliance also burns less wood for the same heat output, leading to additional emissions reductions. However, a recent wood stove changeout study conducted by the University of Montana showed significant reductions in emissions of OC and levoglucosan (a wood-burning tracer) but little or no change in BC emissions from the changeout (Ward et al., 2011).

10.3.3 Emissions Standards for New Wood-burning Units

EPA has authority to establish NSPS emissions standards for new RWC sources, such as fireplaces, wood stoves, and hydronic heaters. These standards establish manufacturing requirements to limit emissions from new units. Such standards can be updated over time as new technologies become available. Since 1988, EPA has regulated $PM_{2.5}$

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[2] For a list of such appliances, see *http://www.epa.gov/burnwise/owhhlist.html*.

emissions from new residential wood heaters sold in the United States. The Residential Wood Heaters NSPS (also referred to as the wood stove NSPS) defines a wood heater as an enclosed, wood burning appliance capable of and intended for space heating or domestic water heating that meets specific criteria, including an air-to-fuel ratio in the combustion chamber averaging less than 35-to-1; a usable firebox volume of less than 0.57 cubic meters (20 cubic feet); a minimum burn rate of less than 5 kg/hr (11 lb/hr) tested by at an accredited laboratory; and a maximum weight of 800 kg (1,760 lb). Many types of sources are exempt from the existing NSPS, including:

- Wood heaters used solely for research and development purposes
- Wood heaters manufactured for export (partially exempt)
- Coal-only heaters
- Open masonry fireplaces constructed on site
- Boilers
- Furnaces
- Cookstoves

The Residential Wood Heaters NSPS is unusual in that it applies to mass-produced consumer items and compliance for model lines can be certified "pre-sale" by the manufacturers. A traditional NSPS approach that imposes emissions standards and then requires a unit-specific compliance demonstration would have been very costly and inefficient. Therefore, the NSPS was designed to allow manufacturers of wood heaters to avoid having each unit tested by allowing, as an alternative, a certification program that is used to test representative wood heaters on a model line basis. Once a model unit is certified, all of the individual units within the model line are subject to similar labeling and operational requirements.

EPA is currently in the process of revising the Residential Wood Heaters NSPS. Specifically, the Agency is considering tightening the air pollution emission limits, adding limits for all pellet stoves, reducing the exemptions, and adding regulations for more source categories, including hydronic heaters and furnaces. EPA expects to propose appropriate revisions in 2012, and finalize revisions in 2013. The tightening of the wood heater NSPS has the potential to help reduce future residential wood burning emissions throughout the United States.

10.3.4 Mitigation Opportunities for In-Use RWC Sources

A fundamental limitation of the standards for new sources discussed above is that they cannot influence emissions from units that were purchased prior to establishment of the NSPS. It can take a long time for NSPS to actually reduce emissions, depending on the rate of replacement of existing units—and in many cases, these units can remain in service for decades. Thus, alternative mitigation strategies are needed to reduce emissions from existing sources.

In 2004, a panel convened by the National Academies of Science made several recommendations to the EPA for improving air quality management in the United States. One of their recommendations was to develop and support programs to address residential wood smoke. Since 2005, EPA has developed a residential wood smoke reduction initiative that has various components to support state, local, and tribal communities in addressing their wood smoke challenges. This initiative focuses on ensuring that wood burning is as clean and efficient as possible to help reduce emissions of harmful pollutants, the amount of fuel used, and the risk of chimney fires from creosote that builds up due to incomplete combustion. In general, these programs were developed to reduce $PM_{2.5}$ and toxic air pollutants, but can be employed to help reduce BC and other GHG (e.g., CH_4 and CO_2) from RWC. The initiative has the following key components.

10.3.4.1 Great American Wood Stove Changeout Program

The hearth industry estimates that of the 12 million wood stoves in U.S. homes today, 75% are wood stoves built before 1990. EPA is working with the hearth products industry and others to help state, local, and tribal agencies create campaigns to promote replacement of old wood stoves and wood-burning fireplaces with new, cleaner-burning and more energy efficient appliances. Programs vary from one community to another, with some areas focusing on changing out old wood stoves and others on retrofitting open fireplaces with cleaner burning options (e.g., gas stoves). The campaigns are typically led by local government or non-profit organizations at the county or regional level.

Residents of participating communities generally receive incentives such as cash rebates, low/no interest loans and discounts to replace their old, conventional wood stoves and fireplace inserts with cleaner-burning, more efficient EPA-certified

New Wood Stoves and Pollution Reduction

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 CO_2
- 53 tons of CH_4
- 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.)

gas, pellet, electric, wood stoves and fireplaces or even geothermal heat pumps. A new EPA-certified wood stove, new flue, and professional installation cost, on average, $3,500 (2010$). Some areas have provided cash incentives to low-income participants only, while others have provided incentives to everyone in the community. The local agency leading the replacement program will sometimes include weatherization programs which insulate homes to help reduce heat loss and reduce fuel consumption. Households that participate in these programs are required to surrender their old wood stoves to be recycled.

Some of the benefits of replacing inefficient wood stoves include:

- Reduction in $PM_{2.5}$ and toxic air pollutants (e.g., benzeno(a)pyrene) by 70%
- Reduction in indoor $PM_{2.5}$ emissions by 70% according to University of Montana[3]
- Improvement in energy efficiency by 50%, using one-third less wood
- Reduction in CH_4, BC, and CO_2 from improved combustion efficiency and use of less fuel wood.

A variety of examples of state and local efforts to reduce emissions from older appliances are available at EPA's Burn Wise website: *http://www.epa.gov/burnwise/casestudies.html*.

EPA's wood stove changeout effort has focused primarily on counties at or near nonattainment for $PM_{2.5}$ where wood smoke is an important local source. EPA estimates that through 2011, the Great American Woodstove Changeout Program has helped changeout or retrofit nearly 24,000 wood stoves and fireplaces in 50 areas. From 2010 on, this program is anticipated to reduce approximately 370 tons of $PM_{2.5}$ and 63 tons of hazardous air pollutants (HAPs) each year after 2010, providing approximately $120 million to $330 million (2010$) in estimated health benefits in the U.S.[4]

The best available cost-effectiveness information on residential wood smoke mitigation comes from a

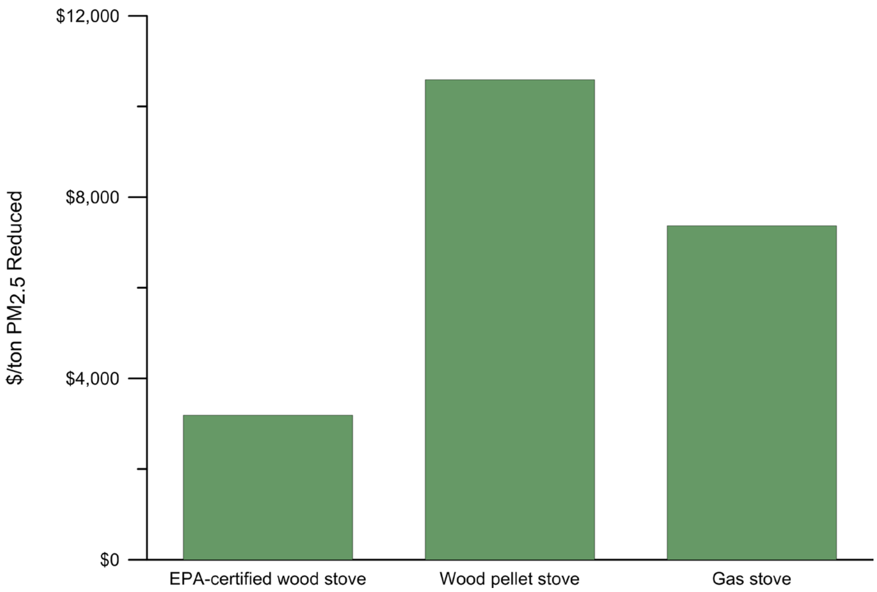

Figure 10-2. Cost Per Ton $PM_{2.5}$ Reduced for Replacing Non-EPA-Certified Wood Stove with EPA-Certified Woodstove (in 2010$). (Source: U.S. EPA, based on data from *http://www.marama.org/visibility/ResWoodCombustion/RWC_FinalReport_121906.pdf*)

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[3] For more information, see: *http://www.ncbi.nlm.nih.gov/pubmed/18665872*.

[4] These estimates reflect national average benefit-per-ton estimates for directly emitted carbonaceous particles from area sources from *http://www.epa.gov/air/benmap/bpt.html* (data accessed February 2011) and Fann, Fulcher, Hubbell 2009 methodology. These estimates have been inflated from 2006$ to 2010$.

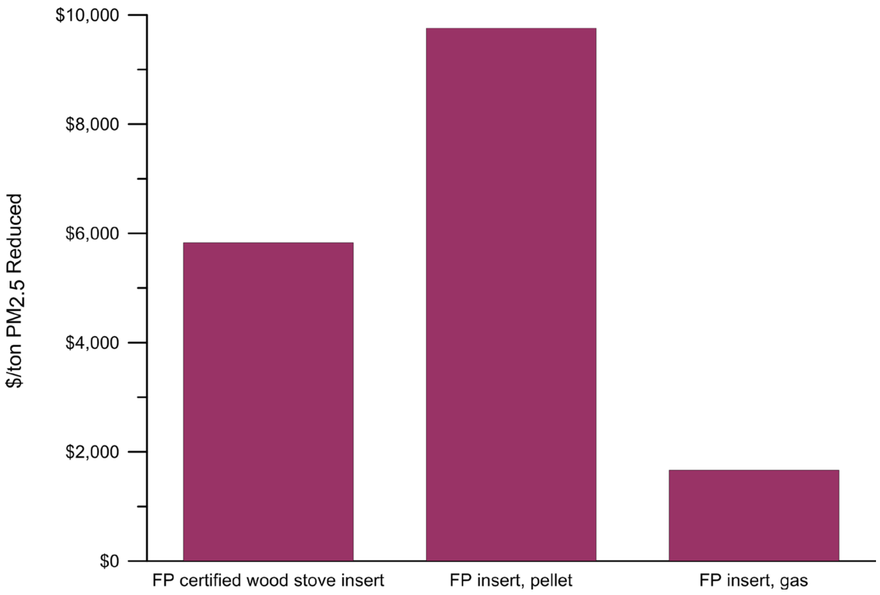

Figure 10-3. Cost Per Ton $PM_{2.5}$ Reduced ($/Ton) for the Addition of an Insert into a Fireplace (2010$). (Source: U.S. EPA, data courtesy of MARAMA)

Mid-Atlantic Regional Air Management Association (MARAMA) document called *Control Analysis and Documentation for Residential Wood Combustion in the MANE-VU Region (2006)*. This document focused on the costs of total $PM_{2.5}$ mitigation. The results suggest that the cost per ton of $PM_{2.5}$ emissions reduced from wood stove changeouts and fireplace retrofits is similar to the cost of other $PM_{2.5}$ controls discussed in previous chapters of this report. Figures 10-2 and 10-3 summarize the MARAMA estimates of the cost per ton $PM_{2.5}$ reduced from these measures; these costs vary depending on the type of wood burning appliance being replaced (old wood stove vs. open fireplace) and on the replacement technology (e.g., EPA-certified wood stove vs. wood pellet stove).

10.3.4.2 Outdoor Wood-Fired Hydronic Heater Program

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.[5] EPA has now expanded the program to include indoor models and hydronic heaters that are fueled by other kinds of solid biomass (e.g., wood pellets).

As of 2011, nearly 10,000 EPA-qualified units had been sold; 24 manufacturing partners had agreed to produce units 70%-90% cleaner; and 22 models had been placed on the market, reducing approximately 6,100 tons of $PM_{2.5}$ emissions each year after 2011 and providing approximately $2.2 billion to $5.4 billion (2010$) in estimated health benefits in the U.S.[6]

10.3.4.3 New Construction Wood-Burning Fireplace Program

The EPA voluntary Wood-burning Fireplace Program is modeled after the Hydronic Heater Program and helps reduce wood smoke emissions growth in areas with $PM_{2.5}$ air-quality problems. The two-phase program covers new installation of low mass (i.e., pre-manufactured) and masonry fireplaces, and is expected to drive technology improvements much sooner than possible through regulation. The program qualifies models achieving a Phase 1 (34% reduction) or a Phase 2 (54% reduction) $PM_{2.5}$ emission level. EPA has worked closely with the hearth products industry to develop this program; however, growth in the program has been hampered by the slowdown in new home construction in the United States.

10.3.5 Additional Regulatory Approaches to Limiting Wood Smoke Emissions

A variety of regulatory programs, including wood burning curtailment programs and requirements to remove old stoves upon resale of a home, have proven effective in helping to address wood smoke.

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[5] Use of Phase 1 labels was prohibited after March 31, 2010. See *http://www.epa.gov/burnwise/owhhlist.html*.

[6] These estimates reflect national average benefit-per-ton estimates for directly emitted carbonaceous particles from area sources from *http://www.epa.gov/air/benmap/bpt.html* (data accessed February 2011) and Fann, Fulcher, Hubbell 2009 methodology. These estimates have been inflated from 2006$ to 2010$.

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 where government officials believed heating with wood was very common. Even though they did not calculate emissions reductions, a follow-up survey of 174 people indicated that

- 3% 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.

Wood Burning Curtailment Programs: One of the most effective ways a community can reduce wood smoke is by developing a mandatory curtailment program or instituting "burn bans." Some communities implement both a voluntary and mandatory curtailment program depending on the severity of their problem. Curtailment programs often have two stages, with Stage 1 allowing EPA-certified wood stoves to operate and Stage 2 banning all wood burning appliances, unless it is the homeowner's only source of heat. Although curtailment programs are not always popular with the public, this measure can be highly effective at reducing wood smoke. As an example, the Sacramento Air Quality Management District's Stage 2 program, implemented in 2008-2009, reduced $PM_{2.5}$ levels by 12 $\mu g/m^3$. The cost effectiveness was estimated to be approximately $6,300 - $11,100 per ton of $PM_{2.5}$ reduced (2010$) (SMAQMD, 2009). To increase the likelihood of success, curtailment programs should include a forecasting and public notification system. In addition, an enforcement component is important to ensure the public takes the program seriously.

Removal of Old Wood Stove Upon Re-Sale of a Home: Old wood stoves are usually made of metal, weigh 250 to 500 pounds, last for decades, and can continue to pollute for just as long. As a result, homeowners are less likely to replace old stoves with a new, cleaner burning technology or remove the old stove, especially if they are not using it. To help get these old stoves "off-line," the state of Oregon and some local communities in other states have required the removal and destruction of old wood stoves upon the resale of a home. Specifically, this requirement has proven very effective in locations like Mammoth Lakes, CA and Washoe County, NV.[7]

10.3.6 Wood Smoke Reduction Resource Guide

In October 2009, EPA released a resource guide called *Strategies for Reducing Residential Wood Smoke*[8] that was written for state, local, and tribal air pollution control officials so they would have a comprehensive list of strategies to help reduce wood smoke from residential heating. The guide provides education and outreach tools, information on regulatory approaches (e.g., burn bans) to reduce wood smoke, as well as voluntary programs to change out old, inefficient wood stoves and

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[7] For more information, see: *http://www.gbuapcd.org/rulesandregulations/PDF/Reg4.pdf.*

[8] *http://www.epa.gov/ttncaaa1/t1/memoranda/strategies-doc-8-11-09.pdf.*

fireplaces. It also notes the upcoming wood heater NSPS has the potential to help reduce future residential wood burning emission throughout the United States. Several state and local communities have effectively implemented residential wood smoke control strategies and have significantly reduced harmful wood smoke pollution. For example, Lincoln County, MT and Sacramento Metropolitan Air Quality Management District have encouraged comprehensive wood smoke reduction strategies to help these areas clean the air and protect public health.

10.4 Residential Cookstoves in Developing Countries

More than 3 billion people worldwide cook their food or heat their homes by burning biomass (e.g., wood, dung, crop residues, and charcoal) or coal in polluting and inefficient traditional stoves (International Energy Agency, 2010). As discussed in Chapter 4, BC emissions from these sources are estimated to account for 21% of the total global inventory. This use of solid fuels also represents a significant part of energy use in developing regions—including nearly 50% of total primary energy supply in Africa, and about 27% in India (International Energy Agency, 2009). Use of biomass and waste in developing nations—nearly all of which is for household cooking and heating—accounts for about 60% of global renewable energy use (International Energy Agency, 2009, Annex A). About 82% of those who rely on traditional biomass fuels for cooking live in rural areas; however, in Sub-Saharan Africa, nearly 60% of people living in urban areas also rely on biomass (International Energy Agency, 2010).

As discussed in Chapter 3, several decades of research document the significant risks to public health associated with traditional cookstoves. Exposure to cookstove emissions leads to an estimated 2 million deaths each year and ranks as the sixth highest mortality risk factor and fifth highest disease risk factor overall (in terms of disability adjusted life years—DALYs) in poor developing countries (World Health Organization, 2009). Emissions from cookstove use have been linked to adverse respiratory, cardiovascular, and neonatal outcomes and to cancer (Smith et al., 2004). Of the total global mortality associated with exposure to cookstove smoke, Sub-Saharan Africa, China and India each account for approximately 25-30%, with the remainder of deaths attributable to cookstoves occurring elsewhere in Asia and Latin America.

The contribution of this source category to emissions of BC and other aerosols has been the focus of growing interest, especially in terms of impacts on sensitive regions such as the Himalayas. A recent study on BC emissions from cookstoves in northern India indicated that cooking with solid fuels is a major source of ambient BC in the region, with peak ambient BC concentrations of about 100 $\mu g/m^3$ (Rehman et al., 2011). Furthermore, the study indicated that OC concentrations (which exceeded BC concentrations by a factor of five) contained significant absorbing BrC, and suggested that previous estimates of atmospheric solar heating in the region due to particles from cookstove emissions should be increased by a factor of two or more. However, there remains significant uncertainty about the extent of BC and associated emissions from cookstoves, and the effect of those emissions on climate. Given the complex emissions mixture resulting from cookstove use, further study is needed to pinpoint the most beneficial strategies for reducing BC emissions from this source. Unquestionably, however, this sector represents the area of largest potential public health benefit of any of the sectors considered in this report. Mitigation of emissions from cookstoves offers a tremendous opportunity to protect health, improve livelihoods, and promote economic development—particularly for women and children. For this reason alone, irrespective of the additional climate benefits that may potentially be achieved, mitigation of cookstove emissions is a pressing priority.

Mitigating BC emissions from cookstoves depends first on identifying technologies that are proven effective in reducing BC emissions, and second on encouraging adoption of these technologies on a large scale. As discussed below, very few improved stoves have been designed specifically to reduce BC emissions, and while some improved technologies are emerging, no advanced stoves that burn solid fuel have yet been adopted on a broad scale (though LPG has been widely disseminated as a clean cooking fuel, and China (see below) implemented a very large earlier stoves program using intermediate stoves). The problem is complicated by the fact that both the impacts of cookstoves and the solutions are regionally dependent. Specifically, the extent of achievable BC reductions, and the impact of those reductions, will depend on the type of stove, the type of fuel used, and the location of emissions. Improved cookstoves and fuels must satisfy the needs of local users, enabling them to cook local foods at the time and in the manner they prefer, using the fuels that are available and affordable. Given the array of different technologies and fuels currently in use, and the sheer number of sources involved, mitigation of BC emissions from cookstoves represents an enormous challenge. However, given their significant contribution to the global inventory of emissions, and the increasing availability of cost-effective and locally appropriate solutions and several other factors noted in section 10.4.3, this sector represents one of the most promising opportunities for mitigation of BC internationally.

10.4.1 Emissions from Cookstoves

Currently, residential cookstoves contribute approximately 21% of the global BC emissions inventory, with emissions concentrated in Sub-Saharan Africa, China, India, and other developing regions of Asia. Dependence on traditional biomass fuels is highly correlated with poverty; countries with higher household income also tend to have a higher share of modern fuels for residential consumption. While the percentage of people relying on traditional biomass fuels for basic household energy needs is expected to decrease in most areas over the coming decades, the aggregate number of people relying on biomass for cooking and heating is expected to increase by 100 million people by 2030 due to population growth (International Energy Agency, 2010). IEA projects that the fastest shift toward modern fuels will occur in India, and the slowest shift will occur in Sub-Saharan Africa (International Energy Agency, 2010). The impact of these changes on emissions is still unclear: as discussed in Chapter 7, under most scenarios, residential emissions are projected to decline significantly by 2030 and further still by 2050 (Streets et al., 2004). However, a decline in emissions, and the rate of decline will depend on rates of adoption of cleaner fuels and cooking technologies described below, and some regions may experience near-term increases in emissions.

10.4.2 Technologies and Approaches for Controlling Emissions from Cookstoves

Because cooking is such a variable, individual-specific activity, there are many complexities related to achieving reductions in BC emissions from improved cookstoves. The type of fuel and its moisture content, the type of stove, the purpose for which it is used (heating vs. cooking), and the manner in which the stove is tended all affect composition of emissions (MacCarty et al., 2008). Cooking practices vary both daily and seasonally due to variation in available foods and fuels, and variation in fuel quality. Additionally, there may be significant variation in the efficiency and durability of stoves, even those that are mass produced.

There is currently no formal definition of what constitutes an “improved” stove. In the past, “improved” stoves typically meant low-cost, locally made stoves aimed at improving efficiency and reducing fuel use. A primary motivation for the use of improved stoves was to reduce demand for fuel wood, thereby reducing pressures on forests as well as the time spent by women and children gathering fuel (Graham et al., 2005; Partnership for Clean Indoor Air, 2005). However, not all such stoves functioned as intended. For example, stoves that have a large amount of heated mass, such as the Lorena stove, may remove smoke with a chimney, prevent burns, and help warm a house, but may not save fuel compared with an open fire (USAID, 2007).

Over the last ten years, a new suite of more effective stoves has been introduced to the marketplace. As a group, these new improved stoves are designed to be much more efficient and clean (as well as safe), and utilize a variety of different technologies and fuels. Most are produced locally for the nearby market, while there are a few that are mass produced internationally and can be shipped anywhere in the world. The stoves span a wide range of cost, durability, and performance, and are designed for different types of staple foods. Importantly, however, these stoves are generally designed to reduce fuel use and emissions of $PM_{2.5}$ and CO (as proxies for the broader suite of emissions from these stoves). Few of the stoves currently on the market were designed to reduce BC specifically (the new Turbococina stove is an exception). In laboratory settings, most of these stoves achieve $PM_{2.5}$ reductions of 40% to 70%. Results from MacCarty et al. (2008) indicate that some non-advanced stoves may not substantially reduce BC emissions, but some gasifier and forced draft (or “fan”) stoves significantly reduce BC emissions compared to the open fire. Field testing has begun to determine whether stoves perform as well in actual (real-world) conditions as in the laboratory. Results from field tests indicate that stove performance under actual conditions varies (Roden et al., 2009; USAID, 2011). Much more such testing is needed, as well as additional research and development in stove design to determine if the stoves are reducing BC in addition to total $PM_{2.5}$.

Among the new technologies now emerging on the cookstove market, there are a few advanced solutions that have been shown to reduce $PM_{2.5}$ by 90% or more in laboratory settings; the limited lab testing performed on these stoves to date indicates that some stoves reduce BC by a similar percentage (MacCarty et al., 2008). These new technologies include advanced forced-draft stoves and “gasifier” stoves (Roth, 2011) that use various solid biomass fuels (including wood, pellets, crop residues, etc.); biogas stoves; and liquid-fuel stoves

Figure 10-4. The Turbococina Stove, which burns wood, is made of a stainless steel cylinder fitted with 10 air injectors, an internal fan that runs on electricity, and a steel plate that regulates the air flow to improve combustion efficiency. This stove is currently available primarily for institutional settings. (Photo courtesy of René Nuñez Suárez, Turbococina)

that burn ethanol, plant oil, or other biomass fuels. It is important to note, however, that few of these stoves are widely commercially available and their performance in the field has not been fully evaluated. Some stoves require electricity to drive a fan (see Figure 10-4), while others have been designed to convert waste heat to electricity to power a fan, which enables excellent emissions performance (including BC emissions reductions) without the need for access to electricity (see Figure 10-5).[9] Some of these stoves are being further tested for emissions in both the lab and the field. These new stove technologies have the potential to reduce emissions from cookstoves nearly to the levels of clean fuels such as LPG (Wilkinson et al., 2009), but many (though not all) require specific and/or highly processed fuels, which increases the total cost of use (Venkataraman et al., 2010). In some cases, clean fuels (ethanol, biogas) are not widely commercially available for cooking. Forced-draft stoves that do not generate their own electricity require access to another electricity source, which can be costly. Even battery-powered fan stoves require intermittent access to electricity. Some gasifier stove designs use natural draft (natural convection) and do not require a fan.

While the basic outlines of lab and field tests have been in place for decades, it is only in the past five years that organizations funding household energy interventions have begun requiring emissions pre-testing, or that performance benchmarks (even informal ones) have been established. Recent testing in both lab and field settings (see below) demonstrate that this new generation of stoves is achieving real and measurable results. Developing globally recognized standards that are widely accepted by the cookstove community and adopted by country governments could spur wider development of clean cookstoves.

Based on available performance and cost data, currently available technologies exhibit a wide range of performance. These options include:

- electric stoves
- gas and liquid fuels
- processed solid fuels
- advanced biomass stoves
- rocket stoves

Figure 10-5. Woman Prepares Banku on a BioLite HomeStove in Kintampo, Ghana. BioLite stoves convert their waste heat into electricity, which is used to power the fan and can be used to charge a battery for a mobile phone (as shown in photo), LED light, or other low-power purpose. (Photo courtesy of Jonathan Cedar, Biolite)

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[9] 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).

- simple stoves
- solar cookers
- behavioral and structural solutions

10.4.2.1 Electric Stoves

Cooking with electricity produces zero emissions within a household, and therefore is highly effective at reducing personal exposures of stove users. However, from a broader public health or climate perspective, emissions associated with the increase in power production must also be considered. The ongoing electricity costs for these stoves can vary substantially by region.

10.4.2.2 Gas and Liquid Fuels

Switching from solid fuels to gaseous or liquid fuels is often the easiest means of dramatically lowering emissions from cooking. In laboratory testing, the Aprovecho Research Center (Aprovecho) found that using liquid petroleum gas (LPG) decreased the amount of energy used by 69%, the mass of fuel used by 89%, particle emissions by over 99%, CO emissions by 98%, and time to boil by 40%, as compared to cooking over an open fire (MacCarty et al., 2010). Field research in Guatemala showed that LPG stoves could reduce indoor 24-hr $PM_{2.5}$ concentrations by over 90% (Naeher et al., 2000). Liquid fuels such as ethanol, kerosene, and plant oils are also options (see Figure 10-6). Aprovecho's lab tests found that cooking with well-made ethanol or kerosene stoves decreased the mass of fuel used by 75% and 82% respectively and particle emissions by over 99% for each; CO emissions by 92% and 87% (MacCarty et al., 2010).

Figure 10-6. CleanCook Ethanol Stove. Currently these stoves are being deployed in refugee camps in Ethiopia and Kenya, with over 3,800 stoves already in use. Deployment of additional stoves is dependent partly on the availability of ethanol. (Photo courtesy of Harry Stokes, Project Gaia)

Biogas derived from waste biomass is potentially as clean as LPG, and in addition it is renewably derived (reducing CO_2 impacts) and requires no distribution infrastructure. Emissions testing of biogas stoves to date suggests that these stoves perform significantly better than solid fuel/stove combinations with regard to emissions of methane, CO, VOC, and CO_2 (Smith et al., 2000b). Plant oils are another liquid fuel being used today for cooking, but independent testing results for stoves using these fuels have not yet been published. Stoves using gas and liquid fuels involve an upfront cost of \$5 to \$50 per stove, as well as an ongoing cost for the fuel that varies substantially by region, fuel, and changing economic conditions. LPG stoves can also require significant deposits on the cylinders, another serious barrier for the very poor. It is also important to note that poorly made kerosene stoves in particular pose safety concerns, including the potential for severe burns and injury associated with accidental fires (Peck et al., 2008).

10.4.2.3 Processed Solid Fuels

For much of the developing world, the advanced solutions described above may be unavailable or simply too costly to use in the near-term. Stoves utilizing processed solid fuels in the form of charcoal, pellets, prepared wood, and briquettes, may be more accessible, and these can also represent very clean solutions. However, like the clean fuels noted above, using processed fuels also involves an ongoing operating cost, which may serve as a barrier for these solutions, especially in regions where fuel wood can be collected free of charge. However, in markets where fuel is purchased, stoves that increase combustion efficiency by 50% are often the easiest stoves to market, since the consumer can expect a quick payback period on the initial investment.

Charcoal is the most common processed solid fuel used today. A number of charcoal stove models are available (see Figure 10-7). Lab tests of charcoal

Figure 10-7. Charcoal Stoves. (a) Charcoal Stove by Burn Design. (Photo courtesy of Peter Scott, Burn Design) (b) Charcoal Zoom Versa Stove produced by EcoZoom. (Photo courtesy of Ben West, EcoZoom) (c) Prakti Charcoal Stove. (Photo courtesy of Mouhsine Serrar, Prakti Design) (d) Envirofit CH-4400 Charcoal Stove. (Photo courtesy of Envirofit) (e) Toyola Charcoal Stove in use in a village in Ghana. (Photo courtesy of Suraj Wahab, Toyola Energy)

stoves for climate forcing emissions found that these stoves—relative to an open fire—reduced the BC/OC ratio somewhat, and reduced total particles by about two thirds (MacCarty et al., 2008). It is important to note that the laboratory emissions tests do not account for emissions in the charcoal production process, which is highly inefficient and polluting, with significant net climate impacts (Bailis et al., 2005). Aprovecho has tested many charcoal stoves for $PM_{2.5}$, CO, and fuel use, finding that $PM_{2.5}$ emissions were 90% lower than for a 3-stone fire and fuel use savings ranged from 45% to 65%. Most charcoal stoves cut time to boil, though only modestly. However, CO emissions increased for all stoves except one (MacCarty et al., 2010). In 2007, EPA tested two charcoal stoves and found that relative to a 3-stone fire, $PM_{2.5}$ emissions from the charcoal stoves fell by over 90% from a hot start but increased when operated from a raw, cold start, and both stoves increased CO emissions (Jetter and Kariher, 2008).

Creating pellets from biomass or briquettes from either coal or biomass can lead to substantial improvements in efficiency and emissions when pellets are burned in well-designed stoves. The Oorja stove (developed by BP and now owned by First Energy of India) is an example of a very clean-burning pellet stove—in this case the pellets are made from crop residues by a partner company. More than 400,000 Oorja stoves have been sold and between 250,000 and 350,000 are in use every day. However, given the cost of pellets, this stove competes with LPG. Other examples include project-based work that have developed briquettes from waste biomass (Haiti, Ghana, and Uganda), stoves designed to burn pellets made from locally available waste biomass (West Africa and elsewhere), and a stove that burns rice hulls (Philippines), though EPA is not aware of any examples where this work has been carried to a large scale. With regard to coal cooking, laboratory measurements indicate that the combination of using improved stoves with processed coal briquettes could have a dramatic impact on aerosol emissions. Zhi et al. measured reductions in particles of 63%—with OC decreasing 61% and BC decreasing 98%. This reduced the BC/OC ratio by about 97%, from 0.49 to 0.016 (Zhi et al., 2009).

10.4.2.4 Advanced Biomass Stoves

There are two types of advanced biomass stoves that can achieve high levels of performance: forced

Figure 10-8. Philips Woodstove (forced draft) Manufactured in Lesotho. (Photo courtesy of Stephen Walker, African Clean Energy Ltd.)

draft and gasifier stoves. Gasifier stoves can be forced-draft or natural draft. These stoves can burn processed or raw biomass, though it is likely the case (field testing data forthcoming) that those using processed fuels will perform better in the field, since processed fuels eliminate a major variable in real-world use of the stoves. It is also likely that lab and field emissions test results will be more consistent for stoves that burn processed fuels. Lab testing of advanced biomass stoves to date generally confirms that these advanced biomass stoves can achieve remarkable emissions reductions—up to 93% lower than traditional stoves (Venkataraman et al., 2010). One study found that these stoves achieved substantial reductions in both overall particles and BC specifically, with the fan stove significantly reducing particle emissions and the gasifier stoves reducing total particles by about two-thirds (as well as reducing the BC:OC ratio). The study also showed that the fan stove was able to reduce time to boil, at least under the lab conditions (MacCarty et al., 2008).

Under Aprovecho's broader lab testing, forced draft fan stoves all reduced (relative to a 3-stone fire) fuel use (by 37% to 63%), CO emissions (in all cases by over 85%), $PM_{2.5}$ emissions (from 82-98+%), and time to boil (11% to 65%). Similarly, the gasifier stoves tested by Aprovecho saved on fuel use, reduced CO emissions, achieved dramatic reductions in particle emissions (with one exception), and cut the time to boil, though generally all to a lesser extent than the fan stoves (MacCarty et al., 2010). In EPA's 2007 testing, the one advanced fan stove tested (Philips) had the best overall performance and the lowest pollutant emissions, reducing emissions of key pollutants such as $PM_{2.5}$ and CO by about 90%. Notably, of the wood burning stoves tested, this stove was also the one that required the least attention to operate (Jetter and Kariher, 2008), although it required fuel with short (<10 cm) lengths (see Figure 10-8). A forthcoming study from investigators in India of recently completed field testing of two forced draft cookstoves and indicated that both stoves substantially reduced BC emissions in both the breathing zone (85% and 49% BC reduction respectively, compared to the traditional mud cookstove) and in the plume zone (86% and 64% respectively) (Kar et al., 2012). Indoor cooking-time BC concentrations were reduced to 5-100 $\mu g/m^3$ by the top-performing forced draft stove (as compared to 50-1000 $\mu g/m^3$ for the mud stove).

It is important to note that, while very promising in terms of performance, most of these models are still in the research and development stage, though a few have been introduced in the market today. These stoves are typically more costly than other biomass stoves, currently costing in the range of $25-100 per unit (plus any processed fuel costs, which can be substantial[10]), though prototypes for newer versions have been developed that manufacturers estimate will cost in the $40-60 range at full production.

10.4.2.5 Rocket Stoves

Where advanced stoves are not widely available in the marketplace, or are not affordable, rocket type stoves are typically the most efficient and clean biomass-burning alternative (see Figure 10-9). Rocket stoves have a combustion chamber designed to allow for better mixing of combustion gases and higher combustion temperatures which slightly improves combustion efficiency (compared to the open fire) and reduces emissions. Additionally, rocket stoves have substantially better heat transfer efficiency, so as they save fuel, they reduce emissions (for a given cooking task). Thus, rocket stoves substantially reduce emissions without relying on electricity or other sophisticated components. However, rocket stoves have not been designed to date to reduce BC emissions. MacCarty et al. (2008) found that the rocket stove reduced total particle emissions by about 40%, but that nearly all of the emissions reductions were of organic matter; BC emissions for this stove did not decrease (and thus the BC:OC fraction increased dramatically). The study also showed that the rocket stove was able to

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[10] For example, the pellets for the Oorja stove in India cost roughly 7 Rupees (~15¢) per kilogram of pellets.

Figure 10-9. Rocket Stoves. (a) Envirofit G3300 Biomass Cookstove in Use in Tanzania. (Photo courtesy of Nancy Ryden, Envirofit) (b) Zoom Dura Biomass Cookstove produced by EcoZoom. (Photo courtesy of EcoZoom)

reduce time to boil, though to a lesser extent than an advanced fan stove.

Aprovecho tested a wide variety of rocket stoves and found important variability in performance indicating that design is critical. Most saved significantly (but not equally) on fuel use (26% to 51% savings relative to a 3-stone fire), though two failed to cut fuel use. All rocket stoves cut CO emissions by 70% or more, while performance on $PM_{2.5}$ emissions varied much more widely (one actually increased emissions), with 60% of those tested achieving reductions of over 50%. Some of the rocket stoves actually increased the time to boil, though most cut it modestly (MacCarty et al., 2010). In EPA's 2007 testing, several non-advanced wood stoves were tested and results varied depending on the design and the stage of operation. Generally, emissions were lower than the 3-stone fire, with faster times to boil. For example, the UCODEA wood stove—now called Ugastove—reduced $PM_{2.5}$ and CO emissions by 48% to 65% when operated at high power, and 35% to 50% at low power (Jetter and Kariher, 2008).

The U.S. Agency for International Development (USAID) recently conducted extensive field testing of five non-advanced biomass stoves in the Dadaab refugee settlements in Kenya. They tested for fuel use, time to boil, and several user preferences—but did not test for BC or any other emissions—and concluded that "all five tested stoves outperformed the open fire, requiring significantly less fuel to cook the test meal....with savings ranging from 32% to 65" (USAID, 2010b). Additional testing of two manufactured rocket stoves by Columbia University researchers demonstrated "substantial and statistically significant fuel savings relative to the three-stone fire" (38% and 46% on average, for the two stoves), but further stressed that fuel savings is just one factor that affects suitability of any given stove in a particular community. Other relevant factors include stove size, ease of use, and cooking time (Adkins et al., 2010).

USAID (2011) also completed a recent round of cookstove testing in the field in Uganda that did examine BC and other emissions and factors, comparing a traditional stove to a leading rocket stove. The study found that with the greater fuel efficiency of the rocket stove (42% savings were measured) came lesser emissions of $PM_{4.0}$ and carbon monoxide. However, the rocket stove (which was not designed to reduce BC emissions) had more than twice the fractional BC content in its PM emissions (15.5%) compared to the traditional stove (7.2%).

Recent field testing in India (Kar et al., 2012) noted above found preliminary results showing that the natural draft stoves had much wider variation in performance than the forced draft stoves, and did not achieve nearly as great reductions in BC. For example, these natural draft stoves reduced BC emissions in the breathing zone by a factor of 1.5 on average (22% to 55%), as compared to a factor of 4 for forced draft stoves. However, BC reductions varied significantly among natural draft stoves: only micro-gasification stoves were shown by Kar et al. (2012) to be effective in reducing BC, while other models occasionally emitted more BC than a traditional cookstove. BC emissions were shown to vary significantly among cooking cycles with same

Figure 10-10. Prakti Double-Pot Woodstove with Chimney. (Photo courtesy of Mouhsine Serrar, Prakti Design)

stove, and the use of mixed fuel (reflective of local practices) was shown to increase plume zone BC concentration (compared to hard wood) by a factor of 2 to 3 across the stoves tested.

Several studies have measured changes in indoor concentrations of $PM_{2.5}$ (but not BC) in kitchens in Latin America due to the transition from a traditional open fire to the use of a griddle stove (known in Latin America as a *plancha* stove)—a raised wood-burning stove with a chimney, typically designed with a flat griddle to make tortillas. Naeher et al. (2000) reported reductions in 24-hour $PM_{2.5}$ concentrations of over 80%, and reported earlier measurements that achieved reductions ranging from 57% to 82%. (See Figure 10-10 for a picture of an improved wood-burning stove with a chimney.) Masera et al. found that CO and $PM_{2.5}$ concentrations in the kitchens using a so-called *Patsari* stove were reduced by 66% and 67%, respectively, compared to traditional cooking methods (Masera et al., 2007). Johnson et al. (2007) further reported that while *Patsari* stoves reduce overall particulate emissions in homes (including net BC emissions), the BC/OC ratio went up, making the net warming implication more ambiguous. McCracken et al. measured personal exposures (always less than reductions in indoor air concentrations since individuals do not spend all of their time in kitchens) and reported reductions in daily average exposure to $PM_{2.5}$ of over 60% (McCracken et al., 2007).

These stoves typically cost anywhere from $8-$100 per unit, depending on the design, quality of materials, performance, use of a chimney, use of a metallic *plancha* (for making tortillas), and durability. Certain models of these stoves have combustion chambers that might also be used to build quality-controlled mud stoves—the combustion chambers themselves may cost as little as $4 to produce.

10.4.2.6 Simple Stoves

Aprovecho test results for a wide variety of simple stoves without a rocket or other improved combustion chamber indicated that the performance of these stoves varies enormously, with only two of seven tested achieving meaningful emissions and fuels use reductions. Most achieved some fuel savings, but increased particle emissions (MacCarty et al., 2010). These stoves typically cost only $2 to $10 per unit, but may last only a few months due to use of less durable materials and lower quality construction.

10.4.2.7 Solar Cookers

Solar cookstoves are emissions free, and thus the cleanest solution. However, the constraints of current solar cookers are significant: they have limited use in the early morning, late afternoon, or on cloudy or rainy days; they can greatly increase cooking time; and they are not suitable for cooking many foods. For this reason, the potential for current solar cookers is best thought of as part of an integrated solution. EPA is not aware of any example of solar cookers (which range in cost from $20 to $75 per stove, including the pot) being adopted at a large scale in a given region. However, with additional advances, such as improvements in energy storage capacity, it is conceivable that solar stoves could be an effective tool for this field in the future.

10.4.2.8 Behavioral and Structural Solutions

Many behavioral and structural steps can be taken to reduce human exposures to cookstove smoke. These include cooking outdoors, keeping children away from cooking stoves, adding ventilation to the kitchen, preparing fuel (drying and cutting to a smaller size), tending stoves more carefully, lighting stoves with improved techniques, or requiring stoves to have chimneys. Each of these solutions will diminish immediate human exposures to cookstove smoke, and are thus to be encouraged as much as possible, though the net benefit to human health may be tempered non-trivially by worsened ambient air quality when use of a chimney alone is the intervention. For purposes of this report, it is also critical to note that some of these behavioral and structural solutions will have little impact on BC emissions or related climate impacts (and may increase forcing by increasing direct emissions to the atmosphere), while others (such as preparing the fuel, tending stoves more carefully, and using improved lighting techniques), may reduce climate forcing emissions.

10.4.3 Programmatic Considerations for Cookstove Mitigation

As this extended discussion of currently available technologies indicates, there are a number of promising opportunities in the cookstove field. Advanced stoves can provide dramatic improvements in public health, and may also offer opportunities to reduce BC emissions. However, with the important exception of widespread adoption of LPG as a cooking fuel, the current scale of total stove replacements is limited, and the number of advanced stoves deployed as part of these programs is very small.

There have been many efforts to bring improved cookstoves to different parts of the world, ranging from large-scale government efforts in both China and India to countless small non-governmental organization-led efforts in communities across the globe. These efforts have had varying degrees of success. By far the most successful effort historically in terms of level of penetration of improved stoves was China's National Improved Stove Program (NISP), introduced by China's Ministry of Agriculture in the 1980s. The NISP targeted 860 of China's 2,126 counties, and the government statistics indicate that from 1982 to 1992, 129 million improved stoves had been installed in rural households (Graham et al., 2005). Gradually, the Chinese government shifted to focus on supporting stove manufacturers (Sinton et al., 2004), and follow-on programs increased total penetration to close to 200 million households (Graham et al., 2005). This program was primarily designed to reduce fuel use (to prevent deforestation). Thus, while the use of chimneys allowed China to lower indoor pollution somewhat, they were not able to reduce overall air pollution and GHG emissions (Wilkinson et al., 2009). It is not clear what impact, if any, this effort may have had on BC emissions.

In 1983, the government of India launched its National Program of Improved Chuhlas (NPIC). Over the next 17 years, the program introduced about 32 million improved biomass stoves to rural households around the country (Barnes and Kumar, 2002). While results varied substantially from region to region, "A 1995-96 survey conducted by the National Council of Applied Economic Research (NCAER) in 18 states indicated that 71% of the cookstoves were in working order and 60% were in use" (Sinha, 2002). Like the Chinese program, India's NPIC was designed to lower demand for fuel wood. The removal of indoor smoke was a secondary priority (Partnership for Clean Indoor Air, 2005). The NPIC has several shortcomings that limited its long-term success, including poorly designed subsidies, poorly designed stoves developed without user input, poor maintenance programs, and—in most regions—no commercial basis for sustained results (much greater success resulted where a commercial model was followed) (Partnership for Clean Indoor Air, 2005). In spite of its shortcomings, India's earlier program remains—after China's NISP—the largest cookstove program ever implemented (Barnes and Kumar, 2002).

Ethiopia, Indonesia, Mexico, Nepal, Nigeria, and Peru are examples of countries that have launched national stove programs; many other countries are actively working in this field. In December 2009, the government of India announced that it would launch a new National Biomass Cookstove Initiative to build on India's earlier national program, but be based almost entirely on a commercial business model in close cooperation with leading manufacturers of clean stoves and fuels in India. India will also seek to catalyze further stove and fuel innovations, for example via a global stove design prize.

The United States has been an active participant in the effort to address the many health risks associated with traditional cookstoves. At the 2002 World Summit on Sustainable Development, U.S. EPA brought together leaders from the government, private, academic, and nongovernmental sectors to launch the PCIA. Through 2010, key PCIA Partners have reported helping 6.6 million households adopt clean cooking and heating practices, reducing harmful exposures for more than 30 million people. PCIA has found that succeeding with sustainable household energy and health programs in developing countries requires focusing on four essential elements: meeting social and behavioral needs of users; developing market-based solutions; improving technology design and performance; and monitoring impacts of interventions.

Over time, the scale and pace of cookstove replacements have been increasing worldwide. Based on reporting from its network of more than

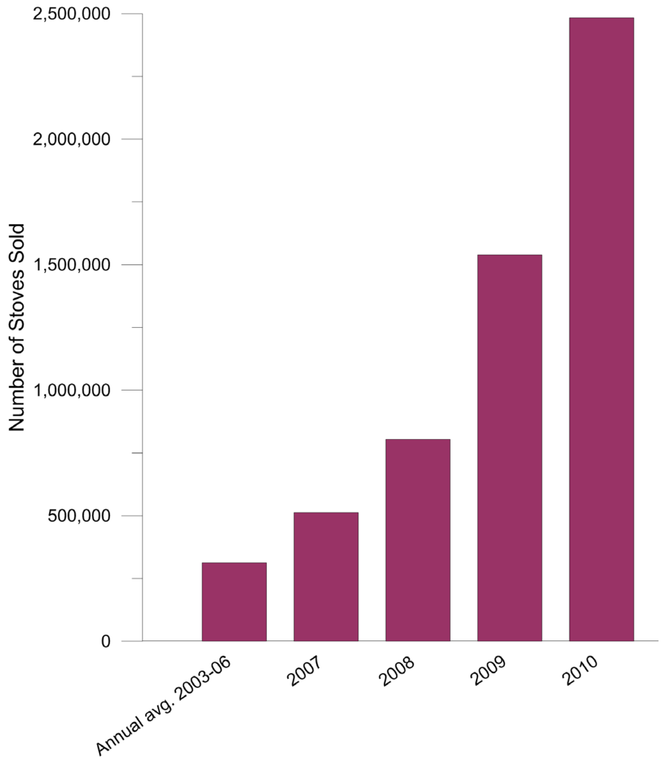


Figure 10-11. Number of Improved Stoves Sold by PCIA Partners, 2003-2010. (Source: U.S. EPA)

500 partners, PCIA indicates that partners sold 2.5 million stoves in 2010 (see Figure 10-11). Based on the latest survey results, PCIA Partners are more than doubling their stove sales every other year. This does not include the internal Chinese stove market and independent manufacturers that make and sell different versions of the so-called "Jiko" charcoal stove across Africa. Including these sales, the total number could be as high as 5 million to 10 million stoves per year, though there is not reliable international data on the quality or performance of many of these stoves, limiting the assessment of climate and health benefits. Despite this progress, the total impact of the cookstove replacements to date has been small, given that the total stove market is on the order of 500 million to 800 million homes.

In addition to the design and fuel innovations noted above, a number of recent developments point to a much greater potential for making large-scale progress in the cookstove sector. These include:

- *Growth of Existing Businesses and Business Models:* An increasing number of businesses are manufacturing and/or selling improved stoves and fuels, utilizing a wide range of business models. These models include non-governmental organizations (NGOs) working to catalyze local businesses around a common and tested stove design (e.g., GERES Cambodia's local partners just sold their 1.5-millionth stove); working to develop local businesses to make and sell artisanal stoves (e.g. GIZ's global efforts to provide over 4 million homes with improved stoves over the past 5 years); a local factory selling directly (e.g., HELPS/Guatemala has grown rapidly and sold over 100,000 stoves); international manufactures with local distributors (e.g., a partnership between the Aprovecho Research Center in Oregon on design, Shengzhou Stove Manufactures in China, and Colorado-based EnviroFit International on sales); and major corporations building their business in emerging markets (e.g., Philips, Bosch-Siemens).

- *New Scalable Technologies:* Many of the stoves noted above represent a new suite of stove technologies that are well designed and durable, and for which extensive emissions testing has been conducted. Such stoves could be mass produced, which would improve the scalability of these solutions (Venkataraman et al., 2010).

- *Carbon Financing:* Cookstove businesses are increasingly leveraging carbon financing in both the formal and voluntary markets to provide capital and increase public awareness. The financing arrangements vary substantially, but typically yield about 0.5 to 2 tons of CO_2-equivalents per stove per year for improved wood and charcoal stoves, and up to 3 to 5 tons of CO_2-equivalents per stove per year for improved coal stoves. Importantly, however, these credits are based on GHG (mostly CO_2) emissions reductions, as measured by reductions in fuel use during in-field tests. Additional work would be required to establish credits for BC reductions. Carbon financing is already transforming financing of cookstove efforts into more rigorous financial transactions with rigor and accountability for stoves sold, stove performance in the field, and stove utilization over time. The high transactions costs involved in obtaining project approval also incentivize large-scale projects and encourage the continued use of approved stoves for many years to generate ongoing credits. Impact investing is a separate, but important opportunity to bring social capital investments to this field, and examples of this tool applied to the cookstove field are beginning to emerge.

- *New Testing and Monitoring Tools:* The demand for rigorous monitoring for carbon and other financing, research, and other needs has also led to the development of less expensive and more effective monitoring technologies that greatly improve our ability to measure and interpret field results. These include relatively inexpensive $PM_{2.5}$ monitors, BC monitors, personal exposure monitors for CO and $PM_{2.5}$, portable stove emissions testing hoods, stove use monitors, and cell-phone based wireless monitoring tools.

In spite of this progress, achieving large-scale adoption of clean cooking solutions will not be easy, and many remaining barriers must be addressed. A recent World Bank study has summarized some of the key challenges, emphasizing the need for a range of stoves that meet users' needs, with demonstrated ability to reduce fuel use and indoor smoke, while maintaining durability and safety. The report also notes that successful programs require functioning commercial markets in order to reach and maintain a large scale of success globally. Innovative financing techniques and well-constructed monitoring and evaluation programs were other tools highlighted as critical to success in reaching the poor (World Bank, 2010). Other major considerations include:

- *Institutional Barriers:* Such barriers include the lack of accepted international standards for different stove-fuel combinations, the lack of independent stove testing facilities in market places around the world, and the lack of health guidelines regarding what interim targets on what is considered a "clean" stove.

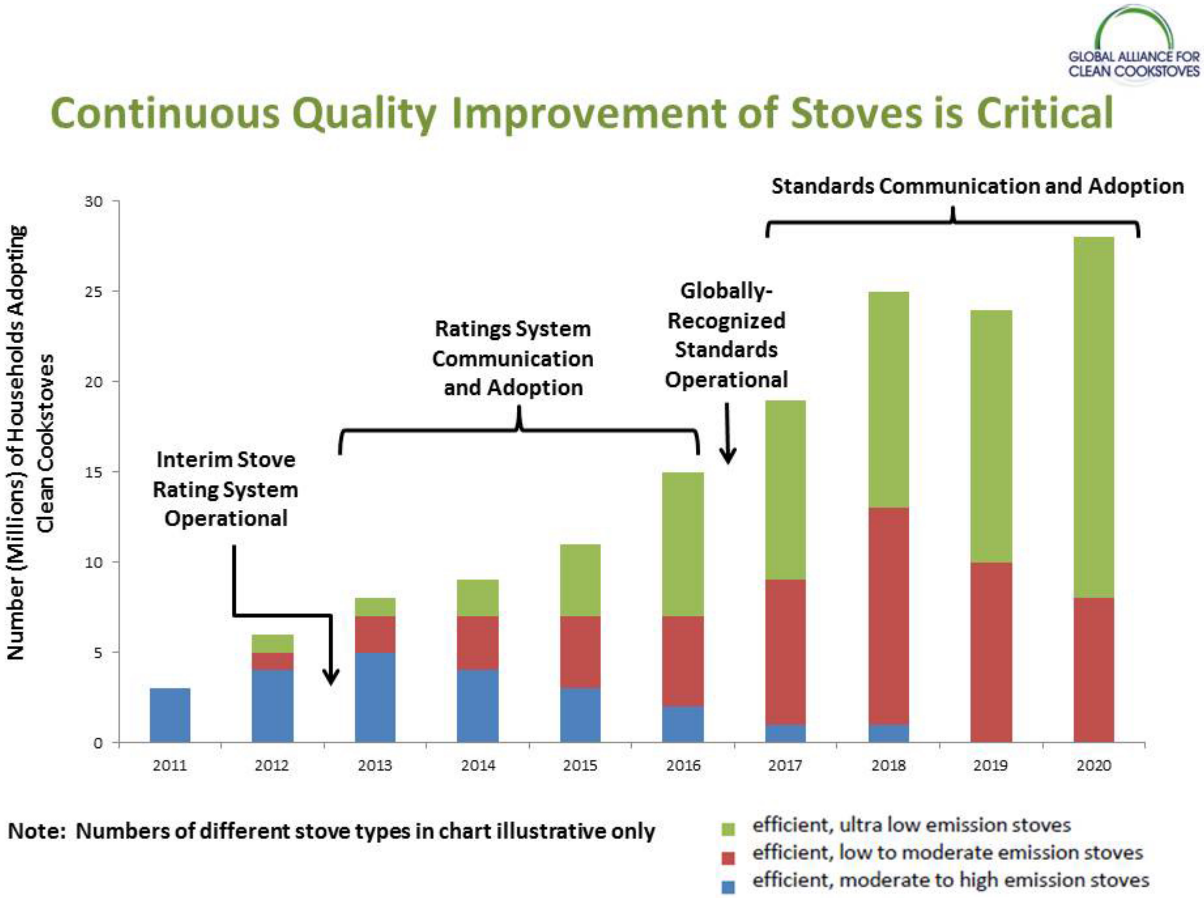
- *Cost:* The cost of improved stoves and fuels alone pose a major challenge for many households. Additional financial barriers include tariffs and duties to import stoves, the large investment needed to take a prototype stove to mass production, the cost and difficulty of developing distribution chains in target markets, the high transactional costs of carbon financing, and the costs of managing an inventory for a widely fluctuating market during business start-up. Separate financing tools are needed make advanced stoves affordable for the poorest populations.

- *Social Barriers:* Cooking solutions must be designed to meet local cooking needs – cooking the local food, in the timeframes needed, with locally available fuels. Solutions for one part of the world may not be applicable in other parts of the world. Past "improved" stoves have not always been designed with the needs and social practices of end users in mind. By extensively testing prototype stoves with users, commercial businesses have been able to lessen these risks. Experience indicates that a full portfolio of solutions will be needed to meet the many cooking needs of the developing world – including preferences for a variety of cooking options (just as most kitchens in the developed world use not use a gas or electric stove, but an oven, a microwave oven, an outdoor grill, a toaster, and many more specialized cooking devices).

- *Global Leadership:* Coordination and cross-disciplinary leadership is needed to pursue integrated solutions that address each of the climate, health, gender, forestry, energy, agricultural, and other dimensions of the cookstove issue. In the past decade, several new efforts have emerged that have brought new focus to the health and climate risks of cookstoves, and new rigor to solutions to these risks. These include the U.S. EPA-led PCIA, the Shell Foundation's Breathing Space program, GIZ's HERA program, and SNV's global biogas efforts, as well as more isolated investments by the World Bank, USAID, and several agencies focused on refugee camps (e.g., United Nations High Commissioner for Refugees and World Food Programme).

In September 2010, the United Nations Foundation and nineteen founding partners launched the Global Alliance for Clean Cookstoves. The Alliance is a new public-private initiative whose mission is "to save lives, improve livelihoods, empower women, and combat climate change by creating a thriving global market for clean and efficient household cooking solutions." The Alliance will work closely with private, non-governmental, UN and other partners to expand efforts to address the global and local barriers that have limited the scope of cookstove replacements. The Alliance has set an interim goal of having 100 million new homes adopt clean and safe cooking solutions by 2020. Achieving this goal will likely entail the sale through commercial distribution channels of well over 100 million stoves in total, with both the quantity of sales and the quality of performance growing substantially over time (see Figure 10-12).

The development of standards for what constitutes a low-emitting stove is essential for ensuring improvements in performance over time. PCIA and the Alliance have joined together with the cookstove community to pursue the development of voluntary global standards through an inclusive, transparent process with the International Standards Organization (ISO). The development of these standards will proceed in parallel with growth in the global stove market. In the early years, distribution chains and businesses will be built around the sale



Note: Numbers of different stove types in chart illustrative only

Figure 10-12. Potential Growth in the Number of Households Adopting Clean Cookstoves Globally through 2020. The Global Alliance anticipates that the market for clean cookstoves will continue to evolve, in parallel with development of cookstove standards. The Alliance has set a goal of 100 million clean stoves by 2020. Numbers of different stove types in the chart are illustrative only. (Reproduced from U.N. Foundation, 2011.)

of available, mostly mid-range stoves. As these distribution channels are built, however, newer advanced solutions will supplant them as the early purchases wear out and are replaced. By the end of the decade, it is the Alliance's goal that most stoves sold will be of the advanced (efficient and very low emission) variety. It is these more advanced solutions that are likely to achieve the more significant BC reductions, as well as the more dramatic health benefits.

While open fires or crude stoves are not a significant source of BC emissions in the United States, the U.S. government has been at the forefront of the effort to establish the Alliance and is a leading partner to the Alliance. The U.S. Department of State is leading Alliance diplomacy to raise the visibility of the issue and engage new country and other partners, and several agencies (EPA, U.S. Department of Health and Human Services [HHS, including the National Institutes of Health and the Centers for Disease Control and Prevention], DOE, USAID, Overseas Private Investment Corporation, Peace Corps, U.S. Department of Agriculture, and the National Oceanic and Atmospheric Administration) are contributing substantially to the Alliance through applied research (on technology, health, stove testing, distribution, adoption, climate, biofuels, forestry), financing, and distribution.

Since its launch , the Alliance has identified up to $120 million in partner commitments, including about $20 million for operations, over $50 million for research, and up to $50 million in financing; brought on more than 275 partners, including 28 country partners; catalyzed the process to develop international consensus standards for clean cookstoves; funded a Kenya study to assess various cookstoves to determine the benefits for children's health; supported regional Alliances in Asia, Latin

America, and Africa to spur local businesses and solutions; built up technical capacity of regional stove testing centers in China, Ethiopia, and other countries; executed comprehensive market analyses of the clean cookstove sector in five countries; supported the development of indoor air guidelines by the World Health Organization; ensured inclusion of household air pollution as a risk factor for non-communicable diseases in the Political Declaration for the UN General Assembly; sponsored the Fifth Biennial PCIA Forum and two international technical research workshops; begun integration of the Alliance with the Partnership for Clean Indoor Air; and worked with UN agencies to improve collaboration among UN cookstoves and fuel programs. The Alliance recently released a first-ever sector-wide strategy report (called *Igniting Change*; U.N. Foundation, 2011) that lays out a strategy for universal adoption of clean cookstoves and fuels, and its 10-year business plan is forthcoming in 2012.

Solutions on this scale are needed to resolve the tremendous human health and environmental burden—including the climate impacts—of traditional cookstove use. As the above discussions indicate, large scale success in this field may be within reach. Substantial reductions in BC on the order of 90% to 95% per household likely depend on switching to cleaner fuels or advanced biomass stoves. Such highly efficient, clean stoves help meet multiple goals, including fuel efficiency, health protection, low climate impacts, and reduction of outdoor pollution (Venkataraman et al., 2010).

Currently, simple unimproved stoves dominate the marketplace. Most current improved stove sales are of the intermediate variety – rocket stoves or other solutions that achieve important health[11] and fuel use benefits, but will not achieve the large health and BC benefits sought. As the Alliance advances towards its interim goal of reaching 100 million homes, solutions will need to evolve towards cleaner fuels and more advanced stoves to ensure that substantial public health and BC benefits are achieved. Additional research and innovations are needed to bring these very clean solutions to massive populations and to move as rapidly as possible to achieve the health and climate benefits that advanced stoves can bring to families and the environment.

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[11] 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).

Chapter 11

Mitigation Approaches for Open Biomass Burning

11.1 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.
  - Wildfires account 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.
- Expanded wildfire prevention efforts may help to reduce BC emissions from wildfire both domestically and globally. 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.

11.2 Introduction

This chapter presents currently available information regarding mitigation efforts and techniques that may help reduce particle emissions from open biomass burning (agricultural burning, prescribed burning, and wildfires). The effectiveness of these controls on emissions of BC and OC (including brown carbon) requires further study. In addition, given the importance of planned fire as a land management tool, there are important tradeoffs that must be considered in evaluating mitigation options for open biomass burning.

11.3 Emissions from Open Biomass Burning

Open biomass burning, as discussed in this report, encompasses three main categories of burning: agricultural burning, prescribed burning, and wildfire.[1] Table 11-1 describes each type of open biomass burning, the land types on which they may occur, and examples of typical resource management objectives each burning type is designed to achieve. In some cases, there are slight differences in how these terms apply to domestic and international burning practices.

The Joint Research Centre of the European Commission estimates that 350 million hectares (865 million acres) of land were affected by

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[1] Categories of contained biomass combustion, including residential heating and cooking and industrial biomass combustion, are addressed in previous chapters.

Table 11-1. 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, invasive species, pests, and disease; manage lands for endangered species; promote various vegetation responses; reduce fuel loading to reduce catastrophic wildfire risk.
Wildfire	An unplanned 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.

fire, worldwide, in 2000 (Food and Agriculture Organization of the United Nations, 2007). However, given the lack of an international standard for fire terminology and the lack of consistent data reporting and collection, it is not possible to distinguish among the fractions of land area that were subject to agricultural versus prescribed burning or wildfire (Food and Agriculture Organization of the United Nations, 2007). Generally, the mass of BC emitted from open biomass burning will depend on the size and duration of the fire, fuel type, fuel conditions, fire phase, and the meteorological conditions on the day of the burn. The emissions estimates presented in Chapter 4 indicate that open biomass burning represents a potentially large, though poorly quantified portion of the U.S. BC emissions inventory. As with the international fire emissions inventories, available data are limited regarding the percentage of land area affected by different types of burning. It is also important to note that emissions of OC are seven times higher than BC emissions from this sector. Preliminary research suggests that the OC fraction may be dominated by BrC, which also absorbs light. More focused research is needed to clarify the composition and quantity of emissions from different types of fires.

As the estimates in Chapter 4 indicate, open biomass burning is the largest BC source in Africa, Central and South America, and Asia, and is one of the largest sources of BC in Russia/Central Asia (the former USSR) and North America. However, there is considerable variation in the type of open burning that dominates in different regions. Fires in sub-Saharan Africa are primarily due to slash-and-burn practices for clearing agricultural sites, burning of crop residues, escaped planned burning, acts of carelessness, and arson (Food and Agriculture Organization of the United Nations, 2007). The primary causes of fire in Central and South America include large-scale conversion of moist tropical forest to rangeland and agriculture, arson, negligence, and hunting (Food and Agriculture Organization of the United Nations, 2007). Available information suggests that the majority of fires in China and other East Asian countries are uncontrolled wildfires, typically caused during land conversion, or by arson and acts of carelessness (Food and Agriculture Organization of the United Nations, 2007). Prescribed burning is used to some degree in China to reduce catastrophic wildfire risk (Morgan, 2009). In India, and other South and Southeast Asian countries, fire emissions stem from agricultural burning, rangeland clearing, escaped planned burning, or acts of carelessness (Food and Agriculture Organization of the United Nations, 2007). Agricultural burning in Kazakhstan, southern Russia, Central and Eastern Europe is a seasonal occurrence, typically starting at the end of April and lasting for a few weeks (Warneke et al., 2009; Stohl et al., 2007). Wildfires in Russia (Siberia) are primarily caused by lightning, escaped planned burning, or acts of carelessness (Food and Agriculture Organization of the United Nations, 2007), and occur from late April throughout the summer (Warneke et al., 2009; Generoso et al., 2007). Russia experiences many smoldering fires in drained or dry peatlands that burn for long periods and produce large quantities of smoke (Food and Agriculture Organization of the United Nations, 2007). In the Far East and southern Siberian portions of Russia, extensive prescribed burning of the grasslands has been used in the spring to reduce highly flammable surface fuels (Food and Agriculture Organization of the United Nations, 2007).

As described in Chapters 2 and 4, there is strong evidence to suggest that emissions from fires in one world region can significantly impact other world regions through transport and deposition processes. Reduced snow and ice albedo, and increased rates of melting in the Arctic, the Himalayas, and other snow- and ice-covered regions of the world are major impacts of BC deposition, with implications for freshwater resources in regions dependent on snow-fed or glacier-fed water systems. Most of the BC that reaches the Arctic has been traced to sources in the Northern mid-latitudes (AMAP, 2009), with open biomass burning as one of the largest of the sources. A primary determinant of the downwind impact of a large fire on snow- and ice-covered regions is the height to which the plume rises (i.e., its injection height). Fire plumes observed by satellite between 1978 and 2009 have shown that more dense wildfire plumes rose to the level of the free troposphere (i.e., 8 km), where long-range transport can occur more readily, over North America than over Australia, or Russia and Northeast Asia (Guan et al., 2010). This difference has been attributed to the type of wildfire that dominates in North America (i.e., boreal crown fires[2] that are large and very high in temperature). In general, between 5 and 28% of the plumes from large wildfires in North America rise into the free troposphere (Val Martin et al., 2010).

Current emissions projections suggest that direct PM emissions from open biomass burning will continue to dominate global BC inventories. In addition, several major climate change science assessments have concluded that large, catastrophic wildfires will likely increase in frequency over the next several decades because of climate warming (Field, 2007; Ryan et al., 2008; Wiedinmyer and Hurteau, 2010; Littell et al., 2010). General climate warming encourages wildfires by extending the summer period that dries fuels and promoting easier ignition and faster spread (Field, 2007). Earlier spring snowmelt has led to longer growing seasons and drought, especially at higher elevations where the increase in wildfire activity has been greatest (Field, 2007). Increased temperature in the future will likely extend fire seasons throughout the western United States, with more wildfires occurring both earlier and later than is currently typical, and will increase the total area burned in some regions (Field, 2007). Within Arctic regions, climate change is expected to shift the treeline northward, with forests replacing a significant portion of land that is currently tundra and tundra vegetation moving into currently unvegetated polar deserts (ACIA, 2004). Changes in Arctic climate are also expected to increase the frequency, severity, and duration of wildfires in boreal forests and dry peat lands, particularly after melting of permafrost (Schneider et al., 2007; ACIA, 2004).[3] These climate-related changes in wildfire location, duration, and frequency will affect both BC and OC emissions.

11.4 Fire as a Resource Management Tool

Fires play an important ecological role across the globe, benefiting those plant and animal species that depend upon natural fires for propagation, habitat restoration, and reproduction. Most North American plant communities evolved with recurring fire and are dependent on recurring fire for maintenance. Ecosystem fire regime analysis includes information about the necessary fire return interval which may vary from one to two years for prairies, three to seven years for some long-needle pine species, 30-50 years for species such as California chaparral, and over 100 years for species such as Lodgepole pine and coastal Douglas-fir.

Natural fires also reduce fuel load, unnatural understory, and tree density, helping to reduce the risk of catastrophic wildfires. In many parts of the United States, historical land management practices during the late 19th and early 20th centuries (e.g., fire suppression, logging, and livestock grazing)

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[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.

[3] Peat fires are also becoming more common in the lower 48 states, as illustrated by the Evans Road Fire (2008) and Pains Bay Fire (2011), both of which occurred in North Carolina. Peat fires have very high emissions relative to fires involving other types of fuels.

have altered the natural fire regime, changed forest structure, and led to heavy fuel accumulation in forests. This, in turn, has increased the size of wildfires and total area burned (Miller et al., 2009; Noss et al., 2006; Allen et al., 2002; McKelvey and Busse, 1996). Accumulated fuel loads will likely continue to affect the size and frequency of large wildfires in the coming decades.

In the United States, prescribed burning is one of the basic tools relied upon by land owners and managers to achieve multiple management objectives in fire-dependent ecosystems. When one management objective is to maintain a fire-dependent ecosystem, the effects of fire cannot be duplicated by other tools. Prescribed fire can also be used to reduce heavy fuel loads, which has the benefit of helping to prevent catastrophic wildfires.

The following section includes an outline of strategies that can be used for conducting prescribed and agricultural burning in a manner that protects air quality by reducing smoke emissions, and managing burning conditions to protect downwind populations. In addition, the importance of fire prevention is discussed. These methods may also be applied with the goal of reducing BC emissions overall, and/or the goal of reducing downwind deposition of BC on snow and ice. As will be discussed, the techniques listed may be more useful in some ecosystems than in others. Further study is needed to identify appropriate strategies to apply under each circumstance.

11.5 Smoke Mitigation Technologies and Approaches in the United States

Appropriate mitigation of BC from open biomass burning depends on the type, timing, and location of burning and must balance multiple objectives including resource management, climate protection, and health protection. Currently available literature is extremely limited regarding the effectiveness of any given mitigation practice for reducing BC emissions from the three general types of burning. More research is needed to better understand the efficacy, potential unintended consequences, and cumulative effects arising from the implementation of any proposed mitigation techniques.

As a starting point, however, it is appropriate to consider how approaches currently used to manage the air quality impacts of open biomass burning may be applicable to BC. Most U.S. domestic policies and programs at the local, state, and federal level focus on protecting air quality and public health by managing smoke and minimizing PM emissions.

There are two basic approaches that are commonly applied to manage the air quality impacts from open biomass burning: (1) use techniques that reduce the emissions produced for a given area; and/or (2) redistribute the emissions through meteorological scheduling and by sharing the airshed (Ottmar et al., 2001).

One common approach in the United States for limiting the impacts of open biomass burning is the development and application of smoke management programs. The Interim Air Quality Policy on Wildland and Prescribed Fires (U.S. EPA, 1998)[4] recognizes the role fire plays as a resource management tool. The policy addresses wildland and prescribed burning managed for resource benefits on public, tribal, and privately-owned wildlands. The policy integrates two public policy goals: (1) to allow fire to function, as nearly as possible, in its natural role in maintaining healthy wildland ecosystems and, (2) to protect public health and welfare by mitigating the impacts of fire emissions on air quality and visibility. The policy encourages state and tribal authorities to adopt and implement smoke management programs to mitigate the public health and welfare impacts from prescribed fires and promote communication and coordination of prescribed burning among land owners. A smoke management program can be extensive and detailed, or can simply identify basic smoke management practices for minimizing emissions and controlling impacts from a prescribed fire.

Based on regulations, the EPA allows the use of basic smoke management practices in lieu of smoke management programs, where appropriate. The Agency intends to issue guidance on the use of basic smoke management practices in the revised Air Quality Policy on Wildland and Prescribed Fires when it is finalized. Basic smoke management practices could include, among other practices, steps to minimize air pollutant emissions during and after the burn, evaluate dispersion conditions to minimize exposure of sensitive populations, and identify procedures to ensure that burners are using basic smoke management practices. USDA recently issued a technical document outlining some potential basic smoke management practices.[5]

A smoke management program establishes a basic framework of procedures and requirements for

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[4] 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 is currently under revision.

[5] See http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1046311.pdf.

planning and managing smoke from prescribed fires. It is typically developed by a state/tribal agency with cooperation and participation by various stakeholders (e.g., public/private land owners/managers, the public). If a state/tribe determines that a smoke management program is needed, they may choose to develop a program using an array of smoke management practices/emissions reduction techniques that they believe will prevent air quality violations and address visibility impairment. A smoke management program can range from a purely voluntary program to a program where prescribed fires are regulated by a permitting authority that analyzes meteorological conditions and air quality considerations and authorizes burning by time of day, fire location/size and anticipated duration. The more-structured program may include enforceable requirements on who may burn and when burning may occur.

The basic elements of a smoke management program include guidelines or requirements regarding authorization to burn, coordination and scheduling, and air quality assessment (U.S. EPA, 1998). In cases where smoke management programs are developed, these generally focus on: (1) actions to minimize emissions (emission reduction techniques); (2) evaluation of predicted smoke dispersion; (3) public notification; (4) contingency measures to reduce exposure; and (5) fire monitoring and plume dispersion characteristics. In addition, smoke management programs frequently lay out guidelines or requirements for recordkeeping and reporting; public education and awareness; surveillance and enforcement; and program evaluation.

In developing a smoke management program, authorities have a number of options available for reducing emissions (e.g., emissions reduction techniques (ERTs)) and managing smoke that can be applied under different circumstances. It is important to note, however, that decisions regarding the appropriate use of different techniques are influenced by a number of considerations—including but not limited to air quality impacts, water quality impacts, Endangered Species Act requirements, and basic resource management objectives. It is also important to note that land managers take safety into consideration when choosing smoke management strategies and ERTs. The following section provides an overview of the current practices employed for mitigating air quality impacts.

11.5.1 Managing Smoke

Many methods for managing smoke, including emissions reduction techniques, may offer the benefits of reduced BC emissions and reduced downwind impacts related to BC deposition on snow and ice, although there are significant uncertainties regarding transport of prescribed fire emissions to the Arctic regions. However, there is still substantial uncertainty about the applicability and effectiveness of these emissions reduction techniques for reducing BC under diverse, site-specific burning conditions. The appropriateness of a given mitigation practice and its effectiveness at reducing $PM_{2.5}$ and/or BC will depend on the type of fuel being burned (e.g., crop residue or forest), the management objectives of the burn, and the seasonal timing and geographic location of the burn. An additional consideration is that open biomass burning occurs on land under various ownership (i.e., federal, state, tribal, and private), which affects management decisions and the types of burning practices implemented on those lands. Currently available literature identifies a number of current fire management practices to address air quality impacts of PM emissions from agricultural and prescribed burning. These practices are listed below.

11.5.1.1 Agricultural Burning PM Mitigation Techniques

- Reduce the number of acres burned
    - Reduce burning through conservation tillage, soil incorporation, or collecting and hauling crop residues to central processing sites (WRAP, 2002).
    - Apply alternate year burning which involves alternating open field burning with various mechanical removal techniques. The period may involve burning every other year or every third year (U.S. EPA, 1992).
- Increase combustion efficiency
    - Use bale/stack for agricultural residue. The bale/stack burning technique is designed to increase the fire efficiency by stacking or baling the fuel before burning. Burning in piles or stacks tends to foster more complete combustion, thereby reducing PM emissions. This control is applicable to field burning where the entire field would be set on fire, and can be applied to all crop types (U.S. EPA, 2005b).
    - Use propane flamers as an alternative to open field burning.

  - Use backing fires ("backburning"). Flaming combustion is cleaner than smoldering combustion. Backburning ensures more fuel is consumed in the flaming phase (Ottmar et al., 2001).

- Reduce fuel loadings
  - Remove straw/stubble before the burn.

- Change burn timing from early spring to either winter or summer to reduce higher impact of BC on snow/ice. Quinn et al. (2008) suggest that this technique may be especially important for mitigating climate impacts in the Arctic, to reduce springtime deposition when the snow and icepack is large. Applicability of this technique will be limited by the type of crop, the resource objectives sought, and biological and operational constraints.

- Convert Land Use
  - Convert from a crop that requires burning to a crop that does not.
  - Convert land to non-agricultural use.

- Educate Farmers
  - Provide training to farmers on proper burning techniques that reduce emissions.

11.5.1.2 Prescribed Burning PM Mitigation

- Reduce the area burned
  - Use mosaic burning. Landscapes often contain a variety of fuel types that are non-continuous and vary in fuel moisture content. Prescribed fire prescriptions and lighting patterns can be assigned to use this heterogeneity in fuel and fuel moisture to mimic a natural wildfire and create patches of unburned areas or burn only selected fuels (Ottmar et al., 2001).

- Reduce fuel consumed (Ottmar et al., 2001)
  - Burn fuel when moisture content is high. Fuel consumption and smoldering can be minimized by burning under conditions of high fuel moisture of duff, litter, and large woody fuels.
  - Conduct burns before precipitation. Scheduling a prescribed burn before a precipitation event may limit the consumption of large woody material, snags, stumps, and/or organic ground matter.

- Reduce fuel loadings (Ottmar et al., 2001)
  - Burn outside the growing season, burn after timber harvest, and burn frequently. Prescribed burning at appropriate times can help reduce the size and magnitude of wildfires.
  - Expand the use of biomass. Harvesting and selling or trading the biomass is one alternative to prescribed burning. Woody biomass can be used in various industries such as pulp and paper, methanol production, and garden bedding. This alternative is most applicable in areas that have large diameter woody biomass and the biomass is plentiful and accessible so as to make biomass utilization economically viable. Small-diameter biomass can be used as posts, poles, or tree stakes. Neary and Zieroth (2007) documented a successful USDA Forest Service project in Arizona to remove and sell small-diameter trees for use in small power plants that burn wood fuel pellets. Biomass can also be pyrolyzed to produce biochar, a fine-grained charcoal, for use as a soil amendment (i.e., to improve physical properties of the soil, such as water retention, permeability, water infiltration, drainage, aeration and structure).[6]
  - Use other fuel treatments such as mechanical treatments/removal. Mechanical treatments may be appropriate when management objectives are to reduce fuel density to reduce a wildfire hazard, or to remove logging waste materials (slash) to prepare a site for replanting or natural regeneration. On-site chipping or crushing of woody material, removal of slash for off-site burning or biomass utilization, whole tree harvesting, and yarding (pulling out) of unmerchantable material may accomplish these goals. Mechanical treatments are normally limited to accessible areas, terrain that is not excessively rough, slopes of 40% or less, sites that are not wet, areas not designated as national parks or wilderness, areas not protected for threatened and endangered species, and areas without cultural or paleological resources.

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[6] See *http://www.biochar.org.*

  - Use chemical treatments. When the management objective is to preclude, reduce, or remove live vegetation and/or specific plant species from a site, chemical treatments may be appropriate tools. However, other potential environmental impacts caused by applying chemicals must also be considered.

  - Use animal grazers. Increasing grazing by sheep, cattle, or goats before burning on rangelands and other lands can reduce grassy or brushy fuels prior to burning, and can help reduce burn frequency.

- Increase combustion efficiency

  - Use mass-ignition techniques that produce short-duration fires (e.g., aerial ignition). Mass ignition can shorten the duration of the smoldering phase and reduce the amount of fuel consumed. However, mass-ignition may also increase plume rise. Therefore, all methods should be evaluated specifically for BC.

  - Use backing fires (see above).

  - Burn piles or windrows. Fuels concentrated into clean and dry piles or windrows generate greater heat and burn more efficiently.

  - Use air curtain incinerators, which are large metal containers or pits with a powerful fan device to force additional oxygen into the fire, to produce a very hot and efficient fire with very little smoke. Air curtain incinerators offer a useful alternative to current fuel reduction and disposal methods, providing the benefits of producing lower smoke emissions compared to pile or broadcast burning; burning a greater variety, amount, and size of materials from dead to green vegetation; reducing fire risk; operating with fewer restrictions in weather and burn conditions; and containing burn area to a specific site.[7]

- Education for Resource Managers

  - Train resource managers on proper burning techniques to reduce emissions.

Currently available literature is extremely limited regarding the cost of reducing BC emissions from agricultural and prescribed fire. Many of the PM emission reduction techniques described above require substantial infrastructure and resource investment (e.g., roads, machinery, etc.) or the existence of a market for biomass utilization products (e.g., wood pellets or biochar). The availability of the required infrastructure, resources, and markets will vary across the country, making the cost of potential mitigation options highly uncertain and dependent on the technique(s) and the site-specific environmental conditions in which the technique(s) are applied. A recent study (Sarofim, 2010) surveyed currently available literature to develop rough cost estimates for the major categories of PM emission reduction techniques described above (i.e., increase combustion efficiency, reduce fuel consumed, reduce fuel loadings, and reduce the area burned).[8] The authors found that these techniques are on the whole likely to be quite expensive for the amount of BC reduced, although there may be potential for lower cost mitigation approaches in locations where markets for biomass utilization exist.

11.5.2 Fire Prevention Techniques

While wildfires are part of the natural functioning of many ecosystems, increasing fuel loads within the United States over the past century have made wildfires harder to control and more expensive to suppress. In addition, wildfires often pose a dangerous threat to the lives and property of civilians and firefighters. Fire prevention techniques can be effective in helping to prevent unplanned human-caused fires. Wildfire prevention efforts in general can be seen as an important strategy for limiting BC emissions both domestically and globally. A number of studies have discussed the timing and structure of prevention efforts to ensure optimal effectiveness in limiting the extent and severity of wildfires (Prestemon et al., 2010; Butry et al., 2010a; Butry et al., 2010b).

Efforts by the U.S. Forest Service and other resource management agencies are currently underway to turn fire suppression programs into more proactive fire management programs that effectively apply fire prevention and hazardous fuels reduction techniques, extensive public education, and law enforcement (National Interagency Fire Center, 2011).

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[7] See *http://www.fs.fed.us/eng/pubs/html/05511303/05511303.html*.

[8] 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.

Fire prevention approaches involve a combination of engineering, education, and enforcement.[9] Education strategies often represent low-cost approaches for preventing unplanned fires. Such strategies must include clear planning and communications with regard to subjects such as fire-prone areas where access is closed or restricted; appropriate use of campfires, smoking, and fireworks; and managing the burning of trash and debris. Raising public awareness through education and outreach, including utilizing media such as newspapers, radio, and television, is also important. Such educational campaigns can be highly effective in preventing unwanted fires: the U.S. Forest Service's long-standing Smokey Bear campaign is among the most successful fire prevention awareness and education campaigns ever conducted (National Wildfire Coordinating Group, 2007).

11.6 Mitigation Technologies and Approaches Globally

As discussed in Chapter 2, a number of recent studies have pointed to the importance of reducing international BC emissions from open biomass burning to alleviate effects on the Arctic, the Himalayas, and other key snow and ice-covered regions. Many of the mitigation techniques and approaches described above could also be applied internationally, and such strategies could provide important climate benefits. However, the practical mitigation options available on the ground in different regions are limited for a number of reasons. Critical barriers to implementing mitigation measures internationally fall within three areas: (1) weak governance (e.g., requisite laws and policies at all levels of government to authorize and enforce fire management practices); (2) lack of local capacity (e.g., requisite funding, training, equipment, and human resources to implement fire management); and (3) lack of support infrastructure (e.g., roads and other infrastructure to access rural areas prone to wildfire, monitoring and early warning systems to detect and track fires).

According to the Food and Agriculture Organization of the United Nations (2007), many African countries particularly in sub-Saharan Africa have no central government fire management policy, and there is a widespread lack of support infrastructure, funding, equipment, and adequately trained human resources for fire management. While most countries in Central, South and Southeast Asia have a government fire policy, limited funding resources restrict their ability to establish or maintain effective fire management programs (Food and Agriculture Organization of the United Nations, 2007). According to Morgan (2009), the Association of Southeast Asian Nations instituted a "zero burning" policy in 1999, but it has been largely ineffective. China, Japan, and South Korea have advanced fire detection systems, including the use of remote sensing (Morgan, 2009), but often at the local level, villages and communities lack resources, adequate training, and professional expertise to control large wildfires (Food and Agriculture Organization of the United Nations, 2007). In many countries in South America, illegal burning even on state-protected lands is widespread due to the absence of enforcement and criminal penalties (Food and Agriculture Organization of the United Nations, 2007). Russia has well-defined laws regulating forest burning practices, but lacks strong enforcement (Food and Agriculture Organization of the United Nations, 2007).

Given these challenges, addressing fundamental barriers to implementation may be just as or more important than identifying and promoting more technological forms of mitigation such as specific burning techniques. Capacity-building efforts may include building basic fire management infrastructure, strengthening governance structures to create and enforce fire policies, and developing economic alternatives to slash-and-burn agriculture. In addition, fire prevention efforts may be important for mitigating wildfire globally. Fire prevention education for the general public and training for workers in the agricultural and forestry sectors in the controlled use of fire will also be important.

There is relatively little information regarding costs of open biomass burning mitigation internationally. Mitigation costs will vary according to country, and will likely be higher in developing countries due to more extensive barriers to implementation as described above. These costs will depend on local environmental conditions, ecosystem type, fire management capacity, and support infrastructure. Costs may also vary within individual countries, according to locality, because authority and responsibility for fire management is often decentralized and is left up to local or regional authorities (Food and Agriculture Organization of the United Nations, 2007).

To address the impact of open biomass burning internationally, the United States has recently initiated research efforts and other international cooperative activities to evaluate and reduce BC emissions from open biomass burning in and around

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[9] Additional information on each of these strategies is available on the National Wildfire Coordinating Group's publications page at http://www.nwcg.gov/teams/wfewt/products.htm.

the Arctic. The U.S. State Department is coordinating a $5 million Arctic Black Carbon Initiative that will fund a number of activities, including a project by the U.S. Department of Agriculture (USDA) to address biomass burning emissions in Eurasia. USDA's multi-agency program contains the following components (USDA, 2010):

- **Research Activities:** USDA scientists (led by the U.S. Forest Service and Agricultural Research Service) will seek to improve estimation of emissions and transport of BC from agricultural burning and forest fires by quantifying spatial and temporal patterns of these emissions in Eurasia and conducting an assessment of long-range transport of BC from fires in Russia and adjoining regions to the Arctic. The research will identify meteorological conditions and potential source locations for Arctic transport of smoke and analyze agronomic practices in Eurasia to identify opportunities for reduced use of agricultural burning. Initial results from this project are shown in section 4.5 of this report, which discusses long-range transport of emissions. By examining the ability of the atmosphere to transport emissions to the Arctic, the project can identify which source regions are most likely to contribute to emissions transport to the Arctic. One of the benefits of this project is that the results are independent of source type, and therefore applicable beyond biomass burning. Injection height is shown to be critical (see section 4.5 for more detail).

- **Technical Exchange and Other Cooperative Activities:** The U.S. Forest Service and Foreign Agricultural Service will implement technical exchanges and cooperation between U.S. and Russian experts on BC, agricultural burning, and fire management. These efforts will support training activities and the development and implementation of innovative local-level "pilot" programs designed to illustrate strategies and practices that could be more broadly applied to reduce any negative environmental impacts of agricultural and forest fires. Key issues include interagency cooperation on fire management, fire budgets, and GIS and remote sensing. USDA will also facilitate public-private partnerships to develop local-level fire wardens and fire brigades in Russia and outreach to farmers in Russia to increase awareness of approaches to reduce BC emissions from agricultural burning.

Chapter 12

Key Black Carbon Mitigation Opportunities and Areas for Further Research

12.1 Summary of Key Messages

- Mitigation of BC offers a clear opportunity: continued reductions in BC emissions can provide significant near-term benefits for climate, public health, and the environment.

- Effective control technologies and approaches are available to reduce BC emissions from a number of key source categories.

  - BC emissions reductions are generally achieved by applying technologies and strategies to improve combustion and/or control direct $PM_{2.5}$ emissions from sources. These and other mitigation approaches could be utilized to achieve further BC reductions, both in the United States and globally.

  - BC mitigation solutions vary significantly by region, and must be adapted based on the specific needs and implementation challenges faced by individual countries. Some source categories, such as mobile diesel engines, can be controlled with similar strategies around the world. Other source categories, such as improved stoves for residential heating and cooking, will require tailored solutions designed to address local needs and challenges. Further work is needed to refine these options to identify which might be best-suited to particular locations or situations.

- Achieving further BC reductions, both domestically and globally, will require adding a specific focus on reducing *direct* $PM_{2.5}$ emissions to overarching fine particle control programs.

  - BC reductions that have occurred to date (largely in developed countries) are mainly due to control programs aimed at $PM_{2.5}$, not targeted efforts to reduce BC per se. Greater attention to BC-focused strategies has the potential to help protect the climate (via the BC reductions achieved through direct $PM_{2.5}$ controls) while ensuring continued improvements in public health (via control of direct $PM_{2.5}$ in highly populated areas). Even if such controls are more costly than controls on secondary PM precursors, the combined public health and climate benefits may justify the expense.

- The options identified in this report for reducing BC emissions are consistent with control opportunities emphasized in other recent assessments. These represent important mitigation opportunities for key world regions, including the United States.

  - **United States:** Based on current inventories and control technologies, mobile diesel engines (on-road and nonroad, commercial marine, locomotives) represent the largest potential area of BC mitigation in the United States. Forthcoming controls on new mobile diesel engines are expected to reduce these emissions by 86% by 2030. Diesel retrofit programs for in-use mobile sources are a valuable complement to new engine standards for reducing emissions. Other source categories, including emissions from stationary sources (ICI boilers, stationary diesel engines, uncontrolled coal-fired EGUs) and residential wood combustion (hydronic heaters and woodstoves), also offer potential opportunities, but on a smaller scale due to fewer remaining emissions in these categories, or limits on control strategies that are cost-effective or easy to implement.

  - **Globally:** The suite of options for reducing BC emissions globally is broader than those used in the United States, and will vary by region. Key BC emissions reduction opportunities globally include residential cookstoves in all regions; brick kilns and coke ovens in Asia; and mobile diesels in all regions. While a variety of other opportunities may exist in individual countries or regions, these sectors account for a large portion of BC emissions, and studies have clearly indicated the benefits of mitigating these sources.

  - **Sensitive Regions:** To address impacts in the Arctic, other assessments have identified the transportation sector (land-based diesel engines and Arctic shipping); residential heating (wood-fired stoves and boilers); and forest, grassland and agricultural burning as primary mitigation opportunities. In the Himalayas, studies have focused on residential cooking; industrial sources (especially coal-fired brick kilns); and transportation, primarily on-road and off-road diesel engines.

- A variety of other options may also be suitable and cost-effective for reducing BC emissions, but these can only be identified with a tailored assessment that accounts for individual countries' resources and needs.

  - Some potential sectors of interest include agricultural burning, oil and gas flaring, and stationary diesel engines in the Arctic far north.

- Key remaining uncertainties include:

  - Atmospheric processes affecting BC concentrations (e.g., transport and deposition)
  - Aerosol-cloud interactions (e.g., radiative and precipitation effects)
  - Climate effects of aerosol mixing state
  - Emissions of BC and co-emitted pollutants from specific regions, sources
  - Warming effect of non-BC aerosols in Arctic
  - Impacts of BC on snow and ice albedo
  - BrC climate impacts
  - Shape and magnitude of PM health impact function
  - Differential toxicity of PM components and mixtures
  - Impacts of BC on ecosystems and crops (dimming)

- Important policy-relevant research needs include:

  - Continued investigation of basic microphysical and atmospheric processes affecting BC and other aerosol species to support the development of improved estimates of radiative impacts, particularly indirect effects.
  - Improving global, regional, and domestic BC 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 climate impacts of brown carbon (BrC).
  - Research on the impact of aerosols in snow- and ice-covered regions such as the Arctic.
  - Standardized definitions and improved instrumentation and measurement techniques for light-absorbing PM, coupled with expanded observations.
  - Continued investigation of the differential toxicity of PM components and mixtures and the shape and magnitude of the PM health impact function.
  - More detailed analysis and comparison of the costs and benefits of mitigating BC from specific types of sources in specific locations.
  - Refinement of policy-driven metrics relevant for BC and other short-lived climate forcers.
  - Analysis of key uncertainties.

12.2 Introduction

Based on currently available evidence, mitigation of BC offers a clear opportunity: continued reductions in BC emissions can provide significant near-term benefits for climate, public health, and the environment. The existing literature indicates that 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 and environmental 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 and other mitigation approaches could be utilized to achieve further BC reductions. In the United States and Europe, significant reductions in BC emissions are already expected to occur over the coming decades as existing regulations such as those on diesel emissions are implemented. Some of these same approaches could help reduce emissions in developing countries, although the source mix is significantly different and additional approaches are needed to address developing countries' needs.

As discussed in Chapter 6, studies looking at global or regional BC emissions reduction scenarios have clearly demonstrated large potential climate and health benefits. The framework discussed in Chapter 7 suggests that policymakers have to consider a number of factors in designing control programs to achieve those benefits. There is no one-size-fits-all mitigation prescription; rather, there is a portfolio of mitigation opportunities, all of which have the potential to provide climate, health and environmental benefits if applied in appropriate locations. Chapters 8-11 illustrate the range of sources, technologies, and control options that policymakers can consider for purposes of BC mitigation.

This chapter brings all of this information together to identify paths forward on BC, based on the best available information about remaining emissions and cost-effective mitigation opportunities. The options identified in this report for reducing BC emissions are consistent with control opportunities emphasized in other assessments, including UNEP/WMO (2011a), LRTAP (2010), USAID (2010a), the Arctic Council (2011), Quinn et al. (2011), and U.S. EPA (2011b). BC mitigation solutions vary significantly by region, and must be adapted based on the specific needs and implementation challenges faced by individual countries. Some source categories, such as improved stoves for residential heating and cooking, will require tailored solutions designed to address local needs and challenges. Further work is needed to refine these options to identify which might be best-suited to particular locations or situations. The recently completed UNEP synthesis report, *Actions for Controlling Short-Lived Climate Forcers*, that identifies regionally appropriate mitigation options for BC can provide important information toward this goal (see UNEP, 2011).

12.3 Controlling Black Carbon as Part of Broader $PM_{2.5}$ Mitigation Program

As discussed in Chapter 7, policymakers designing BC mitigation strategies have a number of separate goals (health, climate, environment) and a variety of alternative approaches to consider. Many of the BC emissions reductions currently being achieved are the result of broader programs or strategies aimed at reducing overall $PM_{2.5}$, primarily for purposes of protecting public health. This raises a very important question, namely: are existing $PM_{2.5}$ control programs and strategies (widely implemented) sufficient to control BC to protect climate, or are alternative strategies and approaches needed to guarantee BC reductions? Asked another way: what might policymakers do differently with regard to BC controls in order to jointly maximize climate and public health benefits? The degree of overlap between programs and measures driven by public health protection vs. climate goals is of critical importance.

In the United States and other developed countries, $PM_{2.5}$ control programs have focused very heavily on reductions in emissions of NO_x and SO_x, which contribute to secondary $PM_{2.5}$ formation. There are several reasons for this: first, NO_x and SO_x emissions from utilities and other major industrial sources contribute the bulk of $PM_{2.5}$ mass in most areas; second, they are the pollutants principally responsible for regional transport of $PM_{2.5}$, and therefore have been the focus of national rules; third, reductions in NO_x and SO_x are highly cost-effective. Widespread reductions in NO_x and SO_x emissions have been credited with substantial decreases in ambient $PM_{2.5}$ concentrations, reductions in acid deposition, and dramatic improvements in public health and the environment (U.S. EPA, 2011c, e). In both the United States and Europe, programs to reduce emissions of NO_x and SO_x continue to serve as cornerstones of $PM_{2.5}$ air quality improvement programs. However, these programs have relatively little impact on BC emissions. This is due to a difference in major emitting sources, and also the types of controls applied, which may not be highly effective in reducing direct particle emissions (including BC and OC).

The other major component of existing $PM_{2.5}$ control programs is the effort to reduce emissions from mobile sources, including diesel vehicles (on-road diesels, nonroad diesels, commercial marine, and locomotives) but also gasoline vehicles. These programs, which have been adopted in both the United States and Europe, have led to stringent controls on NO_x emissions, contributing to the overall decline in NO_x and SO_x emissions discussed above. In addition, controls on mobile sources (particularly diesels) have been highly effective in controlling BC emissions. Mobile source control programs are principally responsible for the dramatic reductions in BC emissions projected to occur in the United States and Europe by 2030. These reductions have been tied to substantial public health benefits (U.S. EPA, 2007). In addition, controls on certain industrial processes have contributed substantially to reductions in direct $PM_{2.5}$ emissions (U.S. EPA, 2004b).

As $PM_{2.5}$ control programs in developed countries have matured, the mix of sources controlled has gradually changed, and can be expected to change further in coming decades. In the United States, state and local areas have already been turning to controls on direct $PM_{2.5}$ emissions as a means of dealing with persistent nonattainment problems. Areas such as Detroit, Michigan; St. Louis, Illinois; Pittsburgh, Pennsylvania (Liberty-Clairton); and Birmingham, Alabama have either included or have been planning to include emissions controls of direct $PM_{2.5}$ in their State Implementation Plans (SIPs) to demonstrate future compliance of the PM NAAQS.[1,2,3,4] These control strategies include emissions controls on industrial sources such as steel mills, coke oven batteries, and foundries, and will reduce BC, OC and toxic metals. Because many of these sources are also located in urban areas where significant numbers of people reside, substantial public health benefits are also expected from these emissions controls (Fann et al., 2011). As discussed in Chapter 6, EPA's health-based $ per ton benefits estimates indicate that these reductions typically can provide higher levels of health benefits per ton of emissions reduction as compared to reductions from other sources.

For developed countries, then, it appears that control programs are gradually evolving in a direction toward controls that achieve large BC reductions, and that these countries are on track to have very stringent controls on, and significant reductions from, many source categories. Some of these reductions will not be fully in place for several decades, which creates opportunities for incentive programs to accelerate the reductions. Thus, there may be hybrid strategies the United States and other countries can adopt that will both increase public health benefits (via control of direct $PM_{2.5}$ in highly populated areas) and protect the climate (via the BC reductions achieved through direct $PM_{2.5}$ controls). Even if such controls are more costly than controls on secondary PM precursors, the extra public health and climate benefits may justify the expense.

In developing countries, the $PM_{2.5}$ challenges are somewhat different. Overall ambient $PM_{2.5}$ and BC concentrations are generally higher than in developed countries, and $PM_{2.5}$ control programs are less advanced. For example, some megacities in Asia have annual average $PM_{2.5}$ concentrations above 100 $\mu g/m^3$, more than six times the air quality standard in the United States (ESMAP, 2004). Moreover, the mix of sources is quite different, requiring different mitigation strategies. For example, mobile sources often have relatively little emissions control currently. Furthermore, there are many more small BC sources like kilns and cookstoves that could be replaced entirely with more advanced technologies. Coal use is very high, and significant air quality improvements could be obtained through switching to cleaner fuels. These changes have already occurred in developed countries. In general, the public health opportunities of $PM_{2.5}$ reductions in developing countries are large, and controls on BC and other directly emitted $PM_{2.5}$ could provide significant public health improvements. This presents a real win-win opportunity, since BC reductions will provide benefits to climate as well as to public health.

It is clear, therefore, that emphasizing BC reductions within broader $PM_{2.5}$ control programs might be a beneficial strategy for health and climate in both developed and developing countries. However, because of the significant differences in the sources, emissions, technologies, costs, and implementation challenges for BC reductions in different countries and regions, the top-tier mitigation options—those that can reliably achieve the largest climate, health and environmental benefits at low cost—will vary among regions. In some cases, options can clearly be discounted in some regions due to low effectiveness or high costs. In other cases, barriers to implementation or simply insufficient information about potential effectiveness can render certain choices less attractive. When all these factors are considered, the number of clearly beneficial mitigation options can be narrowed down significantly. The remaining mitigation options present the clearest near-term opportunities to achieve benefits for climate and public health in specific regions.

12.4 Key Black Carbon Mitigation Opportunities

The control strategies described in detail in Chapters 8-11 reflect the range of existing mitigation options for BC emissions. Based on the BC emissions profile of key world regions, the technologies available (including their cost and demonstrated effectiveness), and the implementation challenges discussed in these earlier chapters, a wide variety of $PM_{2.5}$ mitigation measures appear likely to provide substantial benefits to human health and

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[1] Michigan Department of Environmental Quality Air Quality Division Draft State Implementation Plan Submittal for Fine Particulate Matter. Available online at: *http://michigan.gov/documents/deq/deq-aqd-air-aqe-sip-pm25-1-14-08_223446_7.pdf.*

[2] United States Steel Corporation Granite City Works and Illinois EPA Memorandum of Understanding.

[3] Revision to the Allegheny County Portion of the Pennsylvania State Implementation Plan; Attainment Demonstration for the Liberty-Clariton $PM_{2.5}$ Nonattainment Area. Available online at: *http://www.achd.net/airqual/Liberty-Clairton_PM2.5_SIP-Apr2011.pdf.*

[4] Alabama State Implementation Plan for $PM_{2.5}$ for Birmingham, Alabama.

the environment. A smaller subset of measures—those that affect BC-rich sources and/or emissions in sensitive regions—are the most likely to simultaneously provide climate benefits. Measures that promise both climate and health benefits can be considered "win-win" opportunities, and the lower the uncertainty around those measures (with regard to effectiveness, cost, etc.), the greater the confidence with which policymakers can move forward. Low-cost measures for which there is relatively high certainty of climate as well as health and environmental benefits can be considered "top tier" opportunities. This section discusses potential top-tier mitigation options for key world regions, including the United States. A variety of other options may also be suitable and cost-effective for reducing BC emissions, so this discussion should not be seen as limiting or discouraging further exploration or adoption of other measures. Rather, it is intended to help clarify the "low-hanging fruit" of BC mitigation opportunities. Importantly, these opportunities are described in general terms, aimed at highlighting sectors and regions where mitigation holds particular promise. Individual countries' strategies are likely to look different based on their application of the factors outlined in the mitigation framework in Chapter 7. Each country will have a variety of sector- and technology-specific opportunities that can only be identified with a tailored assessment.

12.4.1 U.S. Black Carbon Mitigation Opportunities

Based on current inventories and control technologies, mobile diesel engines represent the largest potential area of BC mitigation in the United States. Other source categories, stationary sources and residential wood combustion, also offer potential emissions reduction opportunities, but on a smaller scale due to smaller remaining emissions in these categories, or limits on control strategies that are cost effective or easy to implement. Due to U.S. air quality regulations, a number of controls on all of these source categories are already planned over the next decade or two. However, further reductions may be possible, for example with more dedicated funding for diesel retrofits on existing engines. Chapter 6 indicated that BC reductions in the U.S. have the potential to provide large public health benefits domestically, and climate benefits in terms of global average radiative forcing and impacts in the Arctic.

- **Mobile engines: On-road and Nonroad, Commercial Marine, Locomotives:** The United States has already enacted stringent standards for *new mobile source diesel engines* that are projected to reduce BC emissions by 86% between 2005 and 2030 as the existing diesel fleet is replaced by these new engines. Though the standards do not dictate the use of a specific technology, the mobile source reductions will be achieved mostly from DPFs on new diesel engines, in conjunction with ultra low sulfur diesel fuel. EPA has estimated the cost of controlling $PM_{2.5}$ from diesel engines via new engine requirements at about $14,000/ton (2010$) (including the cost of ULSD fuel). The public health benefits of these reductions have been estimated at $290 billion annually in 2030(2010$).

  For the existing legacy *in-use mobile diesel fleet* in the United States, programs such as EPA's National Clean Diesel Campaign and the SmartWay Transport Partnership Program can help achieve additional emissions reductions through retrofits on an estimated 11 million engines. DPFs can reduce $PM_{2.5}$ emissions by up to 99%, at a cost of $8,000 to $15,000 for passive DPFs, and $20,000 to $50,000 for active DPF systems (2010$). However, not all engines are good candidates for DPFs because of old age or poor maintenance.

- **Stationary sources: ICI Boilers, Stationary Diesel Engines, Uncontrolled Coal-Fired EGUs:** Stationary source BC emissions in the United States have declined dramatically in the last century. Remaining emissions constitute 8% of the inventory and come primarily from coal combustion (utilities, industrial/commercial boilers, other industrial processes) and stationary diesel engines. Available control technologies and strategies include direct $PM_{2.5}$ reduction technologies such as fabric filters (also known as baghouses), electrostatic precipitators, and DPFs. These strategies range in cost from as little as $48 per ton $PM_{2.5}$ to over $24,000 per ton $PM_{2.5}$ (2010$), depending on the source category. They may also involve millions of dollars in initial capital costs. As mentioned above, controls on direct $PM_{2.5}$ from industrial sources are being considered by a number of states as a strategy for addressing persistent nonattainment problems in key areas.

- **Residential Wood Combustion: Hydronic Heaters and Wood Stoves:** *Residential wood combustion* (RWC) represents a small portion (3%) of the U.S. BC inventory, but mitigation opportunities are available. In part because seasonal use of these sources is concentrated in northern areas, reducing emissions may reduce deposition on snow and ice. EPA has already established emissions standards for new residential wood stoves and is working to revise and expand these standards to include all residential wood heaters, including hydronic heaters and furnaces, as well as stoves. Mitigation strategies for RWC sources include providing alternatives to wood, replacing inefficient units (wood stoves, hydronic heaters) with newer, cleaner units through voluntary or subsidized change-out programs, or retrofitting existing units to enable use of alternative fuels such as natural gas (fireplaces). New EPA-certified wood stoves have a cost of about \$3,600 per ton $PM_{2.5}$, while gas fireplace inserts average \$1,800 per ton $PM_{2.5}$ (2010\$).

12.4.2 Global Black Carbon Mitigation Opportunities

Outside of the United States, BC emissions are increasing in some locations as a result of rapid economic development coupled with the lack of serious $PM_{2.5}$ mitigation measures. The magnitude of these emissions translates into very substantial impacts to climate, public health, and the environment, particularly in Asia. Global BC mitigation opportunities can be evaluated in two ways: first, in terms of major source types within key emitting regions (i.e., the largest reduction opportunities on a total emissions basis), and second, by focusing on emissions affecting sensitive regions (i.e., the best targeted opportunities in terms of avoiding critical impacts). It is important to note that effective BC mitigation at the global scale will depend on application of *a variety of strategies* rather than widespread adoption of a single strategy.

In terms of major emitting sectors/regions, there are significant differences in emissions and mitigation potential based on the level of development of the emitting country(s). Opportunities in developed regions such as Europe and Russia are in many ways similar to those in the United States: mobile diesel engines are very important, as are emissions from boilers, stationary diesel engines, and residential wood combustion (though the order of importance varies by country). In general, European countries have emissions patterns and control programs that are similar to the United States, though the timing of planned emissions reductions may vary. Russia is distinctive in that it may also have significant mitigation potential in the area of agricultural burning: emissions from this source category in Russia are thought to be large and potentially very significant for the Arctic (CATF, 2009a). However, significant uncertainties hinder mitigation actions in this area at present.

In the developing world, the main BC-emitting sectors internationally are residential solid fuel combustion (cookstoves), industry, and transportation. Mitigation measures to reduce emissions from each of these sectors are presently available and feasible on wide scales. Opportunities in Asia range from mobile diesel engines to industrial sources (especially brick kilns) to residential cooking on traditional stoves. This region is of particular importance for BC mitigation, both because it accounts for approximately 40% of global BC emissions and because of the effect of these on the Himalayas and the Arctic. In Africa and Latin America, emissions come largely from open biomass burning and residential cookstoves. While large, these emissions may be challenging to mitigate, particularly those from open biomass burning (see Chapter 11).

In considering which source categories and locations offer the largest opportunities, it is important to remember that there are significant differences in control opportunities between developed and developing countries. Options that are feasible in the United States and Europe may not be appropriate for application in developing countries due to cost constraints or other barriers. For example, advanced industrial emissions control systems may not be applicable to source types in developing countries, or the costs of installing such systems may be prohibitive. It is also important to note that the opportunities for BC reductions may change as countries develop. For example, currently 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. Finally, it is important to remember that cost, emissions and other relevant data are not as widely available internationally, which makes assessment of top-tier mitigation options outside the United States and Europe more difficult.

Several groups have developed preliminary analyses of global strategies to reduce BC emissions (Rypdal et al., 2005; Kandlikar et al., 2009; Cofala et al., 2007; Baron et al., 2009; UNEP/WMO, 2011; Shindell, 2012). These assessments have generally found the largest BC reductions at lower costs in Asia, which has high emissions from residential cooking and heating as well as poorly controlled transportation and small industrial sources. Some of these assessments also included measures to reduce biomass burning in Africa and South America. BC strategies ranged from improved combustion to add-on particle controls. The actual benefits to global or regional climate resulting from such strategies can, however, vary significantly with the nature of the co-pollutants emitted from different sources, their location, and the nature of the controls.

Based on currently available information, key emissions reduction opportunities internationally may include:

- **Residential Cookstoves in All Regions:** Replacing traditional cookstoves, which account for 25% of total global BC emissions, with more advanced technologies is one of the most promising opportunities internationally, in large part because of the enormous public health benefits that could result from cooking with cleaner fuels and stoves. This strategy may provide particularly large benefits in Asia (including China and India), given the proximity of these sources to both large populations and climate-sensitive regions (the Himalayas). Exposures to cookstove emissions currently lead to an estimated 2 million deaths each year. In Asia, these emissions are also linked to the devastating impacts of ABCs, including the disruption of the Indian monsoon. While there is still uncertainty about BC emissions emanating from different stove and fuel types, cookstoves also emit CO_2 and CH_4, contributing to climate effects, and a variety of aerosols and gases (SO_2, NO_x, CO, OC) which contribute to adverse health and environmental effects. Reducing cookstove emissions helps to alleviate all of these impacts. The magnitude of the health benefits alone justifies the cost of cookstove replacement on large scales.

  While the world-wide stove market is approximately 500-800 million households, current programs likely replace only approximately 5-10 million improved stoves per year. Significant expansion of such programs utilizing locally appropriate advanced stove technologies and fuels could achieve large-scale climate, health and environmental benefits. The costs range from $8-$100+ per stove. Improved cookstove technologies all face important supply, cost, performance, usability, marketability and other barriers that have impeded progress in the past. However, a number of factors point to much greater potential to achieve large-scale success in this sector today. Along with many corporations and world governments, the U.S. government has made a substantial diplomatic, research, and financing commitment to the Global Alliance for Clean Cookstoves, which has an interim goal of having 100 million homes adopt clean and efficient cooking solutions by 2020.[5] Continued investment in the Alliance will help achieve this goal with great public health benefits.

- **Brick Kilns and Coke Ovens in Asia:** Emissions from stationary sources represent 20% of the global inventory, and several recent studies have identified brick kilns and coke ovens as two large contributing source categories for which new, clean technologies are available to reduce BC emissions at reasonable cost. Brick kilns can be replaced by cleaner vertical shaft kilns or Hoffman kilns. Coke ovens can be designed with modern recovery ovens or the use of end-of-pipe control technologies. Since the ratio of BC to other PM constituents co-emitted by these sources (such as OC, SO_2 and NO_x) is high, mitigation measures for brick kilns and coke ovens are likely to result in both climate and health benefits. Furthermore, the health benefits of emissions reductions in these sectors—both for workers and for exposed local populations—may offset mitigation costs.

- **Mobile Diesel Engines in All Regions:** Mobile sources account for approximately 18% of the global BC inventory, and both new engine standards and retrofits of existing engines/vehicles could help reduce these emissions. In the United States and Europe, mitigation programs are already underway. Many other countries have begun the phase-in of emissions standards and ULSD fuel, which is a prerequisite for the proper functioning of DPFs. However, these requirements lag behind in some regions, as do on-the-ground deployment of DPFs and ULSD. As a result, there remains significant opportunity internationally to accelerate the deployment of clean engines and fuels. Similar to stationary sources, these emissions are often co-located with large populations, making them a prime target for mitigation programs motivated by both climate and public health concerns. Mobile source mitigation opportunities are likely to vary among countries. In some regions, on-road sources such as diesel trucks may be the dominant mobile BC emissions source, while in other regions, non-road sources such as locomotives may be particularly important. In general, other countries could achieve further BC reductions by implementing standards for nonroad diesels, locomotive, and commercial marine vessels (categories 1 and 2).

In terms of mitigation measures aimed at protecting particularly sensitive regions, most of the attention

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[5] See *http://cleancookstoves.org*.

to date has been directed at two main world regions: the Arctic and the Himalayas. Available studies suggest that there are considerable differences between the prime mitigation opportunities for these two regions.

In the **Arctic**, recent work by the Arctic Monitoring and Assessment Program (AMAP) has indicated that mitigation measures *near or within* the Arctic will provide the largest benefits for the region. This is because direct forcing and snow/ice albedo forcing in the Arctic per unit of emissions were shown to be highest for emissions originating near to or within the Arctic region, with the Nordic countries (i.e. highest latitudes) having the highest forcings per unit of emissions, followed by Russia, Canada, and the United States (Quinn et al., 2011, p. 95). Importantly, studies have also found that reductions in OC emissions will provide benefits in the Arctic too, since OC emissions appear to exert positive radiative forcing over snow- and ice-covered surfaces. Similarly, sulfate aerosol, which normally exerts a cooling influence, appears to have a much weaker effect over snow and ice (Quinn et al., 2011, p. 101-2). This means that a wide array of measures aimed at reducing PM in or near the Arctic will provide benefits to the Arctic region. In accord with the modeling results from AMAP (Quinn et al., 2011), the Arctic Council Task Force on SLCF has specifically recommended that Arctic Council countries—including the United States, Canada, Russia, Sweden, Norway, Denmark, and Iceland—pursue additional emissions reductions from the following sources (Arctic Council, 2011):

- **Transportation, including land-based diesel engines and Arctic shipping:** While noting that most Arctic nations have already adopted and begun implementing regulations for new diesel engines, the Task Force stressed the benefits of retrofitting or retiring older vehicles and equipment, accelerating the phase-in of ultra low sulfur diesel fuels, and expanding control requirements to additional source categories, such as marine vessels and locomotives, as appropriate. The Task Force noted that although BC emissions from marine shipping are currently small, they occur in direct proximity to snow and ice and are expected to increase in future due to declining summer sea ice in the Arctic. As a result, the Task Force recommended a number of measures aimed specifically at limiting BC emissions from Arctic shipping.

- **Residential heating:** Emissions inventories for Arctic countries indicate that wood-fired stoves and boilers are a major source of BC in the Arctic. The Task Force recommended Arctic countries consider implementing stringent BC emissions standards for stoves and boilers, adopt point-of-manufacture performance standards, and work to develop new technologies and incentive programs to replace old stoves and boilers.

- **Forest, grassland and agricultural burning:** The Task Force identified a variety of techniques that could be utilized to reduce emissions from open biomass burning, a leading source of BC in the Arctic. Available measures include technical assistance, demonstration projects, and financing to encourage no-burn methods, such as conservation tillage or soil incorporation; information campaigns to prevent accidental wildfires and unnecessary use of fire as a land management technique; and expansion of resources for fire monitoring, fire management, and fire response.

Reductions outside of the Arctic will also provide benefits to the Arctic region. The AMAP (Quinn et al., 2011) modeling indicates that BC emissions from outside the Arctic Council nations are contributing (in aggregate) the most to total radiative forcing (warming) in the region (see Chapter 2). This suggests that reductions in BC emissions from regions other than the Arctic itself (60°-90°N) will be effective in reducing Arctic warming; thus, BC reductions anywhere—including the measures listed earlier in this chapter—can be expected to provide benefits specifically to the Arctic region.

For the **Himalayas**, most of the attention has focused on the three source categories mentioned above as important for mitigation globally: residential cookstoves, brick kilns, and mobile diesel engines. USAID (2010a) and U.S. EPA (201b) have both identified a number of measures that could provide substantial climate, health and environmental benefits in this region. Specifically, these opportunities include:

- **Residential cooking:** Cookstoves are the largest source of BC emissions in Asia, and therefore offer the largest overall emissions reduction opportunity in the region. Furthermore, according to USAID, "household fuel and stove interventions … appear to consistently achieve the highest reduction in black carbon emissions per unit cost. This finding holds true for all stove and fuel interventions examined" (USAID, 2010a, p. 3). The study found that these interventions "yield the highest net benefits per unit intervention cost if health benefits are included" (p. 4) and that the GHG co-benefits are also large. All of these factors indicate that the residential sector is a prime opportunity for BC mitigation.

- **Industrial sources, particularly coal-fired brick kilns:** Small-scale industrial sources have been identified as major emitters of BC, with particular attention given to mitigation potential from traditional brick kilns. Adopting more efficient technologies, using cleaner fuels, and improving operating practices may help to reduce BC emissions from this sector. In addition, product substitution may allow for greater efficiency and cleaner production (Heierli and Maithel, 2008). The largest BC emissions reductions can be achieved through substitution of Vertical Shaft Brick Kilns (VSBKs), Hoffmann kilns, or tunnel kilns for traditional Bull's Trench Kilns (BTKs) (U.S. EPA, 2011b). However, even traditional kilns can produce lower emissions with proper fuels and operating procedures.

- **Transportation, primarily on-road and off-road diesel engines:** In Asia, the main focus in the mobile sector is land-based vehicles, especially long-haul trucks and buses (U.S. EPA, 2011b). USAID (2010a) identified a number of measures aimed at mobile sources as falling in the "next tier" of BC abatement measures after residential cookstove interventions. Specifically, the study pointed to the need for more stringent emissions standards, greater use of DPFs for new and existing (retrofitted) diesel vehicles, and a shift toward cleaner fuels such as compressed natural gas (CNG) or liquefied petroleum gas (LPG). However, the study also noted that ULSD fuel is not available in Asia except in a few major metropolitan areas and called for efforts to promote desulfurization in the region (USAID, 2010a). U.S. EPA (2011b) noted that substantial efficiency improvements and reductions in BC emissions could be achieved through improvements in fleet logistics, such as eliminating "deadheading" (i.e. trips with no load).

12.4.3 Other Mitigation Options

The source categories listed above for the United States and other world regions offer the greatest certainty of BC reductions that are likely to be beneficial for climate. Mitigation measures aimed at reducing BC emissions in sectors beyond those listed above require further investigation. The effectiveness of controls in other BC emissions source categories is far less certain, and the costs and implementation barriers potentially higher. Some may be highly effective if designed and implemented correctly.

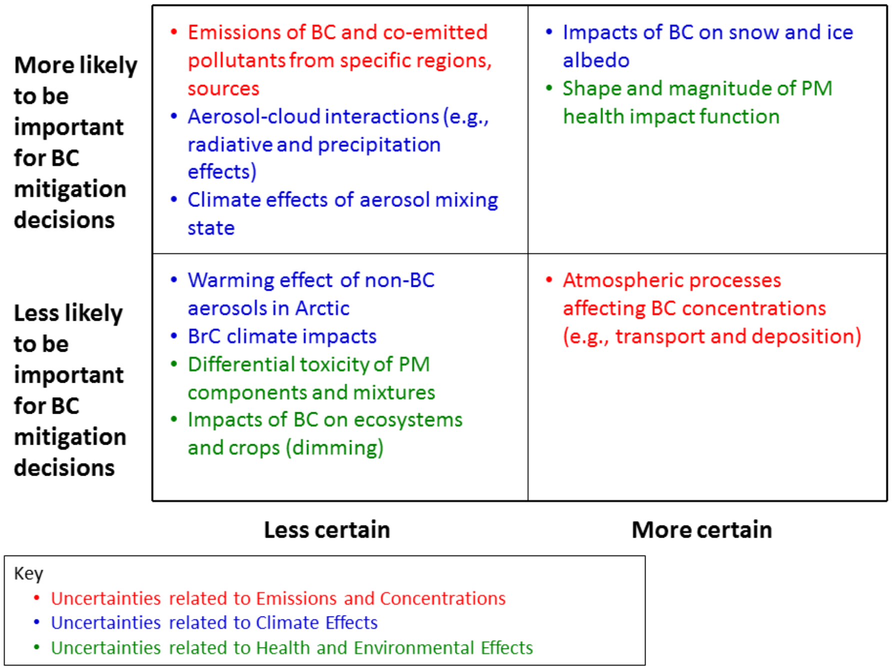
Several studies have suggested that controlling **agricultural burning** could provide important climate benefits (Warneke et al., 2010; CATF, 2009a). Since open biomass burning is the largest source of BC emissions globally, accounting for over 35% of the inventory, actions to reduce emissions from this sector have the potential to make a significant difference. However, it is also important to recognize that emissions of OC are seven times higher than BC emissions from this sector, and that a large portion of emissions come from wildfires. In the United States, for example, wildfires contribute about 68% of total BC emissions from open biomass burning. Presently, data on the percent of land area affected by different types of burning are very limited, and little is known about how specific measures would impact climate, both globally and regionally. Appropriate mitigation measures are highly dependent on a number of variables, including timing and location of burning, resource management objectives, vegetation type, and available resources. The costs of mitigation measures are uncertain and potentially high, as they depend on various site-specific factors. Thus, despite the significant contribution of open biomass burning to BC emissions worldwide, further investigation into specific options is needed to identify feasible mitigation opportunities in appropriate locations. 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 and appropriate use of prescribed fire.

Another sector that has drawn attention, especially with regard to potential impacts in the Arctic, is **oil and gas flaring**. Despite higher uncertainty about the magnitude and impact of BC emissions from flaring, the Arctic Council Task Force on SLCF recommended that this source be evaluated carefully due to the expansion of oil and gas extraction activities in the Arctic region (Arctic Council, 2011).

Other categories of interest include **stationary diesel engines** in the Arctic far North. These sources are located in proximity to snow and ice, are often very expensive to operate due to fuel transport requirements, and may adversely impact the health of indigenous communities.

12.5 Key Policy-Relevant Scientific Uncertainties

Many uncertainties are associated with understanding the effects of BC emissions on air quality, climate, health, and the environment. Some of these may be important and large enough to influence BC mitigation decisions – including the decision about whether to invest in BC mitigation at all as well as the selection of specific mitigation



	Less certain	More certain
More likely to be important for BC mitigation decisions	• Emissions of BC and co-emitted pollutants from specific regions, sources • Aerosol-cloud interactions (e.g., radiative and precipitation effects) • Climate effects of aerosol mixing state	• Impacts of BC on snow and ice albedo • Shape and magnitude of PM health impact function
Less likely to be important for BC mitigation decisions	• Warming effect of non-BC aerosols in Arctic • BrC climate impacts • Differential toxicity of PM components and mixtures • Impacts of BC on ecosystems and crops (dimming)	• Atmospheric processes affecting BC concentrations (e.g., transport and deposition)

Key
- Uncertainties related to Emissions and Concentrations
- Uncertainties related to Climate Effects
- Uncertainties related to Health and Environmental Effects

Figure 12-1. Key Policy-Relevant Scientific Uncertainties Related to BC.
(Source: U.S. EPA)

strategies. Other uncertainties may be important scientific questions, but would be unlikely to influence BC mitigation decisions.

Figure 12-1 roughly groups key uncertainties identified in this report by level of uncertainty and likelihood to affect BC mitigation decisions. Each of these uncertainties pertains to an important aspect of BC – emissions and concentrations, climate effects, or health and environmental effects. Uncertainties categorized as "less certain" and "more likely to be important for BC mitigation decisions" are those for which new information can change our understanding of the direction and effectiveness of BC mitigation in terms of climate, health, and environmental benefits. These are the uncertainties that are the highest priorities for future research. For example, understanding the magnitude of BC emissions from specific regions and sectors, along with co-emitted pollutants (e.g. OC and BrC), is critical to estimating the net warming or cooling effect of that source. New information showing a larger fraction of warming agents among the emitted mixture from a particular source may make mitigation more attractive from a climate perspective; similarly, a smaller relative contribution of warming agents from a source could make it a less attractive target for mitigation. Impacts of BC and other aerosols on clouds, both in terms of radiative forcing and impacts on precipitation, as well as the climate effects of different aerosol mixing states (e.g., possible enhanced absorption by BC when coated by sulfate), are also highly uncertain. Additional research in these areas has the potential to shift the current assessment of the climate benefits of BC mitigation in important ways.

Uncertainties categorized as "less certain" and "less likely to be important for BC mitigation decisions" are those that are not well-understood but for which new information is unlikely to fundamentally change our understanding of BC mitigation impacts. In some cases, the outcome of new information may be to change our understanding of the magnitude of BC mitigation impacts of climate, health, and the environment, but not the direction of these benefits. For example, new information may improve understanding of the effects of non-BC aerosols (e.g., OC, BrC, sulfate) in the Arctic, both in terms of impacts on snow and ice albedo, as well as the pace of melting. Since all particles are generally darker than snow and ice, additional information could change the magnitude of estimated BC mitigation benefits in the Arctic. Similarly, if a broader range of aerosols are confirmed to contribute to warming in the Arctic, it would strengthen evidence that mitigating BC and related co-emitted pollutants could provide significant benefits to this region; however, such evidence would not necessarily change the highest priority mitigation measures aimed at BC-rich sources. Although differential toxicity of $PM_{2.5}$ components and mixtures is critical for understanding the magnitude of the health benefits achieved by controlling $PM_{2.5}$, the current literature points to health effects of BC that are similar to those of other $PM_{2.5}$ components. Since it is very likely that reductions in BC will be found to be health beneficial, new information on differential toxicity is less likely to affect BC mitigation decisions.

Uncertainties categorized as "more certain" are still significant, even if there is relatively more evidence on these topics as compared to those categorized as "less certain." For example, the impacts of BC on snow and ice are characterized by a small but generally consistent body of literature. There is a strong need for additional research in this area, even though currently available studies already suggest BC mitigation efforts will provide benefits in snow/ice covered areas. In addition, there remains some uncertainty regarding the full extent of $PM_{2.5}$ health effects, especially in the developing world. The health effects of $PM_{2.5}$ are generally well characterized as a result of numerous major long-term epidemiological studies conducted in developed countries and a number of short-term studies conducted in locations around the world. Thus the health effects of $PM_{2.5}$ are generally well-known and provide a strong basis for controlling emissions of PM precursors and components, including BC. However, the magnitude and shape of the concentration-response function in developing countries where exposure is significantly higher remains under investigation.

12.6 High Priority Research Needs for Black Carbon

There are a number of high priority research topics that could help advance efforts to control emissions from BC sources, clarify the impacts and benefits of BC mitigation, and reduce key remaining uncertainties, particularly those summarized in section 12.5 and in Figure 12-1. Some of these research topics pertain to key scientific aspects of BC, including composition, morphology, fate, and transport. Other important areas for further research relate directly to information and tools necessary for policymaking about BC. Research on these topics would allow for more informed policy decisions regarding mitigation of emissions of BC and related species. Based on the information reviewed for this report, EPA concludes that priority should be given to research in the following areas:

1. **Continued investigation of basic microphysical and atmospheric processes affecting BC and other aerosol species to support the development of improved estimates of radiative impacts, particularly indirect effects.**

Some of the basic microphysical and atmospheric processes that BC and other aerosol species undergo are not very well understood. This includes the mixing of BC with other aerosol species, the atmospheric aging of BC and how aging affects BC's climatic and health impacts, and interactions with cloud droplets and the hydrologic cycle in general. Incomplete understanding of the *climate effects of aerosol mixing state* and of *aerosol-cloud interactions* limits the scientific community's ability to model BC in the atmosphere and estimate its impacts. Few global models are now able to resolve the cloud microphysics which are of importance in determining climatic effects. Direct radiative forcing from BC is clearly positive and results in warming. However, early results indicate BC emissions lead to a net positive cloud absorption effect but both positive (warming) and negative (cooling) semi-direct and negative indirect effects. The net result may be negative enough to offset some of the warming due to the direct effects of BC. The net effect of BC on cloud absorption, semi-direct and indirect cloud feedbacks depends on many factors, among them aerosol hygroscopicity, absorptivity, and number concentration relative to background particles. Improvements in our understanding of these basic properties through controlled experiments and atmospheric observations could improve climate models, and could also inform ongoing efforts to investigate the health effects of PM constituents, including BC.

2. **Improving global, regional, and domestic BC 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.**

Given the diversity and ubiquity of sources of BC, accurately measuring and tracking *emissions of BC and its co-pollutants from specific source categories* is a very difficult undertaking. Emissions inventories in the United States and other developed countries account for most source categories, but considerable uncertainty remains. More information on both emission factors and usage would be helpful. In particular, emissions from key industrial sources, flaring, residential heating, and open biomass burning remain poorly characterized. In general, mobile source emissions are among the best characterized (especially in developed countries), but improved information is still needed for some sectors, such as nonroad mobile sources (aircraft, locomotives, ocean-going vessels); newer technology on-road diesel/gasoline vehicles; high-emitting vehicles; and vehicles operating at low temperatures.

Uncertainties are larger for BC inventories in developing countries and globally. For these inventories, priorities include better characterization of emissions from residential cookstoves, in-use mobile sources, small fires, smaller industrial sources such as brick kilns, and flaring emissions. For sources such as cookstoves, improved characterization depends critically on field-based measurements of emissions from in-use sources. In addition, usage patterns need to be reviewed to ensure that appropriate "activity" levels are applied to emission factors to arrive at final emissions estimates. Finally, fuller incorporation of regional inventories into global inventories could improve country- and region- specific emissions estimates.

Quantifying and reducing the uncertainties in global, regional, and domestic emissions inventories requires collecting source-specific emissions data from lab or field-based measurements and gathering information on activity levels, operating conditions, and technological configurations. For hard-to reach areas, improvements in estimation techniques could significantly improve global and regional inventories. Systematic collection and sharing of emissions data and meta-data is important for both scientific and policy purposes.

3. **Focused investigations of the climate impacts of brown carbon (BrC).**

The role of BrC is important for determining the potential climate benefit of mitigating sources with high OC emissions such as biomass burning. BrC may also play a stronger role in the Arctic and snow- and ice-covered regions, through deposition on ice and snow. Several aspects of the BrC issue warrant research in order to better characterize the *climate impacts of BrC*. First, multi-wavelength measurements are needed to separate and characterize BrC and BC. Reporting column data by wavelength may aid model-observation comparison, as BC and BrC differ in terms of peak absorption. Second, BrC should be incorporated into climate models, and the impact of BrC on net forcing estimated independently from the impact of BC. Atmospheric observations of BrC and experimental methodologies for determining BrC emissions are also needed, in conjunction with the improvements in BrC measurements and expanded observations, particularly in ice- and snow-covered regions (described further below).

4. **Research on the impact of aerosols in snow- and ice-covered regions such as the Arctic.**

Clarifying the role of BC, BrC, and other aerosols in sensitive regions such as the Arctic and the Himalayas is critically important for proper characterization of the role of different species in climate change in these regions. Additional work on BC impacts should be supplemented with expanded research on non-BC aerosols, including sulfate, nitrate, and OC (including BrC). An expanded observational record coupled with more thorough treatment of different aerosol species in climate models (both described elsewhere in this section) could help clarify the contribution of different species to net warming or cooling in the Arctic and could provide insight into effective mitigation measures. Key topics include the *impact of BC on snow and ice albedo* and the *warming impact of non-BC aerosols in the Arctic*.

5. **Standardized definitions and improved instrumentation and measurement techniques for light-absorbing PM, coupled with expanded observations.**

In order to accurately assess the impacts of BC (and co-pollutant) emissions, it is essential to have a clear understanding of the optical properties of atmospheric aerosols and be able to trace those to emissions from specific sources. Precise and consistent definitions and measurements of BC and other carbonaceous aerosols would ensure more accurate assessment of BC emissions, climate and public health impacts, and mitigation options. Additional research is needed to improve instrumentation and measurement techniques to quantify accurately the light-absorption properties of BC, BrC, and other aerosols; to harmonize measurement and reference methods to standardize definitions of BC, BrC and other compounds; and to refine measurement techniques to collect more data on light-absorption capacity of emissions from specific sources. Additional representative source measurements to better characterize BC emissions by source category, fuel type and combustion conditions can help improve emissions inventories and reduce modeling uncertainties.

It is equally important to expand the observational record for BC, including observations of atmospheric concentrations of BC and BC deposition. Existing measurements of BC are sparse in both spatial and temporal coverage, even in countries with more advanced monitoring programs such as the United States. An expanded observational record based on standardized measurement techniques and instruments could provide important information about BC transport, vertical distribution, atmospheric interactions, and deposition. Observations of BC and BrC deposition are important for furthering understanding of the role of *deposition on snow and ice*. Such data could be used to inform climate models and verify impacts.

6. **Continued investigation of the differential toxicity of PM components and mixtures and the shape and magnitude of the PM health impact function.**

A great deal of research on the health impacts of $PM_{2.5}$ and specific PM components has been conducted over the past 15 years, and these topics have already been identified as priorities by EPA in the context of its periodic reviews of the U.S.

national ambient air quality standards for PM. While the scientific record is robust in many respects, there are still important unanswered questions about the relative toxicity of different constituents. Also, research continues to inform the overall understanding of the magnitude and nature of PM health impacts, including more precise quantitative information about the relationship between indoor and ambient concentrations and health impacts. Continued investment in this research is important, and there is a particular need for more studies in developing countries.

7. More detailed analysis and comparison of the costs and benefits of mitigating BC from specific types of sources in specific locations.

More work is needed to link BC sources in specific regions to climate, health and environmental impacts, all the way through the causal chain. Improved characterization of BC control strategies, their costs, and their net impact on radiative forcing, as influenced by location, will help ensure maximum climate benefits. This depends in large part on improved *emissions characterization and measurement*, as described above, but also more refined modeling techniques capable of evaluating regional or local scale impacts. Greater attention should be paid to the location of the proposed change in emissions, especially for near-Arctic or near-Himalayan emissions. Similarly, the links between *non-radiative impacts of BC, such changes in rain, snow, and water resources*, and specific source classes or regions have not yet been well established; the ability to relate non-radiative effects to aerosol (and precursor) emissions needs further development.

Health impacts will also vary by source type and location. As discussed above, more work on PM components is need and this can support further analysis to help identify emissions reductions with maximum co-benefits for public health and climate. While the relationship between BC and visibility is relatively well understood, the *impacts of BC dimming on other welfare outcomes, such as ecosystem and crop health*, have not been well quantified. Research on the human health and environmental consequences of reductions in specific sectors, coupled with development of appropriate metrics (#8, below) would allow for comparison between source sectors and regions and improved ability to develop and meet policy goals.

8. Refinement of policy-driven metrics relevant for BC and other short-lived climate forcers.

Some of the fundamental assumptions that go into the calculation of policy-relevant climate metrics for long-lived GHGs do not apply to BC; therefore, it is difficult to apply metrics developed for GHGs to BC and other short-lived forcers. Though "alternative" metrics have been proposed for BC, none is yet widely utilized. Appropriately tailored metrics for BC are needed in order to quantify and communicate BC's impacts and properly characterize the costs and benefits of BC mitigation. Improved metrics could incorporate non-radiative impacts of BC, such as impacts on precipitation. Similarly, given BC's (and other aerosols') direct impacts on human health, health outcomes could also be incorporated into such a metric. Developing methods to quantify the benefits of BC mitigation on both climate and health would encourage policy decisions that factor in climate and health considerations simultaneously, within a unified framework. Analysis is also needed to examine how utilizing alternative metrics would affect policy priorities and preferred mitigation options.

9. Analysis of key uncertainties.

Systematic analysis of key remaining uncertainties and technical gaps regarding BC could help prioritize future research and investment by clarifying which of these factors exert the largest influence on modeled outcomes. Such analysis would involve both: (1) Model intercomparison of BC radiative forcing and climate impacts between global and regional models, along with comparisons to ambient measurements (including remote sensing and tracer-based analyses); and (2) Sensitivity analysis of the factors influencing models' representation of (a) the net effect of a given mitigation measure, considering all co-emitted pollutants; and (b) the overall global and regional contribution of BC and OC to radiative forcing and temperature change.

Global and regional models give a different range of predictions of BC's RF and climate impacts due to different model configurations, parameterizations and assumptions (model resolution, chemical and physical processes, aging/mixing, deposition, etc.). A comparison of model results and diagnostic analysis will reduce remaining uncertainties regarding BC impacts. Additional sensitivity analysis should consider the importance of: (1) emissions inventories utilized (including type and magnitude of co-emissions represented); (2) model representation of transport and vertical distribution of emissions, aging and mixing of particles, radiative properties of particles, and particle interactions with clouds and snow; and (3) the use of observational data to constrain model results and emissions estimates.

Appendix 1

Ambient and Emissions Measurement of Black Carbon

A1.1 Introduction

Measurements of BC and other PM constituents are critical to understanding the climate impacts of these substances, as well as evaluating human health and environmental effects. These measurements serve as important inputs to air quality forecasting and climate models, source apportionment models, and emissions inventories. Deposition measurements are also needed to judge impacts on snow and ice.

Observational data for BC comes from two main sources: ambient measurements and source-based emissions measurements. These measurements involve both sample collection and sample analysis procedures, with each step having important impacts on reported measurements. Most estimates of BC are based on thermal-optical and filter-based optical techniques, which classify the measured quantity as apparent elemental carbon (ECa) and apparent black carbon (BCa). While the terms "black carbon" and "elemental carbon" are frequently used as labels for quantities produced, the addition of the term "apparent" clarifies that these are considered to be estimates of BC concentrations. This appendix describes the most common sample analysis methods (thermal-optical and optical), the types of instruments that can be used for these methods, and key limitations in current measurement methods, approaches, and instruments. This appendix also describes the key sources of ECa and BCa measurement data in the United States, in terms of the types of ambient data collected and the information gathered from testing of both stationary and mobile sources. Next, this appendix describes key applications of source-testing data, particularly for constructing U.S. emissions inventories. Data from other countries is reported where available and applicable.

A1.2 Ambient Black Carbon Measurements

BC mass concentration estimates are routinely measured at ground-level in the ambient air or in deposited materials, but can also be taken in aircraft and on remote sampling platforms. Globally, a significant amount of ambient data has been compiled from the following types of measurements:

- *Ground-based ambient air measurements* are taken in near real time using field analyzers or obtained in a laboratory following collection of PM onto a filter. This is by far the largest source of observational data on BCa and ECa. Details on some of the key ambient air monitoring networks producing these data are described in Table A1-1.
- *Ice core measurements* of BCa and ECa have been conducted in glaciers around the world, providing a historical record of BC concentrations.
- *Surface snow measurements* have been conducted to quantify recent BC in snow based on BCa and ECa concentrations in locations around the world. Snow data is much more limited in spatial and temporal coverage in comparison to ambient monitoring.

The concentration of carbon in PM is regularly measured using methods based on the chemical, physical, and light absorption properties of the particles. The chemical and physical properties of carbonaceous PM vary in terms of both refractivity (the inertness of the carbon at high temperatures) and light absorption. Each carbon measurement technique provides unique information about these properties. All current analysis methods are operationally defined, meaning that there is no universally accepted standard measurement. When developing these methods and operational criteria, some scientists use PM's optical properties or light-absorbing characteristics (optical or light absorption methods), some use its thermal and chemical stability (thermal-optical methods), while others use its morphology or microstructure or nanostructure (microscopy methods). One major class of methods, thermal or thermal-optical techniques, distinguishes refractory and non-refractory carbon as ECa and OCa, respectively (Figure A1-1). The second major class of methods, optical methods, quantifies the light absorbing component of particles as BCa,

Table A1-1. Global Monitoring Activities. (Source: U.S. EPA)

Worldwide Air Monitoring Networks$^{a, b}$ 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—PM$_{2.5}$ Speciation Trends Network^{c}	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/rnspa-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 Tranmittance – 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*

Worldwide Air Monitoring Networks[a, b] for Black Carbon

Network	Country/ Agency	Years of Data	BC - Indicator	Number of Sites	Measurement Method	Location of Information and/ or Data
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*

[a] 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.

[b] 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.

[c] 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.

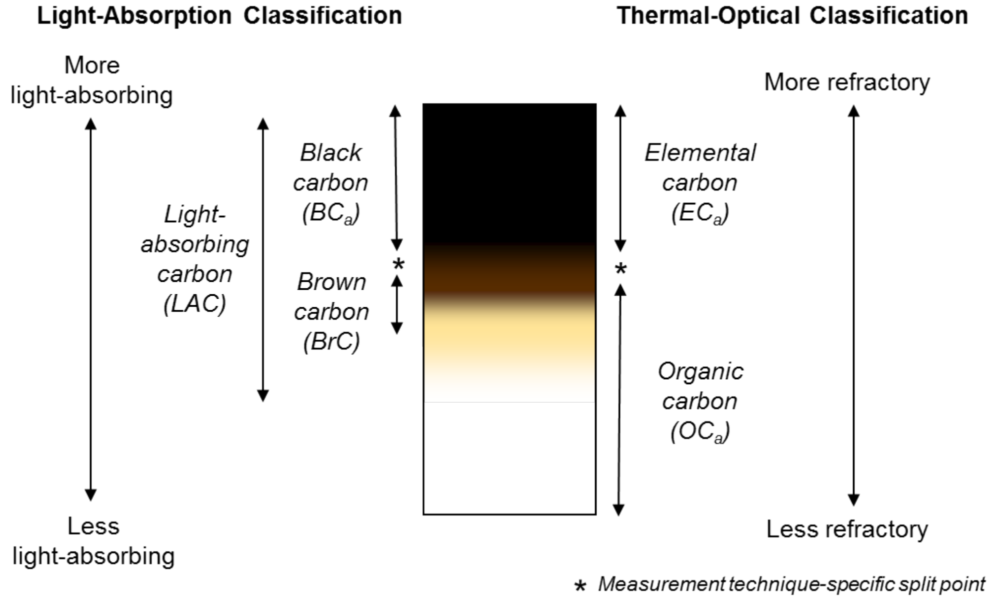


Figure A1-1. Measurement of the Carbonaceous Components of Particles. Black carbon and other types of light-absorbing materials can be characterized by measuring their specific light-absorbing properties, as seen on the left side of the figure (BCa/BrC/LAC). This contrasts with other approaches to characterizing 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). (This figure is also as shown as Figure 5-1.) (Source: U.S. EPA) which can be used to estimate BC concentrations and can also indicate the existence of components that absorb in the near-UV (i.e., brown carbon, BrC). Light absorbing carbon (LAC) is a term used for light absorbing substances in the atmosphere, which includes soot and its components, BC and BrC. There is a lack of consensus and standardization regarding the operational criteria used, calibration materials used, and defining characteristics or properties of the BC measured. The methods used to measure ECa and BCa require standardization and re-evaluation for climate and regulatory uses.

A1.2.1 Thermal-Optical Methods, ECa

As noted in Chapter 5, thermal-optical methods are by far the most commonly used. Since 1982, thermal-optical analysis methods have been applied to measure the ECa and OCa component of ambient and source aerosols (Huntzicker et al., 1982; NIOSH, 1999; Birch and Cary, 1996; Chow et al., 1993; Peterson and Richards, 2002; Chow et al., 2007). 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 ECa. Thermal optical-reflectance (TOR) methods use reflectance for char correction and separation of ECa from OCa, while thermal-optical transmittance (TOT) uses transmittance. Long-standing reliance on these methods—which measure ECa, rather than BC—has resulted in an extensive observational record based on ECa and OCa splits, and the frequent substitution of ECa data for BCa data, since availability of the latter is limited. In addition to laboratory-based thermal-optical methods for ECa, semi-continuous or near real-time thermal-optical methods for ECa and OCa are commercially available. The semi-continuous analyzer provides hourly in-field measurements of ECa and OCa. This semi-continuous analyzer also provides a measure of light absorbing or optical BCa.

A1.2.2 Light-Absorption Measurements, BCa

Currently, light-absorption or “optical” measurements of BCa are not consistently deployed in routine monitoring programs in the United States. The one program area in which light-absorption methods are used is in assessing visibility impairment in national parks and wilderness areas via the Interagency Monitoring of Protected Visual Environments (IMPROVE) program. To date, optical methods have not been widely used in urban monitoring networks. However, such approaches are commercially available and could be more widely deployed. These approaches fall into two general categories—optically absorbing and incandescent (thermal emission of light) measurement. Relative to the incandescence techniques, optical techniques for BCa are in wider use. A listing of a variety of commercially available instruments used for monitoring ambient or source concentrations of BCa and the wavelength selected for measurement is provided in the Table A1-2.

Modern light-absorbing techniques rely on passing a laser beam at a specific wavelength through a particle sample, either in an air volume or deposited onto a filter, and observing how much light is absorbed by the particles. BCa is typically measured over the green to infrared wavelengths, where it absorbs more strongly than other LAC. BrC may also absorb light at shorter wavelengths (near-UV and UV). Many BCa instruments can measure at multiple wavelengths, sometimes simultaneously depending on the exact instrument configuration. This provides information about components that absorb light over different parts of the UV/Visible spectrum. Thus, these instruments may be used to distinguish between BCa and BrC; however, in many cases researchers have not been careful to distinguish how much of the measured light-absorbing carbon falls into each category. In order to convert light absorption to a BC mass concentration, a mass absorption coefficient or similar conversion factor is used. The conversion factor is based on experiments that simultaneously measure light absorption at a specific wavelength and BC mass (either as ECa from ambient measurements or particle mass from soot generation experiments). It is recommended that light absorption be reported in the original units of absorption along with any mass absorption coefficients or conversion factors used to convert absorption to BC mass concentration.

Incandescence is the second approach used to quantify BCa. Laser induced incandescence (LII) subjects particles in an air stream to a high-intensity laser in the infrared. Some LII techniques can measure individual particles, providing data on particle size, BCa mass concentration (based on an assumed BC density), and an indication of the mixing state of the particles. LII is currently used in limited research applications in the United States.

A1.2.3 Inter-comparisons Among Optical BCa and Thermal-Optical ECa Measurements

Given that ECa concentrations are commonly used to represent BCa, and vice versa, the relationship between BCa and ECa is important to characterize.

Table A1-2. Examples of Commercially Available Optical BCa Measurement Techniques. (Source: U.S. EPA)

Instrument (Manufacturer)[a]	Real-time (R) or Off-line (O)	Filter (F) or Air Stream (A)	Wavelengths Measured (See Chapter 2)[b]
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

[a] The use of commercial trade names or vendor names does not constitute an endorsement by the U.S. EPA.

[b] A variety of mass absorption coefficients (MACs) or similar conversion factors are used to convert light absorption at a particular wavelength to BC mass concentration. See the BCa:ECa comparison in Table A1-3 for the MACs used in the comparison studies referenced.

It should be noted that the two measurements are not always entirely independent, as the selected conversion factor to estimate BCa is sometimes based on experiments establishing a relationship between light absorption and ECa. A number of inter-comparison studies have examined several different BCa or ECa measurement approaches simultaneously to evaluate how well they agreed (Table A1-3). Recent studies, published in the year 2000 or later, that compare ambient BCa and ECa measurements were reviewed (Chow et al., 2009; Bae et al., 2007; Hitzenberger et al., 2006; Snyder and Schauer, 2007; Sharma et al., 2002; Sahu et al., 2009; Yang et al., 2006; Miyazaki et al., 2008; Babich et al., 2000; Ram et al., 2010; Husain et al., 2007; Jeong et al., 2004; Lim et al., 2003; Hagler et al., 2007a). In a wide variety of environments, ranging from the remote Arctic to urban cities, BCa and ECa measurements were reported to have consistently high correlation (average R = 0.86 +/- 0.11). In addition, Figure A1–2 shows that ratios of BCa/ECa are typically near 1 (BCa/ECa = 0.7-1.3, or within 30%, for 70% of studies), however there do exist studies reporting very low BCa/ECa ratios (~0.5) and very high BCa/ECa ratios (~2).

The ratio of BCa to ECa and the consistency of the relationship may depend on the aerosol mixture and/or the specific method used. The difference in BCa and ECa concentration may also be largely influenced by the conversion factors used to change light absorption into mass concentrations for optical methods as well as corrections for measurement artifacts. The differences between BCa and ECa may also be due to a lack of consistency in the post-processing of the raw measurements among studies (Venkatachari et al., 2006; Collaud Coen et al., 2010; Park et al., 2010; Virkkula et al., 2007; Chow et al., 2009; Bond et al., 1999; Chen et al., 2004; Jimenez et al., 2007). It should be noted that these inter-comparison data are based on ambient measurements and similar data are needed for source measurements.

Table A1-3. Inter-comparison of Ambient BCa and ECa Measurements. Comparisons include (a) BCa measurements using different instruments; (b) BCa measurements using different instruments after application of correction algorithm; (c) ECa measurements using different instruments; and (d) BCa measurements compared to ECa measurements. (Source: U.S. EPA)

Table A1-3 (a) 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 and Schauer (2007)	Aethalometer	PSAP	0.93	0.86	1.41	Slope of line (intercept small)
					2.68	Overall Average Ratio

Table A1-3 (b) 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
					1.03	Overall Average Ratio

Table A1-3 (c) ECa–ECa comparisons.

Citation	High Measurement	Low Measurement	r	n	Ratio (High/Low)	Notes
Bae et al. (2007)	NIER-EC NIOSH TOT	UT-EC NIOSH TOT	0.99	223	1.11	Semicontinuous Sunset with different temperature protocols: NIER - shortened protocols, UT: nine-step
Bae et al. (2009)	IMPROVE TOR	ACE-Asia TOT	0.79	709	2.14	St. Louis Supersite, MO
Cheng et al. (2010)	IMPROVE_A TOR	IMPROVE_A TOT	0.95	89	1.74	TOT and TOR from single DRI Model 2001 analyzer with denuder
Cheng et al. (2010)	IMPROVE_A TOR	IMPROVE_A TOT	0.95	89	1.83	TOT and TOR from single DRI Model 2001 analyzer without denuder
Cheng et al. (2011)	IMPROVE TOR	NIOSH TOT	0.97	81	1.72	Birmingham, AL urban SEARCH site
Cheng et al. (2011)	IMPROVE TOR	NIOSH TOT	0.92	97	2.78	Jefferson Street, Atlanta, GA urban SEARCH site
Cheng et al. (2011)	IMPROVE TOR	NIOSH TOT	0.85	75	3.57	Pensacola, FL coastal SEARCH site
Cheng et al. (2011)	IMPROVE TOR	NIOSH TOT	0.81	80	4.00	Centerville, AL rural SEARCH site
Chow et al. (2006)	IMPROVE_A TOR	STN NIOSH TOT		17	1.87	Fresno Hi-Vol Summer

Citation	High Measurement	Low Measurement	r	n	Ratio (High/Low)	Notes
Chow et al. (2006)	IMPROVE_A TOR	STN NIOSH TOT		8	2.86	Fresno Hi-Vol Winter
Chow et al. (2006)	IMPROVE_A TOR	STN NIOSH TOT		17	1.49	Fresno RAAS Summer
Chow et al. (2006)	IMPROVE_A TOR	STN NIOSH TOT		5	2.17	Fresno RAAS Winter
Chow et al. (2006)	IMPROVE_A TOR	STN NIOSH TOT		17	1.46	Fresno RAAS Summer with denuder
Chow et al. (2006)	IMPROVE_A TOR	STN NIOSH TOT		5	1.69	Fresno RAAS Winter with denuder
Chow et al. (2009)	IMPROVE_A_TOR_EC	IMPROVE_A_TOT EC	0.95	49	1.30	Fresno Supersite, CA
Chow et al. (2009)	STN_TOR EC	STN_TOT EC	0.9	18	1.41	Fresno Supersite, CA
Chow et al. (2009)	IMPROVE_A TOR EC	STNTOR EC	0.94	18	1.10	Fresno Supersite, CA
Chow et al. (2009)	IMPROVE_A TOR EC	French two step EC	0.9	8	1.03	Fresno Supersite, CA
Chow et al. (2009)	IMPROVE_A TOR EC	Sunset Field EC TOT	0.87	48	1.82	Fresno Supersite, CA
Fujita et al. (2007)	IMPROVE TOR	NIOSH TOT	0.94	14	1.60	Ambient (urban)
Fujita et al. (2007)	IMPROVE TOR	NIOSH TOT	0.99	21	1.20	Ambient (on-road)
Gan et al. (2010)	IMPROVE TOR	NIOSH TOT		6	1.09	Submarine diesel PM
Klouda et al. (2005)	IMPROVE TOR	STN-NIOSH TOT		99	1.66	RM 8785 suspended PM
					1.89	**Overall Average Ratio**

Table A1-3 (d) is on the following two pages.

Table A1-3 (d) BCa–ECa comparisons.

Citation	BCa Method	MAC ($m^2 g^{-1}$)	l, 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[a]	0.4[a]	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[a]	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[a]	0.4[a]	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[a]	0.4[a]	1.00	Philadelphia, PA
Hagler et al. (2007)	PSAP	[b]	565	NIOSH TOT	0.95		7ngm^{-3}	[b]	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[c]	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[c]	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[c]	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 and Schauer (2007)	PSAP	[b]	565	Sunset $PM_{2.5}$ hourly	0.91			[b]	Riverside, CA
Snyder and Schauer (2007)	Aethalometer AE-31	[b]	880	Sunset $PM_{2.5}$ hourly	0.93			[b]	Riverside, CA
Sharma et al. (2002)	Aethalometer AE-11 $PM_{2.5}$	19	880	IMPROVE TOR $PM_{2.5}$	0.89		0.58[a]		Egbert, Canada
Sharma et al. (2002)	PSAP $PM_{2.5}$	10	565	IMPROVE TOR $PM_{2.5}$	0.99		0.58[a]		Egbert, Canada
Sharma et al. (2002)	Aethalometer AE-11 $PM_{2.5}$	19	880	IMPROVE TOR $PM_{2.5}$	0.98		1.42[a]		Downsview, Canada

Citation	BCa Method	MAC ($m^2 g^{-1}$)	l, nm	EC Method	r	Avg BCa	Avg EC	Ratio BCa/EC	Location
Sharma et al. (2002)	PSAP $PM_{2.5}$	10	565	IMPROVE TOR $PM_{2.5}$	0.69		1.42[a]		Downsview, Canada
Sharma et al. (2002)	Aethalometer AE-11	19	880	Cachier two step EC 1100C in O_2	0.91		0.087, 0.012[a]		Alert, Canada (Arctic)
Sharma et al. (2002)	PSAP	10	565	Cachier two step EC 1100C in O_2	0.93		0.087, 0.012[a]		Alert, Canada (Arctic)
Sharma et al. (2002)	Aethalometer AE-11 $PM_{2.5}$	19	880	NIOSH TOT $PM_{2.5}$	0.92		1.95[a]		Evans Ave, Canada
Sharma et al. (2002)	PSAP $PM_{2.5}$	10	565	NIOSH TOT $PM_{2.5}$	0.96		1.95[a]		Evans Ave, Canada
Sharma et al. (2002)	Aethalometer AE-11 $PM_{2.5}$	19	880	NIOSH TOT $PM_{2.5}$	0.89		1.82[a]		Palmerston, Canada
Sharma et al. (2002)	PSAP $PM_{2.5}$	10	565	NIOSH TOT $PM_{2.5}$	0.89		1.82[a]		Palmerston, Canada
Sharma et al. (2002)	Aethalometer AE-11 $PM_{2.5}$	19	880	NIOSH TOT $PM_{2.5}$	0.92		1.48[a]		Winchester, Canada
Sharma et al. (2002)	PSAP $PM_{2.5}$	10	565	NIOSH TOT $PM_{2.5}$	0.54		1.48[a]		Winchester, Canada
Venkatachari et al. (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 et al. (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 et al. (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 et al. (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 et al. (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 et al. (2008)	COSMOS	9.8	565	Sunset $PM_{1.0}$ hourly	0.96	n/a	n/a	n/a	Thailand
Babich et al. (2000)	Aethalometer AE-20	19	880	IMPROVE TOR $PM_{2.5}$	0.87	1.1	1.4	0.79	Bakersfield, CA
Babich et al. (2000)	Aethalometer AE-20	19	880	IMPROVE TOR $PM_{2.5}$	0.98	1.2	1.5	0.80	Chicago, IL
Babich et al. (2000)	Aethalometer AE-20	19	880	IMPROVE TOR $PM_{2.5}$	0.95	0.8	1.3	0.62	Dallas, TX
Babich et al. (2000)	Aethalometer AE-20	19	880	IMPROVE TOR $PM_{2.5}$	0.95	1.1	1.5	0.73	Philadelphia, PA
Babich et al. (2000)	Aethalometer AE-20	19	880	IMPROVE TOR $PM_{2.5}$	0.96	3.1	3.9	0.79	Phoenix, AZ
Babich et al. (2000)	Aethalometer AE-20	19	880	IMPROVE TOR $PM_{2.5}$	0.92	1.6	1.9	0.84	Riverside,CA
Ram et al. (2010)	Aethalometer	16.6	880	Sunset TOT NIOSH PM_{10}	0.79	4.45	3.84	1.2	Kanpur, India
Husain et al. (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 et al. (2003)	PSAP $PM_{2.5}$	10	565	R&P 5400 $PM_{2.5}$ hourly	n/a	1.26	2.8	0.5	Atlanta, GA
Lim et al. (2003)	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 et al. (2003)	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 et al. (2003)	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

[a] Median concentration.

[b] 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.

[c] Calibration curve based on dissolved carbon black.

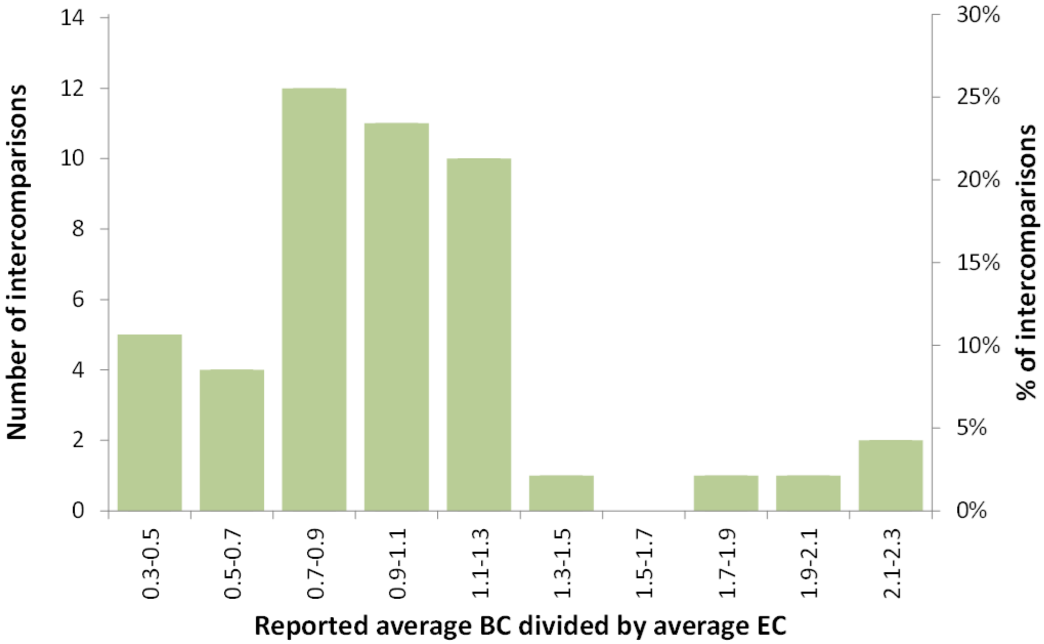


Figure A1-2. Reported BCa/ECa Ratios for a Pair of Measurement Techniques Reported in Ambient Field Studies. (Source: U.S. EPA)

Instruments measuring light absorption are often capable of measuring light absorption at additional wavelengths in the near-UV or UV, which may help indicate the presence of BrC. Such approaches are currently used to attempt separation of the influence of wood smoke aerosols, which tend to be rich in BrC, from those that are dominated by diesel emissions and other fossil fuel combustion, which are rich in BC.

The disagreement among BCa measurements may be due in part to differing instrument sensitivities and responses to other PM components (Slowik et al., 2007), filter-loading artifacts, or the use of an incorrect light-absorption-to-BC mass concentration conversion factor for studies reporting BCa in terms of their mass concentrations. Chow et al. (2009) found that applying post-processing algorithms greatly improved the agreement among different filter-based BCa techniques (Refer to Table A1-3 above). EPA and other researchers (Collaud Coen et al., 2010; Park et al., 2010; Virkkula et al., 2007; Bond et al., 1999; Chen et al., 2004; Jimenez et al., 2007) are similarly assessing whether post-processing algorithms and site-specific conversion factors may also be beneficial to better understand the differences among BCa and ECa measurements. In addition to issues with the various BCa and ECa measurement techniques mentioned above, BCa-to-ECa ratios are likely to be affected by the presence of other light-absorbing species (e.g., BrC and dust). The specific inter-comparison circumstances (e.g., location, season, sample collection and analysis procedures, optical wavelength, data corrections, and aerosol mixture[1]) may be important to understand and reconcile reported differences. A summary of the data presented in Figure A1-2 and comparisons of BCa/BCa (with and without correction algorithms) and ECa/ECa along with the circumstances for the inter-comparison measurements is found in Table A1-3 (Inter-comparison of Ambient BCa and ECa Measurements).

A1.2.4 Inter-comparisons Among Thermal-Optical ECa Measurements

The differentiation between OCa and ECa by thermal-optical methods is highly dependent on the specific measurement method operational procedures used. Different instrument operating procedures, thermal temperature profiles, and correction for char can lead to differences in ECa. Previous method comparison studies summarized

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[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.

in Watson et al. (2005) showed differences of a factor of 2 to be common for measurements of ECa. Method differences have been found not only to depend on the operational definitions of the methods, but also to depend on the chemical composition and source of aerosol collected (i.e., location and seasons), particle loading on the filter, and uniformity of the filter deposit. Samples containing biomass smoke present more difficulties due to components (inorganic compounds) that result in an overestimation of ECa (Novakov and Corrigan, 1995a). The presence of BrC can also affect the measurement of ECa. Reisinger et al. (2008) evaluated the BrC content of samples and the impact on the comparison of thermal-optical methods and found TOT methods to be less sensitive to the presence of BrC than other thermal methods tested.

There are currently no reference materials that reflect the variety of aerosol types in the atmosphere and there is no standardized method protocol. There is one consensus-based standard reference material (SRM 8785) available from the National Institute of Standards and Technology (NIST) that has values assigned for OCa and ECa as measured by the two EPA protocols discussed below (IMPROVE TOR and NIOSH-like TOT). The ECa assigned by the two protocols (IMPROVE: NIOSH) in the NIST SRM differ by a factor of 1.7 (Klouda et al., 2005).

Results from thermal-optical ECa:ECa comparison studies that use methods similar to the two EPA protocols that are not summarized in Watson et al., (2005) and published since 2005 were reviewed and summarized in Table A1-3 (Klouda et al., 2005; Bae et al., 2007; 2009; Cheng et al., 2010; 2011; Chow et al., 2006; 2009; Fujita et al., 2007; Gan et al., 2010). The average ratio of ECa:ECa from these studies is 1.9. Chow et al. (2006) also compared ECa measurements by IMPROVE and NIOSH-like TOT methods for a variety of lab generated source samples. The IMPROVE:NIOSH-like TOT ECa range of ratios were from 1.01 to 1.04 for diesel, acetylene flame, carbon black, and graphite source samples, and 1.13 for electric arc samples. The biggest difference was found for the wood smoke source sample (ratio of 1.88).

A1.2.5 Inter-comparison of Two EPA ECa Measurement Protocols

The IMPROVE TOR and NIOSH-like TOT methods have been widely used in the EPA's national urban Chemical Speciation (CSN) and rural IMPROVE ambient monitoring networks. EPA has transitioned the urban CSN from the NIOSH-type TOT method to the IMPROVE_A TOR method. The transition began in May 2007 and was completed in October 2009 and includes a change to the sampling system as well as the analytical method. The major difference in the sampling method is the sampling flow rate (increased to ~22 LPM from ~6.7 LPM) and sample filter diameter (reduced from 46.2 mm to 25 mm), which results in an overall increase in pressure drop across the filter during sampling. The combination of these changes results in a reduction in the OC measured, which is most likely related to a change in sampling artifacts. The rationale for the transition of the urban CSN to IMPROVE-like sampling and analysis method was to institute consistency in the carbon measurements across the EPA's national particulate monitoring networks.

To understand the differences between the two carbon monitoring protocols, EPA established pairs of old and new CSN monitors at 11 sampling locations and collected parallel carbon measurements for 12 months from May 2009 to April 1010. Most other CSN sites also collected 2 months of parallel measurements when they initially transitioned between May 2007 and October 2009. In addition, lower flow rate CSN samplers whose carbon was measured with the NIOSH-type TOT method were used to collect data at 14 urban sites in the IMPROVE network.

The comparison between the previous CSN TOT data and the current CSN IMPROVE TOR data indicates that measured EC is reasonably consistent between the methods at the 11 locations that produced 2009-2010 data (Figure A1-3). These data suggest that both monitoring protocols could be interchangeably used to evaluate BC aerosols predicted by climate and air quality models, and to evaluate trends. The seasonal differences in these EC differences are modest, and may be related to the combined effect of sampling rates and analytical protocol and the resulting differences in measured OC as described above. However, when all the parallel EC measurements are considered, a different pattern emerges.

The comparison of urban EC derived with the IMPROVE TOR method to that from the CSN NIOSH-like TOT method in Figure A1-4 shows the ratio of monthly values decreasing from approximately 1.5 in 2005 to approximately 1.0 towards the end of 2009. The higher monthly ratios observed between 2005-2006 are consistent with the finding that CSN EC is on average 30% lower than IMPROVE EC (Hand et al., 2011) and the ratios shown for the last 12 months are consistent with the data presented

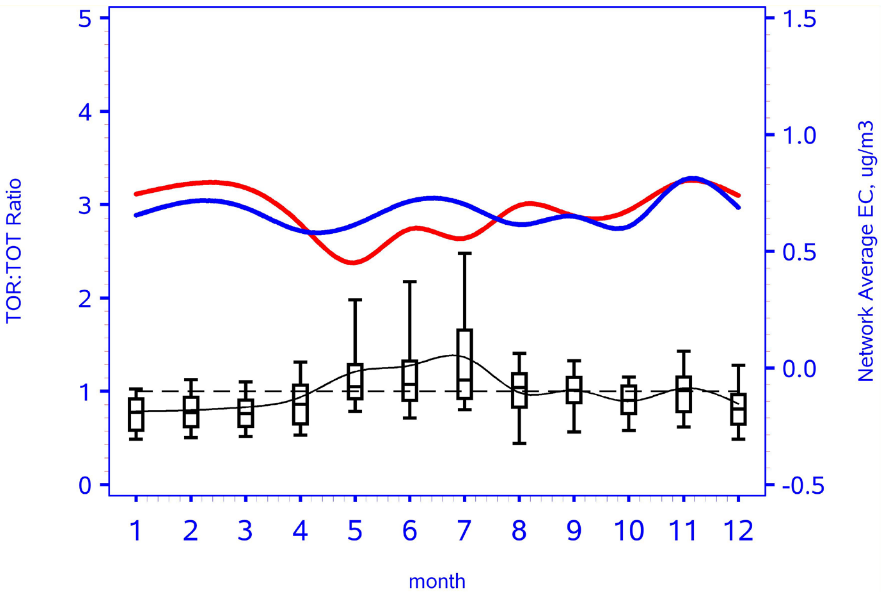

Figure A1-3. Monthly Distribution of ECa/ECa Ratio For Two EPA methods (TOR/TOT) From 897 Collocated Measurements Among 11 Urban CSN Locations. Average CSN NIOSH ECa concentration is red; IMPROVE TOR ECa is blue; the distribution of daily ratios is presented as box plots (black). (Source: U.S. EPA, AQS)

in Figure A1-3. Although the number of included monitors varies from month to month, the pattern from a consistent set of 5 collocated IMPROVE–CSN sites from 2005 thru 2009 reveals the same general trend. Although the ratio appears to have decreased during this five year period, the cyclical behavior suggests that relatively higher ratios often occur during the warmer months. EPA will continue to evaluate the differences between the two measurement protocols and possible connections to changes in the way the measurements were conducted as well as the potential influence of changes in other collected aerosols.

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 A1-5 shows the monthly distribution of OC/BC among 897 measurements at 11 urban monitoring sites[2] 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), 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 et al., 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 aerosol is critical for differentiating among reflecting vs. absorbing particles for assessment of radiative forcing, where OC is assumed to be mainly light scattering. While the IMPROVE TOR OCa is adjusted for sampling artifact with backup filters, the CSN NIOSH TOT protocol is only adjusted with nominal network value of 1 $\mu g/m^3$ which may be too low (Chow et al., 2010a). On the other hand the much more suppressed seasonal behavior in the TOR carbon ratios could be related to the higher flow rate IMPROVE protocol samplers which may not fully retain semi-volatile OC particles. The latter will require further study to understand its implications for using these measurements to develop emissions

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[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.

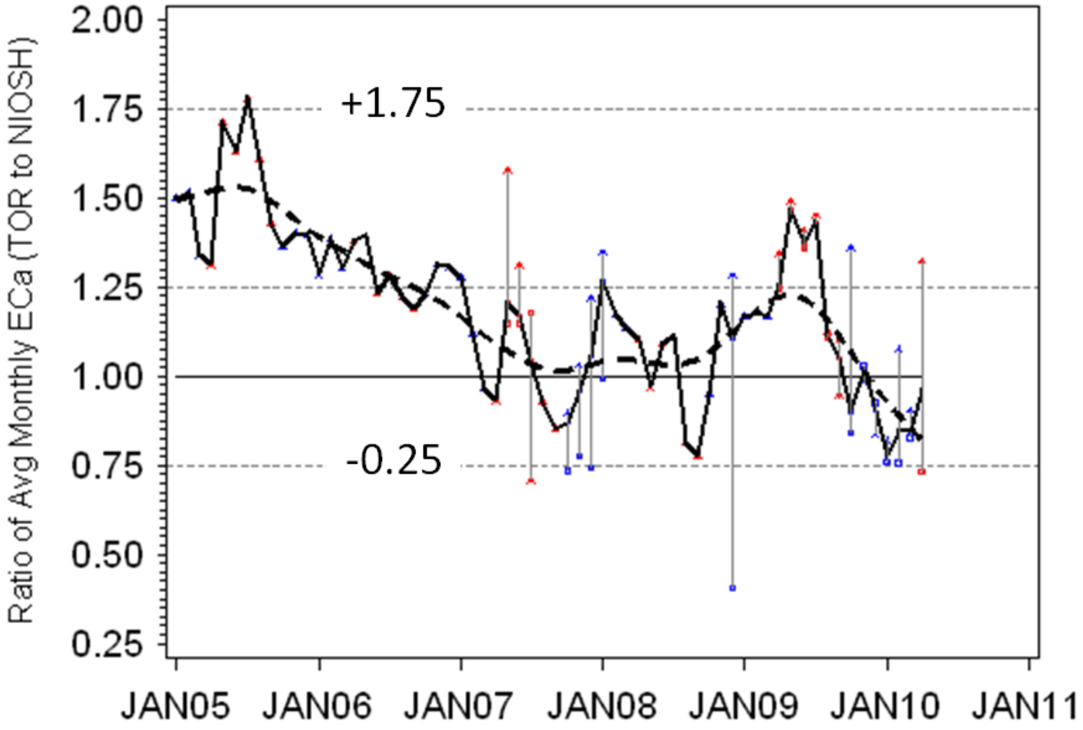

Figure A1-4. Trend in Ratio of Urban EC Derived From IMPROVE TOR and CSN NIOSH-like TOT Methods. The ratio of monthly averages derived from collocated IMPROVE (or CSN IMPROVE-like) with CSN MetOne samplers is shown, where the former measurements are produced with the IMPROVE protocol and the latter measurements are produced with the NIOSH-like TOT protocol. The results provided by ECa from the IMPROVE network samplers are shown with (*) and those from the CSN are represented by squares. Months October-March are denoted in blue and months April-September are shown in red. The dotted line is a spline fit through the monthly data. (Source: U.S. EPA, AQS)

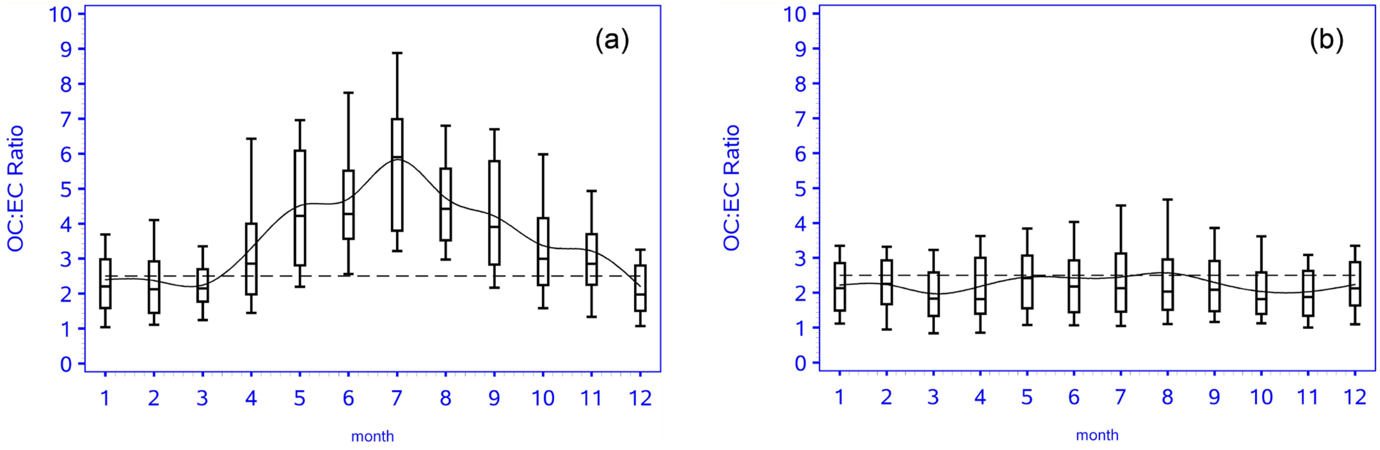

Figure A1-5. Monthly Distribution of OC/BC ratios for 11 CSN Sites Produced With (a) the NIOSH-like TOT and (b) IMPROVE_A TOR Monitoring Protocols, 2009-10. Nominal OC sampling corrections of 1 µg/m3 for CSN NIOSH type TOT have been used (Frank, 2006). The IMPROVE protocol data are adjusted with backup filters. Due in part to inability to adequately correct the CSN NIOSH OC sampling artifacts (Chow et al., 2010a), these data may in fact overstate ambient OC/BC and imply a seasonal pattern which may be an artifact of the monitoring method. (Source: U.S. EPA) inventories and to evaluate climate modeling data.

A1.2.6 Implications of Changes in ECa Monitoring on Estimated Concentration Trends

The urban ECa measurements as reported by the CSN in Figure A1-6 reveal a different picture than the one presented in Chapter 5, Figure 5-12. (For convenience, Figure 5-12 is also re-produced as Figure A1-7.) The reported EC data based on the NIOSH-type measurements in Figure A1-6 appear to depict a slight upward progression from 2002 thru 2006. However, when the data are adjusted using a 5-month moving average of the monthly ratios for 2005-2010 shown in Figure A1-4, together with the 2005 average ratio for earlier data, an estimated downward trend in EC is revealed.

A1.2.7 Other Measurements

Microscopy (the use of microscopes to view the structure of particles) and spectroscopy (measurement of a chemical as a function of wavelength) provide additional information about the physical and chemical structure of carbonaceous PM. An advantage of these methods is that they provide detailed information about how particles age and transform from the point of emission to the atmosphere. A variety of microscopy techniques have been applied to investigate carbon particles. Scanning electron microscopy (SEM) with energy dispersive X-ray spectroscopy (EDX), transmission electron microscopy (TEM), and Raman Microspectroscopy (RM) are the most widely used and have provided the most significant information about carbon aerosol composition to date. Like the thermal-optical

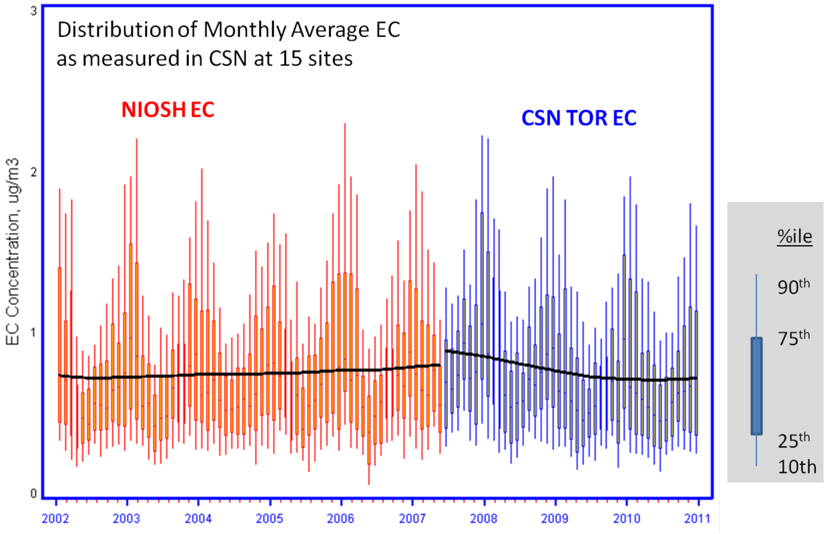

Figure A1-6. Monthly Distribution of Reported CSN EC at 15 sites From 2002 Through 2010. The ECa data produced with the CSN NIOSH-like TOT is shown in red and the ECa data produced with the IMPROVE_A TOR monitoring protocol is shown in blue. (Source: U.S. EPA, AQS)

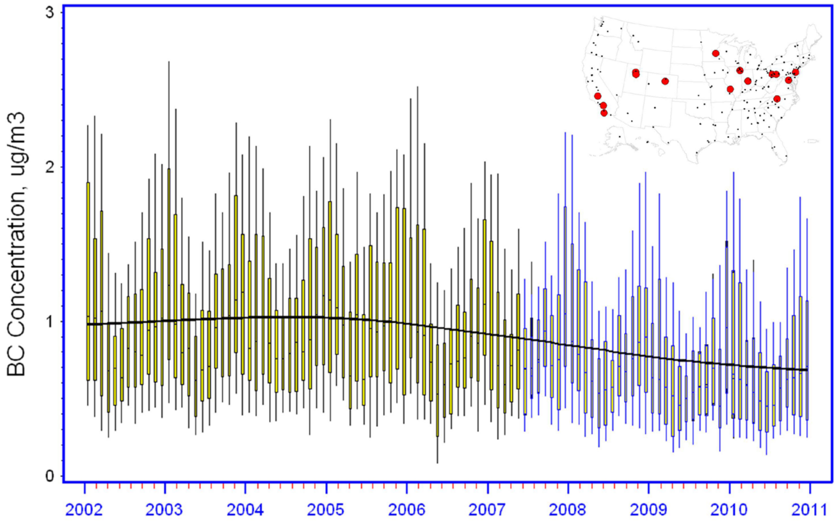

Figure A1-7. Ambient BC Trends (2002-2010), Based on Monthly Distribution of Average ECa Concentrations Among 15 CSN Monitoring Locations in the United States. The map shows the location of the 15 monitoring sites. (This figure is also as shown as Figure 5-12.) (Source: U.S. EPA) and light absorption measurement methods, microscopy has limitations and is subject to artifacts and interpretation issues, but these techniques do provide additional information not gathered by the thermal and optical measurement techniques.

A1.2.8 Limitations of Ambient Measurement Methods

Specific operating conditions, such as the heating temperature, time of heating, and char correction procedures, can influence thermal-optical measurement results. Chemical composition and emissions sources of the measured aerosol, filter loading, and uniformity of the filter particle deposit can also influence OCa and ECa values obtained. Studies suggest that EC measurements for some types of emissions (biomass smoke, dust) may be more strongly affected than traffic-related (e.g., diesel) samples, in part because of higher levels of inorganic components and BrC (Novakov and Corrigan, 1995b). A summary of the comparison of optical BCa to thermal ECa measurements is provided above.

Currently, there are no reference standards for assessing the accuracy of OCa or ECa measurements by thermal methods, nor is there a standardized method protocol for distinguishing between OCa and ECa. Development of standard reference materials and the consensus on standardized method protocols (including data reporting procedures) will be important in the future for the consistent measurement of OCa and ECa for climate purposes.

All optical BC measurements share a fundamental limitation in that they do not directly measure BC mass concentration. Instead, conversion factors (e.g., mass absorption efficiency or mass absorption cross-section) are necessary to generate BCa mass concentrations from the different optical measurements. In addition, the most commonly used filter-based methods are prone to artifacts during sampling. The extent of filter loading can influence particle scattering and shadowing effects which bias results (Park et al., 2010; Bond et al., 1999; Weingartner et al., 2003). While several filter-loading-based correction algorithms have been introduced (e.g., Virkkula et al., 2007), it is uncertain as to whether a correction algorithm should be universally applied as the artifacts may depend upon the particle composition and concentration. Because the aerosol absorption and derived BCa depend on wavelength, it should be noted that some reported BC that is based on wavelengths in the visible spectrum may include other LAC.

A1.2.9 Critical Gaps and Research Needs in Ambient Measurement Methods

In light of the limitations discussed above, the following research can help improve the ambient measurement of BC and LAC in the future and reduce the uncertainty:

- Further comparisons of the predominant thermal and optical methods in use today are needed to better understand and characterize the differences and uncertainties. As comparisons are made, it is important to clearly document the operational conditions of the methods used.

- Having a consistent, well-defined "BC" reference material would help to better understand method differences and define the uncertainties in the various measurement methods.

- It is important to agree on a standardized method of operation and calibration for those methods identified as most important for measuring BC in support of climate and health.

- There is a need to develop methods capable of quantifying particulate components, referred to as BrC or (collectively with BC) as LAC, that provide additional light absorption in the near-UV and UV wavelengths.

- To ensure proper use of measurements, consistent data reporting (including metadata) of the sampling and analysis protocols and data adjustments must be provided.

- Continued research and further development is needed for continuous or real-time single particle measurements (e.g., aerosol time-of-flight mass spectrometry and single particle soot photometers) to enhance our knowledge of particle composition and mixing state.

A1.3 Black Carbon Emissions Source Measurements

Source measurements are used for a variety of purposes, including regulatory compliance. However, in the United States and elsewhere, such Measurements generally focus on total $PM_{2.5}$ mass. measurements of specific components are not required in the United States as part of regulatory testing, and EPA does not have an official source measurement method. Instead, PM composition is measured largely for research purposes, including development of EPA's emissions models. Available source measurements are also used to develop and verify emissions inventories, refine standard measurement approaches, and assess control technologies and mitigation approaches. Due to the limited amount of source emissions data for the carbonaceous content of PM, EPA often must rely on data and methodologies for total PM mass, or substitute emissions models.

A1.3.1 Stationary Source Emissions Measurement Methods

Most current federal stationary source emission standards are focused on the regulation of filterable total PM mass. For most stationary sources in the current inventory, $PM_{2.5}$ emissions are derived from use of a scaling factor applied to collection of filterable total PM and the PM_{10} size fractions. Some local/state and site specific standards also require testing for PM_{10} and $PM_{2.5}$ mass, which sometimes includes both size fractions of filterable and condensable PM. The latter allows for inclusion of certain semi-volatile particles. EPA has recently promulgated a stationary method for $PM_{2.5}$ mass and refined the condensable stationary source

Table A1-4. Stationary Source Emissions Measurement Methods. (Source: U.S. EPA)

Method	PM Type	Filtration Temperature (°F)	Purpose	CFR Reference
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 10 µm	Stack Temperature	General – Particle Sizing	40 CFR 51 Appendix M
EPA Method 201A	Filterable 10 µm/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 10 µm/2.5 µm (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/testmeth/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 µm/2.5 µm			

measurement protocol (U.S. EPA, 2010f); over time this will help ensure greater consistency in stationary source emissions measurements. However, stationary source data currently available for $PM_{2.5}$ inventory purposes are based on non-standardized methods and procedures for PM_{10} and total filterable PM.

Due to the complex nature and variety of sources, regulatory and other standardized source PM methods are mainly designed to provide consistent results across a certain category of sources and not necessarily the entire universe of sources (Myers, 2006). Thus, compilations of source emissions measurements for total PM mass exist such as EPA's AP-42 (compilation of EPA's emission factors) and the U.S. National Emissions Inventory (NEI). However, none of these compilations reflects routine sampling required by regulation for all sources in the inventory. Measurement of carbonaceous PM components including BC or EC are not required as part of compliance testing. Such results are generally available only in the academic literature.

There are a large variety of methods for the measurement of PM mass from stationary sources, many of which measure both the filterable and condensable fractions of $PM_{2.5}$. These methods vary due to operational differences such as filtration temperature and conditioning and treatment of the different components of PM. Table A1-4 provides a list of commonly used stationary source methods and some examples of operational differences for determining PM mass from a variety of sources.

A1.3.2 Mobile Source Emissions Measurement Methods

Mobile sources consist of a diverse group of vehicles and engines, including light-duty gasoline vehicles, heavy-duty diesel trucks, gasoline-powered nonroad engines (e.g., lawnmowers, snowmobiles, recreational boats), nonroad diesel engines (e.g., excavators, locomotives, and marine vessels), and turbine and propeller-driven aircraft. Due to their diverse technologies and applications for highway and nonroad uses, there is considerable variability in BC emissions from mobile sources.

In the United States, particles in mobile source exhaust emissions are measured for compliance with PM emission standards and are expressed on a mass per unit work (g/bhp-hr) or mass per distance traveled (g/mi) basis. For regulatory certification, diesel exhaust particle emissions are measured using procedures described in 40 CFR Part 1065, which employs an engine dynamometer paired with a dilution sampling system collecting samples on Teflon filters at temperatures of about 125°F (which reduces water condensation, yet allows for condensation of organic compounds). The filters are then conditioned at a specific temperature and humidity[3] and weighed. This procedure is commonly used to measure PM from non-diesel mobile sources for research purposes.

Mobile source emissions of BC are almost always measured as ECa. Unlike PM measurements, however, ECa measurements are not routinely taken and EPA does not presently have an official (or even recommended) EC measurement method for mobile sources for regulatory purposes. However, EPA does measure BC in its mobile source emissions characterization programs. There, BC is measured as a particulate matter (PM) component for both gasoline vehicles such as light-duty cars/trucks and diesel vehicles such as heavy-duty diesel trucks (up to 80,000 lbs. gross vehicle weight). It is also measured to a more limited extent from nonroad diesel and even gasoline engines (both 2-stroke cycle engines which have lubricating oil mixed with the fuel) and 4-stroke cycle engines. It is also measured in PM from locomotives, commercial marine, and aircraft.

Sampling temperature has a major effect on the quantity and even the composition of PM. PM emissions are collected on a filter from diluted exhaust. The general methodology for measuring mobile source PM involves diluting the vehicle exhaust with ambient air roughly at a 10/1 dilution ratio (although the dilution ratio varies greatly depending on engine operating mode) using a stainless steel dilution tunnel. The filter temperature is about 125°F, which is high enough to prevent water condensation on the filter from the copious amounts of water vapor present in vehicle exhaust formed from fuel combustion. This temperature also allows some condensation of the organic hydrocarbon compounds present in vehicle exhaust. This general method was developed and has been in use since about 1970 for both diesel and gasoline exhaust. This methodology is also used for EPA emission standards for exhaust from diesel engines including on-road trucks and more recently nonroad diesel engines. This measurement system, known as constant volume sampling of an exhaust stream that is diluted with varying amounts of dilution air, allows for accurate mass weighting of emissions over transient driving conditions (accelerations,

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[3] Mobile source measurements are made at 45%RH, while ambient measurements and many other source tests use 35%RH.

decelerations, steady-state cruise, and idle) where exhaust volume varies. In the ambient air though, vehicle exhaust is rapidly diluted to about 1,000:1 which results in somewhat different condensation of the hydrocarbon compounds into particulate.

The PM measurement method is more developed for diesel PM than for gasoline PM. Numerous studies have been done measuring diesel PM starting with the first EPA emission standard for the 1970 model year for visible smoke from diesel engines.

A1.3.3 Use of Emissions Source Test Data

Though carbonaceous components of PM are not systematically measured across all categories, both EPA and external researchers have measured these components from some source categories. EPA has compiled all available source emissions data into a database called SPECIATE, which currently contains 3,326 raw PM profiles. Because many of these measurements are drawn from research on emissions measurements, the data comes from a variety of sampling and analytical technologies (e.g., see Chang and England, 2004). Despite the uncertainties and limited size of the testing dataset compared to the total number of sources, the SPECIATE database represents the best compiled source of data available. A subset of these data was selected to characterize the source profiles for 15 source categories reported in Chapter 4, Figure 4-1.[4] The number of individual profiles by source category can be quite limited and sometimes only a single value was used. Similar summaries are available elsewhere (Chow et al., 2010a). Note that for some sources, the sum of BC and OC is less than 100% of $PM_{2.5}$ mass. The raw data used to compile Figure 4-1 is available in Table A1-5, along with the percent of estimated non-carbon PM and the OC/BC ratios.

As discussed in Chapter 4, however, EPA does not use any of these profiles for on-road vehicles, since the mobile MOVES model directly calculates EC emissions (U.S. EPA, 2010c). Mobile sources have more variability in emissions than stationary sources, because mobile-source EC varies with driving mode, specific model mix, and other conditions. MOVES is designed to capture this variability. Currently, EPA still uses speciation profiles for nonroad diesel.

A1.3.4 Limitations of Source Emissions Measurement Methods

To estimate EC emissions for a specific source category, EC is typically assumed to be a specific fraction of $PM_{2.5}$ and then total $PM_{2.5}$ mass is used as the starting point. Thus, the measurement and/or estimate of $PM_{2.5}$ mass is one very important source of potential uncertainty. There are inconsistencies in the way $PM_{2.5}$ is measured among source categories, including in the approach for determining filterable and condensable mass, filter equilibration conditions (including laboratory relative humidity), temperature of testing, and dilution and related procedures for semi-volatile PM. Some of these variables can also affect the measurement of carbon components. Because of the way estimates of PM components are generated, both the PM and carbon-specific measurements can affect estimates of BC and OC emissions for a given source category.

Current $PM_{2.5}$ estimation methods based on PM_{10} and total filterable PM can produce variable results, particularly the methods that include condensable PM. For certain stationary source categories, this can produce measurement artifacts that can overestimate the condensable PM emissions by an order of magnitude.[5]

The use of scaling factors applied to filterable total PM and/or PM_{10} to generate estimates of $PM_{2.5}$ introduces additional uncertainty to the estimated emission rate (National Research Council, 2004).

Finally, the representativeness of a particular source profile based on a limited number of source tests is questionable, and derived composite profiles applied to a large number of sources is another source of uncertainty. For both PM and speciation test data, there are the related representativeness issues of tests conducted with actual vs. allowable emissions from the stacks and effluents; tests conducted at facilities of varying age and with different degree and type of controls; and tests affected by other operating conditions. These factors are often not taken into account when BC profiles are applied to $PM_{2.5}$ emissions. There are also potential issues regarding $PM_{2.5}$ mass closure (including treatment of volatile components, particle

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[4] Following the procedures of Reff et al. (2009), 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.

[5] Example artifacts include the potential conversion of sulfur dioxide gas into sulfate particles, affecting the reported PM mass.

bound water) and comparison of BC data based on different measurement methods.

A1.3.5 Critical Gaps and Research Needs in BC Emissions Sampling and Measurement Methods

In light of the limitations discussed above, the following research can help shed light on amounts of BC and LAC emitted by various sources and lessen the uncertainty in developing an inventory of emissions:

1. **For all source measurements**

- Understand how the source EC values relate to source BC values based on currently available techniques.
- Develop high-quality source profiles for sources that need improved characterization for BC, including research into how to quantify the additional light-absorbing components in the near-UV or UV spectrum that are referred to as BrC or, collectively with BC, as LAC.
- Develop a standard "BC" reference material and establish a standard measurement method to report source data as BC.

2. **Stationary source measurements**

- Understand the effect of varying source test methods and conditions on measured $PM_{2.5}$ and BC; and standardization of PM source testing procedures for filterable and condensable PM.
- Perform uncertainty analysis of all source profiles that exist in SPECIATE and how the total mass from the SPECIATE collection methods relates to the total mass from the methods used in the emissions inventory.
- Increase the quantity and quality of meta-data available in the databases that better explain how $PM_{2.5}$ and EC fractions were derived for the various sources in EPA's inventories.

3. **Mobile source measurements**

- Develop standard measurement methods for BC for both on-road and nonroad engines, especially diesels but also gasoline vehicles/engines.
- Establish more routine measurement procedures for BC, including ones that can measure these quantities over short time periods (even instantaneously) as well as over an entire driving cycle.

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. (Source: U.S. EPA)

Stats (%ile)	Agricultural Burning	Bituminous Combustion	Charbroiling	Distillate Oil Combustion	HDDV Exhaust	LDDV Exhaust	Natural Gas Combustion	Noncatalyst Gasoline Exhaust	Onroad Gasoline Exhaust	Prescribed Burning	ProcessGas Combustion	Residential Wood Combustion: HardSoft	SubBituminous Combustion	Wildfires	Wood Fired Boiler
OC Fraction of $PM_{2.5}$															
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	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	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	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 $PM_{2.5}$															
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	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	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	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
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%

Appendix 2

Black Carbon Emissions Inventory Methods and Comparisons

A2.1 Introduction

This appendix provides specific details on the approach used to generate domestic inventories for stationary, area, and mobile sources, and compares that approach to methods used in compiling international inventories. It explores key methodological similarities and differences between inventories, and also outlines the specific methodologies and data inputs used to construct key global and regional inventories currently available.

In general, existing inventories for BC are based on calculations rather than actual emissions measurements. Direct emissions measurements of BC and other PM constituents are rare, and no known inventory is based on direct BC emissions data. Instead, "BC" inventories are calculated mathematically from $PM_{2.5}$ inventories. These calculations divide the direct carbonaceous particle emissions from the $PM_{2.5}$ inventory into two categories: EC and OC. Thus most "BC" inventories are actually EC inventories. Though sometimes the terms EC and BC are used interchangeably, EC is actually more narrowly defined (see Chapter 2 and Appendix 1). By tracking only EC, current inventories fail to account for the portion of primary OC emissions that is light absorbing (including some BC and also BrC). As discussed in Chapter 4, this means current domestic and international inventories systematically underestimate total LAC; however, the magnitude of this gap has not been adequately quantified to date.

Most inventories use a **bottom-up approach**, first pairing $PM_{2.5}$ emission factors with activity level data for the source category to generate $PM_{2.5}$ emissions estimates, and then applying a speciation factor to estimate the amount of BC (or other constituents) contained in the total mass of $PM_{2.5}$ emissions. The BC emissions for individual source categories are then aggregated to form total BC emissions estimates. The speciation factors for an individual source category relate emissions of specific constituents to total $PM_{2.5}$ mass—for example, the amount of BC to total $PM_{2.5}$. $PM_{2.5}$ emission factors and the speciation factors for particular constituents can be based on either fuel consumption data (i.e., estimated emissions of total $PM_{2.5}$ or specific constituents per unit of fuel) or actual measured source emissions from emissions testing (see Appendix 1). Some inventories use a combination of these different approaches, depending on the information available for each source category.

In a few cases, emissions may be back-calculated from remote sensing of primary $PM_{2.5}$ emissions or measured ambient data of the amount of carbonaceous aerosols in the atmosphere. This kind of **top-down approach** is far less common; currently only a few regional inventories in Asia rely on such methods.

Additional information on approaches used to calculate the U.S. emissions inventory and other global/regional inventories is provided in the following text.

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 (see *http://www.epa.gov/ttn/chief/eiinformation.html*). EPA's 2002 Consolidated Emissions Reporting Rule (*http://www.epa.gov/ttn/chief/cerr/cerr.pdf*) 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 on-road 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.

A2.2.1 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:

$$E = A \times EF\ (1 - ER/100) \qquad \text{(Equation 1)}$$

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 4 of this report.

A2.2.2 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 (U.S. EPA, 2010d). It currently contains thousands of records of (mostly) filterable $PM_{2.5}$ EFs updated through calendar year 2004.[1]

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 estimates. EFs can be as simple as an average rate per unit process input. In most cases, EFs depend on many variables such as process parameters, effluent temperatures, ambient temperatures, wind speed, and soil moisture. In these cases the formula is applied to estimate emissions for a particular set of conditions. Under some circumstances in the inventorying of $PM_{2.5}$, EFs and estimation techniques are applied for analyses other than those for which they were developed. The accuracy and representativeness of the EFs are determined by the reliability of the testing methodology, how uniformly it is applied across sources, or the engineering process information used to derive the EFs.

EFs for some emissionS categories are more reliable than others. In some cases an EF may not be available for a source category because of insufficient or unacceptable data for generalization across source type. Often it is difficult to determine precisely what the certainty in the EF is. Thus, the application of EFs requires subjectivity and judgment from knowledgeable technical staff for the application of concern. As discussed in a previous chapter, users of EFs in national-, regional-, and urban- scale studies should be cognizant of their potential limitations, and other techniques should be considered to improve the confidence in PM emissions inventories. Several such approaches have been developed and some continue to be explored: continuous emission-monitoring sensors, material balances, specialized source profiling for composition and compositional material balances, source sampling to obtain improved particle-size distributions and location-specific emission rates, near-source ambient characterization, and source apportionment techniques. It is important to note that the reliability of EF estimation decreases when only a few source tests are used as the basis for the factor, or when judgmental decisions are made from analogy between technologies. Differences in EF estimates also can develop if the current operations or processes are significantly different from those upon which the original EFs were derived.

When most people think of PM, they envision solid particles that exist in the exhaust stream. However, $PM_{2.5}$ is composed of both a filterable fraction and a condensable fraction (see earlier discussion). The filterable fraction already exists in solid particle form in the exhaust. The condensable fraction exists in gaseous form in the exhaust stream but condenses rapidly in the ambient air to form tiny liquid droplets. Together, the filterable and condensable fractions are referred to as direct emissions of $PM_{2.5}$, and the summed number is what is reported in the NEI for all source categories. Most AP-42 EFs do not quantify the condensable fraction of total $PM_{2.5}$ emissions. "Gap filling" techniques are used to estimate condensable $PM_{2.5}$ for many stationary and area source categories. This introduces some uncertainties in the emissions estimates.

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[1] *http://cfpub.epa.gov/webfire*.

A2.2.3 Emissions Reduction Factors

The emissions-reduction factor (ER) in Equation 1 accounts for emissions controls employed on a source. For example, these include various effluent exit devices such as bag house filters and electrostatic precipitators for removal of PM. Like other process equipment, emissions controls have variable operating performance depending on their design, maintenance, and nature of the process controlled. Thus, like EFs, values of ERs are overall averages for specific processes and emissions-control designs based on limited testing. Actual values of ERs vary in time and by process in an undocumented manner, adding significant uncertainty to emissions estimates. Note that if no emissions controls are applied, the abatement efficiency equals zero (ER=0) and the emissions calculation is reduced to the product of activity and the emission factor, EF.

A2.2.4 Activity Levels

The last piece of information needed in equation 1 to estimate $PM_{2.5}$ emissions for sources is activity patterns. Activity patterns (AP) describe average temporal operating characteristics of a process, including estimates of the down time for maintenance or process failure. Values of AP for point and non-point sources are each obtained in different ways owing to the differing nature of the sources.

Most point sources or industrial sources operate with local permits, and these require information about process emissions, including temporal characteristics. For sources with CEMs for monitoring opacity (roughly proportional to fine PM loading), such as large utility boilers, real-time data are available to derive activity patterns, and deduce emissions variability over extended time periods. Further, point sources keep and report records of output during operating periods, and maintenance or other down times.

There is a great deal of complexity in acquiring activity data for nonpoint sources, which are diverse in character, individually small, and often intermittent, but collectively significant. Though such sources are difficult to characterize, they are generally important to PM emissions estimation because their aggregated mass emissions can be large and their chemical composition (e.g., BC) may be important for estimating source attribution. One good example of such a category is forest fires, burning of land-clearing debris, and agricultural burning.

Temporal resolution depends on the allocation of emissions aggregated seasonally, weekly, daily or by diurnal variation, depending on use and industry activity patterns. The temporal allocations allow for improved approximation of the actual temporal patterns that can be important not only for precise annual averaging using seasonal or daily allocations, but also for short-term impairment taken over periods of 24 hours or less. “Typical” temporal variations for different sources have been developed from surveys, activity analyses, and expert consensus. These temporal models are approximations that may deviate substantially from actual emissions in a given location. Depending on the requirements for precision in estimations, local testing through observations and activity data may be required, not only for large point or nonpoint sources, but for smaller ones that may be of special interest.

For nonpoint sources, emissions can be estimated coarsely from “top-down” measures of activity at the state or national level, including demographics, land use, and economic activity. The construction industry, for example, is based on the total annual expenditures at the regional level. These estimates are then allocated by county, using a procedure linked with construction costs and estimated area under construction. Because of their potential importance as PM sources, considerable effort has been devoted recently to the characterization of emissions and activity patterns for non-point sources. Another example is estimation of emissions from fires, which depends upon knowledge of the time, location, and areal extent of the burn, fuel loading, types of combustible material and moisture content. Recent efforts by EPA include the use of process modeling and remote sensing data to better estimate fire activity patterns and emissions from fires (Blue Sky Framework, 2010). Finally, residential wood burning is also an important local source of PM and BC. Quantification of emissions from this source category has been approached through acquisition of data on how much fuel is burned in fireplaces and woodstoves using national consumption estimates. Where this source is a large contributor to PM, local surveys of firewood use are used to supplement and improve activity level estimates. For all burning categories, the PM emissions reported via AP-42 contain both condensable and filterable emissions, so that the uncertainties involved with arriving at total $PM_{2.5}$ is less compared to other point and non-point sources.

A2.2.5 Estimating BC and OC Emissions

Next, these $PM_{2.5}$ emissions can be converted to BC and OC by proper application of speciation factors from the SPECIATE database. (See Appendix 1 for details on SPECIATE.) The equation used is quite simple: $PM_{2.5}$ Emissions (in tons) * fraction of $PM_{2.5}$ that is BC = BC emissions. This can be difficult given that there are thousands of $PM_{2.5}$ source categories but only a limited number of speciation profiles. Therefore, special attention must be given in mapping specific profiles to source categories. These details are explained in Reff et al. (2009). Application of these methods to the inventories results in the 90 source categories for which BC and OC emissions are reported in Chapter 4.

While the process for compiling BC emissions inventories is reasonably straightforward, there are important limitations in this process that introduce uncertainties in final BC emissions estimates. These include:

- The reliability of the $PM_{2.5}$ emission factors used in Equation 1. Some emission factors for point and non-point sources are more reliable than others (NARSTO, 2005).

- The reliability of condensable PM estimates by source category. Some sources include PM condensables as part of their testing protocol (fires, residential wood combustion). Others do not, and a generic emission factor (via AP-42) is applied to estimate the amount of condensable PM each source emits. This introduces a level of uncertainty in determining final BC emissions that is not currently accounted for. The source measurements section of this report gives a clearer indication of what the issues are and how they can be improved.

- The reliability of activity levels used in Equation 1. Some activity levels are generated using process models (Blue Sky Framework, 2010), while some are generated using surrogate information (U.S. EPA, 2005a).

- Finally, many “augmentations” are done in the emissions inventory processing steps. These augmentations include scaling measured PM to $PM_{2.5}$ as well as assigning condensable emissions estimates to point and nonpoint sources that are not available via source testing. Some of the impacts of the uncertainties in doing this have been explored (NARSTO, 2005), but the issue has not been dealt with holistically.

A2.2.6 Mobile Sources

In the U.S. inventory, mobile sources consist of the following general categories of vehicles and engines:

- *On-road gasoline*, such as passenger cars and light-duty trucks

- *On-road diesel*, including light-duty passenger cars, light-duty trucks, and heavy-duty trucks. Unlike in Europe, very few diesel passenger cars are sold in the United States, making heavy-duty diesel trucks the dominant vehicle type in this category.

- *Nonroad diesel*, including construction, agricultural, and other equipment

- *Nonroad gasoline*, including both 2-stroke and 4-stroke cycle engines such as those used in lawn/garden equipment and recreational marine

- *Commercial marine*, classified by engine displacement as categories C1, C2, and C3 (ocean-going)

- *Locomotives*

- *Aircraft*, which are generally turbine aircraft rather than the smaller piston gasoline-powered aircraft

BC emissions from on-road vehicles, both gasoline and diesel, are now calculated directly using EPA’s new MOVES 2010 model. For other mobile source categories, BC emissions are calculated using methods similar to those described above for stationary sources. EPA has released a number of technical reports on MOVES. These give the structure of the model including fleet and activity data such as vehicle miles traveled, the default national vehicle population, and vehicle activity patterns.[2] They also give information on exhaust emission rates and deterioration by model year for light-duty and heavy-duty vehicles.[3] Similar information is available on the EPA NONROAD

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[2] EPA (November 2010) Report EPA-420-R-10-026, MOVES 2010 Highway Vehicle Population and Activity Data, *http://www.epa.gov/otaq/models/moves/420r10026.pdf*.

[3] EPA (April 2001) Report 420-R-01-007 EPA’s New Generation Mobile Source Emissions Model: Initial Proposal and issues. MOVES, *http://www.epa.gov/otaq/models/moves/movesback.htm*.

EPA (February 2003) Report 420-R-03-005 Investigation for the Physical Emission Rate Estimator to be Used in MOVES, *http://www.epa.gov/otaq/models/moves/movesback.htm*.

model for things like equipment population, emission factors, and engine turn over.[4]

The inventories given below for the United States include all 50 states. They also include all marine activity (both domestic and foreign) within 200 nautical miles of shore.

A2.2.6.1 On-road Gasoline and Diesel

For on-road gasoline and diesel vehicles, EPA's emissions models directly calculate both total $PM_{2.5}$ emissions and BC emissions. Recent improvements in EPA's new MOVES 2010 model (U.S. EPA, 2010e) as compared to the earlier MOBILE6.2 model (U.S. EPA, 2003) include accounting for high emitters, deterioration of PM emissions (i.e., increase in PM mass) with higher mileage, and increased PM emissions at lower temperatures.[5] This model directly calculates BC emissions (as well as other exhaust PM components such as sulfates and OC), and accounts for the significantly reduced BC fraction emitted from on-road diesels due to application of diesel particulate filters (DPFs) (required for heavy-duty diesel trucks up to 80,000 pounds GVW beginning with the 2007 model year). An important input for the gasoline vehicle $PM_{2.5}$ portion of the MOVES model is a recent study examining PM emissions from about 500 in-use vehicles (Coordinating Research Council, 2008). MOVES accounts for the lower EC/PM fraction (about 10%) for diesels with DPFs. Several studies such as the Health Effects Institute Coordinating Research Council study (Khalek et al., 2009) have evaluated the EC/PM fraction of heavy-duty diesel engines with DPFs finding a fraction of about 10% versus the more typical 70-80%. EPA plans to perform an uncertainty analysis with MOVES, considering the various inputs and what the range of uncertainty might be, and what the overall uncertainty might be.

Gasoline OC and BC emissions increase dramatically at lower ambient temperatures. To calculate this increase for gasoline vehicles, we used calculations done for EPA rulemaking packages for gasoline PM, for which an hourly grid-cell temperature adjustment was done as part of emissions processing at the county level for each of the over 3,200 counties. As a general rule, diesel PM emissions are less sensitive to temperature for a variety of reasons (lower importance of cold start since many diesels trucks do not operate on short trips; easier engine warm up since older diesels do not have catalysts which take a finite time to warm up during which emissions are higher). This means that BC emissions from diesel vehicles are not projected to increase as much at lower temperatures as would be the case with gasoline vehicles.

MOVES can also be used to calculate tire and brake wear $PM_{2.5}$, with speciation factors applied to calculate BC. Only a small fraction of the PM from tire and brake wear is in the $PM_{2.5}$ range, so estimated BC emissions from these categories are fairly small. However, a large fraction of tire wear PM (about 22%) is BC. In the U.S. inventories reported in Chapters 4 and 8, these detailed calculations at the county level were done for 2005 and projection years (2020, 2030) along with some less detailed calculations (at the national level) for 1990. One important thing to note is that the PM, BC, and OC are relatively high from the on-road gasoline vehicle fleet for 1990 due to the presence of a large number of non-catalyst vehicles still remaining in the fleet.

A2.2.6.2 Nonroad Gasoline and Diesel

For nonroad engines (both gasoline and diesel powered), EPA calculates BC emissions based on PM emissions estimates from the NONROAD model (U.S. EPA, 2008b). Also, the National Mobile Inventory Model (NMIM) uses the current version of the NONROAD model (NONROAD2008) to calculate emissions inventories. The model incorporates emission factors (in g/BHP-hr – that is, grams per brake horsepower-hour), engine output (BHP-hr), and usage data for a wide number of NONROAD sources. For gasoline engines, 2-stroke cycle engines are a separate category from 4-stroke cycle engines. These engines have lubricating oil mixed with the fuel so the exhaust VOC (and PM) will be markedly different from that for the more standard 4-stroke cycle engines. VOC from 2-stroke

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[4] EPA (December 2005) Report EPA 420-R-05-016 Exhaust Emission Effects of Fuel Sulfur and Oxygen on Gasoline Nonroad Engines, *http://www.epa.gov/oms/nonrdmdl.htm#model*.

EPA (July 2010) Report EPA-420R-10-017 Nonroad Engine Population Estimates, *http://www.epa.gov/oms/nonrdmdl.htm#model*.

EPA (December 2005) EPA Report 420-R-05-017 Seasonal and Monthly Activity Allocation Fractions for Nonroad Engine Emissions Modeling, *http://www.epa.gov/oms/nonrdmdl.htm#model*.

EPA (July 2010) EP{A Report 420-R-10-015 Median Life, Annual Activity, and Load Factor Values for Nonroad Engine Emissions Modeling, *http://www.epa.gov/oms/nonrdmdl.htm#model*.

EPA (December 2005) EPA Report 420-R-05-018 Calculation of Age Distributions in the Nonroad Model: Growth and Scrappage, *http://www.epa.gov/oms/nonrdmdl.htm#model*.

EPA (July 2010) EPA Report 420-R-10-018 Exhaust and Crankcase Emission Factors for Nonroad Engine Modeling – Compression Ignition, *http://www.epa.gov/oms/nonrdmdl.htm#model*.

EPA (December 2005) EPA Report 420-R-05-021 Geographic Allocation of Nonroad Engine Population Data to the State and County Level, *http://www.epa.gov/oms/nonrdmdl.htm#model*.

[5] 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 (U.S. EPA, 2010e).

cycle engines will contain higher molecular weight compounds from the oil mixed with the gasoline; oil compounds are typically in the C20-C37 range. For these engines, the profile used to derive the emissions estimates in Chapter 4 is that used for non-catalyst equipped gasoline-powered motor vehicles since these nonroad gasoline engines do not have catalysts. This profile (92049) comes from the EPA SPECIATE database and shows 10% of the PM being BC. Admittedly, data specific for nonroad gasoline engines, especially 2-stroke engines with their oil combustion, are needed. EPA plans to add NONROAD into MOVES at some point in the future. Also, at some point possibly beyond that, EPA would perform an uncertainty analysis on the nonroad inventories.

BC emissions are then calculated by using speciation factors denoting the percent of PM emissions represented by BC. A speciation factor for nonroad diesel engines not equipped with DPFs comes from EPA’s SPECIATE database (U.S. EPA, 2008a). The profile used to derive the emissions estimates in Chapter 4 (Profile 92035) is actually derived from heavy-duty on-road diesels and has 77% of the PM being BC. Beginning in calendar year 2012, many if not most newly manufactured nonroad diesels for that “model year” will be equipped with DPFs. This technology reduces exhaust PM mass by over 90%, and the small amount of PM remaining has relatively little BC. In effect, DPFs preferentially reduce BC. Roughly 10% of the PM from a diesel with a DPF consists of BC based on a large emissions characterization program on four 2007 model year on-road diesel truck engines equipped with DPFs. The testing was done by Southwest Research Institute for this program conducted by the Coordinating Research Council (Khalek et al., 2009).

A critical factor in compiling BC inventories for nonroad diesels is to correctly apportion the BC emissions between pre-trap-equipped diesel engines and trap-equipped diesels in any given calendar year. The NONROAD model correctly calculates the combined PM mass in a given calendar year accounting for pre-trap and trap-equipped diesels. Though it does not presently calculate BC emissions separately, a later version of the model under development will do so. Meanwhile, when NONROAD is run, one can get a model year emissions output for specific calendar years. One can then probably manually take that model year input and apply the higher BC speciation percent (77%) to the pre-trap equipped engines and the lower percentage (roughly 10%) to the new diesel engines equipped with DPFs. The inventory numbers presented account for this difference. For nonroad gasoline, a speciation profile of 10% of the PM being BC is used based on tests on older non-catalyst light-duty vehicles. Most nonroad engines do not have catalysts. Since almost no or limited PM speciation has been obtained for the exhaust of these engines, the most appropriate factor to apply is based on older non-catalyst vehicles (of the types produced before introduction of catalysts with the 1975 model year). It is also important to note, however, that 2-stroke cycle engine production will be changed with the advent of new EPA emission standards.

A2.2.6.3 Commercial Marine, Locomotive, and Aircraft

Commercial marine, locomotive, and aircraft emissions are calculated separately in spreadsheet models, with separate BC speciation factors for C1/C2 commercial marine and C3 commercial marine (the larger ocean-going vessels). For the smaller vessels, the profile for nonroad diesel engines is applied even though the higher sulfur content of the fuel will lead to the PM containing higher sulfate emissions than for nonroad diesels. DPFs will be required for these vessels starting in 2014, reducing the BC fraction to about 10% of the PM. However, DPFs will only be used on some engine classes, and implementation dates will vary (depending on factors such an engine size). Thus, there is a need for a model to correctly account for the implementation of these standards. For now, a model year break-out of PM emissions was done for both 2020 and 2030. Separate BC/PM speciation factors were applied to the PM emissions from the diesels with and without DPFs. Currently, the diesel BC speciation factor of 77% BC/PM is used for C1/C2 commercial marine for all years of analysis: 2005 as well as the non-DPF equipped engines in 2020 and 2030. Evidence from recent studies (Lack et al., 2009) suggests that a lower BC speciation factor may be more appropriate for C1/C2 marine.

PM emissions from C3 Marine have substantially different PM speciation profiles than smaller diesel engines used in C1/C2 Marine and on-road and nonroad diesel. C3 marine diesels burn a high molecular weight residual oil that contains very high sulfur levels (up to 45,000 ppm versus the 15 ppm in on-road and nonroad diesel fuel). Past EPA evaluations of C3 marine have used the EPA PM SPECIATE profile of Residual Oil Combustion (U.S. EPA, 2008c), which estimates a 1% BC speciation factor.

For this report, an updated BC speciation profile was estimated from studies available in the literature. Results from relevant studies that measured BC

Table A2-1. Summary of Recent Studies that Measured BC and PM Emission Rates from C3 Marine. (Source: U.S. EPA)

Study	Vessel	Fuel	Fuel Sulfur Content	BC/PM[a]
Murphy et al. (2009)	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)[b]	Medium Speed Diesel Engine	Heavy Fuel Oil	22,100 ppm	2.63%
Lack et al. (2009)	Slow-speed diesel vessels[c]	Variety	Variety[d]	7.33%
Lack et al. (2009)	Medium-speed diesel vessels[e]	Variety	Variety[f]	28.00%

[a] Lack et al. (2009) measured BC not EC with Light-Absorption Measurements (photoacoustic aerosol spectrometer with adsorption at 532 nm), other studies measured EC using thermal-optical methods, but are referred to as BC for comparison.

[b] Engine test, with engine load 85-110%.

[c] PM measurements come from 29 SSD ships; BC emissions come from 52 SSD ships.

[d] Mostly high sulfur fuel (>5,000 ppm).

[e] PM measurements come from 12 vessels; BC emissions come from 51 vessels.

[f] Mostly low sulfur fuel (<5,000 ppm).

Table A2-2. Summary of Speciation Ratios of PM from Relevant Marine Studies. (Source: U.S. EPA)

Reference	BC	EC	OC or OM	BC Measurement	EC/OC Measurement
Agrawal et al. (2008)		0.5%	11%		Thermal optical transmittance (TOT) NIOSH 5040
Murphy et al. (2008)		0.3%	7.4%		
Petzhold et al. (2010)	0.85%	2.6%	21%	Multi-Angle Absorption Photometer, MAAP	Thermal-Optical Method,VDI 2465 Part 2
(SSD) Lack et al. (2009)	7.3%		23%	Photoacoustic Aerosol Spectrometer with adsorption at 532 nm	Organic Matter (OM) calculated by subtracting SO$_4$,NO$_3$, and NH$_4$ from aerosol mass spectrometer (AMS) measurements
(MSD) Lack et al. (2009)	28%		16%		
(>5,000 ppm) Lack et al. (2009)	7.1%		19%		
(<5,000 ppm) Lack et al. (2009)	51%		35%		

and PM emission rates from marine sources are summarized in Table A2-1.

As noted in Table A2-1, there is substantial variation in the reported BC/PM emission profiles from these studies. Table A2-2 displays the BC, EC, and OC speciation rates from the relevant marine studies. Large variations between BC (light-absorption measurements) and EC (thermal-optical measurements) are observed for C3 Marine emissions. Petzold et al. (2010) noted differences in the EC and BC emissions up to a factor of 3 for a medium speed diesel engine measured in the laboratory. The discrepancies among different BC emission factors for marine sources are additionally noted in the literature (Petzold et al., 2010). Considering the uncertainty of the values, EPA selected a BC/PM speciation factor of 3% which falls in the middle of the range of reported values. Alternatively, one can also consider which study has vessels that are most representative of the C3 marine population or even a population-weighted average. However, the differences in the measurement techniques and definitions of BC in the cited C3 Marine studies make it difficult to combine the data across studies. EPA recognizes that this is an area of active research and recommends further work be conducted.

In Table A2-2, the results from the Lack et al. (2009) study are subdivided according to vessel type: slow-speed diesel (SSD) and medium speed diesel (MSD). Lack et al. (2009) also grouped the BC emission observations according to fuel sulfur content. Fifty-one ship observations had low sulfur fuel content (<5,000 ppm) and 42 ship observations of vessels had fuel sulfur content greater than 5,000 ppm. From the available data, BC and OC speciation factors were calculated for each of the subcategories. The ships in the Lack et al. (2009) study with low sulfur content had much lower sulfate speciation factors and higher BC speciation factors than the other studies.

To estimate the BC/PM factor for future years (2020 and 2030), the international fuel sulfur limits were considered (Table A4-2) as well as the speciation rates from the studies evaluated. Lowering the fuel sulfur content is an effective method to reduce the particulate sulfate, which comprises the majority of the PM from marine vessels using heavy fuel oil. Due to the substantial drop in fuel sulfur levels, the BC speciation factor should rise in 2020 and 2030. Due to limited data, the EPA chose a C3 marine BC speciation factor of 6% for 2020 and 2030. For now, EPA is choosing 11% as an OC/PM speciation fraction for 1990 and 2005 with a higher fraction (58.6%) for 2020 and 2030 when fuel sulfur reductions occur, especially in ECA areas.

Various inventories have also been prepared for C3 marine using fuel consumption, emissions, and vessel activity (Paxian et al., 2010).

For locomotives, as for C1/C2 marine, the HDDV on-road profile (77% BC) is presently being used for pre-2014 engines although available data suggest this number might be too high. DPFs will be used in subsequent years, reducing BC to about 10%. For 2020 and 2030, the PM model outputs are obtained by calendar year and for the years when the standards take effect, the 10% number is used.

For purposes of emissions inventory estimates, aircraft operations are often broken into two basic portions. The first portion, landing and take-off (LTO) cycle, is normally defined to include aircraft ground operations (taxi/idle) as well as aircraft operations below 3,000 feet elevation in the local airport terminal area. The second portion is referred to as non-LTO that includes climb (above 3,000 feet) to cruise altitude and descent from cruise to 3,000 feet. Together these portions comprise what is called "full-flight" emissions.

Emissions for the LTO portion are fairly well characterized. Engine emission rates are measured in jet engine test cells during FAA certification testing; it is believed that these measurements reasonably predict engine emissions rates for aircraft in actual LTO operations. Programs for evaluating and controlling LTO emissions have been in place in the United States for about thirty years. Today there are LTO engine emissions standards for hydrocarbons, carbon monoxide, oxides of nitrogen and smoke number. While work is now underway to develop a sampling and measurement procedure and certification requirement for aircraft jet turbine engine PM emissions, there are not yet specific engine emission standards for PM. To address this shortfall on at least an interim basis, FAA, working with EPA, industry and academic experts developed a methodology to estimate LTO PM emissions. This methodology, known as the "First Order Approximation" (FOA), uses information on smoke number and other engine and fuel parameters to estimate LTO PM emission rates for each engine model (Kinsey and Wayson, 2007). This information is then matched with airframe information on number of engines to get a per LTO emission rate for each aircraft type. Using the airport specific information and the aircraft activity for each airframe/ engine combination, the LTO PM Inventory estimates are made. The total PM emission rate includes all types of compounds contributing to the PM mass. It is estimated that only about 13% of the PM mass is BC; the remainder is composed of sulfates and organics. The average BC PM emission index (EI) is in the range of 0.04-0.05 g/kg fuel burned for the LTO portion.

The estimation of non-LTO BC emissions depends on a very limited set of measurements. Emissions testing in jet engine test cells does not fully characterize PM emission rates at altitude because they are conducted at sea level static conditions and have to be carefully extrapolated to altitude conditions, due to the differences in the atmospheric environment and engine operating conditions outside of the LTO—including cruise altitudes. Although there are research models available to estimate non-LTO BC, there is not yet a consensus approach for estimating non-LTO PM emissions as exists for LTO PM emissions. This is an area of ongoing research within the scientific and technical aviation communities.

However, two important points should be recognized with regard to non-LTO PM BC emissions. First, results from FAA's model entitled "System for Assessing Aviation's Global Emissions" (SAGE) indicate that total fuel burn during non-LTO operations is about ten times that during the LTO.[6] Since the PM emission inventory is linked to fuel burn, overall PM emissions during the non-LTO portion of the "full flight" would be expected to be larger than those during the LTO portion. Second, this is an area of ongoing research and to-date there are no less than six researchers who have used various methods to estimate the EI for PM BC

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[6] The FAA SAGE website is *http://www.faa.gov/about/office_org/headquarters_offices/apl/research/models/sage*.

Table A2-3. Estimates of Aircraft BC Emissions. (Source: U.S. EPA)

Reference	Aircraft	Engine(s)	Measurement Condition	BC EI (g/kg fuel)
Kinsey et al. (2011)	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 et al. (1999a)	ATTAS	Rolls-Royce/SNECMA M45HMk501	Non-LTO thrust levels at SLS In-flight	0.118-0.149 0.11-0.15
Pueschel et al. (1997)	Concorde	Olympus 593	In-flight 16300 m altitude	0.07-0.11
Petzold et al. (1999b)	B737-300	CFM56-3B1	In-flight 7925 m altitude	0.01
Petzold et al. (1999b)	A310-300	CF6-80C2A2	In-flight 10670 m altitude	0.021
Petzold et al. (1999b)	VFW 614	M45H	In-flight 7925 m altitude	0.07-0.11
Lilenfield et al. (1995)	DC8	GE 404	All thrust levels at SLS	0.03-0.4
Anderson et al. (1998)	Multiple	Multiple	In-flight; mass EIs estimated from number EIs based on average particle volume and mass	0.01-0.35
Dopelheuer (2001)	n/a	CF6-50C2	Modeled Cruise Simulation - DLR Method using empirical calculation	0.015

emissions during the non-LTO portion of "full flight" (Lilenfield et al., 1995; Pueschel et al., 1997; Anderson et al., 1998; Petzold et al., 1999a; 1999b; Dopelheuer, 2001; Kinsey et al., 2011). Some researchers have used equivalent non-LTO thrust levels on the ground while the estimates of others were based on in-situ plume measurements of aircraft in flight. It is difficult to make direct comparisons among these values or to use this data to derive a point estimate for the non-LTO BC EI since they were developed on different airframe/engine models of different technology vintages using different measurement approaches. While data from the published researchers ranges from about 0.01-0.35 g/kg fuel burned, the majority of the data lies in the range of about 0.02-0.11 g/kg fuel burned. Each study has its relative strengths and weaknesses and most of the older engines with higher EIs are no longer in service. Table A2-3 summarizes the publicly available literature on this issue.

A2.3 Development of International Emissions Inventories for Black and Organic Carbon

There are a number of methodological differences between the approaches used to compile domestic and international inventories. Specifically, in contrast to EPA's method of using emission factors paired with activity levels to estimate BC and OC emissions, the most widely used global emissions inventory described in Chapter 4 (Bond et al., 2004) incorporates other factors to derive estimated BC emissions, including fuel type, combustion source technology type, and emissions controls. There is extensive usage data on emissions from specific vehicles and engines in the United States which is used for input for EPA models. These data exists to a lesser extent outside the United States. Fuel consumption data, which are often used as a substitute in global inventory calculations, are also useful but do not have the detail that vehicle/engine usage data have.

Like the U.S. inventory, global inventories typically rely on Equation 1 (outlined above), for estimating BC and OC emissions. That is, country-specific emission factors are combined with appropriate activity level information to yield an estimate of emissions. For example, on-road cars, trucks, buses, and all on-road mobile sources are generally assessed through travel-based emission factors and vehicle miles traveled (VMT). This approach associates mobile source emissions with traffic patterns, providing spatial and temporal information about the distribution of emissions that can be used in a variety of applications (for example, air quality modeling). However, motor vehicle emission factors are highly variable and uncertain because of different vehicle types, ages, maintenance, and operating conditions (Cadle et al., 2006). Fuel composition data often can be obtained more easily and accurately than activity measures such as VMT. Fuel-based emission factors for fossil fuel and bio-fuel combustion, for example, can be derived easily from diluted in-plume measurements, using simultaneous CO_2 measurements to determine dilution ratios and to relate other pollutants to the weight of carbon in the consumed fuel (Chow et al., 2010c). Fuel-based emission factors are very common in global and large-scale inventories where detailed information on source activity is very limited.

Global BC and OC inventories are complicated by a lack of specific, detailed information on source types, emission factors, activities, and controls, especially in the developing world. In such cases a simple equation for calculating emissions based solely on emission factors and activity levels cannot be applied rigorously. Therefore, certain proxies have been used to estimate BC and OC emissions.

A2.3.1 Specific Approaches Used in Global Inventories

Most global inventories attempt to estimate BC emissions, even though some of the emission factors used seemingly represent testing on EC. There are important differences in the way various global estimates were generated, resulting in some variation in the total emissions estimates generated in different studies. It is useful to compare the approaches in more detail.

The Bond inventories, which are the most extensively used global inventories, were characterized in Chapter 4. Bond et al. (2004) identified about 50 different combinations of fuel type and usage and subdivided these into processes with different emissions characteristics. This approach is based on combining fuel composition data and assumptions of combustion technologies and emissions controls, and is very similar to earlier work done in the literature (Klimont et al., 2002). Emissions for a fuel/sector combination are calculated as an aggregate of the contributions of all technologies within that sector. The total emissions for each country, in turn, are the sum across all fuel/sector combinations. The reader is referred to published literature for more details on the methods used and the uncertainties inherent in their methodology (Bond et al., 2004; 2007). An overview of the Bond estimation procedure is given in Figure A2-1. Using this method, global BC emissions were estimated at about 8.9 million tons per year, with an uncertainty range of 4.8 -24 million tons/year. The United States accounted for about 6% of global BC emissions in this inventory.

Other authors have compiled inventories based on alternative methods. Penner et al. (1993) looked at developing BC emission in two ways: first, based on fuel consumption estimates, and second based on BC/SO_2 ratios. In examining the relationship between ambient BC and SO_2 concentrations in urban areas around the world, the authors found strong correlations in source areas and also that various sources had characteristic BC/SO_2 ratios. Site-specific BC/SO_2 ratios were transformed to BC emissions using available SO_2 emissions estimates for each country/world region. The result was a global BC emissions estimate of about 26 million tons per year from urban fuel use. Penner et al. (1993) also calculated global BC emissions on the basis of fuel consumption, assuming constant emission factors for commercial and domestic coal, diesel fuel, wood, and bagasse combustion, yielding a total of 14 million tons BC/yr, with 7.3 million tons/year from fossil fuel combustion and 6.7 million tons/year from wood and bagasse fuel burning. Even though global estimates from the two methods differed by a factor of about 2, larger differences were found for individual countries and regions.

Cooke and Wilson (1996) compiled published estimates for biomass areal density, amounts above ground, the fractions burned, and emission factors for different fuel types (e.g., forests, savanna). Agricultural burning and biomass combustion for heating and cooking were not included. Country-specific fuel consumption rates were compiled for industrial, domestic, and combined sectors for solid, liquid, and gaseous fuels. Country-wide

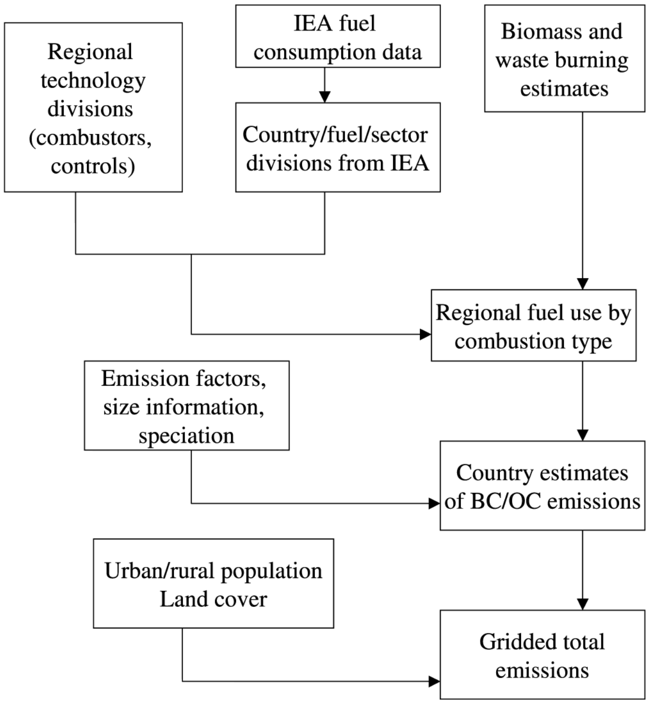

Figure A2-1. Bond et al. Methodology for Developing Emissions Estimates. (Source: Bond et al., 2004) emissions were distributed to grids according to population density. The Cooke and Wilson global BC inventory of about 15 million tons was comprised of 9 million tons and 6 million tons from fossil fuel and biomass combustion, respectively. The fossil fuel component of 9 million tons was approximately one-third that estimated by Penner using the BC/SO_2 ratio approach (i.e. 26 million tons), but similar to emissions based on Penner's fuel consumption approach (i.e., 7.3 million tons from fossil fuel combustion).

Liousse et al. (1996) reported global BC and OM (organic mass) emissions. OM is divided by 1.3 and converted to measured elements in organic molecules. Their inventory includes categories for biomass (i.e., savanna and forest fires), agricultural waste, wood fuel, and dung combustion as well as domestic coal and diesel fuel combustion. Global fossil fuel combustion (7.3 million tons BC /year) and biomass burning (6.2 million tons EC /year) total 13.5 million tons EC/year, lower than the estimate of 15 million tons BC/year from Cooke and Wilson that excluded agricultural burning and biomass combustion for fuel and energy production. Louisse et al. (1996) also estimated global OC emissions of 69 million tons/year, with 24 million tons OC/year from fossil fuel and 45 million tons/year from biomass burning.

Then, Cooke et al. (1999) refined Cooke and Wilson (1996) by considering the relative ages of vehicles in developed and developing countries and particle size differences for controlled and uncontrolled combustion processes. The estimated global BC inventory from fossil fuel combustion of 7 million tons in Cooke et al. (1999) was consistent with estimates of 9 million tons BC/year by Cooke and Wilson (1996) and 7.3 million tons BC/year by Louisse et al. (1996). The Cooke et al. (1999) estimate of global OC emissions was about 11 million tons, about half that of Louisse et al. (1996) (24 million tons OC/year).

Cofala et al. (2007) have used a global version of the Regional Air Pollution Information and Simulation (RAINS) model to estimate anthropogenic emissions of BC and OC (along with numerous other pollutants). The authors rely on the RAINS methodology (Klimont et al., 2002) for particle emissions, which they modify to capture regional and country-specific characteristics of BC and OC emissions as laid out in other references (Kupiainen and Klimont, 2007) and extend it to developing regions with data from Bond et al. (2004). Their methods result in an estimate of 6 million tons of BC emitted globally in 1995 and about 5.9 million tons of BC emitted in 2000.

A2.3.2 Specific Approaches Used in Regional Inventories

Chapter 4 also discusses the information available from alternative inventories available for specific countries or world regions. Table A2-4 provides a comparison of key differences in data and methods between some of these inventories.

A comparison among the regional inventories listed in Table A2-4 yields some interesting information. It is also helpful to compare the emissions estimates from these inventories with estimates in the appropriate portion(s) of the Bond/Streets global inventory. Specifically:

- Cao et al. (2006) used emission factors from Cooke's 1999 global inventory and Streets' inventory of China BC emissions for the year 2000, along with Andreae and Merlet (2001) emission factors for biofuels. They developed a local inventory based on specific emission factors for crop straw used in cooking stoves by testing five different types of straw in a combustion tower designed to simulate Chinese cooking stoves. Their national BC emissions estimate for 2000 in China is about 1.7 tons, which is somewhat higher than the Bond estimate of 1.6 million tons and the Streets estimate of 1.2 million tons. The authors attributed this to their inclusion of coal combustion in rural industry and rural residential sources, which they noted are often underestimated in more global estimates. They outlined the residential and industrial sectors as being the most important in contributing to Chinese BC emissions.
- Streets et al. (2001) measured Chinese BC emissions for 1995, using mostly emission factors from other literature sources. Their inventory focuses on submicron BC emissions rather than bulk emissions, because submicron emissions are more relevant to radiative transfer calculations. Their study noted also that most inventories assume that the fraction of BC that makes up $PM_{2.5}$ remains constant throughout the combustion process. Streets et al. (2001) states that smoldering combustion, while releasing a higher amount of particulate matter, does not exhibit temperatures high enough to produce the same proportion of BC. They propose a negative correlation between particulate emissions and the fraction of BC emitted. Other observations by this study were that removal efficiency of particles for the industrial sector is lower and less documented than that of the power sector, making emissions from the Chinese industrial sector more uncertain and variable, and that

Table A2-4. Regional Inventories of BC and OC Emissions. (Source: U.S. EPA)

Carbon Emission Inventory Reference	Region, Base Year (Resolution, if available)	Emission Source Categories	Source of Emission Factors	Source(s) of Activity Information[a]
Reddy and Venkataraman (2002b, Fossil Fuel) Reddy and Venkataraman (2002a, Biomass Combustion) Black Carbon and Organic Carbon	India, 1996-1997 (.25 x .25)	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, Cement Manufacturers' Association, Centre for Monitoring of India Economy, The Fertilizer Association of India, Ministry of Coal, Ministry of Industry, and Ministry of Petroleum and Natural Gas, Statistics for Iron and Steel Industry in India. Biofuel consumption in rural and urban from Tata Research Institute and National Sample Survey and Forest coverage from the Forest Survey of India.
Streets et al. (2003) Black Carbon Only	Asia, 2000 (1 x 1 to .08 x .08)	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 compiled on a provincial basis. For the transportation sector, used World Road Statistics and World Motor Vehicle Data.
Cao et al. (2006) Black Carbon and Organic Carbon	China, 2000 (.2 x .2)	Includes 5 sectors (i.e. power generation, industry, residential, transportation, and 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.	Literature review Laboratory tests of biofuel emissions from cooking stoves	Point source activity from State Power Corporation of China and Editorial Board of China Rural Energy Yearbook. Area sources activity from National Bureau of Statistics and various government agencies, mainly at the county level.
Streets et al. (2001) Black Carbon	China, 1995	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 (Hordijk et al., 1995). Generated by China's Energy Research Institute.
Parashar et al. (2005) Black Carbon and Organic Carbon	India, 1995	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 (2002b).
Sahu et al. (2008) Black Carbon	India, 2001 (1 x 1)	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, 1991 and 2001	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.

Carbon Emission Inventory Reference	Region, Base Year (Resolution, if available)	Emission Source Categories	Source of Emission Factors	Source(s) of Activity Information[a]
Lamarque et al. (2010) Black Carbon and Organic Carbon	Global, 2000 (.5 x .5)	12 different sectors over 40 different regions	Literature Review (Junker and Liousse, 2008 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, 1996	7 different fuel types	Literature Review (from Cooke and Wilson, 1996)	
Mayol-Bracero et al. (2002) Black Carbon	India, 2000	8 different fuel types across four (five?) sectors	Literature Review (Streets et al., 2001))	RAINS-Asia model.
Ohara et al. (2007) Black Carbon and Organic Carbon	Asia, 2000 (.5 x .5)	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 RAINS-Asia.
Derwent et al. (2001)	Western Europe,1995-1998	All sectors	Back calculated using dispersion modeling and ambient data	NA

[a] Unless otherwise noted, refer to main Emission Inventory Reference for a list of specific publications and databases from which underlying data were drawn.

domestic emissions in China are responsible for over 80% of Chinese BC emissions. The final estimate of BC emissions was about 1.5 million tons, which is lower than the Bond estimate of 1.6 tons but higher than emissions estimated via other regional Asian studies REAS and TRACE-P (1.3 million tons and 1.0 million tons, respectively).

- Reddy and Venkataraman (2002a; 2002b) estimated BC emissions in India for the year 1996 by developing emission factors using Indian fuel composition and indigenous pollution control technology. Emission factors of coal for the power and industrial sector were derived from those of the EPA. Domestic emission factors were taken from Indian literature sources (Gray, 1986). Transportation factors were taken from an average of PM emission factors from countries with similar transportation statistics. The study noted that most of the BC emissions from India were from the transportation sector (almost 60%). Overall they predicted BC emissions from 1996–1997 to be 0.36 million tons annually. This estimate is lower than almost all other estimates of Indian BC emissions. The authors claimed their lower estimate was due to different emission factors – other emission factors used in more global inventories were too high due to improper differentiation of fuel composition, combustion type, and $PM_{2.5}$ composition differences.

- Parashar et al. (2005) estimated Indian BC emissions for 1995 and used Bond emission factors for fossil fuel combustion. Biomass combustion emission factors were determined by actually combusting different types of fuels in a U shaped chimney. They found that dung cakes released a particularly high amount of particles due to a smoldering combustion, which releases more particles than other types of burning. Their final emissions estimate for India was 0.92 million tons BC, which is higher than Bond's estimate of 0.64 million tons of BC. The higher emissions could be due to the higher emission factor they found for dung cakes, which accounted for a higher than proportional amount of domestic emissions.

- Mitra et al. (2002) estimated Indian BC emissions for the year 1996 using Cooke's 1999 emission factors for "under developed" countries. They only calculated emissions for four fossil fuel categories: coal, diesel, gasoline, kerosene. Their annual emission estimate was approximately

1.1 million tons of BC. This is much higher than Bond's estimate of 0.63 million tons BC in India. The higher emissions could be due to the use of emission factors for under developed countries, in that Bond et al. in their estimates may have used emission factors from more developed countries to represent fossil fuel combustion characteristics in India.

- Sahu et al. (2008) estimated BC emissions from India for the years 1991 and 2001. They used Cooke's 1999 emission factors for "under developed countries" for fossil fuel combustion and Reddy's regionalized 2002 emission factors for biofuel combustion. Their final estimate of BC emission in India was about 1.5 million tons per year, higher than any other inventory despite the fact that they did not inventory small industry. Their high estimates could be due to using under-developed country emission factors for all fossil fuel combustion sectors and also from use of diesel activity information which did not represent current conditions.

- Mayol-Bracero et al. (2002) calculated BC emissions in India using the Chinese emission factors developed by Streets et al. (2001) and national activity data from the GAINS model. Their final estimate of BC emissions in India (2000) was 0.5 million tons, slightly lower than Bond's estimate of about 0.6 million tons.

- Ohara et al. (2007) developed REAS (Regional Emission Inventory for many parts of Asia) for several pollutants including black carbon for the period 1980 – 2003. Emissions were calculated as a product of activity data, emission factors, and removal efficiency of controls. BC emission factors were taken from Streets et al. (2003a), and characterized into developed countries and countries with no known emission controls (this category included India and China). The emission factors for developed countries changed several times over the time period of the inventory. Chinese BC emissions in 2000 were 1.2 million tons and Indian emissions were estimated to be about 0.9 million tons. These numbers compare fairly well to other estimates for those countries. The inventory noted the domestic sector as the main contributor to BC emissions.

- Dickerson et al. (2002) used two different approaches to measuring BC emissions in India and other South Asian countries. They first did a bottom up inventory using emission factors and activity level data. They assumed that South Asia source types of BC were similar to those of China, and they obtained energy use information from the RAINS-Asia model. For residential biofuel combustion, they used the emission factor 1 g/kg, which was taken from measurements in the literature (Muhlbaier-Dasch, 1982) and was similar to that used in the Reddy and Venkataraman study outlined earlier. Their estimates of BC emissions from power plants were lower than Reddy and Venkataraman because of smaller emission factors due to a high level of ash in the particulate emissions. Indian vehicles were assumed to be similar to Chinese vehicles; the authors used emission factors from Streets' 2001 work. The final BC estimate for India was 0.56 million tons. Their estimates differ from Penner et al. (1993), Cooke and Wilson (1996), and Cooke and Wilson (1996) as well as Bond et al. (2004), because of possible inclusion of ash in the emission factors, omission of biofuels, and difference in time periods. For the second method, CO emissions were used as surrogates to estimate BC emissions. They found that that total BC emissions for India using this method were more on the order of 2 million tons. The team concluded that bottom up inventory estimates produce much smaller values of BC emissions than do actual in field observations (or "top down" estimates), which could imply errors in calculating these inventories "bottom up."

- Zhang et al. (2009) focus on the INTEX-B mission, the goals of which were to quantify transport and evolution of Asian pollution to North America and assess its implications for regional air quality and climate. The inventory improved China's emissions estimates by balancing the spread of new and old technology in China's industrial sector and improving energy statistics. For other Asian countries, the mission used IEA energy statistics and emission factors documented in Klimont et al. (2002). The INTEX-B mission also incorporated inventories that were thought to be more accurate representations of individual countries, such as the Indian inventory from Reddy et al. (2002a; 2002b), the Japan inventory from Kannari et al. (2008), and the South Korean inventory from Park et al. (2001). China emissions were 1811 Gg and India emissions were 344 Gg for 2006. INTEX-B also included small industry emissions, but noted that they were uncertain of the numbers. The authors also noted that for Southeast Asia, the activity level data was extrapolated and there were few local emission factors, so the data may not be very accurate. This mission was seen as an improvement on the previous TRACE-P due to the updated methodology and collaboration with local inventory efforts.

Appendix 3

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

Geographic Scale	Results of Study	Mitigation Measures	Pollutants	Reference
Studies of Mitigation Strategies for Ambient Reductions in BC				
Global, Arctic	Fossil fuel soot (FS) and biofuel 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 greater population exposures to BSG.	Elimination of global anthropogenic FS and BSG.	$PM_{2.5}$ from fossil fuel soot, biofuel 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	Relative to no extra controls, imposing tighter vehicle emission standards in developing countries avoids 120,000-280,000 premature air-pollution related deaths in 2030.	Imposition of tighter vehicle emission standards (e.g. Euro 6 standards for light-duty vehicles) in China, India, Africa, Middle East, Brazil, and the rest of Latin America by 2030.	SO_2, OC, BC, NO_3, O_3	Shindell et al. (2011)
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 Asian 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. (2011)
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 vast majority achieved by BC emissions controls. 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).	SO_2, OC, BC, NO_3, O_3	UNEP and WMO (2011a)

Geographic Scale	Results of Study	Mitigation Measures	Pollutants	Reference
Studies of Indoor and Ambient Mitigation Strategies for BC				
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 mitigation strategies involving indoor BC stove emissions and diesel BC emissions 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 emissions from diesel vehicles used in urban cities.	BC, OC, SO_2	Kandlikar et al. (2009)
Studies of Indoor Mitigation Strategies to Reduce BC				
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_2	Smith et al. (2008)
India and UK	Low emission stoves in India result in 12,500 fewer DALYs annually and energy efficiency in the UK households results in 850 fewer DALYs per year.	Energy efficiency in UK household heating and 150 million low-emission cookstoves in India.	BC, OC, sulfates	Wilkinson et al. (2009)

Appendix 4

Efforts to Limit Diesel Fuel Sulfur Levels

As discussed in Chapter 8, the availability of low-sulfur diesel fuel is imperative for many emissions control strategies. Sulfur in fuel will poison the catalysts that are built into passive DPFs, thus rendering them ineffective. DPFs work ideally with 50 ppm or less sulfur diesel fuel (“low-sulfur diesel”). Thus, nations that have adopted low sulfur requirements for diesel fuel of 50 ppm or less are best positioned to adopt more stringent emission standards for new motor vehicles, and have more flexibility to target emissions from in-use vehicles. Nations with established standards of 500 ppm or less have more limited institutional and technological potential for further reductions. Nations with nominal or no limits on sulfur in diesel fuel are unable to adopt technology-based standards or controls on in-use engines that would offer significant reductions in BC.

Aside from the United States, Canada, Japan, and the European Union, 50 ppm or less sulfur diesel fuel is not common. Only a few metropolitan areas in developing Asia have 50 ppm sulfur diesel available (USAID, 2010a). However, several countries around the world have adopted schedules that require the use of lower sulfur diesel fuel between 2010 and 2015:

- Africa: Morocco established limits of 50 ppm in 2009, and Tunisia will require 50 ppm fuel in 2014-2015.
- Americas and Caribbean: Mexico adopted ULSD (< 15 ppm) in 2009 nationwide, while Chile and Brazil have mandated ULSD in urban areas between 2009 and 2013. Several other nations have established requirements for diesel fuel with 50 ppm sulfur, either nationwide (Columbia 2013, Chile 2010, Uruguay 2010) or in large urban areas (Argentina 2012, Colombia 2010).
- Caucasus and Central Asia: Armenia and Kazakhstan both introduced requirements for 10 ppm diesel fuel in 2010. Georgia adopted national standards for 50 ppm diesel fuel in 2010.
- East Asia and Pacific Islands: Malaysia required 50 ppm diesel fuel in 2010 and is requiring 10 ppm diesel fuel in 2015. Singapore, Malaysia, and the Republic of Korea have established national sulfur standards of 50 ppm in diesel fuel between 2007 and 2010. Thailand is limiting diesel fuel to 50 ppm sulfur in 2012. Malaysia and the Republic of Korea plan to adopt 10-15 ppm sulfur limits between 2010 and 2015.
- Eastern Europe: lbania and Belarus plan to require 10 ppm sulfur in diesel fuel in 2011-2012. Croatia, Russia, and Turkey have adopted standards of 50 ppm between 2008 and 2010, though numerous fuel grades continue to be sold.
- South Asia: China limits diesel sulfur to 50 ppm in Beijing (2008), Hong Kong, and Macao; diesel fuel in Taiwan is limited to 50 ppm sulfur after 2005 and 10 ppm starting in 2011. For selected urban areas, India is requiring the use of 50 ppm sulfur diesel fuel in 2010.
- Southwest Asia/Middle East: Israel required 10 ppm sulfur in diesel fuel in 2009, while Qatar is requiring it in 2012. Saudi Arabia and Syria will require 50 ppm fuel in 2014-2015.

Numerous other countries have established diesel sulfur limits of 500 ppm prior to 2015, including Azerbaijan, Brazil (outside urban areas), Ecuador, Fiji, India, Malawi, Mozambique, Oman, Pakistan, the Philippines, South Africa, Sri Lanka, Thailand, Vietnam, and Zimbabwe.

Among nations with less stringent standards on fuel sulfur (e.g., 2,000-10,000 ppm) in either all or part of their territory, some have lowered the limits in recent years. For example, outside urban areas, Argentina and Peru are reducing allowable sulfur to 1500 ppm between 2010 and 2012, from levels of 2500-3000 ppm introduced in 2006. Venezuela reduced allowable sulfur from a standard of 5,000 ppm established to a new standard of 2,000 ppm in 2010. Notable among nations of sub-Saharan Africa, Mauritius established a diesel fuel sulfur standard of 2500 in 2001. Moving to lower sulfur levels in these regions is hampered by economic and technical barriers.

Among nations without sulfur standards, some include oil producing nations, such as Egypt, Iran, and Kuwait. Many sub-Saharan African nations lack national sulfur standards. In the former Soviet Union, many central Asian countries base their national standards on Russia's GOST 305/82 standard for diesel fuel (2,000 ppm). Nevertheless, some nations have diesel fuel with sulfur levels that meet the national standards of countries from which they export. For example, diesel fuel in Lesotho, Namibia, Swaziland, and Botswana meets the 500 ppm national standard established in South Africa, from which they import their fuel.

Through the Partnership for Clean Fuels and Vehicles (PCFV) (*http://www.unep.org/transport/pcfv/*), UNEP continues to work with developing nations to identify opportunities and build capacity to establish lower sulfur levels. For example, the PCFV holds workshops in Africa, Asia, and the Americas, gathering local scientists, engineers, and officials to discuss scientific evidence and economic impacts of how diesel fuel sulfur levels affect cities in developing countries. These meetings follow on PCFV's successful campaign to eliminate lead in gasoline, which recently celebrated the complete phase-out of lead in African gasoline.

Several regional intergovernmental agreements have also been signed by representatives at the ministerial level. In February 2008, environmental ministerial officials from Latin America and the Caribbean in Santo Domingo, Dominican Republic agreed to promote sulfur reduction in fuel throughout the region, with a target goal of 50 ppm. In July 2009, several west and central African environmental ministers signed a regional framework agreement on air pollution, including goals to adopt 3500 ppm fuel sulfur limits by the end of 2011, with a goal of 50 ppm fuel by 2020. Though non-binding on governments, these agreements suggest that there is significant impetus to reduce sulfur levels in fuels used in the developing world.

In addition to governmental and intergovernmental efforts to reduce diesel fuel sulfur levels, several private sector initiatives also exist. Vehicle industries around the world have recognized the value of reduced sulfur for enabling lower-emissions vehicles and high-efficiency combustion technologies. In 2002, vehicle and engine manufacturers from the United States, Europe, and Japan published a report on worldwide fuels harmonization, which promoted lower sulfur levels in gasoline and diesel fuel. More recently, the African Refiners Association has developed a set of "AFRI" fuel specifications (AFRI-1 through AFRI-4) as a developmental pathway for African development of ≤50 ppm sulfur.

Table A4-1 gives recent information on national standards for on-road diesel sulfur limits, and estimates of current sulfur levels. In addition to the efforts described above, Chapter 8 also mentions the limits on sulfur content of marine fuel being phased in under requirements from the IMO. Table A4-2 provides details regarding the fuel sulfur levels allowed for C3 marine fuel within ECAs and globally outside of ECAs, and the schedule for phase-in of tighter limits on sulfur content of this fuel. For this table, the Global and ECA fuel standards shown are the maximum fuel sulfur levels allowed under MARPOL Annex VI for ships with engines over 130 kW.[1] The date on which the ECA requirements become enforceable for a specific geographic area depends on the date the treaty amendment incorporating the ECA enters into force.

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[1] MARPOL is an abbreviation of "marine pollution," and is the acronym used to refer to the International Convention on the Prevention of Pollution from Ships.

Table A4-1. International Regulations and International Agreements on Diesel Fuel Sulfur Levels (in ppm). (Source: U.S. EPA)

Region	Country	Year 2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	Current Maximum Level
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									50
	Argentina (Non-Urban)	?	?	?	?	?	?	2500	2500	2500	2500	2500	2500	1500									1500
	Barbados	No existing or planned standards.																					(5000)
	Bolivia	?	?	?	?	?	?	?	?	?	?	(2000)											5000
	Brazil (Non-Urban)	?	3500	3500	3500	3500	3500	3500	3500	3500	3500	1800	1800	1800	1800	500							2000
	Brazil (Metropolitan)	?	2000	2000	2000	2000	500	500	500	500	50	50	50	50	10								50
	Chile (National)	?	?	?	?	?	?	350	350	350	350	50											50
	Chile (Santiago)	?	?	?	?	?	?	50	50	50	50	10											10
	Colombia (National)	?	?	?	?	?	?	2500	2500	2500	2500	500			50								500
	Colombia (Bogota)	?	?	?	?	?	?	?	?	?	?	50											
	Costa Rica	No existing or planned standards.																					(500)
	Cuba	No existing or planned standards.																					(8000)
	Dominican Republic	No existing or planned standards.																					7500
	Ecuador (National)	?	?	?	?	?	?	?	(500)	500	500	50											
	Ecuador (Urban)	?	?	?	?	?	?	5000															500
	Ecuador (Non-Urban)	?	?	?	?	?	?	7000															
	El Salvador	?	?	?	?	?	?	?	?	?	?	500											
	Guatemala	?	?	?	?	?	?	?	?	?	?	(500)											5000
	Honduras	?	?	?	?	?	?	?	?	?	?	(500)											5000
	Panama	?	?	?	?	?	?	?	?	?	?	(1000)											5000
	Peru (Urban)	?	?	?	?	?	?	1500	1500	1500	1500	50											

Region	Country	Year																					Current Maximum Level
		2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	
Americas and Caribean	Peru (Non-Urban)	?	?	?	?	?	?	3000	3000	3000	3000	1500											
	Uruguay	?	?	?	?	?	?	8000	8000	8000	8000	50											
	Venezuela	?	?	?	?	?	?	5000	5000	5000	5000	2000											5000
East Asia, Australia, and Pacific Islands	Australia	?	?	?	500	500	500	50	50	50	10												10
	Cambodia	2000	?	?	?	?	?																1500
	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)											4000
	Japan	100	100	100	100	100	50	50	50	50	50	50											
	Malaysia	3000	3000	(500)	(500)	(500)	(500)	(500)	(500)	(500)	(500)	50	50	50	50	50	10						
	Nepal	10000	10000	10000	10000	10000	10000	10000	10000	10000	10000	10000											500
	Philippines	?	?	?	?	?	?	?	?	?	?	500											
	Republic of Korea	500	500	500	500	500	500	500	500	500	500	50											
	Singapore	500	500	500	500	500	500	500	50	50	50	50											
	Thailand	500	500	500	500	500	500	500	500	500	500	500	500	50									
	Vietnam	10000	10000	2000	2000	2000	500	500	500	500	500	500											
Eastern Europe, Caucasus, and Central Asia	Albania	?	?	?	?	?	?	?	?	?	?	350	150	10									2000
	Belarus	?	?	?	?	?	?	?	?	?	?	50	10										350
	Bosnia & Herzegovina	?	?	?	?	?	?	?	?	?	?	350											350
	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)											2000-5000
	Serbia	?	?	?	?	?	?	?	?	?	10000	(350)											10000
	Uzbekistan	?	?	?	?	?	?	?	?	?	?	5000											400- 2000

Region	Country	Year 2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	Current Maximum Level
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	?	?	?	?	?	?	?	?	?	?	?	?	500									
	Sri Lanka	10000	10000	10000	3000	?	?	?	?	?	?	500											
Southwest Asia and North Africa	Algeria	No existing or planned standards.																					900
	Bahrain	?	?	?	?	?	?	?	?	?	?	(500)											500
	Egypt	?	?	?	?	?	?	?	?	?	?	(5000)											10000
	Iran	?	?	?	?	?	?	?	?	?	?	(5000)											10000
	Iraq	?	?	?	?	?	?	?	?	?	?	(10000)											25000
	Israel	?	?	?	?	?	?	?	?	?	10												10
	Jordan	?	?	?	?	?	?	?	?	?	?	350											(7000-10000)
	Kuwait	?	?	?	?	?	?	?	?	?	(50)	(50)	(50)	(50)	(10)								2000
	Lebanon	No existing or planned standards.																					(5,005,000)
	Libya	?	?	?	?	?	?	?	?	?	?	(1000)											1500
	Morocco	?	?	?	?	?	?	?	?	?	50	50											50
	Oman	?	?	?	?	?	?	?	?	(50)	(50)	(50)											500
	Palestinian territories	No existing or planned standards.																					(10000)
	Qatar	?	?	?	?	?	?	?	?	?	?	500	500	10									500
	Saudia Arabia	?	?	?	?	?	?	?	?	?	?	?	?	?	?	50							5000
	Syria	?	?	?	?	?	?	?	?	?	(50)	(50)	(50)	(50)	(50)	(50)	50						7000
	Tunisia	?	?	?	?	?	?	10000	350	350	10												5000
	United Arab Emirates	?	?	?	?	?	?	?	?	?	(50)/ 500												350
	Yemen	No existing or planned standards.																					10000
Sub-Saharan Africa	Angola	?	?	?	?	?	?	3000															
	Benin	No existing standards.											3500	3500	3500	3500	3500	3500	3500	3500	3500	50	(10000)
	Botswana	No existing or planned standards.																					(500)
	Burkina Faso	No existing standards.											*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*50*	(5000)
	Burundi	No existing or planned standards.																					(5000)
	Cameroon	?	?	?	?	?	?	?	?	?	?	5000											5000

Region	Country	Year																					Current Maximum Level
		2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	
Sub-Saharan Africa	Cape Verde	No existing or planned standards.																					(3000)
	Central African Republic	No existing or planned standards.																					(3000-5000)
	Chad	No existing standards.											3500	3500	3500	3500	3500	3500	3500	3500	3500	50	5000
	Côte d'Ivoire	?	?	?	?	?	?	?	?	?	?	5000	3500	3500	3500	3500	3500	3500	3500	3500	3500	50	5000
	Dem. Rep. of Congo												*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*50*	(5000)
	Djibouti	?	?	?	?	?	?	?	?	?	?	5000											5000
	Equitorial Guinea	No existing or planned standards.																					(5000-8000)
	Eritrea	?	?	?	?	?	?	?	?	?	?	7000											7000
	Ethiopia	?	?	?	?	?	?	?	?	?	?	(5000)											10000
	Gabon	No existing or planned standards.																					(8000)
	Gambia	No existing or planned standards.																					
	Ghana	?	?	?	?	?	?	?	?	?	?	5000	3500	3500	3500	3500	3500	3500	3500	3500	3500	50	5000
	Guinea	No existing standards.											*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*50*	(5000)
	Guinea Bissau	No existing or planned standards.																					(5000)
	Kenya	?	?	?	?	?	?	?	?	(5000)													10000
	Liberia	No existing standards.											*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*50*	(5000)
	Madagascar	?	?	?	?	?	?	?	?	?	?	5000											500
	Malawi	?	?	?	?	?	?	?	?	500													500
	Mali	No existing standards.											3500	3500	3500	3500	3500	3500	3500	3500	3500	50	(10000)
	Mauritania	No existing or planned standards.																					(5000)
	Mauritius	?	2500																				2500
	Mozambique	?	?	?	?	?	?	?	?	500													500
	Namibia	No existing or planned standards.																					(500)
	Niger	?	?	?	?	?	?	?	?	?	?		*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*50*	10000
	Nigeria	?	?	?	?	?	?	?	?	?	?	(1330)											5000
	Republic of the Congo	?	?	?	?	?	?	?	?	?	?	(1000)	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*50*	10000
	Senegal	?	?	?	?	?	?	?	?	?	?												5000
	Sierra Leone	No existing standards.											3500	3500	3500	3500	3500	3500	3500	3500	3500	50	(5000)
	South Africa	?	?	?	?	?	?	500	-50														500
	Tanzania	?	?	?	?	?	?	?	?	?	?	5000											5000
	Togo	?	?	?	?	?	?	?	?	?	?	10000	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*3500*	*50*	10000
	Uganda	?	?	?	?	?	?	?	?	?	?	5000											5000

Region	Country	Year																					Current Maximum Level
		2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	
	Zambia	?	?	?	?	?	?	?	?	?	?	7500											7500
	Zimbabwe	?	?	?	?	?	?	?	?	500													

Notes:

1. Parentheses indicates that fuel of a given sulfur level is available or sold in that country, though the national standard may differ.
2. Strikethrough numbers indicate that fuel with sulfur in excess of the local standard is commonly sold.
3. Gray-shaded numbers indicate intergovernmental agreement on future standards.
4. Underlined numbers refer to agreements made by national ministerial-level officials.
5. Italicized numbers refer to agreements made by national officials below ministerial level.

Source Material:

1. Partnership for Clean Fuels and Vehicles, United Nations Environmental Programme (*http://www.unep.org/pcfv*).
2. Krylov et al. (2005) Current problems. Low-sulfur diesel fuels: pluses and minuses. Chemistry and Technology of Fuels and Oils 41: 423-428.
3. Regional Environmental Centre for the Caucasus. (2008) Fuel quality and vehicle emission standards overview for the Azerbaijan Republic, Georgia, the Kyrgyz Republic, the Republic of Armenia, the Republic of Kazakhstan, the Republic of Moldova, the Republic of Turkmenistan, the Republic of Uzbekistan, and the Russian Federation. (*http://www.unep.org/pcfv*).
4. Rojas-Bracho, L. (2009) An international strategy for black carbon controls in the transport sector. The case of Mexico. Presented at International Workshop on Black Carbon, November 30, 2009. (*http://www.theicct.org/information/workshops/international_workshop_on_black_carbon*).
5. West and Central African Regional Framework Agreement on Air Pollution, Abidjan, Ivory Coast, July 20-22, 2009.
6. UNEP (2008b). Final Report of the 16th Meeting of the Forum of Ministers of the Environment of Latin America and the Caribbean. UNEP/LAC-IGWG.XVI/9.

Table A4-2. International Fuel Sulfur Limits for C3 Marine Fuel, by Target Year. (Source: U.S. EPA)

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

Appendix 5

U.S. Emission Standards for Mobile Sources

Table A5-1. On-Road Regulations for PM Control. (Source: U.S. EPA)

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

Table A5-2. On-Road Gasoline-Vehicle Regulations. (Source: U.S. EPA)

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

Table A5-3. Nonroad Diesel Regulations. (Source: U.S. EPA)

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

Table A5-4. Locomotive Regulations.[a] (Source: U.S. EPA)

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

[a] Note that in the 2008 Locomotive/Marine Rule, EPA revised its emission standards for remanufactured locomotives to redesignate as Tier 1 most locomotives originally built between 1993 and 2001. EPA determined that these locomotives could be readily retrofitted to meet the more stringent Tier 1 standards when remanufactured.

Table A5-5. Commercial Marine Regulations. (Source: U.S. EPA)

Regulation	Sector	Model Years Applicable	PM standard	Units
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
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

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, locomotives and commercial marine can also be significant. The following discussion addresses emission standards in other parts of the world.

Outside the United States, 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 United States, Australia also bases its national standards on those developed in the United States, 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 4, 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.[1] 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 scatter around the trend line of each country reflects differences in standards based on vehicle type (truck vs. bus), test procedure (e.g., operating cycle), and/or location (e.g., urban vs. rural).

Beyond nations that have regulations with emission standards, other nations have been addressing vehicle emissions in some manner. Some other nations are adopting emission standards for light-duty vehicles, generally based on EURO standards. Others have eliminated or are scheduled to eliminate lead from gasoline, which enables the implementation of standards to reduce tailpipe emissions using catalytic aftertreatment. An example of this progress is found in Africa, where all nations have eliminated lead in gasoline. Others have banned the import of light-duty vehicles without a catalytic converter or established opacity testing requirements for cars, trucks, or scooters. This progress suggests room for additional technology-based approaches to reducing BC emissions.

Many other countries lack any emission standards. The reasons for their lack of emission standards may be attributable to several causes, including insufficient governmental capacity, poverty and other economic factors, and government policy. Many such countries face many other problems related to economic development, public health, violence, and authoritarian rule. Addressing BC from motor vehicle emissions in these locations may requires attention to factors other than technology.

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[1] See *http://www.dieselnet.com/standards* for more information.

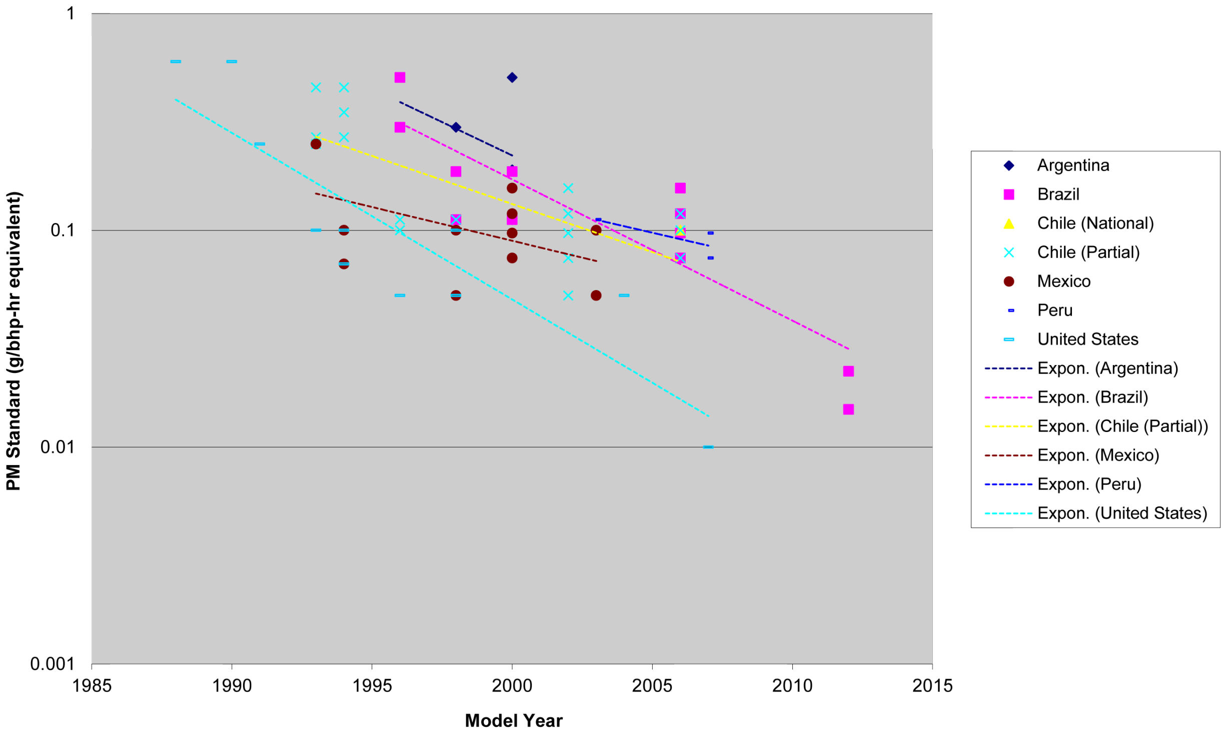

Figure A6-1. Heavy-Duty Highway Diesel PM Emissions Standards in the Americas and the Caribbean (Logarithmic Scale). (Source: U.S. EPA)

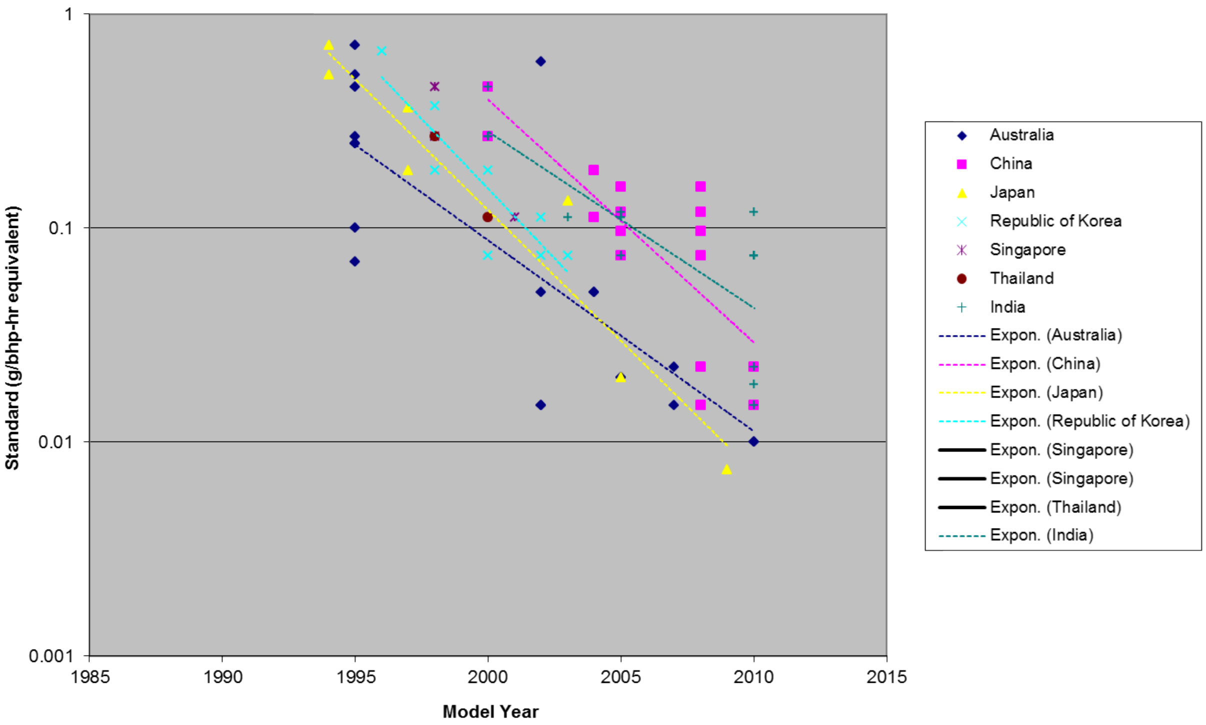

Figure A6-2. Heavy-Duty Highway Diesel PM Emissions Standards in Asia and Australia (Logarithmic Scale). (Source: U.S. EPA)

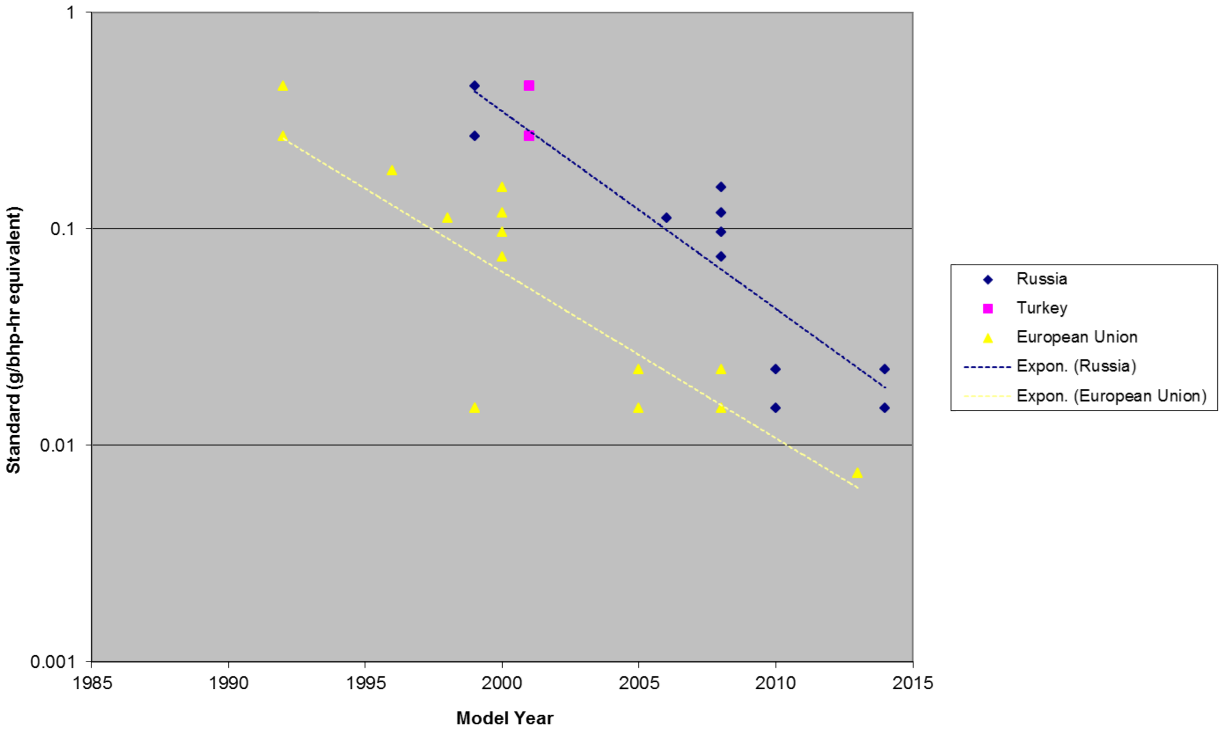

Figure A6-3. Heavy-Duty Highway Diesel PM Emissions Standards in Europe (Logarithmic Scale).
(Source: U.S. EPA)

Appendix 7

Research Needs

The chapters of this report have laid out much of what is known about BC. While there is a great body of knowledge about BC, its emissions, its atmospheric properties, its effects on climate, and its impacts on health, more information on these topics would lead to enhanced capability to design policies and control strategies that would be most beneficial for the environment and for human health. Many of these research priorities have been highlighted in this report; these research needs are summarized here, with the relevant section(s) of the report calling for each research topic given in parentheses. These descriptions also give more detail on the uncertainties and research needs discussed in sections **12.5** and **12.6**.

A7.1 Basic Atmospheric Chemistry of BC, LAC, and Measurement Approaches

Establishing standardized definitions of terms related to carbonaceous aerosols and harmonizing measurement approaches will reduce the uncertainty in emissions inventories and result in improved and more consistent air quality and climate estimates. Standardized definitions will also encourage the use of preferred monitoring approaches and more consistent, less uncertain data, resulting in better climate and health assessments; currently, a lack of consistency in connections between BC and health endpoints could be caused by uncertain or inconsistently produced data. These definitions can provide the basis for future reference and equivalent measurement methods for a regulatory program if ever needed.

- It is important to develop standardized definitions of BC, BrC, non-light-absorbing OC and other light-absorbing aerosols (**2.3**).

- It is also important to establish corresponding reference materials for needed particle properties (**2.3**, **5.2**).

- Interpretation and harmonization of existing ambient and emissions measurements are necessary to conform to the specified definitions and properties to meet the needs of climate and health assessments (**5.2**).

While BC absorbs light across the entire solar spectrum, BrC absorbs primarily in the ultraviolet range. The quantity of solar energy absorbed depends upon the molecular structure and the mass of the relevant BrC compounds present in a combustion plume or aerosol mixture. OC is usually the most significant co-emitted pollutant, by mass, among the major BC emitting sources.

- Estimates are needed to separately characterize BrC and the non-light-absorbing (scattering) portions of OC. Characterizations of the particulate, semi-volatile and perhaps some volatile (“intermediate VOC”) components of non-light-absorbing (scattering) portions of OC are especially important to improve inventories, to improve model estimates of SOA, and to evaluate models.

- It is necessary to develop and/or improve instrumentation and measurement techniques to identify and quantify the physical and chemical properties (optical, size, number, composition, and mass) of BrC and LAC, and through modeling, to estimate the radiative forcing of BrC.

- It is also important to develop the ability to track the radiative forcing and cloud droplet-forming potential of particles, from various emissions sources, as they age in the atmosphere (**5.2, 2.6**).

There is also a need to improve instrumentation and measurement techniques to identify and quantify the properties of BC (optical, size, number, and mass) taking into consideration different emissions sources, combustion conditions and varying particle mixtures (**5.2**, **4.3**, **4.4**). Further study is needed to understand the differences among existing thermal and optical monitoring and measurement methods for BC and how they are affected by (a) the emissions sources and resulting aerosol mixtures (considering variations in the combustion process and different fuels for source measurements), (b) atmospheric processing (aging), and (c) the manner in which monitoring is performed. Preferred protocols for future measurements and monitoring are desired to ensure more consistent and useful data collection. Improved source measurement techniques will facilitate the opportunity for more measurements, better emission factors and then better inventories. Multiple methods may be needed to serve different data needs.

A7.2 Ambient and Emissions Source Measurement

Emissions source measurements are much more limited than ambient measurements. Additional representative source measurements are needed to better characterize BC emissions (and speciated profiles) by emissions source, fuel type and combustion conditions. These data are needed to improve BC emission factors and inventories, to create BrC emission factors and inventories, and to help develop emissions and modeling uncertainties. Research can also help guide a new focused data collection effort.

- Methods for using source measurements to develop emission factors for light-absorbing carbon (including BrC) – particularly for the major source categories like open biomass combustion and mobile sources – need to be explored (5.2, 4.3, 4.4).

Regular speciated PM measurements, including EC (or BC) measurements, currently occur at very coarse spatial and temporal scales. More spatially dense ambient monitoring can help identify unaccounted for emissions and sources. More temporal estimates (within and among years) as well as semi-continuous data will help account for the impacts of emissions trends. Understanding the effect of historical emissions changes will be useful in estimating the need for future emissions reductions to meet various policy goals.

- More ground-level ambient measurements are needed, with improved spatial and temporal coverage, particularly outside the United States. These data would be useful for a multitude of reasons, including better understanding of pollution trends and better linkages between PM components, such as BC, and health, climate, ecosystem, and visibility outcomes (**5.3**, **5.4**, **and 5.7**).

A consolidated global database of BC and OC ambient and source measurements would be useful for facilitating the analysis of all existing data and for ensuring consistent development of inventories and evaluation of global climate models. A protocol should be established to improve the quality and to guide the reporting of meta-data available both for ambient measurements and source profiles (e.g. identifying conversions of absorption measurements to mass and adjustments for sampling artifacts and or material balance). Consistent reporting of measurements is desired because the variability among existing measurement approaches and data contribute to the uncertainty of domestic and global emissions estimates and impacts.

- Mechanisms for archiving, consolidating, and sharing existing measured data on light absorption and scattering of particles, available globally, are needed to reduce the need for new measurements and to produce better air quality and climate characterizations (**5.2**, **5.7**).

Full chemical composition emissions measurements for the important carbonaceous combustion sources, including the identification of carbonaceous co-pollutants in addition to BC (OC, BrC) measurements would promote better understanding the atmospheric aging and climate-relevant properties of carbonaceous aerosols as well as would be essential in constructing the emissions inventories needed for assessment of any co-benefits and tradeoffs from BC mitigation efforts (**5.2**, **4.3**, **4.4**, **6.4**).

A7.3 Emissions Inventories

A7.3.1 Global

Given the importance of emissions inventories in understanding the impacts of BC and control measures, more accurate representations of BC emissions globally is critical for more accurate estimates of BC's impacts. Sources of BC outside the United States are responsible for 94% or so of the current emissions globally and this number is expected to increase in the future. The work of Bond et al. (i.e., 2004) is widely recognized as the "best currently available" global BC inventory. That work is based on combinations of fuel, combustion type, and emissions controls and their prevalence on a regional basis. The 2004 work (which represented 1996 fuel-use data) has since been updated to reflect more recent years and to improve the emission factors and usage patterns for some sources. However, there remain significant uncertainties for BC inventories in developing countries and globally. There are several ways in which these inventories can be improved. Suggested areas for further research largely focus on improving these emissions by gathering more data both on the emissions side and on the "usage" (activity level) side as well on determining how best to allocate these emissions both spatially and temporally:

- For mobile sources, there is a need to better characterize the on-road and nonroad fleet in urban and rural areas for different countries including commercial marine and locomotives.
- Emissions estimates for nonroad applications need to be developed and collated separately from on-road operations (**4.4.1**).
- In the case of biomass burning, all fires need to be better captured for their emissions characteristics and activity level.
- A careful review of usage patterns needs to be conducted to ensure that appropriate "activity" levels are applied to emission factors to arrive at final emissions estimates (**4.4.3**).
- Small(er) sources are especially poorly represented on a regional basis, and better characterization of emissions from residential cookstoves, in-use mobile sources, small fires, smaller industrial sources such as brick kilns, and flaring emissions is needed.
    - Many of these "small" sources could have relatively high BC emission factors (**4.4.1** and **4.4.3**).
    - For sources such as cookstoves, improved characterization depends critically on field-based measurements of emissions from in-use sources.
    - In addition, usage patterns need to be reviewed to ensure that appropriate "activity" levels are applied to emission factors to arrive at final emissions estimates.
    - Finally, fuller incorporation of regional inventories into global inventories could improve country- and region- specific emissions estimates.
- Steps need to be taken to better engage international scientists, governments, and regulatory agencies in collaborations of technology assessments to help improve base-year and out-year global emissions.
    - Merging regional inventories with global estimates should be explored in an attempt to improve country- and region-specific emissions estimates in global inventories (**4.4.3** and **4.4.4**).
    - Initiating an international forum in which scientists and governments can more readily and routinely engage to help facilitate and share this research would expedite emissions inventory improvement and help to harmonize estimation methods across world regions.

A7.3.1.1 Domestic

While domestic emissions of BC and PM are generally better characterized than global emissions, considerable uncertainty remains for these estimates and there are several aspects of domestic inventories that need improvement.

- More information on both emission factors and usage would be helpful. In particular, emissions from key industrial sources, flaring, residential heating, and open biomass burning remain poorly characterized.
- In general, mobile source emissions are among the best characterized (especially in developed countries), but improved information is still needed for some sectors, most likely through increased testing and data acquisition. (These needs are all discussed in Section **4.3.2**.)
    - Special attention should be given to PM mass and composition from nonroad sources (gasoline and diesel), aircraft (including in-flight emissions), commercial marine especially C3 (ocean going), and locomotives including both current and, where available, future technologies.
    - Characterization of emissions from newer-technology on-road diesel and gasoline vehicles would also be useful, as would better emissions characterization (including fleet fraction) of high-emitting vehicles/engines (so-called "super-emitters").
    - Finally, the improved characterization of emissions at low ambient temperature and with different fuels (including renewables) would improve the understanding of present and future emissions from mobile sources.

These data can then be used to develop newer and more accurate emission factors for these key areas within mobile sources. These new data could also enable construction of emissions models specifically for EC for nonroad sources similar to what exists for on-road (and rely less on using $PM_{2.5}$ models and translating to EC via a separate database in which speciated emissions from sources are archived).

A7.3.2 Uncertainty Analysis

For both global and domestic EC/BC emissions, analyses of uncertainty would aid in determining which sectors have the strongest estimates of BC emissions, which in turn could lead to easier decisions on mitigation options. Bond et al. (2004) have done a Monte Carlo-type uncertainty analysis for global emissions and have estimated that uncertainty in BC emissions inventories is generally on the order of a "factor of 2," and more work along these lines is needed. A starting point on the domestic side would be to qualitatively rank sources by considering the strengths and weaknesses of how the BC emissions were assembled. This could lead to a research task of doing a more rigorous uncertainty analysis on the global estimates. This method could also be applied to future-year projections of emissions.

- Quantitative measures for describing uncertainties in emissions should be advanced, as was suggested in Section **4.4.4**.

- As a first step, a qualitative description of uncertainty in emissions estimates, sector-by-sector, for domestic emissions inventories including models should be performed.

- The uncertainty that stems from combining BC data sets collected by several measurement techniques (for regional and global inventories) should be estimated.

- Finally, the Monte Carlo-type work by Bond et al. (2004) should be extended to determine uncertainties in global emissions estimates.

A7.4 Ambient Observations, Including Deposition

While many kinds of additional measurements would improve our understanding of BC in the atmosphere, a few specific types of measurements were highlighted in this report as able to fill some important gaps in the current understanding of BC. BC's vertical distribution (and its impact on surface and atmospheric radiative forcing and cloud formation) is one of the important uncertainties in assessing BC's overall impact. Similarly, the deposition of BC is a source of uncertainty in determining BC's overall impacts. For example, a recent NOAA/GFDL modeling study (Liu et al., 2011) showed that the simulated vertical column of BC concentrations over the Arctic is highly sensitive to different parameterizations of deposition properties.

- Better characterizing the vertical distribution of BC would allow for a fuller understanding of its climatic impacts, which are dependent on its vertical distribution, and would further understanding of the discrepancies between models and observations (**5.5** and **5.7**).

- Research to inform the characterization of BC wet scavenging and dry deposition rates, deposition on snow and ice and resulting radiative forcing, albedo, and hydrological changes (**2.6**, **5.6.2**, **5.6.5**) would improve the modeling of these important processes.

A7.5 Modeling

Some aspects of the modeling of BC in the atmosphere, and its effects on climate, are better understood than others. BC's aging/mixing states (internal, external, or core-shell) and its indirect and semi-direct effects are likely the largest uncertainty in assessing BC's RF and climate impacts.

- Improving the modeling of BC's direct effects due to aging/mixing states assumptions, semi-direct effects (on vertical mixing, clouds, and differential heating at surface and the atmosphere), and indirect effects (on cloud formation, lifetime, albedo, etc.) would be very useful for improving representations of BC's environmental impacts (**2.6**).

BrC is also a heretofore under-studied aspect of the impacts of carbonaceous aerosols. Inclusion of BrC in climate models, and analysis of implications for net forcing, including a sensitivity analysis of the upper and lower reasonable bounds for BrC absorption, would enable a fuller accounting of the climatic impacts of carbonaceous aerosol sources.

- Reporting column data by wavelength may aid model-observation comparison as BC and BrC differ in terms of peak absorption (**2.3**, **5.6.2**).

- Coupled with experimental estimates of BrC emissions and laboratory estimates of BrC scattering and absorption, this should clarify the magnitude of the cooling offset due to organic carbon co-emissions, which is important for determining net forcing of abatement measures.

A7.6 Climate Impacts

Further research on the impacts of BC and carbonaceous aerosols on climate, with an eye toward filling in the continuum from source to impact, would enable policies that are more likely to have beneficial impacts. Several aspects of this issue were highlighted in the chapters of this report. Emissions of other pollutants from the same sources can lead to difficulties in determining whether a control strategy will result in radiative cooling or warming. In addition, there are indications that near the Arctic and other snow-covered regions, the net effect of any mitigation measure including measurable amounts of BC is much more likely to be net cooling.

- Better characterization of the sources as well as the "total" radiative effect of the control measure will reduce the possibility of unintentional warming as well as lead to more efficient policy.
- Research is needed to characterize the range of possible control strategies and evaluating the probability that a given measure will result in net cooling (**6.4**). Specific focus should be placed on the location (especially latitude) of the proposed change in emissions, especially for near-Arctic emissions. Specific measures would be the goal, but sectorial-level analysis would also be useful.

One important aspect of BC's impacts on climate is its role in snow and ice melting. This is of particular relevance in areas where BC deposition may affect snow pack that influences the availability of water resources for downstream populations (e.g., California, Himalayas and Tibetan Plateau, Andes, high African mountains) as well as in the Arctic. Focused research on the role of deposited BC could shed light on the effectiveness of mitigation measures for protecting water resources and snow in sensitive regions.

Specific contributors to the overall uncertainty in BC's impacts include aerosol mixing state and cloud impacts. The biggest uncertainty in direct effect calculations is due to aging/mixing of particles, and the biggest uncertainty overall may be cloud interactions. Continued research is needed on the radiative properties of BC and co-emissions, especially regarding cloud interactions and effects of aging/mixing (**2.6**).

Non-radiative impacts of BC are more poorly understood than its radiative impacts; even if the net radiative effect of a given measure is near-zero, there may be other climatic impacts. Continued research is needed on the non-radiative effects of BC and other aerosols, especially regarding precipitation/hydrological interactions and dimming (**2.6.3**).

A7.7 Metrics

It is difficult to apply climate metrics developed for GHGs to BC and other short-lived forcers as many of the fundamental assumptions that go into the calculation of these policy-relevant metrics for long-lived GHGs are not appropriate for application to BC. Though "alternative" metrics have been proposed for BC, none is yet widely utilized. Appropriately tailored metrics for BC are needed in order to quantify and communicate BC's impacts and properly characterize the costs and benefits of BC mitigation. Improved metrics could incorporate non-radiative impacts of BC, such as impacts on precipitation. Similarly, given BC's (and other aerosols') direct impacts on human health, health outcomes could also be incorporated into such a metric. Developing methods to quantify the benefits of BC mitigation on both climate and health would encourage policy decisions that factor in climate and health considerations simultaneously, within a unified framework.

Ways to quantify the various impacts of BC into a unified framework or to compare BC to other SLCFs and GHGs would enable quantification of how BC mitigation leads to the attainment of various policy goals:

- Explore the use of emissions source measurements and/or emissions inventory estimates to determine whether (or how) measures like OC/EC ratios, BC (as measured currently), and other LAC forms can be utilized to inform development of metrics to prioritize emissions sources for mitigating the climate effects of PM emissions (**5.2, 2.7.3**). Simple emission-related measures are useful screening tools that permit mitigation decisions without the need to run complex climate models. Currently, such measures are used but with unknown, but probably large, uncertainty. For example, using directly emitted OC as an indicator of scattering from aerosols does not acknowledge the role of SOA or OC's BrC component.
- There is a need for an analysis of the implications of using different metric choices on future emissions and climate, examining the benefits of near-term versus long-term temperature abatement and therefore short-lived vs. long-lived gas abatement (**2.7.3, 2.7.4**). One example of this would be a system where an economic model is run with different GWP/GTP values

for BC, and then the emissions are fed into a climate model. Key issues involve the relationship between different metrics values and economic impacts, and in terms of the temporal pattern of temperature implications.

- Given the differences in the mechanisms by which BC and GHGs impact climate and human health, there is a need for improved quantification and valuation of specific climate change impacts that could be attributed to BC emissions, including analysis of how these differ from quantification of GHG reduction benefits.

- Additionally, development of approaches to identify and properly account for co-benefits in mitigation benefit analyses would more accurately capture the climatic impacts of policy options (**2.7.3, 2.7.4**). Key differences between BC and other GHGs for valuation purposes include the regional specificity of BC impacts, the differences in vertical distribution of forcing, and cloud interactions. Human health effects and precipitation, visibility, and dimming effects are also cited as benefits of BC control. Including these benefits could improve metrics development and would move these metrics beyond simply climatic endpoints to a whole host of environmental and health goals.

A7.8 Health

A great deal of research on the health impact of $PM_{2.5}$ and specific PM components has been conducted over the past 15 years, and these topics have already been identified as priorities by EPA in the context of its periodic reviews of the U.S. national ambient air quality standards for PM. While the scientific record is robust in many respects, there are still important unanswered questions about the relative toxicity of different constituents. Also, research continues to inform the overall understanding of the magnitude and nature of PM health impacts, including more precise quantitative information about the relationship between indoor and ambient concentrations and health impacts. Continued investment in this research is important, and there is a particular need for more studies in developing countries.

In addition to the ongoing research on the health impacts of PM components such as BC, work is needed on linking the many types of impacts from a particular emissions sector. Research is needed to quantify the integrated climate and health impacts of individual economic sectors and domestic and international mitigation measures, accounting for the full mixture of emissions, both direct and indirect climate effects, and both indoor and outdoor exposure (**2.7.3.5, 2.7.4**, **3.3**, and **3.4**). This includes the complex mix of organic compounds that comprise the co-emitted species from BC sources such as fossil fuel and biomass combustion. This would help elucidate where the greatest opportunities to benefit human health and the environment lie, with respect to BC mitigation.

A7.9 Mitigation Technologies and Measurements

Of key importance is research on which BC mitigation strategies are most cost-effective and beneficial for public health and climate. The necessary continued research on mitigation includes more information on costs and benefits of mitigation by sector, and development of new or improved mitigation technologies for various sectors. This research would in turn allow for comparisons across sectors in terms of costs and benefits for climate and human health. The following research needs are for specific source sectors.

A7.9.1 Stationary

While emissions from controlled domestic sources are relative well understood, there are ways in which estimates could be improved. Additional source testing and development of improved emission factors (**9.8**) would result in higher quality emissions inventories for stationary sources in the developed and the developing world. This would improve our estimates of the effectiveness of traditional control equipment (baghouses, electrostatic precipitators) in reducing the BC fraction of PM. Along with improved activity level estimates, it would improve overall emissions inventory estimates for the industrial sector. Key categories would include coke production, brick kilns, and oil and gas flaring. Profiles for stationary sources in the developing world are especially uncertain.

A7.9.2 Mobile

Mobile sources, especially on an international basis, are an important source of EC emissions and improved, more effective, and more cost-effective control technology would result in wider adoption and likely benefits for both human health and climate change mitigation. It is important to continue the development of current mobile source control technology (such as diesel particulate filters for in-use vehicles) targeted specifically for EC, including the ability to assure adequate durability and lower costs. Also, the applicability to more existing engines, especially nonroad, locomotive, and C1/C2 marine, is needed. Control technology for C3 marine is also needed. The speciated emissions profiles that result from the use of existing control technologies are currently not well characterized (**8.6**).

A7.9.3 Cookstoves

Given the ubiquity of high-emitting cookstoves in the developing world, it is not surprising that there are many opportunities for research into mitigating cookstove emissions. First, linking cookstoves to climate outcomes requires a clear understanding of the emissions from particular stoves and fuels. Currently, such information is lacking, and most of the testing that is occurring is in laboratory settings. Therefore, there is a need for laboratory and field testing to characterize emissions (BC, OC, CH_4, other constituents) from different types of cookstoves, based on stove design, fuel type, and usage patterns (**10.4.1**, **10.4.2**). Expanded lab and in-field testing data are needed to clarify what constitutes a "clean" stove/fuel.

Because of the difference in fuels among regions, as well as the difference in sensitivity of the local and regional environment to climate forcers, there is a need for regional-level studies of the net climate impacts of cookstove emissions from different cooking stove-fuel combinations, including linkages to radiative forcing, glacial melt, and precipitation impacts (**10.4.1, 10.4.2**). Studies evaluating the extent to which emissions from stoves are linked to climate impacts at the local/regional level would help clarify which cookstove mitigation efforts would be beneficial for climate, and what technologies and fuels would be needed to achieve maximum climate benefits.

In order to make good policy decisions about preferred interventions and investments, there is need to better understand the linkages between emissions changes and health benefits. Improved dose/response information would enable policymakers to target specific improved stoves and fuels for development and dissemination. Specifically, research is recommended on examining dose-response relationships between emissions of various cookstove emission constituents and health endpoints of concern (ALRI/pneumonia, COPD, cardiovascular disease, cancer, etc.) (**10.4.1**, **10.4.2**).

A7.9.4 Residential Heating

There are some uncertainties associated with the emissions from residential heating. More research on the composition of particles from residential heating (**10.3.1**) would enable better quantification of the benefits of mitigation from the residential sector. More data on the optical properties of these aerosols would be especially useful. Related to this uncertainty in emissions composition is the effect that different heating technologies and abatement options have on the chemical composition of carbonaceous aerosol emissions (**10.3.1**). There is currently little data on whether the improved stoves used for air quality purposes equally reduce all PM components, or whether some are reduced preferentially. These changes in composition can be measured using existing (or new) techniques for emissions speciation.

Similarly, the performance of residential heating appliances as they age has not been well documented. Given that the lifetime of wood stoves is on the order of several decades, more complete long-term performance data on the emissions from older heating appliances would enable more accurate emissions and impact projections into the future (**10.3.2**).

A7.9.5 Biomass Burning

Globally, a major fraction of BC emissions come from biomass burning, and yet biomass burning emissions are especially uncertain. In order to understand the efficacy of mitigation options, biomass burning emissions should be better characterized. Specifically, there is a need for additional measurements and biomass burning emissions and activity factors as a function of size and duration of the fire, fuel type, fuel conditions, fire phase, and meteorological conditions on the day of the burn and other significant variables (**11.3**). There is a need for more information about total area burned in each fire category in the United States and globally, and also a need for additional fire activity data on a broad scale. A complete impacts assessment of biomass combustion emissions requires inventories that include both BC and BrC, along with co-pollutants that may offset warming, and other materials that are implicated in human health and ecosystems impacts. More information on the plume rise of such fires would also improve their representation in chemical transport models.

Regarding actual mitigation measures for biomass burning, there is a need for an analysis of the efficacy of existing and proposed methods, including analysis of total life-cycle impacts. An assessment of the efficacy, and any unintended consequences due to the implementation, of proposed measures for biomass smoke mitigation, beginning with a synthesis of available research results (**11.5.1**, **11.6**) would be especially useful in considering mitigation options. This type of analysis could conceivably include (1) total life-cycle studies of mitigation methods, both individually and in combination with others, for ecosystems impacts, (2) total life-cycle studies of the net climate forcing impacts arising from the use of these methods, or (3) total life-cycle studies of the economic impacts of the use of these methods.

Bibliography

Abt Associates (2008) BenMAP Technical Appendices. Prepared for the Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. Available on the Internet at *http://www.epa.gov/air/benmap/models/BenMAPappendicesSept08.pdf*.

ACIA (2004) Impacts of a warming Arctic: Arctic climate impact assessment. Cambridge University Press.

Ackerman A.S. and Toon O.B. (1981) Absorption of visible radiation in atmosphere containing mixtures of absorbing and nonabsorbing particles. *Applied Optics,* **20**, 3661-3667.

Ackerman A.S., Toon O.B., Stevens D.E., Heymsfield A.J., Ramanathan V., and Welton E.J. (2000) Reduction of tropical cloudiness by soot. *Science,* **288**, 1042-1047.

Adachi K. and Buseck P.R. (2008) Internally mixed soot, sulfates, and organic matter in aerosol particles from Mexico City. *Atmospheric Chemistry and Physics,* **8**, 6469-6481.

Adams T.T., Walsh J., Brown M., Goodrum J., Sellers J., and Das K. (2002) A demonstration of fat and grease as an industrial boiler fuel. University of Georgia Engineering Outreach Service, Athens, GA.

Adar S.D., Gold D.R., Coull B.A., Schwartz J., Stone P.H., and Suh H. (2007) Focused exposures to airborne traffic particles and heart rate variability in the elderly. *Epidemiology,* **18**, 1, 95-103.

Adar S.D. and Kaufman J.D. (2007) Cardiovascular disease and air pollutants: evaluating and improving epidemiological data implicating traffic exposure. *Inhalation Toxicology,* **19**, 135-149.

Adkins E., Tyler E., Wang J., Siriri D., and Modi V. (2010) Field testing and survey evaluation of household biomass cookstoves in rural sub-Saharan Africa. *Energy for Sustainable Development,* **14**, 172-185.

Adler G., Riziq A.A., Erlick C., and Rudich Y. (2010) Effect of intrinsic organic carbon on the optical properties of fresh diesel soot. *Proceedings of the National Academy of Sciences,* **107**, 6699-6704.

Adler T. (2010) Better burning, better breathing: improving health with cleaner cook stoves. *Environmental Health Perspectives,* **118**, 3.

Agrawal H., Welch W.A., Miller J.W., and Cocker D.R. (2008) Emission measurements from a crude oil tanker at sea. *Environmental Science & Technology,* **42**, 7098-7103.

Albrecht B.A. (1989) Aerosols, cloud microphysics, and fractional cloudiness. *Science*, **245**, 1227–30, doi: 10.1126/science.245.4923.1227.

Alexander D.T.L., Crozier P.A., and Anderson J.R. (2008) Brown carbon spheres in East Asian outflow and their optical properties. *Science,* **321**, 833-836.

Alexeeff S.E., Coull B.A., Gryparis A., Suh H., Sparrow D., Vokonas P.S., and Schwartz J. (2011) Medium-term exposure to traffic-related air pollution and markers of inflammation and endothelial function. *Environmental Health Perspectives,* **119**, 4, 481-486.

Alfaro S.C., Lafon S., Rajot L., Formenti P., Gaudichet A., and Maille M. (2004) Iron oxides and light absorption by pure desert dust: An experimental study. *Journal of Geophysical Research,* **109**, D08208, doi: 10.1029/2003JD004374.

Allen C.D., Savage M., Falk D.A., Suckling K.F., Swetnam T.W., Schulke T., Stacey P.B., Morgan P., Hoffman M., and Klingel J.T. (2002) Ecological restoration of southwestern Ponderosa Pine ecosystems: a broad perspective. *Ecological Applications,* **12**, 5, 1418-1433.

Allen J.O., Dookeran N.M., Smith K.A., Sarofim A.F, Taghizadeh K., and Lafleur A.L. (1996) Measurement of polycyclic aromatic hydrocarbons associated with size-segregated Atmospheric Aerosols in Massachusetts. *Environmental Science & Technology,* **30**, 1023–1031.

Amador-Muñoz O., Villalobos-Pietrini R., Agapito-Nadales M.C., Munive-Colín Z., Hernández-Mena L., Sánchez-Sandoval M., Gómez-Arroyo S., Bravo-Cabrera J.L., and Guzmán-Rincón J. (2010) Solvent extracted organic matter and polycyclic aromatic hydrocarbons distributed in size-segregated airborne particles in a zone of México City: Seasonal behavior and human exposure. *Atmospheric Environment,* **44**, 1, 122-130.

AMAP (2009) *Arctic Monitoring and Assessment Programme 2009 update on selected climate issues of concern*, Arctic Monitoring and Assessment Programme, Oslo, Norway (AMAP Technical Report No. 2). Available on the Internet at *http://www.amap.no/assessment/generalpublic.htm*.

American Transportation Research Institute (2011) Compendium of Idling Regulations. Available on the Internet at *http://www.atri-online.org/research/idling/ATRI_Idling_Compendium.pdf*.

Anderson B.E., Cofer W.R., Bagwell D.R., Barrick J.W., Hudgins C.H., and Brunke K.E. (1998) Airborne observations of aircraft aerosol emissions I: total nonvolatile particle emission indices. *Geophysical Research Letters,* **25**, 10, 1689-1692.

Anderson J. and Sherwood T. (2002) Comparison of EPA and other estimates of mobile source rule costs to actual price changes. Society of Automotive Engineers Paper 2002-01-1980.

Andreae M.O. and Merlet P. (2001) Emission of trace gases and aerosols from biomass burning. *Global Biogeochemical Cycles,* **15**, 955- 966.

Andreae M.O. and Gelencsér A. (2006) Black carbon or brown carbon? The nature of light-absorbing carbonaceous aerosols. *Atmospheric Chemistry and Physics,* **6**, 3131-3148.

Andreae M.O., Schmid O., Yang H., Chand D., Yu J.Z., Zeng L.M., and Zhang Y.H. (2008) Optical properties and chemical composition of the atmospheric aerosol in urban Guangzhou, China. *Atmospheric Environment,* **42**, 25, 6335-6350.

Anenberg S.C., Horowitz, L. W., Tong D.Q., and West J.J. (2010) An estimate of the global burden of anthropogenic ozone and fine particulate matter on premature human mortality using atmospheric modeling. *Environmental Health Perspectives,* **118**, 1189-1195.

Anenberg S.C., Talgo K., Arunachalam S., Dolwick P., Jang C., and West J.J. (2011) Impacts of global, regional, and sectoral black carbon emission reductions on surface air quality and human mortality. *Atmospheric Chemistry and Physics,* **11**, 7253-7267.

Archer D., Eby M., Brovkin V., Ridgwell A., Cao L., Mikolajewicz U., Caldeira K., Matsumoto K., Munhoven G., Montenegro A., and Tokos K. (2009) Atmospheric lifetime of fossil fuel carbon dioxide. *Annual Review of Earth and Planetary Sciences,* **37**, 117-134.

Arctic Council (2009) Arctic marine shipping assessment 2009 report. Tromsø, Norway.

Arctic Council (2011) Progress report and recommendations for ministers, Nuuk Greenland. Arctic Council Task Force on Short-Lived Climate Forcers. Available on the Internet at *http://arctic-council.npolar.no/accms/export/sites/default/en/meetings/2011-nuuk-ministerial/docs/3-0a_TF_SPM_recommendations_2May11_final.pdf*.

Arnott W.P., Hamasha K., Moosmuller H., Sheridan P.J., and Ogren J.A. (2005) Towards aerosol light-absorption measurements with a 7-wavelength Aethalometer: Evaluation with a photoacoustic instrument and 3-wavelength nephelometer. *Aerosol Science and Technology,* **39**, 17-29.

Audet P. and Charest C. (2007) Heavy metal phytoremediation from a meta-analytical perspective. *Environmental Pollution,* **147**, 231-237.

Auffhammer M., Ramanathan V., and Vincent J.R. (2006) Integrated model shows that atmospheric brown clouds and greenhouse gases have reduced rice harvests in India. *Proceedings of the National Academy of Sciences,* **103**, 19668-19672.

Aunan K., Berntsen T.K., Myhre G., Rypdal K., Streets D.G., Woo J.-H., and Smith K.R. (2009) Radiative forcing from household fuel burning in Asia. *Atmospheric Environment,* **43**, 35, 5674-5681, doi: 10.1016/j.atmosenv.2009.07.053.

Babich P., Davey M., Allen G., and Koutrakis P. (2000) Method comparisons for particulate nitrate, elemental carbon, and $PM_{2.5}$ mass in seven U.S. cities. *Journal of the Air & Waste Management Association,* **50**, 7, 1095-1105.

Baccarelli A., Wright R.O., Bollati V., Tarantini L., Litonjua A.A., Suh H.H., Zanobetti A., Sparrow D., Vokonas P.S., and Schwartz J. (2009) Rapid DNA methylation changes after exposure to traffic particle. *American Journal of Respiratory and Critical Care Medicine,* **179**, 572-578.

Bachmann J.D. (2007) Will the circle be unbroken: A history of the US national ambient air quality standards. *Journal of the Air & Waste Management Association,* **57**, 6, 652-697.

Bachmann J.D. (2009) Black carbon: A science/policy primer. Pew Center on Global Climate Change, Arlington, VA.

Bae M.-S., Demerjiana K.L., and Schwaba J.J. (2006) Seasonal estimation of organic mass to organic carbon in $PM_{2.5}$ at rural and urban locations in New York State. *Atmospheric Environment,* **40**, 39, 7467-7479.

Bae M.-S., Hong C.-S., Kim Y.J., Han J.-S., Moon K.-J., Kondo Y., Komazaki Y., and Miyazaki Y. (2007) Intercomparison of two different thermal-optical elemental carbons and optical black carbon during ABC-EAREX2005. *Atmospheric Environment,* **41**, 13, 2791-2803, doi: 10.1016/j.atmosenv.2006.11.040.

Bae M.-S., Schauer J.J., Turner J.R., and Hopke P.K. (2009) Seasonal variations of elemental carbon in urban aerosols as measured by two common thermal-optical carbon methods. *Science of the Total Environment,* **407**, 18, 5176-5183, doi: 10.1016/j.scitotenv.2009.05.035.

Bahadur R., Feng Y., Russell L.M., and Ramanathan V. (2011) Impact of California's air pollution laws on black carbon and their implications for direct radiative forcing. *Atmospheric Environment,* **45**, 1162-1167.

Bailis R., Ezzati M., and Kammen D.M. (2005) Mortality and greenhouse gas impacts of biomass and petroleum energy futures in Africa. *Science,* **308**, 98-103, doi: 10.1126/science.1106881.

Baja E.S., Schwartz J.D., Wellenius G.A., Coull B.A., Zanobetti A., Vokonas P.S., and Suh H.H. (2010) Traffic-related air pollution and QT interval: Modification by diabetes, obesity, and oxidative stress gene polymorphisms in the Normative Aging Study. *Environmental Health Perspectives,* **118**, 6, 840-846.

Ban-Weiss G., Cao L., Bala G., and Caldeira K. (2011) Dependence of climate forcing and response on the altitude of black carbon aerosols. *Climate Dynamics*, doi: 10.1007/s00382-011-1052-y.

Barnes and Kumar (2002) Success factors in improved stoves programmes: lessons from six states in India. *Journal of Environmental Studies and Policy,* **5**, 2, 99-112.

Barnett T.P., Adam J.C., and Lettenmaier D.P. (2005) Potential impacts of a warming climate on water availability in snow-dominated regions. *Nature,* **438**, doi: 10.1038/nature04141.

Baron R.E., Montgomery W.D., and Tuladhar S.D. (2009) An analysis of black carbon mitigation as a response to climate change. Prepared by the Copenhagen Consensus on Climate, August. Available on the Internet at *http://fixtheclimate.com/component-1/the-solutions-new-research/*.

Barth M.J. and Tadi R.R. (1998) Emissions comparison between truck and rail: case study of California I-40. *Transportation Research Record* **1520**, 44-52.

Bauer S.E., Menon S., Koch D., Bond T.C., and Tsigaridis K. (2010) A global modelling study on carbonaceous aerosol microphysical characteristics and radiative effects. *Atmospheric Chemistry and Physics,* **10**, 7439-7456.

Bauer S.E. and Menon S. (2012) Aerosol direct, indirect, semi-direct and surface albedo effects from sector contributions based on the IPCC AR5 emissions for pre-industrial and present day conditions. *Journal of Geophysical Research,* **117** (D01206), doi: 10.1029/2011JD016816.

Bell M., Dominici F., Ebisu K., Zeger S., and Samet J. (2007) Spatial and temporal variation in $PM_{2.5}$ chemical composition in the United States for Health Effects Studies. *Environmental Health Perspectives,* **115**, 989-995.

Bell M., Ebisu K., Peng R., Samet J., and Dominici F. (2009) Hospital admissions and chemical composition of fine particle air pollution. *American Journal of Respiratory and Critical Care Medicine,* **179**, 1115-1120.

Bell T.L., Rosenfeld D., Kim K.-M., Yoo J.-M., Lee M.-I., and Hahnenberger M. (2008) Midweek increase in U.S. summer rain and storm heights suggests air pollution invigorates rainstorms. *Journal of Geophysical Research,* **113**, D02209, doi: 10.1029/2007JD008623.

Berntsen T., Fuglestvedt J., Myhre G., Stordal F., and Berglen T.F. (2006) Abatement of greenhouse gases: does location matter? *Climatic Change,* **74**, 4.

Bevan S.L., North P.R.J., Grey W.M.F., Los S.O., and Plummer S.E. (2009) Impact of atmospheric aerosol from biomass burning on Amazon dry-season drought. *Journal of Geophysical Research,* **114**, D09204, doi: 10.1029/2008JD011112.

Birch M.E. and Cary R.A. (1996) Elemental carbon-based method for monitoring occupational exposures to particulate diesel exhaust. *Aerosol Science and Technology,* **25**, 221-241.

Blue Sky Framework (2010) Blue Sky Modeling Framework: website for a modular modeling system that enables fire information, consumption, and smoke modeling Developed by the U.S. Forest Service AirFire Team and Sonoma Technology, Inc., April 1. Available on the Internet at *http://blueskyframework.org/*.

Bluestein J., Rackley J., and Baum E. (2008) Sources and mitigation opportunities to reduce emissions of short-term arctic climate forcers. Technical Report No. 2 prepared by Arctic Monitoring and Assessment Program (AMAP), Oslo, Norway.

Bonazza A., Sabbioni C., and Ghedini N. (2005) Quantitative data on carbon fractions in interpretation of black crusts and soiling on European built heritage. *Atmospheric Environment,* **39**, 14, 2607-2618, doi: 10.1016/j.atmosenv.2005.01.040.

Bond T.C., Anderson T.L., and Campbell D. (1999) Calibration and Intercomparison of filter based measurements of visible light absorption by aerosols. *Aerosol Science and Technology,* **30**, 582-600 (6), doi: 10.1080/027868299304435.

Bond T.C. (2001) Spectral dependence of visible light absorption by carbonaceous particles emitted from coal combustion. *Geophysical Research Letters,* **28**, 21.

Bond T., Streets D., Yarber K., Nelson S., Woo J., and Klimont Z. (2004) A technology based global inventory of black and organic carbon emissions from combustion. *Journal of Geophysical Research,* **109**, doi: 10.1029/2003JD003697.

Bond T.C. and Sun H.L. (2005) Can reducing BC emissions counteract global warming? *Environmental Science & Technology,* **39**, 16.

Bond T.C. and Bergstrom R.W. (2006) Light absorption by carbonaceous particles: an investigative review. *Aerosol Science and Technology* **40**, 1, 27-67.

Bond T., Habib G., and Bergstrom R.W. (2006a) Limitations in the enhancement of visible light absorption due to mixing state. *Journal of Geophysical Research,* **111**, D20211.

Bond T.C., Wehner B., Plewka A., Wiedensohler A., Heintzenberg J., and Charlson R.J. (2006b) Climate-relevant properties of primary particulate emissions from oil and natural gas combustion. *Atmospheric Environment* **40**, 3574-3587.

Bond T.C. (2007) Can warming particles enter global climate discussions? *Environmental Research Letters,* **2**, 4, 1-9, doi: 10.1088/1748-9326/2/4/045030.

Bond T.C., Bhardwaj E., Dong R., Jogani R., Jung S., Roden C., Streets D.G., and Trautmann. N.M. (2007) Historical emissions of black and organic carbon aerosol from energy-related combustion. *Global Biogeochemical Cycles,* **21**, GB2018, 1850-2000.

Bond T.C. (2008) Targeting black carbon for climate reasons: what do we know about emissions, and is it enough to get started? Presentation to *Princeton University Woodrow Wilson School 59e Policy Workshop, Princeton, NJ*.

Bond T.C., Zarzycki C., Flanner M.G., and Koch D.M. (2011) Quantifying immediate radiative forcing by black carbon and organic matter with the Specific Forcing Pulse. *Atmospheric Chemistry and Physics,* **11**, 1505-1525, doi: 10.5194/acp-11-1505-2011.

Boucher O. and Reddy M.S. (2008) Climate trade-off between BC and carbon dioxide emissions. *Energy Policy,* **36**, 193-200.

Bower J., Broughton G., Connolly C., Cook A., Eaton S., Glynn A., Grice S., Kent A., Loader A., Stedman J., Targa J., Telling S., Tsagatakis I., Vincent K., Willis P., Yap F.-w., and Yardley R. (2009) Air Pollution in the UK, 2008. ISBN 978-0-85521-1899, November. Available on the Internet at *http://uk-air.defra.gov.uk/library/annualreport/viewonline?year=2008*.

Box J.E., Bromwich D.H., and Bai L. (2004) Greenland ice sheet surface mass balance 1991 - 2000: Application of Polar MM5 mesoscale model and in situ data. *Journal of Geophysical Research,* **109**, D16105, doi: 10.1029/2003JD004451.

Boy E., Bruce N., and Delgado H. (2002) Birth weight and exposure to kitchen wood smoke during pregnancy in rural Guatemala. *Environmental Health Perspectives,* **110**, 1, 109-114.

Broadway R.M. and Cass R.W. (1975) Fractional efficiency of a utility boiler baghouse: Nucla Generating Plant. EPA-600/2-75-013-a (NITS PB 246 641), August.

Brown S.G., Frankel A., Raffuse S.M., Roberts P.T., Hafner H.R., and Anderson D.J. (2007) Source apportionment of fine particulate matter in Phoenix, Arizona, using positive matrix factorization. *Journal of the Air & Waste Management Association,* **57**, 741-752 (STI-2675), doi: 10.3155/1047-3289.57.6.741.

Bruce N., Perez-Padilla R., and Albalak R. (2000) Indoor air pollution in developing countries: a major environmental and public health challenge for the new millennium. *Bulletin of the World Health Organization,* **78**, 9, 1078-1092.

Buonicore A.J. and W.T. Davis (eds.) (1992) *Air pollution Engineering Manual.* Air & Waste Management Association, Van Nostrand Reinhold, New York, NY.

Burnet P.G., McCrillis R.C., and Morgan S.J. (1988) Performance of certified wood stoves under field conditions. Report for August 1985-September 1987. In *Transactions, PM_{10}: Implementation of Standards*, C.V. Mathai and D.H. Stonefield, eds., Air Pollution Control Association, Pittsburgh, PA, 664-672.

Burt R., Wilson M.A., Mays M.D., and Lee C.W. (2003) Major and trace elements of selected pedons in the USA. *Journal of Environmental Quality,* **32**, 2109-2121.

Butler A.T. (1988) Control of woodstoves by state regulation as a fine particulate control strategy. In *Transactions, PM_{10}: Implementation of Standards*, C.V. Mathai and D.H. Stonefield, eds., Air Pollution Control Association, Pittsburgh, PA, 654-663.

Butry D.T., Prestemon J.P., and Abt K.L. (2010a) Optimal timing of wildfire prevention education. *Ecology and the Environment*, **137**, 197-206.

Butry D.T., Prestemon J.P., Abt K.L., and Sutphen R. (2010b) Economic optimization of wildfire intervention activities. *International Journal of Wildland Fire* **19**, 659-672.

Butterfield D., Beccaceci S., Sweeney B., Green D., Alexander J., and Grieve A. (2010) 2009 annual report for the UK Black Carbon Network. *NPL Report,* **AS-52**, May.

Buzcu-Guven B., Harriss R., and Hertzmark D. (2010) Gas flaring and venting: extent, impacts, and remedies. Prepared for the Energy Forum of the James A. Baker III Institute of Public Policy, Rice University. Available on the Internet at *http://www.rice.edu/energy/publications/docs/Birnur%20Carbon.pdf*.

Cachier H. and Pertuisot M.H. (1994) Particulate carbon in Arctic ice. *Analusis Magazine,* **22**, 34-37.

Cachier H. (1997) Particulate and dissolved carbon in air and snow at the Summit site. In *Transfer of aerosols and gases to Greenland snow and ice*, J.-L. Jaffrezo ed., Cedex, France, 21-27.

Cadle S.H., Groblicki P.J., and Stroup D.P. (1980) Automated carbon analyzer for particulate samples. *Analytical Chemistry,* **52**, 13, 2201-2206.

Cadle S.H., Bellan T., Black K., Carlock M., Graze R., Minasslan F., Murray H., Nam E., Natarajan M., and Lawson D.R. (2006) Real-world vehicle emissions: A summary of the 15th coordinating research council on-road vehicle emissions workshop. *Journal of the Air & Waste Management Association,* **56**, 121-136.

Cakmak S., Dales R.E., Angelica R.M., and Blanco V.C. (2011) The risk of dying on days of higher air pollution among the socially disadvantaged elderly. *Environmental Research,* **111**, 3, 388-393.

California Air Resources Board (2010) Preliminary discussion paper: proposed amendments to California's low-emission vehicle regulations – particulate matter mass, ultrafine solid particle number, and black carbon emissions. *May.*

Cao G., Zhang X., and Zheng F. (2006) Inventory of black carbon and organic carbon emissions from China. *Atmospheric Environment,* **40**, 34, 6516-6527.

Carlton A.G., Wiedinmyer C., and Kross J.H. (2009) A review of Secondary Organic Aerosol (SOA) formation from isoprene. *Atmospheric Chemistry and Physics,* **9**, 4987-5005.

Carmichael G.R., Adhikary B., Kulkarni S., D'Allura A., Tang Y., Streets D., Zhang Q., Bond T., Ramanathan V., Jamroensan A., and Marrapu P. (2009) Asian aerosols: current and year 2030 distributions and implications to human health and regional climate change. *Environmental Science & Technology,* **43**, 15, 5811-5817.

Cass R.W. and Broadway R.M. (1976) Fractional efficiency of a utility boiler baghouse: Sunbury Steam Electric Station. EPA-600/2-76-077a [NTIS PB253 943], March.

CATF (2009a) Agricultural fires and arctic climate change: a special Clean Air Task Force report. May. Available on the Internet at *http://www.catf.us/resources/publications/files/Agricultural_Fires_and_Arctic_Climate_Change.pdf*.

CATF (2009b) The carbon dioxide-equivalent benefits of reducing black carbon emissions from U.S. Class 8 trucks using diesel particulate filters: a preliminary analysis. Prepared by the Clean Air Task Force, September. Available on the Internet at *http://www.catf.us/resources/publications/files/CATF-BC-DPF-Climate.pdf*.

CCSP (2009) *Atmospheric aerosol properties and climate impacts, a report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research*, M. Chin, R.A. Kahn, and S.E. Schwartz, eds., National Aeronautics and Space Administration, Washington, D.C., USA.

Cerveny R.S. and Balling R.C. (1998) Weekly cycles of air pollutants, precipitation, and tropical cyclones in the coastal NW Atlantic region. *Nature,* **394**, 561-563.

Chakrabarty R.K., Moosmuller H., Chen L.-W.A., Lewis K., Arnott W.P., Mazzoleni C., Dubey C.K., Wold C.E., Hao W.M., and Kreidenweis S.M. (2010) Brown carbon in tar balls from smoldering biomass combustion. *Atmospheric Chemistry and Physics,* **10**, 6363-6370, doi: 10.5194/acp-10-6363-2010.

Chameides W.L., Yu H., Liu S.C., Bergin M., Zhou X., Mearns L., Wang G., Kiang C.S., Saylor R.D., Luo C., Huang Y., Steiner A., and Giorgi F. (1999) Case study of the effects of atmospheric aerosols and regional haze on agriculture: an opportunity to enhance crop yields in China through emission controls. *Proceedings of the National Academy of Sciences,* **96**, 13626-13633.

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., Li S.-M., and Moosmuller H. (2010) Observations of OM/OC and specific attenuation coefficients (SAC) in ambient fine PM at a rural site in central Ontario. *Atmospheric Chemistry and Physics,* **10**, 2393-2411, doi: 10.5194/acp-10-2393-2010.

Chan T.W., Brook J.R., Smallwood G.J., and Lu G. (2011) Time-resolved measurements of black carbon light absorption enhancement in urban and near-urban locations of Southern Ontario, Canada. *Atmospheric Chemistry and Physics,* **11**, 10407-10432, doi: 10.5194/acp-11-10407-2011.

Chang O.M.C. and England G.C. (2004) Development of fine particulate emission factors and speciation profiles for oil and gas-fired combustion systems, update: critical review of source sampling and analysis methodologies for characterizing organic aerosol and fine particulate source emission profiles.

Charlson R.J. (1992) Climate forcing by anthropogenic aerosols. *Science,* **255**, 423.

Chen L.-W., Chow J.C., Watson J.G., Moosmuller H., and Arnott W.P. (2004) Modeling reflectance and transmittance of quartz fiber filter samples containing elemental carbon particles: implications for thermal/optical analysis. *Journal of Aerosol Science,* **35**, 765-780.

Chen W.-T., Lee Y.H., Adams P.J., Nenes A., and Seinfeld J.H. (2010) Will BC mitigation dampen aerosol indirect forcing? *Geophysical Research Letters,* **37**, L09801, doi: 10.1029/2010GL042886.

Cheng Y., He K.B., Dua F.K., Zheng M., Ma Y.L., Tan J.H., and Du Z.Y. (2010) Improved measurement of carbonaceous aerosol: evaluation of the sampling artifacts and inter-comparison of the thermal-optical analysis methods, *Atmospheric Chemistry and Physics*, **10**, 8533-8548, doi: 10.5194/acp-10-8533-2010.

Cheng Y., Zheng M., He K.-B., Chen Y., Yan B., Russell A.G., Shi W., Jiao Z., Sheng G., Fu J., and Edgerton E.S. (2011) Comparison of two thermal-optical methods for the determination of organic carbon and elemental carbon: Results from the southeastern United States. *Atmospheric Environment,* **45**, 11, 1913-1918.

Chi K.H., Lin C.Y., Yang C.F., Wang J.L., Lin N.H., Sheu G.R., and Lee C.T. (2010) PCDD/F measurement at a high-altitude station in central Taiwan: evaluation of long-range transport of PCDD/Fs during the Southeast Asia biomass burning event. *Environmental Science & Technology,* **44**, 8, 2954-2960.

Chin M., Diehl T., Ginoux P., and Malm W. (2007) Intercontinental transport of pollution and dust aerosols: implications for regional air quality. *Atmospheric Chemistry and Physics,* **7**, 21, 5501-5517.

Chou C.C.K., Chen W.-N., Chang S.-Y., Chen T.-K., and Huang S.-H. (2005) Specific absorption cross-section and elemental carbon content of urban aerosols. *Geophysical Research Letters,* **32**, L21808, 1-4, doi: 10.1029/2005gl024301.

Chow J.C., Watson J.G., Pritchett L.C., Pierson W.R., Frazier C.A., and Purcell R.G. (1993) The DRI thermal optical reflectance carbon analysis system description, evaluation and applications in United States air quality studies. *Atmospheric Environment,* **27A**, 8, 1185-1201.

Chow J.C., Watson J.G., Doraiswamy P., Chen L.-W.A., Sodeman D.A., Ho S.S.H., Kohl. S.D., Trimble D.L., and Voepel. H. (2006) Climate change - characterization of black carbon and organic carbon air pollution emissions and evaluation of measurement methods, Phase I: method intercomparison. Report prepared for the California Air Resources Board, Sacramento, CA, by the Desert Research Institute, Reno, NV, Report Number DRI 04-307, July 11. Available on the Internet at *http://www.arb.ca.gov/research/apr/past/04-307_v1.pdf*.

Chow J.C., Watson J.G., Chen L.-W.A., Chang M.C.O., Robinson N.F., Trimble D., and Kohl S. (2007) The IMPROVE-A temperature protocol for thermal/optical carbon analysis: maintaining consistency with a long-term database. *Journal of the Air & Waste Management Association,* **57**, 1014-1023.

Chow J.C., Watson J.G., Doraiswamy P., Chen L.-W.A., Sodeman D.A., Lowenthal D.H., Arnott W.P., and Motallebi N. (2009) Aerosol light absorption, black carbon, and elemental carbon measurements at the Fresno Supersite, California. *Atmospheric Research,* **93**, 4, 874-887.

Chow J.C., Watson J.G., Chen L.-W.A., Rice J., and Frank N.H. (2010a) Quantification of $PM_{2.5}$ organic carbon sampling artifacts in US networks. *Atmospheric Chemistry and Physics,* **10**, 5223-5239, doi: 10.5194/acp-10-5223-2010.

Chow J.C., Watson J.G., Green M.C., and Frank N.H. (2010b) Filter light attenuation as a surrogate for elemental carbon. *Journal of the Air & Waste Management Association,* **60**, November.

Chow J.C., Watson J.G., Lowenthal D.H., Chen L.-W.A., and Motallebi N. (2010c) Black and organic carbon emission inventories: Review and application to California. *Journal of the Air & Waste Management Association,* **60**, 4, 497-507.

Chow J.C., Watson J.G., Lowenthal D.H., Chen L.-W., and Motallebi N. (2011) $PM_{2.5}$ source profiles for black and organic carbon emission inventories. *Atmospheric Environment,* **45**, 5407-5414.

Chu S.-H. (2005) Stable estimate of primary OC/EC ratios in the ECtracer method. *Atmospheric Environment,* **39**, 1383-1392.

Chuang K.J., Coull B.A., Zanobetti A., Suh H., Schwartz J., Stone P.H., Litonjua A., Speizer F.E., and Gold D.R. (2008) Particulate air pollution as a risk factor for ST-segment depression in patients with coronary artery disease. *Circulation,* **118**, 1314-1320.

Chung C., Ramanathan V., Kim D., and Podgorny I. (2005) Global anthropogenic aerosol direct forcing derived from satellite and ground-based observations. *Journal of Geophysical Research,* **110**, D24207, doi: 10.1029/2005JD006356.

Chung C.E. and Zhang G.J. (2004) Impact of absorbing aerosol on precipitation: Dynamic aspects in association with convective available potential energy and convective parameterization closure and dependence on aerosol heating profile. *Journal of Geophysical Research,* **109**, D22103, doi: 10.1029/2004JD004726.

Chung S.H. and Seinfeld J.H. (2002) Global distribution and climate forcing of carbonaceous aerosols. *Journal of Geophysical Research,* **107**, D19.

Chung S.H. and Seinfeld J.H. (2005) Climate response of direct radiative forcing of anthropogenic black carbon. *Journal of Geophysical Research,* **110**, D11102, doi: 10.1029/2004JD005441.

Chylek P., Srivastava V., Cahenzli L., Pinnick R.G., Dod R.L., Novakov T., Cook T.L., and Hinds B.D. (1987) Aerosol and graphitic carbon content of snow. *Journal of Geophysical Research,* **92**, 9801-9809.

Chylek P., Johnson B., and Wu H. (1992) Black carbon concentration in a Greenland Dye-3 ice core. *Geophysical Research Letters,* **19**, 1951-1953.

Chylek P., Johnson B., Damiano P.A., Taylor K.C., and Clement P. (1995) Biomass Burning Record and Black Carbon in the GISP2 Ice Core. *Geophysical Research Letters,* **22**, 89-92, doi: 10.1029/94GL02841.

Chylek P., Kou L., Johnson B., Boudala F., and Lesins G. (1999) Black carbon concentrations in precipitation and near surface air in and near Halifax, Nova Scotia. *Atmospheric Environment,* **33**, 2269-2277.

Clark N., Demers P., Karr C., Koehoorn M., Lencar C., Tamburic L., and Brauer M. (2010) Effect of early life exposure to air pollution on development of childhood asthma. *Environmental Health Perspectives,* **118**, 2, 284-290, doi: 10.1289/ehp.0900916.

Clarke A.D. and Noone K.J. (1985) Soot in the Arctic snowpack: A cause for perturbations in radiative transfer. *Atmospheric Environment,* **19**, 2045-2053.

Clarke L., Edmonds J., Jacoby H., Pitcher H., Reilly J., and Richels R. (2007) Scenarios of greenhouse gas emissions and atmospheric concentrations, sub-report 2.1A. In *Synthesis and Assessment Product 2.1*, U.S. Climate Change Science Program and the Subcommittee on Global Change Research. Department of Energy, Office of Biological & Environmental Research, Washington, D.C., 154.

Cofala J., Amann M., Klimont Z., Kupiainen K., and Hoglund-Isaksson L. (2007) Scenarios of global anthropogenic emissions of air pollutants and methane until 2030. *Atmospheric Environment,* **41**, 8486-8499, doi: 10.1016/j.atmosenv.2007.07.010.

Cohen A.J., Anderson H.R., Ostro B., Pandey K.D., Krzyzanowski M., Kunzli N., K. G., Pope III C.A., Romieu I., Samet J.M., and Smith K.R. (2004) Urban air pollution. In *Comparative quantification of health risks: global and regional burden of disease due to selected major risk factors*, M. Ezzati, A.D. Lopez, A. Rodgers, and C.J.L. Murray, eds., World Health Organization, Geneva, 1353-1434.

Collaud Coen M., Weingartner E., Apituley A., Ceburnis D., Fierz-Schmidhauser R., Flentje H., Henzing J.S., Jennings S.G., Moerman M., Petzold A., Schmid O., and Baltensperger U. (2010) Minimizing light absorption measurement artifacts of the Aethalometer: evaluation of five correction algorithms. *Atmospheric Measurement Techniques,* **3**, 457-474, doi: 10.5194/amt-3-457-2010.

Cooke W.F. and Wilson J.J.N. (1996) A global black carbon aerosol model. *Journal of Geophysical Research,* **101**, D14, 19395-19409.

Cooke W.F., Liousse C., Cachier H., and Feichter J. (1999) Construction of a 1° × 1° fossil fuel emission data set for carbonaceous aerosol and implementation in the ECHAM4 model. *Journal of Geophysical Research,* **104**, 22,137-122,162 (D18), doi: 10.1029/1999JD900187

Coordinating Research Council (2008) Kansas City characterization study. Final Report prepared for the Environmental Protection Agency, Project E69. Based on EPA Contract Report (ERG No. 0133.18.007.001).

Corbett J.J., Lack D.A., Winebrake J.J., Harder S., Silberman J.A., and Gold M. (2010) Arctic shipping emissions inventories and future scenarios. *Atmospheric Chemistry and Physics,* **10**, 19, 9689, doi: 10.5194/acp-10-9689-2010.

Cross E.S., Onasch T.B., Ahern A., Wrobel W., Slowik J.G., Olfert J., Lack D.A., Massoli P., Cappa C.D., Schwarz J.P., Spackman J.R., Fahey D.W., Sedlacek A., Trimborn A., Jayne J.T., Freedman A., Williams L.R., Ng N.L., Mazzoleni C., Dubey M., Brem B., Kok G., Subramanian R., Freitag S., Clarke A., Thornhill D., Marr L.C., Kolb C.E., Worsnop D.R., and Davidovits P. (2010) Soot particle studies—instrument inter-comparison—project overview. *Aerosol Science and Technology,* **44**, 8, 592-611, doi: 10.1080/02786826.2010.482113.

Curry J.A. (1995) Interactions among aerosols, clouds, and climate of the Arctic Ocean. *Science of the Total Environment,* **160/161**, 777-791.

Daniel J.S., Solomon S., Sanford T.J., McFarland M., Fuglestvedt J., and Friedlingstein P. (2011) Limitations of single-basket trading: lessons from the Montreal Protocol for climate policy. *Climatic Change*, doi: 10.1007/s10584-011-0136-3.

DeBell L.J. (2006) Spatial and seasonal patterns and temporal variability of haze and its constituents in the United States. Report IV. IMPROVE report by Colorado State University, Fort Collins, CO, November. Available on the Internet at *http://vista.cira.colostate.edu/improve/publications/Reports/2006/PDF/IMPROVE_Report_IV.pdf*.

Delfino R.J., Staimer N., Gillen D., Tjoa T., Sioutas C., Fung K., George S.C., and Kleinman M.T. (2006) Personal and ambient air pollution is associated with increased exhaled nitric oxide in children with asthma. *Environmental Health Perspectives,* **114**, 1736-1743.

Delfino R.J., Staimer N., Tjoa T., Polidori A., Arhami M., Gillen D.L., Kleinman M.T., Vaziri N.D., Longhurst J., Zaldivar F., and Sioutas C. (2008) Circulating biomarkers of inflammation, antioxidant activity, and platelet activation are associated with primary combustion aerosols in subjects with coronary artery disease. *Environmental Health Perspectives,* **116**, 7, 898-906.

Delfino R.J., Staimer N., Tjoa T., Gillen D.L., Polidori A., Arhami M., Kleinman M.T., Vaziri N.D., Longhurst J., and Sioutas C. (2009) Air pollution exposures and circulating biomarkers of effect in a susceptible population: clues to potential causal component mixtures and mechanisms. *Environmental Health Perspectives,* **117**, 1232-1238.

Delfino R.J., Tjoa T., Gillen D.L., Staimer N., Polidori A., Arhami M., Jamner L., Sioutas C., and Longhurst J. (2010) Traffic-related air pollution and blood pressure in elderly subjects with coronary artery disease. *Epidemiology,* **21**, 3, 396-404, doi: 10.1097/EDE.0b013e3181d5e19b.

Delfino R.J., Gillen D.L., Tjoa T., Staimer N., Polidori A., Arhami M., Sioutas C., and Longhurst J. (2011) Electrocardiographic ST segment depression and exposure to traffic-related aerosols in elderly subjects with coronary artery disease. *Environmental Health Perspectives,* **119**, 2, 196-202.

Delucchi M.A. and Jacobson M.Z. (2011) Providing all global energy with wind, water, and solar power, Part II: reliability, system and transmission costs, and policies. *Energy Policy,* **39**, 3, 1170-1190.

Denman K.L., Brasseur G., Chidthaisong A., Ciais P., Cox P.M., Dickinson R.E., Hauglustaine D., Heinze C., Holland E., Jacob D., Lohmann U., Ramachandran S., Dias P.L.d.S., Wofsy S.C., and Zhang X. (2007) Couplings between changes in the climate system and biogeochemistry. In *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, S. Soloman, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller, eds., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Derwent R.G., Ryall D.B., Jennings S.G., Spain T.G., and Simmonds P.G. (2001) Black Carbon Aerosol and Carbon Monoxide in European Regionally Polluted Air Masses at Mace Head, Ireland during 1995-1998. *Atmospheric Environment,* **35**, 6371-6378.

DeWinter J.L., Raffuse S.M., Larkin N.K., Strand T.T., Brown S.G., Craig K.J., and Roberts P.T. (2011) Mitigating the impact of prescribed burning in the continental United States using trends in synoptic scale transport to the Arctic region. *Air & Waste Management Association Greenhouse Gas Strategies in a Changing Climate, San Francisco, CA, November 16-17* (STI-4148). Available on the Internet at *http://events.awma.org/GHG2011/ghg2011.html*.

Dickerson R.R., Andreae M.O., Campos T., Mayol-Bracero O.L., Neusuess C., and Streets D.G. (2002) Analysis of black carbon and carbon monoxide observed over the Indian Ocean: implications for emissions and photochemistry. *Journal of Geophysical Research,* **107**, D19, 8017, doi: 10.1029/2001JD000501.

Diesel Technology Forum (2006) Retrofitting America's diesel engines. *November*. Available on the Internet at *http://www.dieselforum.org/news-center/pdfs/Retrofitting-America-s-Diesel-Engines-11-2006.pdf/view*.

Diesel Technology Forum (2007) CMAQ funded diesel retrofit projects. *April*. Available on the Internet at *http://www.dieselforum.org/news-center/pdfs/CMAQ-web.pdf/view*.

Dillner A.M., Stein C., Larson S.M., and Hitzenberger R. (2001) Measuring the mass extinction efficiency of elemental carbon in rural aerosol. *Aerosol Science and Technology*, **35**, 6, 1009-1021.

Dockery D.W., Luttmann-Gibson H., Rich D.Q., Link M.S., Mittleman M.A., Gold D.R., Koutrakis P., Schwartz J.D., and Verrier R.L. (2005) Association of air pollution with increased incidence of ventricular tachyarrhythmias recorded by implanted cardioverter defibrillators. *Environmental Health Perspectives,* **113**, 670-674.

Doherty S.J., Warren S.G., Grenfell T.C., Clarke A.D., and Brandt R.E. (2010) Light-absorbing impurities in Arctic snow. *Atmospheric Chemistry and Physics,* **10**, 11647-11680.

Dopelheuer A. (2001) Quantities, characteristics and reduction potentials of aircraft engine emissions. By the Society of Automotive Engineers, Technical Paper 2001-01-3008.

Dubovik O. and King M.D. (2000) A flexible inversion algorithm for retrieval of aerosol optical properties from Sun and sky radiance measurements. *Journal of Geophysical Research,* **105**, 20673-20696.

Dubowsky S.D., Suh H., Schwartz J., Coull B.A., and Gold D.R. (2006) Diabetes, obesity, and hypertension may enhance associations between air pollution and markers of systemic inflammation. *Environmental Health Perspectives,* **114**, 992-998.

Dukan M. (2010) Mitigating industrial black carbon through energy recycling. *Climate Alert,* **20**, 2.

Dusek U., Frank G.P., Hildebrandt L., Curtius J., Schneider J., Walter S., Chand D., Drewnick F., Hings S., Jung D., Borrmann S., and Andreae M.O. (2006) Size matters more than chemistry for cloud-nucleating ability of aerosol particles. *Science,* **312**, 5778, 1375-1378.

Dwyer J.G., Norris J.R., and Ruckstuhl C. (2010) Do climate models reproduce observed solar dimming and brightening over China and Japan? *Journal of Geophysical Research,* **115**, D00k08, doi: 10.1029/2009JD012945.

ESMAP (2004) Toward cleaner urban air in South Asia: tackling transport pollution, understanding sources. Report by the World Bank Energy Sector Management Assistance Programme (ESMAP), ESMAP Report 281/04, March. Available on the Internet at *http://www.worldbank.org/sarubanair*.

Ezzati M., Rodgers A., Lopez A.D., Hoorn S.V., and Murray C.J.L. (2004) Mortality and burden of disease attributable to individual risk factors. In *Comparative Quantification of Health Risks: Global and Regional Burden of Disease Due to Selected Major Risk Factors*, M. Ezzati, A.D. Lopez, A. Rodgers, and C.J.L. Murray, eds., World Health Organization, Geneva, Switzerland, 2141-2166.

Falke S.R., Husar R.B., and Schichtel B.A. (2001) Fusion of SeaWiFS and TOMS satellite data with surface observations and topographic data during extreme aerosol events. *Journal of the Air & Waste Management Association,* **51**, 11, 1579-1585.

Fann N., Fulcher C., and Hubbell B. (2009) The influence of location, source, and emission type in estimates of the human health benefits of reducing a ton of air pollution. *Air Quality, Atmosphere & Health,* **2**, 3, 169-176, doi: 10.1007/s11869-009-0044-0.

Fann N., Lamson A.D., Anenberg S.C., Wesson K., Risley D., and Hubbell B.J. (2011) Estimating the national public health burden associated with exposure to ambient $PM_{2.5}$ and ozone. *Risk Analysis*, **32**, 1, 81-95, doi: 10.1111/j.1539-6924.2011.01630.x.

Fann N. and Risley D. (2011) The public health context for $PM_{2.5}$ and ozone air quality trends. *Air Quality, Atmosphere & Health,* **4**, 1-11, doi: 10.1007/s11869-010-0125-0.

Favez O., Alfaro S.C., Sciare J., Cachier H., and Abdelwahab M.M. (2009) Ambient measurements of light-absorption by agricultural waste burning organic aerosols. *Journal of Aerosol Science,* **40**, 7, 613-620.

Feingold G., Jiang H., and Harrington J.Y. (2005) On smoke suppression of clouds in Amazonia. *Geophysical Research Letters,* **32**, L02804, doi: 10.1029/2004GL021369.

Field C.B. (2007) Contribution of working group II to the fourth assessment report of the intergovernmental panel on climate change. In *Climate Change 2007: Impacts, Adaptation and Vulnerability*, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden, and C.E. Hanson, eds., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Flanner M.G., Zender C.S., Randerson J.T., and Rasch P.J. (2007) Present-day climate forcing and response from BC in snow. *Journal of Geophysical Research,* **112**, D11, doi: 10.1029/2006JD008003.

Flanner M.G., Zender C.S., Hess P.G., Mahowald N.M., Painter T.H., Ramanathan V., and Rasch P.J. (2009) Springtime warming and reduced snow cover from carbonaceous particles. *Atmospheric Chemistry and Physics,* **9**, 2481-2497.

Flanner M.G., Shell K.M., Barlage M., Perovich D.K., and Tschudi M.A. (2011) Radiative forcing and albedo feedback from the Northern Hemisphere cryosphere between 1979 and 2008. *Nature Geoscience*, **4**, 151-155, doi: 10.1038/ngeo1062.

Food and Agriculture Organization of the United Nations (2007) Fire management – global assessment 2006. Rome, Italy, FAO Forestry Paper 151.

Forster P., Ramaswamy V., Artaxo P., Berntsen T., Betts R., Fahey D.W., Haywood J., Lean J., Lowe D.C., Myhre G., Nganga J., Prinn R., Raga G., Schulz M., and R. Van Dorland (2007) Changes in atmospheric constituents and in radiative forcing. In *Climate Change 2007: the Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller, eds., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Frank N. (2006) Carbon Measurements and Adjustments. Presented at Air Quality Data in Health Effects Research. Available on the Internet at *http://vista.cira.colostate.edu/improve/Publications/Workshops/Carbon_Jan2008/Carbon_Measurements_and_Adjustments_Frank.ppt*.

French H.W. (2007) Reports of forced labor at brick kilns unsettle China. *New York Times,* **June 16**.

Frenklach M. (2002) Reaction mechanism of soot formation in flames. *Physical Chemistry Chemical Physics,* **4**, 11, 2028-2037, doi: 10.1039/B110045A.

Fu Q. and Sun W. (2006) Apparent optical properties of spherical particles in absorbing medium. *Journal of Quantitative Spectroscopy and Radiative Transfer,* **100**, 1-3, 137-142, doi: 10.1016/j.jqsrt.2005.11.031.

Fuglestvedt J. (2009) Impacts of metric choice on analyzing climate effects of emissions. IPCC expert meeting on the science of alternative metrics, The Grand Hotel, Oslo, Norway, 18-20 March 2009, 75 pages, G.-K. Plattner, T.F. Stocker, P. Midgley, and M. Tignor, eds., IPCC Working Group I Technical Support Unit, University of Bern, Bern, Switzerland (ISBN 978-92-9169-126-5). Available on the Internet at *http://www.ipcc.ch/pdf/supporting-material/expert-meeting-metrics-oslo.pdf*.

Fuglestvedt J.S., Berntsen T.K., Godal O., Sausen R., Shine K.P., and Skodvin T. (2003) Metrics of climate change: assessing radiative forcing and emission indices. *Climatic Change,* **58**, 267-331.

Fuglestvedt J.S., Shine K.P., Berntsen T., Cook J., Lee D.S., Stenke A., Skeie R.B., Velders G.J.M., and Waitz I.A. (2010) Transport impacts on atmosphere and climate: metrics. *Atmospheric Environment,* **44**, 37, 4648-4677, doi: 10.1016/j.atmosenv.2009.04.044.

Fujino J., Nair R., Kainuma M., Masui T., and Matsuoka Y. (2006) Multi-gas mitigation analysis on stabilization scenarios using AIM global model. *The Energy Journal,* Special Issue Number 3: Multigas Mitigation and Climate Policy, 343-354.

Fujita E.M., Campbell D.E., Arnott W.P., Chow J.C., and Zielinska B. (2007) Evaluations of the chemical mass balance method for determining contributions of gasoline and diesel exhaust to ambient carbonaceous aerosols. *Journal of the Air & Waste Management Association,* **57**, 6, 721-740.

Fuller K.A., Malm W.C., and Kreidenweis S.M. (1999) Effects of mixing on extinction by carbonaceous particles. *Journal of Geophysical Research,* **104**, D13, 15941-15954, doi: 10.1029/1998jd100069.

Gaines L., Vyas A., and Anderson J. (2006) Estimation of fuel use by idling commercial trucks. Paper 06-2567: *85th Annual Meeting of the Transportation Research Board*, Washington, D.C.

Galbavy E.S., Anastasio C., Lefer B.L., and Hall S.R. (2007) Light penetration in the snowpack at Summit, Greenland: part 1, nitrite and hydrogen peroxide photolysis. *Atmospheric Environment,* **41**, 5077-5090.

Gan T.H., Hanhela P., Mazurek W., and Gillett R. (2010) Characteristics of submarine engine diesel particulates in the maritime environment. *Journal of Aerosol Science,* **41**, 1, 23-35, doi: 10.1016/j.jaerosci.2009.09.007.

Gan W., Koehoorn M., Davies H., Demers P., Tamburic L., and Brauer M. (2011) Long-term exposure to traffic-related air pollution and the risk of coronary heart disease hospitalization and mortality. *Environmental Health Perspectives,* **119**, 501-507 (4).

Garrett T.J., Zhao C., Dong X., Mace G.G., and Hobbs P.V. (2004) Effects of varying aerosol regimes on low-level Arctic stratus. *Geophysical Research Letters,* **31**, L17105 (17), doi: 10.1029/2004GL019928.

Gelencsér A. (2004) *Carbonaceous aerosol*, Springer, Dordrecht, The Netherlands.

Generoso S., Bey I., Attie J.-L., and Breon F.-M. (2007) A satellite- and model-based assessment of the 2003 Russian fires: Impact on the Arctic region. *Journal of Geophysical Research,* **112**, D15302, doi: 10.1029/2006JD008344.

Godleski J.J., Clarke R.W., Coull B.A., Saldiva P.H.N., Jiang N.F., Lawrence J., and Koutrakis P. (2002) Composition of inhaled urban air particles determines acute pulmonary responses. *The Annals of Occupational Hygiene,* **46**, 419-424.

Gold D.R., Litonjua A.A., Zanobetti A., Coull B.A., Schwartz J., MacCallum G., Verrier R.L., Nearing B.D., Canner M.J., Suh H., and Stone P.H. (2005) Air pollution and ST-segment depression in elderly subjects. *Environmental Health Perspectives,* **113**, 883-887.

Gomot-de Vaufleury A. and Kerhoas I. (2000) Effects of cadmium on the reproductive system of the land snail Helix aspersa. *Bulletin of Enviornmental Contamination and Toxicology,* **64**, 434-442.

Graber E.R. and Rudich Y. (2006) Atmospheric HULIS: how humic-like are they? A comprehensive and critical review. *Atmospheric Chemistry and Physics,* **6**, 729-753.

Graham S., Bailis R., Charron D. (2005) Household energy, indoor air pollution and health: overview of experiences and lessons in China. 1-108. Center for Entrepreneurship in International Health and Development, School of Public Health, University of California, Berkeley.

Grannas A.M., Shepson P.B., and Filley T.R. (2004) Photochemistry and nature of organic matter in Arctic and Antarctic snow. *Global Biogeochemical Cycles,* **18**, GB1006, doi: 10.1029/2003GB002133.

Gray H. (1986) Control of Atmospheric fine primary carbon particle concentrations. Available on the Internet at *http://resolver.caltech.edu/CaltechEQL:EQL-R-23*.

Grenfell T.C., Perovich D.K., and Ogren J.A. (1981) Spectral albedos of an alpine snow pack. *Cold Regions Science and Technology,* **4**, 121-127.

Grenfell T.C., Warren S.G., and Mullen P.C. (1994) Reflection of solar radiation by the Antarctic snow surface at ultraviolet, visible, and near-infrared wavelengths. *Journal of Geophysical Research,* **99**, 18669-18684.

Grenfell T.C., Light B., and Sturm M. (2002) Spatial distribution and radiative effects of BC in the snow and sea ice during the SHEBA experiment. *Journal of Geophysical Research,* **107**, C10, 8032.

Grieshop A.P., Reynolds C.C.O., Kandlikar M., and Dowlatabadi H. (2009) A black-carbon mitigation wedge. *Nature Geoscience,* **2**, 533-534, doi: 10.1038/ngeo595.

Griffin J.J. and Goldberg E.D. (1983) Impact of fossil fuel combustion on sediments of Lake Michigan: a reprise. *Environmental Science & Technology,* **17**, 244-245, doi: 10.1021/es00110a013.

Guan H., Esswein R., Lopez J., Bergstrom R., A. Warnock, Follette-Cook M., Fromm M., and Iraci L.T. (2010) A multi-decadal history of biomass burning plume heights identified using aerosol index measurements. *Atmospheric Chemistry and Physics,* **10**, 6461-6469, doi:10.5194/acp-10-6461-2010.

Gundel L.A., R.L. D., Rosen H., and Novakov T. (1984) The relationship between optical attenuation and black carbon concentration for ambient and source particles. *The Science of the Total Environment,* **36**, 197-202.

Gupta J. (2003) Informal labour in brick kilns: need for regulation. *Economic and Political Weekly,* **38**, 31, 3282-3292.

Hadley O., Ramanathan V., Carmichael G.R., Tang Y., Corrigan C.E., Roberts G.C., and Mauger G.S. (2007) Trans-Pacific transport of black carbon and fine aerosol (d < 2.5 mm) into North America. *Journal of Geophysical Research,* **112**, D05309, doi: 10.1029/2006JD007632.

Hadley O.L., Corrigan C.E., Kirchstetter T.W., Cliff S.S., and Ramanathan V. (2010) Measured black carbon deposition on the Sierra Nevada snow pack and implication for snow pack retreat, *Atmospheric Chemistry and Physics*, **10**, 7505-7513, doi: 10.5194/acp-10-7505-2010.

Hagler G.S.W., Bergin M.H., Smith E.A., and Dibb J.E. (2007a) A summer time series of particulate carbon in the air and snow at Summit, Greenland. *Journal of Geophysical Research,* **112**, D21309, doi: 10.1029/2007JD008993.

Hagler G.S.W., Bergin M.H., Smith E.A., Dibb J.E., Anderson C., and Steig E.J. (2007b) Particulate and water-soluble carbon measured in recent snow at Summit, Greenland. *Geophysical Research Letters,* **34**, L16505, doi: 10.1029/2007GL030110.

Hamilton R.S. and Mansfield T.A. (1991) Airborne particulate elemental carbon: its sources, transport and contribution to dark smoke and soiling. *Atmospheric Environment,* **25**, 715-723.

Hammer C.U., Clausen H.B., Dansgaard W., Gundestrup N., Johnsen S.J., and Reeh N. (1978) Dating of Greenland ice cores by flow models, isotopes, volcanic debris, and continental dust. *Journal of Glaciology,* **20**, 82, 3-26.

Hand J.L., Copeland S.A., Day D.E., Dillner A.M., Indresand H., Malm W.C., McDade C.E., Moore Jr. C.T.T., Pitchford M.L., Schichtel B.A., and Watson J.G. (2011) Spatial and seasonal patterns and temporal variability of haze and its constituents in the United States. Report V: interagency monitoring of protected visual environments. IMPROVE report by Colorado State University, Fort Collins, CO, June. Available on the Internet at *http://vista.cira.colostate.edu/improve/publications/Reports/2011/PDF/Cover_TOC.pdf*.

Hanna E., Huybrechts P., Janssens I., Cappelen J., Steffen K., and Stephens A. (2005) Runoff and mass balance of the Greenland ice sheet: 1958-2003. *Journal of Geophysical Research,* **110**, D13108, doi: 10.1029/2004JD005641.

Hansen A.D.A., Rosen H., and Novakov T. (1982) Real-time measurement of the absorption coefficient of aerosol particles. *Applied Optics,* **21**, 3060–3062.

Hansen J. and Nazarenko L. (2004) Soot climate forcing via snow and ice albedos. *Proceedings of the National Academy of Sciences,* **101**, 423-428, doi: 10.1073/pnas.2237157100.

Hansen J., Sato M., Ruedy R., and others (2005) Efficacy of climate forcings. *Journal of Geophysical Research,* **110**, D18104, doi: 10.1029/2005JD005776.

Hansen J., Sato M., Kharecha P., Russell G., Lea D.W., and Siddall M. (2007a) Climate change and trace gases. *Philosophical Transactions of the Royal Society A,* **365** (1856), 1925-1954, doi: 10.1098/rsta.2007.2052.

Hansen J., Sato M., Ruedy R., Kharecha P., Lacis A., Miller R., Nazarenko L., Lo K., Schmidt G.A., Russell G., Aleinov I., Bauer S., Baum E., Cairns B., Canuto V., Chandler M., Cheng Y., Cohen A., Del Genio A., Faluvegi G., Fleming E., Friend A., Hall T., Jackman C., Jonas J., Kelley M., Kiang N.Y., Koch D., Labow G., Lerner J., Menon S., Novakov T., Oinas V., Perlwitz J., Perlwitz J., Rind D., Romanou A., Schmunk R., Shindell D., Stone P., Sun S., Streets D., Tausnev N., Thresher D., Unger N., Yao M., and Zhang S. (2007b) Climate simulations for 1880-2003 with GISS modelE. *Climate Dynamics,* **29**, 661–696.

Hansen J., Sato M., Kharecha P., and von Schuckmann K. (2011) Earth's energy imbalance and implications. *Atmospheric Chemistry and Physics,* **11**, 13421-13449, doi: 10.5194/acp-11-13421-2011.

Haynie F.H. (1986) Environmental factors affecting corrosion of weathering steel. In *Materials Degradation Caused by Acid Rain: Developed from the 20th State-of-the-Art Symposium of the American Chemical Society, June 1985, Arlington, VA*., ACS symposium series 318, 163-171, R. Baboian, ed., American Chemical Society, Washington, D.C.

Hays M.D., Smith N.D., Kinsey J.S., Dong Y., and Kariher P. (2003) A polycyclic aromatic hydrocarbon (PAH) size distributions in aerosols from appliances of residential wood combustion as determined by direct thermal desorption. *Journal of Aerosol Science,* **34**, 1061-1084.

Health Effects Institute (2010) Traffic-related air pollution: a critical review of the literature on emissions, exposure, and health effects. Special report, 17, available on the Internet at *http://pubs.healtheffects.org/view.php?id=334*.

Hecobian A., Zhang X., Zheng M., Frank N., Edgerton E.S., and Weber R.J. (2010) Water-soluble organic aerosol material and the light-absorption characteristics of aqueous extracts measured over the Southeastern United States. *Atmospheric Chemistry and Physics,* **10**, 5965–5977, doi: 10.5194/acp-10-5965-2010.

Hedman B., Naslund M., and Marklund S. (2006) Emission of PCDD/F, PCB, and HCB from combustion of firewood and pellets in residential stoves and boilers. *Environmental Science & Technology,* **40**, 4968-4975.

Hegerl G.C., Zwiers F.W., Braconnot P., Gillett N.P., Luo Y., Orsini J.A.M., Nicholls N., Penner J.E., and Stott P.A. (2007) Understanding and attributing climate change. In *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller, eds., Cambridge University Press, Cambridge, United Kingdom and New York, New York, USA.

Hegg D.A., Warren S.G., Grenfell T.C., Doherty S.J., Larson T.V., and Clarke A.D. (2009) Source attribution of black carbon in arctic snow. *Environmental Science & Technology,* 43, 4016–4021.

Hegg D.A., Warren S.G., Grenfell T.C., Doherty S.J., and Clarke A.D. (2010) Sources of light-absorbing aerosol in arctic snow and their seasonal variation. *Atmospheric Chemistry and Physics,* **10**, 10923–10938, doi: 10.5194/acp-10-10923-2010.

Heierli U. and Maithel S. (2008) Brick by brick: the herculean task of cleaning up the Asian brick industry. Swiss Agency for Development and Cooperation, February. Available on the Internet at *http://www.poverty.ch/documents/brick.pdf*.

Hijioka Y., Matsuoka Y., Nishimoto H., Masui M., and Kainuma M. (2008) Global GHG emissions scenarios under GHG concentration stabilization targets. *Journal of Global Environmental Engineering,* **13**, 97-108.

Hinckley J. and Doshi K. (2010) Emission controls for wood-fired boilers. Prepared by the U.S. Forest Service Biomass Energy Resource Center, Montpelier, VT, May.

Hirdman D., Burkhart J.F., Sodemann H., Eckhardt S., Jefferson A., Quinn P.K., Sharma S., Strom J., and Stohl A. (2010) Long-term trends of black carbon and sulphate aerosol in the Arctic: changes in atmospheric transport and source region emissions. *Atmospheric Chemistry and Physics,* **10**, 9351-9368, doi: 10.5194/acp-10-9351-2010.

Hitzenberger R., Petzold A., Bauer H., Ctyroky P., Pouresmaeil P., Laskus L., and Puxbaum H. (2006) Intercomparison of thermal and optical measurement methods for elemental carbon and black carbon at an urban location. *Environmental Science & Technology,* **40**, 20, 6377-6383.

Holben B.N., Eck T.F., Slutsker I., Tanre D., Buis J.P., Setzer A., Vermote E., Reagan J.A., Kaufman Y.J., Nakjima T., Lavenu F., Jankowiak I., and Smirnov A. (1998) AERONET – A federated instrument network and data archive for aerosol characterization. *Remote Sensing of the Environment,* **66**, 1-16.

Holguin F. (2008) Traffic, outdoor air pollution, and asthma. *Immunology and Allergy Clinics of North America,* **28**, 3, 577-588, doi: 10.1016/j.iac.2008.03.008.

Holland M., Bitz C., Tremblay B., and Bailey D. (2008): The role of natural versus forced change in future rapid summer Arctic ice loss. In *Arctic Sea Ice Decline: Observations, Projections, Mechanisms, and Implications*, E. DeWeaver, C. Bitz, and L.-B. Tremblay (eds). *Geophysical Monograph Series*, **180**, American Geophysical Union, 133-150.

Hopke P.K., Ito K., Mar T., Christensen W.F., Eatough D.J., Henry R.C., Kim E., Laden F., Lall R., Larson T.V., Liu H., Neas L., Pinto J., Stolzel M., Suh H., Paatero P., and Thurston G.D. (2006) PM source apportionment and health effects: 1. Intercomparison of source apportionment results. *Journal of Exposure Science and Environemntal Epidemiology,* **16**, 3, 275-286, doi: 10.1038/sj.jea.7500458.

Hordijk L., Foell W., and Shah J. (1995) RAINS-ASIA: An Assessment Model For Acid Deposition In Asia. Available on the Internet at *http://www.iiasa.ac.at/~rains/asia1/chapter1.pdf*.

Horvath H. (1993) Atmospheric light absorption: a review. *Atmospheric Environment,* **27A**, 3, 293-317, doi: 10.1016/0960-1686(93)90104-7.

Hough M.L. (1988) Oregon approach to reducing residential woodsmoke as part of the PM_{10} strategy. In *Transactions, PM_{10}: Implementation of Standards*, C.V. Mathai and D.H. Stonefield, eds., Air Pollution Control Association, Pittsburgh, PA, 654-663.

Huang J., Fu Q., Zhang W., Wang X., Zhang R., Ye H., and Warren S. (2011) Dust and black carbon in seasonal snow across northern China. *Bulletin of the American Meteorological Society,* **92**, 175-181.

Huang Y.-C.T., Ghio A.J., Stonehuerner J., McGee J., Carter J.D., Grambow S.C., and Devlin R.B. (2003) The role of soluble components in ambient fine particles-induced changes in human lungs and blood. *Inhalation Toxicology,* **15**, 327-342.

Hubbell B.J., Fann N., and Levy J.I. (2009) Methodological considerations in developing local-scale health impact assessments: Balancing national, regional and local data. *Air Quality, Atmosphere, and Health,* **2**, 99-110.

Huffman G.P., Huggins F.E., Shah N., Huggins R., Linak W.P., Miller C.A., Pugmire R.J., Meuzelaar H.L.C., Seehra M.S., and Manivannan A. (2000) Characterization of Fine Particulate Matter Produced by Combustion of Residual Fuel Oil. *Journal of the Air and Waste Management Association,* **50**, 1106-1114.

Huntzicker J.J., Johnson R.L., Shah J.J., and Cary R.A. (1982) Analysis of organic and elemental carbon in ambient aerosols by a thermal-optical method. In *Particulate Carbon: Atmospheric Life Cycle*, G.T. Wolff and R.L. Klimisch, eds., Plenum Press, New York, New York, 79-88.

Husain L., Dutkiewicz V.A., Khan A.J., and Ghauri B.M. (2007) Characterization of carbonaceous aerosols in urban air. *Atmospheric Environment,* **41**, 6872–6883.

Husain L., Khan A.J., Ahmed T., Swami K., Bari A., Webber J.S., and Li J. (2008) Trends in atmospheric elemental carbon concentrations from 1835 to 2005. *Journal of Geophysical Research,* **113**, D13102, doi: 10.1029/2007JD009398.

Husar R.B. (2011) Satellite measurements of atmospheric aerosols. In *Aerosol Measurement: Principles, Techniques, and Applications*, Third Edition, P. Kulkarni, P. A. Baron and K. Willeke eds., John Wiley & Sons, Inc., Hoboken, NJ, doi: 10.1002/9781118001684.ch30.

International Energy Agency (2009) *World energy outlook 2009*, International Energy Agency (ISBN: 978-92-64-06130-9).

International Energy Agency (2010) Energy poverty: how to make modern energy access universal. Special early excerpt of the *World energy outlook 2010*, International Energy Agency (ISBN 978-92-64-08624-1). Available on the Internet at *http://www.worldenergyoutlook.org/docs/weo2010/weo2010_poverty.pdf*.

IPCC (2007) *Climate change 2007: the physical science basis, contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change*, S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller, eds., Cambridge University Press, Cambridge United Kingdom and New York, NY, USA, 996.

IPCC (2009) Meeting report of the expert meeting on the science of alternative metrics. 75 pages, G.-K. Plattner, T.F. Stocker, P. Midgley, and M. Tignor, eds., IPCC Working Group I Technical Support Unit, University of Bern, Bern, Switzerland (ISBN 978-92-9169-126-5). Available on the Internet at *http://www.ipcc.ch/pdf/supporting-material/expert-meeting-metrics-oslo.pdf*

Isaksen I.S.A., Granier C., Myhre G., Berntsen T.K., Dalsoren S.B., Gauss M., Klimont Z., Benestad R., Bousquet P., Collins W., Cox T., Eyring V., Fowler D., Fuzzi S., Jockel P., Laj P., Lohmann U., Maione M., Monks P., Prevot A.S.H., Raes F., Richter A., Rognerud B., Schulz M., Shindell D., Stevenson D.S., Storelvmo T., Wang W.-C., van Weele M., Wild M., and Wuebbles D. (2009) Atmospheric composition change: climate-chemistry interactions. *Atmospheric Environment,* **43**, 5138-5192.

Ito A. and Penner J.E. (2005) Historical emissions of carbonaceous aerosols from biomass and fossil fuel burning for the period 1870-2000. *Global Biogeochemical Cycles,* **19**, GB2028 (2), doi: 10.1029/2004GB002374.

Ito K., Christensen W.F., Eatough D.J., Henry R.C., Kim E., Laden F., Lall R., Larson T.V., Neas L., Hopke P.K., and Thurston G.D. (2006) PM source apportionment and health effects: 2. An investigation of intermethod variability in associations between source-apportioned fine particle mass and daily mortality in Washington, DC. *Journal of Exposure Science and Environemntal Epidemiology,* **16**, 4, 300-310, doi: 10.1038/sj.jea.7500464.

Ito K., Mathes R., Ross Z., Nádas A., Thurston G., and Matte T. (2011) Fine particulate matter constituents associated with cardiovascular hospitalizations and mortality in New York City. *Environmental Health Perspectives,* **119**, 4, 467-473, doi: 10.1289.ehp.1002667.

Jackson S.C. (2009) Parallel pursuit of near-term and long-term climate mitigation. *Science,* **326**, 5952, 526-527, doi: 10.1126/science.1177042.

Jacobson M.Z. (1999) Isolating nitrated and aromatic aerosols and nitrated aromatic gases as sources of ultraviolet light absorption. *Journal of Geophysical Research,* **104**, D3, 3527-3542, doi: 10.1029/1998jd100054.

Jacobson M.Z. (2000) A physically-based treatment of elemental carbon optics: Implications for global direct forcing of aerosols. *Geophysical Research Letters,* **27**, 217-220.

Jacobson M.Z. (2001) Strong radiative heating due to the mixing state of BC in atmospheric aerosols. *Nature,* **409**, 695-697.

Jacobson M.Z. (2002) Control of fossil-fuel particulate black carbon and organic matter, possibly the most effective method of slowing global warming. *Journal of Geophysical Research,* **107**, 4410, doi: 10.1029/2001jd001376.

Jacobson M.Z. (2005) Correction to "control of fossil-fuel particulate BC and organic matter, possibly the most effective method of slowing global warming." *Journal of Geophysical Research,* **110**, D14105, doi: 10.1029/2005JD005888.

Jacobson M.Z. (2006) Effects of absorption by soot inclusions within clouds and precipitation on global climate. *Journal of Physical Chemistry,* **110**, 6860-6873.

Jacobson M.Z. (2007) Testimony for the hearing on Black Carbon and Climate Change. U.S. House Committee on Oversight and Government Reform [Cited October 18]. Available on the Internet at *http://oversight.house.gov/images/stories/documents/20071018110606.pdf*.

Jacobson M.Z. and Streets D.G. (2009) Influence of future anthropogenic emissions on climate, natural emissions, and air quality. *Journal of Geophysical Research,* **114**, D8, D08118, doi: 10.1029/2008jd011476.

Jacobson M.Z. (2010) Short-term effects of controlling fossil-fuel soot, biofuel soot and gases, and methane on climate, Arctic ice, and air pollution health. *Journal of Geophysical Research,* **115**, D14209.

Jacobson M.Z. and Delucchi M.A. (2011) Providing all global energy with wind, water, and solar power, Part I: Technologies, energy resources, quantities and areas of infrastructure, and materials. *Energy Policy,* **39**, 3, 1154-1169, doi: 10.1016/j.enpol.2010.11.040.

Jansen K.L., Larson T.V., Koenig J.Q., Mar T.F., Fields C., Stewart J., and Lippmann M. (2005) Associations between health effects and particulate matter and black carbon in subjects with respiratory disease. *Environmental Health Perspectives,* **113**, 1741-1746.

Janssen N.A.H., Hoek G., Simic-Lawson M., Fischer P., van Bree L., ten Brink H., Keuken M., Atkinson R.W., Anderson H.R., Brunekreef B., and Cassee F.R. (2011) Black carbon as an additional indicator of the adverse health effects of airborne particles compared with PM_{10} and $PM_{2.5}$. *Environmental Health Perspectives,* **119**, 1691-1699.

Japar S.M. and Szkarlat A.C. (1980) Measurement of diesel vehicle exhaust particulate using photoacoustic spectroscopy. *Combustion Science and Technology,* **24**, 5-6, 215-219, doi: 10.1080/00102208008952440.

Japar S.M., Szkarlat A.C., and Pierson W.R. (1984) The determination of the optical properties of airborne particle emissions from diesel vehicles. *Science of the Total Environment,* **36**, 121-130.

Jeong C.-H., Hopke P.K., Kim E., and Lee D.-W. (2004) The comparison between thermal-optical transmittance elemental carbon and Aethalometer black carbon measured at multiple monitoring sites. *Atmospheric Environment,* **38**, 31, 5193-5204.

Jeong G.-R. and Wang C. (2010) Climate effects of seasonally varying biomass burning emitted carbonaceous aerosols. *Atmospheric Chemistry and Physics,* **10**, 8373-8389, doi:10.5194/acp-10-8373-2010.

Jerrett M., Burnett R.T., Pope C.A., Ito K., Thurston G., Krewski D., Shi Y., Calle E., and Thun M. (2009) Long-term ozone exposure and mortality. *New England Journal of Medicine,* **360**, 11, 1085-1095, doi: 10.1056/NEJMoa0803894.

Jetter J. and Kariher P. (2008) Solid-fuel household cook stoves: Characterization of performance and emissions. *Biomass and Bioenergy,* **33**, 2, 294-305.

Jimenez J.L., Canagaratna M.R., Donahue N.M., Prevot A.S.H., Zhang Q., Kroll J.H., DeCarlo P.F., Allan J.D., Coe H., Ng N.L., Aiken A.C., Docherty K.D., Ulbrich I.M., Grieshop A.P., Robinson A.L., Duplissy J., Smith J. D., Wilson K.R., Lanz V.A., Hueglin C., Sun Y.L., Tian J., Laaksonen A., Raatikainen T., Rautiainen J., Vaattovaara P., Ehn M., Kulmala M., Tomlinson J.M., Collins D.R., Cubison M.J., Dunlea E.J., Huffman J.A., Onasch T.B., Alfarra M.R., Williams P.I., Bower K., Kondo Y., Schneider J., Drewnick F., Borrmann S., Weimer S., Demerjian K., Salcedo D., Cottrell L., Griffin R., Takami A., Miyoshi T., Hatakeyama S., Shimono A., Sun J.Y, Zhang Y.M., Dzepina K., Kimmel J.R., Sueper D., Jayne J.T., Herndon S.C., Trimborn A.M., Williams L.R., Wood E.C., Kolb C.E., Middlebrook A.M., Baltensperger U., and Worsnop D.R. (2009) Evolution of organic aerosols in the atmosphere, *Science*, **326**, 1525-1529, doi: 10.1126/science.1180353.

Jimenez J., Claiborn C., Larson T., Gould T., Kirchstetter T.W., and Gundel L. (2007) Loading effect correction for real-time aethalometer measurements of fresh diesel soot. *Journal of Air and Waste Management,* **57**, 868-873, doi: 10.3155/1047-289.57.7.868.

Johnson M., Edwards R., Frenk C.A., and Masera O. (2007) In-field greenhouse gas emissions from cook stoves in rural Mexican households. *Atmospheric Environment,* **42**, 6, 1206-1222, doi: 10.1016/j.atmosenv.2007.10.034.

Junker C. and Liousse C. (2008) A global emission inventory of carbonaceous aerosol from historic records of fossil fuel and biofuel consumption for the period 1860-1997. *Atmospheric Chemistry and Physics,* **8**, 1195-1207.

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) Satellite-derived aerosol optical depth over dark water from MISR and MODIS: comparisons with AERONET and implications for climatological studies. *Journal of Geophysical Research,* **112**, D18205, doi: 10.1029/2006JD008175.

Kahn R.A., Nelson D.L., Garay M., Levy R.C., Bull M.A., Diner D.J., Martonchik J.V., Paradise S.R., Hansen E.G., and Remer L.A. (2009) MISR aerosol product attributes, and statistical comparisons with MODIS. *IEEE Transactions on Geoscience and Remote Sensing*, **47**, 12, 4095-4114, doi: 10.1109/TGRS.2009.2023115.

Kahn R.A., Gaitley B.J., Garay M.J., Diner D.J., Eck T., Smirnov A., and Holben B.N. (2010) Multiangle Imaging SpectroRadiometer global aerosol product assessment by comparison with the Aerosol Robotic Network, *Journal of Geophysical Research*, **115**, D23209, doi: 10.1029/2010JD014601.

Kandlikar M., Reynolds C.C.O., and Grieshop A.P. (2009): A perspective paper on black carbon mitigation as a response to climate change. Copenhagen Consensus on Climate.

Kannari A., Streets D.G., Tonooka Y., Murano K., and Baba T. (2008) MICS-Asia II: an inter-comparison study of emission inventories for the Japan Region. *Atmospheric Environment,* **42**, 15, 3584-3591.

Kar A., Rehman I.H., Burney J., Praveen P.S., Suresh R., Singh L., Singh V.K., Ahmed T., Ramanathan N., and Ramanathan V. (2012) Real-time assessment of black carbon pollution in Indian households due to traditional and improved biomass cookstoves. *Environmental Science and Technology* (in press).

Kaspari S.D., Schwikowski M., Gysel M., Flanner M.G., Kang S., Hou S., and Mayewski P.A. (2011) Recent increase in black carbon concentrations from a Mt. Everest ice core spanning 1860–2000 AD. *Geophysical Research Letters,* **38**, L04703, doi: 10.1029/2010GL046096.

Kaufman Y.J., Tanre D., Dubovik O., Karnieli A., and Remer L.A. (2001) Absorption of sunlight by dust as inferred from satellite and groundbased remote sensing. *Geophysical Research Letters,* **28**, 1479-1482.

Kazadzis S., Bais A., Arola A., Krotkov N., Kouremeti N., and Meleti C. (2009) Ozone monitoring instrument spectral UV irradiance products:comparison with ground based measurements at an urban environment. *Atmospheric Chemistry and Physics,* **9**, 585-594, doi: 10.5194/acp-9-585-2009.

Khalek I.A., Bougher T.L., and Merritt P.M. (2009) Coordinating Research Council, Advanced Collaborative Emissions Study, Phase 1.

Kim J.J., Smorodinsky S., Lipsett M., Singer B.C., Hodgson A.T., and Ostro B. (2004) Traffic-related air pollution near busy roads: the East Bay children's Respiratory Health Study. *American Journal of Respiratory and Critical Care Medicine,* **170**, 520-526.

Kim Y., Hatsushika H., Muskett R.R., and Yamazaki K. (2005) Possible effect of boreal wildfire soot on Arctic sea ice and Alaska glaciers. *Atmospheric Environment,* **39**, 3513-3520.

Kinsey J. and Wayson R.L. (2007) EPAct PM Methodology Discussion Paper.

Kinsey J.S., Hays M.D., Dong Y., Williams D.C., and Logan R. (2011) Chemical characterization of the fine particle emissions from commercial aircraft engines during the Aircraft Particle Emissions eXperiment (APEX) 1 to 3. Environmental Science and Technology, **45** (8), 3415–3421, doi: 10.1021/es103880d.

Kiraz K., Kart L., Demir R., Oymak S., Gulmez I., Unalacak M., and Ozesmi M. (2003) Chronic pulmonary disease in rural women exposed to biomass fumes. *Clinical and Investigative Medicine,* **26**, 5, 243-248.

Kirchstetter T.W., Aguiar J., Tonse S., and Novakov T. (2008) Black carbon concentrations and diesel vehicle emission factors derived from coefficient of haze measurements in California: 1967-2003. Prepared by Lawrence Berkeley National Laboratory, Berkeley, CA, LBNL Paper LBNL-63493.

Klimont Z., Cofala J., Bertok I., Amann M., Heyes C., and Gyarfas F. (2002) Modelling Particulate reductions in Europe: A framework to estimate reduction potential and control costs. Interim Rep., IR-02-076.

Klouda G.A., Filliben J.J., Parish H.J., Chow J.C., Watson J.G., and Cary R.A. (2005) Reference Material 8785: Air particulate matter on filter media. *Aerosol Science and Technology,* **39**, 173-183.

Knox A., Evans G.J., Brook J.R., Yao X., Jeong C.H., Godri K.J., Sabaliauskas K., and Slowik J.G. (2009) Mass absorption cross section of ambient black carbon aerosol in relation to chemical age. *Aerosol Science and Technology,* **43**, 6, 522-532, doi: 10.1080/02786820902777207.

Koch D., Bond T.C., Streets D., and Unger N. (2007a) Linking future aerosol radiative forcing to shifts in source activities. *Geophysical Research Letters,* **34**, L05821, doi: 10.1029/2006GL028360, 2007.

Koch D., Bond T.C., Streets D., Unger N., and van der Werf G.R. (2007b) Global impacts of aerosols from particular source regions and sectors. *Journal of Geophysical Research,* **112**, D02205.

Koch D., Schulz M., Kinne S., McNaughton C., Spackman J.R., Balkanski Y., Bauer S., Berntsen T., Bond T.C., Boucher O., Chin M., Clarke A., Luca N.D., Dentener F., Diehl T., Dubovik O., Easter R., Fahey D.W., Feichter J., Fillmore D., Freitag S., Ghan S., Ginoux P., Gong S., Horowitz L., Iversen T., Kirkev°ag A., Klimont Z., Kondo Y., Krol M., Liu X., Miller R., Montanaro V., Moteki N., Myhre G., Penner J.E., Perlwitz J., Pitari G., Reddy S., Sahu L., Sakamoto H., Schuster G., Schwarz J.P., Seland Ø., Stier P., Takegawa N., Takemura T., Textor C., Aardenne J.A.v., and Zhao Y. (2009) Evaluation of black carbon estimations in global aerosol models. *Atmospheric Chemistry and Physics,* **9**, 9001-9026, doi: 10.5194/acp-9-9001-2009.

Koch D. and Del Genio A.D. (2010) Black carbon semi-direct effects on cloud cover: review and synthesis. *Atmospheric Chemistry and Physics*, **10**, 7685-7696, doi: 10.5194/acp-10-7685-2010.

Kopacz M., Mauzerall D.L., Leibensperger E.M., Wang J., Henze D.K., and Singh K. (2011) Origin and radiative forcing of black carbon transported to the Himalayas and Tibetan Plateau, *Atmospheric Chemistry and Physics*, **11**, 2837-2852, doi: 10.5194/acp-11-2837-2011.

Kopp R.E. and Mauzerall D.L. (2010) Assessing the climatic benefits of BC mitigation. *Proceedings of the National Academy of Sciences,* **107**, 26, 11703-11708.

Koren I., Martins J.V., Remer L.A., and Afargan H. (2008) Smoke invigoration versus inhibition of clouds over the Amazon. *Science,* **321**, 946-949.

Kozlowski T. and Keller. T. (1966) Food relations of woody plants. *The Botanical Review,* **32**, 293-382.

Kralovec A.C., Christensen E.R., and Van Camp R.P. (2002) Fossil and wood combustion as recorded by carbon particles in Lake Erie sediments 1850-1998. *Environmental Science & Technology,* **36**, 1405-1413.

Krewski D., Jerrett M., Burnett R.T., Ma R., Hughes E., Shi Y., Turner M.C., Pope III C.A., Thurston G., Calle E.E., Thun M.J., Beckerman B., DeLuca P., Finkelstein N., Ito K., Moore D.K., Newbold K.B., Ramsay T., Ross Z., Shin H., and Tempalski B. (2009) Extended follow-up and spatial analysis of the American Cancer Society study linking particulate air pollution and mortality. *Research Report/Health Effects Institute,* **140**, 5-114; discussion 115-136.

Krylov I.F., Emel'yanov V.E., Nikitina E.A., Vizhgorodskii B.N., and Rudyak K.B. (2005) Low-sulfur diesel fuels: pluses and minuses. *Chemistry and Technology of Fuels and Oils,* **41**, 6, 423-428, doi: 10.1007/s10553-006-0001-7 .

Kucera T., Horakova H., and Sonska A. (2008) Toxic metal ions in photoautotrophic organisms. *Photosynthetica,* **46**, 481-489.

Kuki K., Oliva M., and Pereira E. (2008) Iron ore industry emissions as potential ecological risk factor for tropical coastal vegetation. *Environmental Management,* **42**, 1, 111-121, doi: 10.1007/s00267-008-9093-7.

Kukla G.J. and Kukla H.J. (1974) Increased surface albedo in the northern hemisphere. *Science,* **183**, 709-714.

Kupiainen K. and Klimont Z. (2007) Primary Emissions of Submicron and Carbonaceous Particles in Europe and the Potential for their Control. IIASA Interim Report. Available on the Internet at *http://www.iiasa.ac.at/rains/reports/ir-04-079.pdf*.

Lack D.A., Corbett J.J., Onasch T., Lerner B., Massoli P., Quinn P.K., Bates T.S., Covert D.S., Coffman D., Sierau B., Herndon S., Allan J., Baynard T., Lovejoy E., Ravishankara A.R., and Williams E. (2009) Particulate emissions from commercial shipping: Chemical, physical, and optical properties. *Journal of Geophysical Research,* **114**, D00F04, doi: 10.1029/2008jd011300.

Lack D.A. and Cappa C.D. (2010) Impact of brown and clear carbon on light absorption enhancement, single scattering albedo and absorption wavelength dependence of black carbon. *Atmospheric Chemistry and Physics,* **10**, 4207-4220, doi: 10.5194/acp-10-4207-2010.

Laden F., Schwartz J., Speizer F.E., and Dockery D.W. (2006) Reduction in fine particulate air pollution and mortality: Extended follow-up of the Harvard Six Cities Study. *American Journal of Respiratory and Critical Care Medicine,* **173**, 667-672.

Laird D. (2008) The charcoal vision: A win-win-win scenario for simultaneously producing bioenergy, permanently sequestering carbon, while improving soil and water quality. *Agronomy Journal* **100**, 178-181.

Lamarque J.-F., Bond T.C., Eyring V., Granier C., Heil A., Klimont Z., Lee D., Liousse C., Mieville A., Owen B., Schultz M.G., Shindell D., Smith S.J., Stehfest E., Aardenne J.V., Cooper O.R., Kainuma M., Mahowald N., McConnell J.R., Naik V., Rishi K., and Vuuren D.P.v. (2010) Historical (1850-2000) gridded anthropogenic and biomass burning emissions of reactive gases and aerosols: methodology and application. *Atmospheric Chemistry and Physics,* **10**, 7017-7039.

Larkin N.K., O'Neill S.M., Solomon R., Raffuse S., Strand T.M., Sullivan D.C., Krull C., Rorig M., Peterson J., and Ferguson S.A. (2009) The BlueSky smoke modeling framework. *International Journal of Wildland Fire,* **18**, 8, 906-920 (STI-3784), doi: 10.1071/WF07086.

Larkin N.K., Strand T., Raffuse S., Drury S., Sullivan D., Wheeler N., and Chinkin L. (2010) Developing an improved wildland fire emissions inventory. *19th International Emissions Inventory Conference*, San Antonio, Texas, USA, 27-30 September.

Larkin N.K., Brown S., Craig K., DeWinter J., Raffuse S., Strand T., Roberts P., and Solomon R. (2011) Identifying the potential for Arctic transport of smoke. In proceedings of the *American Meteorological Society's Ninth Fire and Forest Meteorology Symposium, Palm Springs, CA, 17-20 October* (Paper 192366). Available on the Internet at *http://ams.confex.org/ams/9FIRE/webprogram*.

Lau K.M. and Kim K.M. (2006) Observational relationships between aerosol and Asian monsoon rainfall, and circulation. *Geophysical Research Letters,* **33**, L21810, doi: 10.1029/2006GL027546.

Lau W.K.M., Kim M.-K., Kim K.-M., and Lee W.-S. (2010) Enhanced surface warming and accelerated snow melt in the Himalayas and Tibetan Plateau induced by absorbing aerosols. *Environmental Research Letters,* **5**, 2, doi: 10.1088/1748-9326/5/2/025204.

Lauer A. and Hendricks J. (2006) Simulating aerosol microphysics with the ECHAM4/MADE GCM - Part II: results from a first multiannual simulation of the submicrometer aerosol. *Atmospheric Chemistry and Physics,* **6**, 5495-5513, doi:10.5194/acp-6-5495-2006.

Lavanchy V.M.H., Gaggeler H.W., Schotterer U., Schwikowski M., and Baltensperger U. (1999) Historical record of carbonaceous particle concentrations from a European high-alpine glacier (Colle Gnifetti, Switzerland). *Journal of Geophysical Research,* **104**, 21227-21236.

Legrand M., Preunkert S., Schock M., Cerqueira M., Kasper-Giebl A., Afonso J., Pio C., Gelencser A., and Dombrowski-Etchevers I. (2007) Major 20th century changes of carbonaceous aerosol components (EC, WinOC, DOC, HULIS, carboxylic acids, and cellulose) derived from Alpine ice cores. *Journal of Geophysical Research,* **112**, D23S11, doi: 10.1029/2006JD008080.

Lenton T.M., Held H., Kriegler E., Hall J.W., Lucht W., Rahmstorf S., and Schellnhuber H.J. (2009) Tipping elements in the Earth's climate system. *Proceedings of the National Academy of Sciences,* **105**, 1786-1793.

Levy J.I., Baxter L.K., and Schwartz J. (2009) Uncertainty and variability in health-related damages from coal-fired power plants in the United States. *Risk Analysis,* **29**, 1000-1014.

Liepert B.G., Feichter J., Lohmann U., and Roeckner E. (2004) Can aerosols spin down the water cycle in a warmer and moister world? *Geophysical Research Letters,* **31**, L06207.

Lilenfield H., Whitefield P.D., and Hagen D. (1995) Ground testing of emissions from aircraft exhaust. Paper by the American Institute of Aeronautics and Astronautics, Paper AIAA-95-0110.

Lim B. and Cachier H. (1996) Determination of black carbon by chemical oxidation and thermal treatment in recent marine and lake sediments and cretaceous-tertiary clays. *Chemical Geology,* **131**, 143-154.

Lim H.-J., Turpin B.J., Edgerton E., Hering S.V., Allen G., Maring H., and Solomon P. (2003) Semicontinuous aerosol carbon measurements: comparison of Atlanta Supersite measurements. *Journal of Geophysical Research,* **108**, 8419, doi: 10.1029/2001JD001214.

Lim H. (2002) Study of exhaust emissions from idling heavy duty diesel trucks and commercially available idle reducing devices. EPA 420-R-02-025, October.

Lin C.-I., Baker M., and Charlson R.J. (1973) Absorption coefficient of atmospheric aerosol: a method for measurement. *Applied Optics,* **12**, 6,1356-1363.

Liousse C., Cachier H., and Jennings S.G. (1993) Optical and thermal measurements of black carbon aerosol content in different environments: Variation of the specific attenuation cross-section, sigma (σ). *Atmospheric Environment,* **27A**, 8, 1203-1211, doi: 10.1016/0960-1686(93)90246-u.

Liousse C., Penner J.E., Chuang C., Walton J.J., and Eddleman H. (1996) A global three-dimensional study of carbonaceous aerosols. *Journal of Geophysical Research,* **101**, D14, 19411- 19432.

Lipfert F.W., Baty J.D., Wyzga R.E., and Miller J.P. (2006) $PM_{2.5}$ constituents and related air quality variables as predictors of survival in a cohort of U.S. military veterans. *Inhalation Toxicology,* **18**, 645-657.

Lipfert F.W., Wyzga R.E., Baty J.D., and Miller J.P. (2009) Air pollution and survival within the Washington University-EPRI veterans cohort: risks based on modeled estimates of ambient levels of hazardous and criteria air pollutants. *Journal of the Air & Waste Management Association,* **59**, 473-489.

Lippmann M., Ito K., Hwang J.S., Maciejczyk P., and Chen L.C. (2006) Cardiovascular effects of nickel in ambient air. *Environmental Health Perspectives,* **114**, 1662-1669.

Littell J.S., Oneil E.E., McKenzie D., Hicke J.A., Lutz J.A., Norheim R.A., and Elsner M.M. (2010) Forest ecosystems, disturbance, and climactic change in Washington State, USA. *Climatic Change*, 129-158, doi 10.1007/s10584-010-9858-x.

Liu D., Wang Z., Liu Z., Winker D., and Trepte C. (2008a) A height resolved global view of dust aerosols from the first year CALIPSO lidar measurements. *Journal of Geophysical Research,* **113**, D16214, doi: 10.1029/2007JD009776.

Liu J., Fan S., Horowitz L.W., and II H.L. (2011) Evaluation of factors controlling long-range transport of black carbon to the Arctic. *Journal of Geophysical Research,* **116** (D04307), doi: 10.1029/2010JD015145.

Liu S., Zhou Y., Wang X., Wang D., Lu J., Zheng J., Zhong N., and Ran P. (2007) Biomass fuels are the probable risk factor for chronic obstructive pulmonary disease in rural South China. *Thorax,* **62**, 10, 889-897, doi: 10.1136/thx.2006.061457.

Liu X., Xu B., Yao T., Wang N., and Wu G. (2008b) Carbonaceous particles in Muztagh Ata ice core, West Kunlun Mountains, China. *Chinese Science Bulletin,* **53**, 21, 3379-3386.

Lohmann U. and Hoose C. (2009) Sensitivity studies of different aerosol indirect effects in mixed-phase clouds. *Atmospheric Chemistry and Physics,* **9**, 8917-8934.

LRTAP (2010) Report by the Co-Chairs of the Ad Hoc Expert Group on Black Carbon to the Convention on Long-Range Transboundary Air Pollution (LRTAP). 30 September 2010. Available on the Internet at *http://www.unece.org/fileadmin/DAM/env/documents/2010/eb/eb/ece.eb.air.2010.7.e.pdf*.

MacCarty N., Ogle D., Still D., Bond T., and Roden C. (2008) A laboratory comparison of the global warming impact of five major types of biomass cooking stoves. *Energy for Sustainable Development,* **12**, 2, 5-14.

MacCarty N., Still D., and Ogle D. (2010) Fuel use and emissions performance of fifty cooking stoves in the laboratory and related benchmarks of performance. *Energy for Sustainable Development* **14**, 161-171.

MACTEC (2005) Boiler best available retrofit technology (BART) engineering analysis. Lake Michigan Air Directors Consortium (LADCO), March 30.

Madrigano J., Baccarelli A., Wright R., Suh H., Sparrow D., Vokonas P., S., and Schwartz J. (2010) Air pollution, obesity, genes, and cellular adhesion molecules. *Occupational and Environmental Medicine.,* **67**, 5, 312-317.

Magi B.I. (2009) Chemical apportionment of southern African aerosol mass and optical depth. *Atmospheric Chemistry and Physics,* **9**, 7643-7655.

Malm W.C. (1999) *Introduction to Visibility, Revised Edition*, Colorado State University, Fort Collins, CO, USA. Available on the Internet at *http://vista.cira.colostate.edu/improve/Education/IntroToVisinstr.htm*.

Mann J.K., Balmes J.R., Bruckner T.A., Mortimer K.M., Margolis H.G., Pratt B., Hammond S.K., Lurmann F.W., and Tager I.B. (2010) Short-term effects of air pollution on wheeze in asthmatic children in Fresno, California. *Environmental Health Perspectives,* **118**, 10, 1497-1502, doi: 10.1289/ehp.0901292.

Manne A.S. and Richels R.G. (2001) An alternative approach to establishing trade-offs among greenhouse gases. *Nature,* **410**, 6829.

Mar T.F., Ito K., Koenig J.Q., Larson T.V., Eatough D.J., Henry R.C., Kim E., Laden F., Lall R., Neas L., Stolzel M., Paatero P., Hopke P.K., and Thurston G.D. (2006) PM source apportionment and health effects. 3. Investigation of inter-method variations in associations between estimated source contributions of PM2.5 and daily mortality in Phoenix, AZ. Journal of Exposure Science and Environemntal Epidemiology, 16, 4, 311-320, doi: 10.1038/sj.jea.7500465.

MARAMA (2006) Control Analysis and Documentation for Residential Wood Combustion in the MANE-VU Region. Prepared by OMNI Environmental Services, Inc., Beaverton, OR. Available on the Internet at *http://www.marama.org/visibility/ResWoodCombustion/RWC_FinalReport_121906.pdf*.

Marinoni A., Cristofanelli P., Laj P., Duchi R., Calzolari F., Decesari S., Sellegri K., Vuillermoz E., Verza G.P., Villani P., and Bonasoni P. (2010) Aerosol mass and black carbon concentrations, a two year record at NCO-P (5079 m, Southern Himalayas). *Atmospheric Chemistry and Physics*, **10**, 8551-8562, doi: 10.5194/acp-10-8551-2010.

Marmur A., Mulholland J., Kim E., Hopke P., Sarnat J., Klein M., Tolbert P., and Russell A. (2006) Comparing Results from Several $PM_{2.5}$ Source-Apportionment Methods for Use in a Time-Series Health Study. *Epidemiology,* **17**, S200-S200.

Martin W.J., II, Glass R.I., Balbus J.M., and Collins F.S. (2011) A Major Environmental Cause of Death. *Science*, **334**, 6053, 180-181, doi: 10.1126/science.1213088.

Martins J.A., Silva Dias M.A.F., and Gonalves F.L.T. (2009) Impact of biomass burning aerosols on precipitation in the Amazon: I. A modelling study. *Journal of Geophysical Research,* **114**, D02207, doi: 10.1029/2007JD 009587.

Masclet P., Hoyau V., Jaffrezo J.L., and Cachier H. (2000) Polycyclic aromatic hydrocarbon deposition on the ice sheet of Greenland. Part I: superficial snow. *Atmospheric Environment,* **34**, 3195-3207.

Masera O., Edwards R., Arnez C., Berrueta V., Johnson M., Bracho L., Riojasrodriguez H., and Smith K. (2007) Impact of Patsari improved cook stoves on indoor air quality in Michoacán, Mexico. *Energy for Sustainable Development,* **11**, 2, 45-56, doi: 10.1016/S0973-0826(08)60399-3.

Mauritsen T., Sedlar J., Tjernström M., Leck C., Martin M., Shupe M., Sjogren S., Sierau B., Persson P.O.G., Brooks I.M., and Swietlicki E. (2010) Aerosols indirectly warm the Arctic. *Atmospheric Chemistry and Physics,* **10**, 16775-16796, doi: 10.5194/acpd-10-16775-2010.

Mauritsen T., Sedlar J., Tjernström M., Leck C., Martin M., Shupe M., Sjogren S., Sierau B., Persson P.O.G., Brooks I.M., and Swietlicki E. (2011) An Arctic CCN-limited cloud-aerosol regime. *Atmospheric Chemistry and Physics,* **11**, 165-173, doi: 10.5194/acp-11-165-2011.

Mayol-Bracero O.L., Gabriel R., Andreae M.O., Kirchstetter T.W., Novakov T., Ogren J., Sheridan P., and Streets D.G. (2002) Carbonaceous aerosols over the Indian Ocean during the Indian Ocean Experiment (INDOEX): chemical characterization, optical properties, and probable sources. *Journal of Geophysical Research,* **107**, D19, 8030, doi: 10.1029/2000JD000039.

McCarty J.L. (2011) Remote sensing-based estimates of annual and seasonal emissions from crop residue burning in the contiguous United States. *Journal of the Air & Waste Management Association,* **61**, 22-34.

McConnell J.R., Edwards R., Kok G.L., Flanner M.G., Zender C.S., Saltzman E.S., Banta J.R., Pasteris D.R., Carter M.M., and Kahl J.D.W. (2007) 20th-century industrial black carbon emissions altered Arctic climate forcing. *Science,* **317**, 5843, 1381-1384, doi: 10.1126/science.1144856.

McConnell J.R. and Edwards R. (2008) Coal burning leaves toxic heavy metal legacy in the Arctic. *Proceedings of the National Academy of Sciences,* **105**, 34, 12140-12144.

McConnell J.R. (2010) Historical black carbon and other ice core aerosol records in the Arctic for GCM evaluation. *Atmospheric Environment,* **44**, 2665-2666.

McConnell R., Berhane K., Gilliland F., Molitor J., Thomas D., Lurmann F., Edward Avol, Gauderman W.J., and Peters J.M. (2003) Prospective study of air pollution and bronchitic symptoms in children with asthma. *American Journal of Respiratory and Critical Care Medicine,* **168**, 790-797, doi: 10.1164/rccm.200304-466OC.

McCracken J.P., Smith K.R., Díaz A., Mittleman M.A., and Schwartz J. (2007) Chimney stove intervention to reduce long-term wood smoke exposure lowers blood pressure among Guatemalan women. *Environmental Health Perspectives,* **115**, 7, 996-1001, doi: 10.1289/ehp.9888.

McCracken J., Baccarelli A., Hoxha M., Dioni L., Melly S., Coull B., Suh H., Vokonas P., and Schwartz J. (2010) Annual ambient black carbon associated with shorter telomeres in elderly men: Veterans affairs normative aging study. *Environmental Health Perspectives,* **118**, 11, 1564-1570, doi: 10.1289/ehp.0901831.

McKelvey K.S. and Busse K.K. (1996) Twentieth-century fire patterns on Forest Service lands. In *Sierra Nevada Ecosystem Project: Final Report to Congress, Vol. II, Assessments and Scientific Basis for Management Options*, University of California Davis Centers for Water and Wildland Resources, Davis, CA.

McMeeking G.R., Kreidenweis S.M., Carrico C.M., Collett J.L., Day D.E., and Malm W.C. (2005) Observations of smoke-influenced aerosol during the Yosemite Aerosol Characterization Study: 2. Aerosol scattering and absorbing properties. *Journal of Geophysical Research,* **110**, D18, D18209, doi: 10.1029/2004jd005624.

Meehl G.A., Arblaster J.M., and Collins W.D. (2008) Effects of black carbon aerosols on the Indian monsoon. *Journal of Climate*, **21**, 2869-2882, doi: 10.1175/2007JCLI1777.1.

Menon S., Hansen J., Nazarenko L., and Luo Y. (2002) Climate effects of black carbon aerosols in China and India. *Science,* **297**, 2250-2253.

Menon S., Koch D., Beig G., Sahu S., Fasullo J., and Orlikowski D. (2010) Black carbon aerosols and the third polar ice cap. *Atmospheric Chemistry and Physics,* **10**, 4559-4571, doi: 10.5194/acp-10-4559-2010.

MEPC (2010) Prevention of air pollution from ships: reduction of emissions of black carbon from shipping in the Arctic. *60th Session of the Marine Environment Protection Committee* (MEPC 60/4/24). Available on the Internet at *http://www.rina.org.uk/hres/mepc%2060_4_24.pdf.*

Miller J.D., Safford H.D., Crimmins M., and Thode A.E. (2009) Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. *Ecosystems,* **12**, 16–32, doi: 10.1007/s10021-008-9201-9.

Min S.-K., Zhang X., and Zwiers F.W. (2008) Human influence on Arctic sea ice detectable from early 1990s onwards. *Geophysical Research Letters,* **35** (21), doi: 10.1029/2008GL035725.

Ming J., Cachier H., Xiao C., Qin D., Kang S., Hou S., and Xu J. (2008) Black carbon record based on a shallow Himalayan ice core and its climatic implications. *Atmospheric Chemistry and Physics,* **8**, 1343-1352, doi: 10.5194/acp-8-1343-2008.

Ming J., Xiao C., Cachier H., Qin D., Qin X., Li Z., and Pu J. (2009) Black carbon in the snow of glaciers in west China and its potential effects on albedos. *Atmospheric Research,* **92**, 114-123.

Minoura H., Takahashi K., Chow J.C., and Watson J.G. (2006) Multi-year trend in fine and coarse particle mass, carbon, and ions in downtown Tokyo, Japan. *Atmospheric Environment,* **40**, 14, 2478-2487, doi: 10.1016/j.atmosenv.2005.12.029.

Mishchenko M.I., Cairns B., Kopp G., Schueler C.F., Fafaul B.A., Hansen J.E., Hooker R.J., Itchkawich T., Maring H.B., and Travis L.D. (2007) Accurate monitoring of terrestrial aerosols and total solar irradiance: introducing the Glory mission. *Bulletin of the American Meteorological Society,* **88**, 677-691, doi: 10.1175/BAMS-88-5-677.

Mitra A.P. and Sharma C. (2002) Indian aerosols: present status. *Chemosphere,* **49**, 9, 1175–1190.

Miyazaki Y., Kondo Y., Sahu L.K., Imaru J., Fukushima N., and Kano M. (2008) Performance of a newly designed continuous soot monitoring system (COSMOS). *Journal of Environmental Monitoring,* **10**, 1195-1201, doi: 10.1039/b806957c.

Moffet R.C. and Prather K.A. (2009a) *In-situ* measurements of the mixing state and optical properties of soot with implications for radiative forcing estimates. *Proceedings of the National Academy of Sciences,* **106**, 29, 11872-11877.

Moffet R.C. and Prather K.A. (2009b) Climate effects of black carbon aerosols in China and India. *Science,* **297**, 2250 –2253.

Mohr L., Luo S., Mathias E., Tobing R., Homan S., and Sterling D. (2008) Influence of season and temperature on the relationship of elemental carbon air pollution on pediatric asthma emergency room visits. *Journal of Asthma,* **45**, 10, 936-943.

Moosmüller H., Chakrabarty R.K., and Arnott W.P. (2009) Aerosol light absorption and its measurement: A review. *Journal of Quantitative Spectroscopy and Radiative Transfer,* **110**, 11, 844-878, doi: 10.1016/j.jqsrt.2009.02.035.

Mordukhovich I., Wilker E., Suh H., Wright R., Sparrow D., Vokonas P.S., and Schwartz J. (2009) Black carbon exposure, oxidative stress genes, and blood pressure in a repeated-measures study. *Environmental Health Perspectives,* **117**, 11, 1767–1772, doi: 10.1289/ehp.0900591.

Morgan G. (2009) Asia and Australasia wildfire management: a regional perspective. In *Proceedings of the Third International Symposium on Fire Economics, Planning, and Policy: Common Problems and Approaches*, A. González-Cabán ed., USDA Forest Service Pacific Southwest Research Station, Albany, CA (PSW-GTR-227).

Muhlbaier-Dasch J. (1982) Particulate and gaseous emissions from wood-burning fireplaces. *Environmental Science & Technology,* **16**, 639-645.

Muri G., Cermelj B., Faganeli J., and Brancelj A. (2002) Black carbon in Slovenian alpine lacustrine sediments. *Chemosphere,* **46**, 1225-1234.

Muri G., Wakeham S.G., and Faganeli J. (2003) Polycyclic aromatic hydrocarbons and black carbon in sediments of a remote alpine lake (Lake Planina, Northwest Slovenia). *Environmental Toxicology and Chemistry,* **22**, 1009-1016.

Murphy D.M., Chow J.C., Leibensperger E.M., Malm W.C., Pitchford M., Schichtel B.A., Watson J.G., and White W.H. (2011) Decreases in elemental carbon and fine particle mass in the United States. *Atmospheric Chemistry and Physics,* **11**, 4679-4686, doi: 10.5194/acp-11-4679-2011.

Murphy S.M., Agrawal H., Sorooshian A., Padró L.T., Gates H., Hersey S., Welch W.A., Jung H., Miller J.W., Cocker D.R., Nenes A., Jonsson H.H., Flagan R.C., and Seinfeld J.H. (2009) Comprehensive simultaneous shipboard and airborne characterization of exhaust from a modern container ship at sea. *Environmental Science & Technology,* **43**, 13, 4626-4640, doi: 10.1021/es802413j.

Myers R. (2006) Back to the back to the future: stationary source testing for fine PM. *AWMA EM,* **April**, 25-30.

Myhre G., Berglen T.F., Johnsrud M., Hoyle C.R., Berntsen T., Christopher S.A., Fahey D.W., Isaksen I.S.A., Jones T.A., Kahn R.A., Loeb N., Quinn P.K., Remer L., Schwarz J.P., and Yttri K.E. (2009) Modelled radiative forcing of the direct aerosol effect with multi-observation evaluation. *Atmospheric Chemistry and Physics,* **9**, 1365-1392, doi: 10.5194/acp-9-1365-2009.

Naeher L.P., Leaderer B.P., and Smith K.R. (2000) Particulate matter and carbon monoxide in highland Guatemala: indoor and outdoor levels from traditional and improved wood stoves and gas stoves. *Indoor Air,* **10**, 200-205.

Naidoo G. and Chirkoot D. (2004) The effects of coal dust on photosynthetic performance of the mangrove, Avicennia marina in Richards Bay, South Africa. *Environmental Pollution,* **127**, 359-366.

NARSTO (2005) A NARSTO assessment: improving emission inventories for effective air quality management across North America. Prepared by the NARSTO Emission Inventory Assessment Team. Available on the Internet at *http://www.narsto.org/section.src?SID=8*.

National Academies Press (2001) *Evaluating Vehicle Inspections and Maintenance Programs*, Washington, DC.

National Interagency Fire Center (2011) Prevention and education. Available on the Internet at *http://www.nifc.gov/prevention_education.html*

National Research Council (2004) Research priorities for airborne particulate matter: IV. Continuing research progress. Prepared by the National Research Council of the National Academies Committee on Research Priorities for Airborne Particulate Matter, Washington, DC.

National Research Council (2011) *Climate Stabilization Targets: Emissions, Concentrations, and Impacts Over Decades to Millennia*, National Academies Press, Washington, DC.

National Research Council of the National Academies (2005) *Radiative Forcing of Climate Change: Expanding the Concept and Addressing Uncertainties*, The National Academies Press, Washington, DC.

National Wildfire Coordinating Group (2007) Chapter 8. Fire prevention. In *Communicator's Guide for Wildland Fire Management: Fire Education, Prevention, and Mitigation Practices.* Available on the Internet at *http://www.nifc.gov/preved/comm_guide/wildfire/files/pdf%20%20files/linked%20pdfs/8%20fire%20prevention.pdf*.

Nazaroff W.W. and Cass G.R. (1991) Protecting museum collections from soiling due to the deposition of airborne particles. *Atmospheric Environment,* **25**, 841-852.

Neary D.G. and Zieroth E. (2007) Forest bioenergy system to reduce the hazard of wildfires: White Mountains, Arizona. *Biomass and Bioenergy,* **31**, 638-645.

NESCAUM (2005) Assessment of control technology options for BART-eligible sources steam electric boilers, industrial boilers, cement plants and paper and pulp facilities. In *Air Pollution Engineering Manual, Air & Waste Management Association*, A.J. Buonicore and W.T. Davis eds., Van Nostrand Reinhold, New York, NY.

NESCAUM (2009) Applicability and feasibility of NO_x, SO_2, and PM emissions control technologies for industrial, commercial, and institutional (ICI) boilers. January.

NESCAUM (2010) Technologies for control and measurement of mercury emissions from coal-fired power plants in the United States: a status report. July.

NIOSH (1999) National Institute for Occupational Safety and Health method 5040, elemental carbon (diesel particulate), NIOSH manual of analytical methods (NMAM), 4th Ed., Issue 3.

Noone and Clarke (1998) Soot scavenging measurements in arctic snowfall. *Atmospheric Environment,* **22**, 12.

Nordmann S., Birmili W., Weinhold K., Wiedensohler A., Mertes S., Muller K., Gnauk T., Herrmann H., Pitz M., Cyrys J., Flentje H., Ries L., and Wirtz K. (2009) Atmospheric aerosol measurements in the German Ultrafine Aerosol Network (GUAN) Part 2: Comparison of measurements techniques for graphitic, light-absorbing, and elemental carbon, and non-volatile particle volume under field conditions. *Gefahrstoffe Reinhaltung der Luft,* **69**, 11-12, 469-474.

Noss R.F., Franklin J.F., Baker W.L., Schoennagel T., and Moyle P.B. (2006) Managing fire-prone forests in the western United States. *Frontiers in Ecology and the Environment,* **4**, 9.

Notten M., Oosthoek A.J.P., Rozema J., and Aerts R. (2005) Heavy metal concentrations in a soil-plant-snail food chain along a terrestrial soil pollution gradient. *Environmental Pollution,* **138**, 178-190.

Novakov T. (1982) *Soot in the Atmosphere. In Particulate Carbon Atmospheric Life Cycle*, G.T. Wolff and R.L. Klimisch, eds., Plenum Press, New York.

Novakov T. and Corrigan C.E. (1995a) Thermal characterization of biomass smoke particles. *Mikrochim Acta,* **119**, 157-166.

Novakov T. and Corrigan C.E. (1995b) Influence of sample composition on aerosol organic and black carbon determinations. Prepared by Lawrence Berkeley National Laboratory, Berkeley, CA, LBL-37513 UC-407. Available on the Internet at *http://www.osti.gov/bridge/servlets/purl/122026-ZVCNoC/webviewable/122026.pdf*.

Novakov T., Ramanathan V., Hansen J., Kirchstetter T., Sato M., Sinton J., and Sathaye J. (2003) Large historical changes of fossil-fuel black carbon aerosols. *Geophysical Research Letters,* **30**, doi: 10.1029/2002gl016345.

Novakov T., Menon S., Kirchstetter T.W., Koch D., and Hansen J.E. (2005) Aerosol organic carbon to black carbon ratios: analysis of published data and implications for climate forcing. *Journal of Geophysical Research,* **110**, D21205, doi: 10.1029/2005JD005977.

O'Neill B.C. (2000) The jury is still out on global warming potentials. *Climatic Change,* **44**, 427-443.

O'Neill M.S., Veves A., Zanobetti A., Sarnat J.A., Gold D.R., Economides P.A., Horton E.S., and Schwartz J. (2005) Diabetes enhances vulnerability to particulate air pollution-associated impairment in vascular reactivity and endothelial function. *Circulation,* **111**, 22, 2913-2920, doi: 10.1161/CIRCULATIONAHA.104.517110.

O'Neill M.S., Veves A., Sarnat J.A., Zanobetti A., Gold D.R., Economides P.A., Horton E.S., and Schwartz J. (2007) Air pollution and inflammation in type 2 diabetes: a mechanism for susceptibility. *Occupational and Environmental Medicine,* **64**, 373-379, doi: 10.1136/oem.2006.030023.

Ogren J.A., Charlson R.J., and Groblicki P.J. (1983) Determination of elemental carbon in rainwater. *Analytical Chemistry,* **55**, 1569-1572.

Ogren J.A., Groblicki P.J., and Charlson R.J. (1984) Measurement of the removal rate of elemental carbon from the atmosphere. *The Science of the Total Environment,* **36**, 329-338.

Ogren J.A., Sheridan P.J., Andrews E., Jefferson A., Heintzenberg J., and Mertes S. (2001) Measurements of the light absorption efficiency of graphitic carbon in Indian Ocean aerosols. *Journal of Aerosol Science,* **32**, 1001, 677-688.

Ohara T., Akimoto H., Kurokawa K., Horii N., Yamaji K., Yan X., and Hayasaka T. (2007) An Asian emission inventory of anthropogenic emission sources for the period 1980-2020. *Atmospheric Chemistry and Physics,* **7**, 16, 4419-4444.

Orozco-Levi M., Garcia-Aymerich J., Villar J., Ramirez-Sarmiento A., Anto J.M., and Gea J. (2006) Wood smoke exposure and risk of chronic obstructive pulmonary disease. *European Respiratory Journal,* **27**, 3.

Ostro B., Feng W.-Y., Broadwin R., Green S., and Lipsett M. (2007) The effects of components of fine particulate air pollution on mortality in California: results from CALFINE. *Environmental Health Perspectives,* **115**, 1, 13-19, doi: 10.1289/ehp.9281.

Ostro B.D., Feng W.Y., Broadwin R., Malig B.J., Green R.S., and Lipsett M.J. (2008) The impact of components of fine particulate matter on cardiovascular mortality in susceptible subpopulations. *Occupational and Environmental Medicine,* **65**, 750-756.

Ostro B., Roth L., Malig B., and Marty M. (2009) The effects of fine particle components on respiratory hospital admission in children. *Environmental Health Perspectives,* **117**, 475-480.

Ottmar R.D., Peterson J.L., Leenhouts B., and Core J.E. (2001) Smoke management: techniques to reduce or redistribute emissions. In *Smoke Management Guide for Prescribed and Wildland Fire: 2001 edition*, C.C. Hardy, R.D. Ottmar, J.L. Peterson, J.E. Core, and P. Seamon eds., National Wildire Coordination Group.

Pacala S. and Socolow R. (2004) Stabilization wedges: solving the climate problem for the next 50 years with current technologies. *Science,* **305**, 5686, 968-972.

Painter T.H., Barrett A.P., Landry C.C., Neff J.C., Cassidy M.P., Lawrence C.R., McBride K.E., and Farmer G.L. (2007) Impact of disturbed desert soils on duration of mountain snow cover. *Geophysical Research Letters,* **34**, 12, L12502, doi:10.1029/2007GL030284.

Parashar D.C., Gadi R., Mandal T.K., and Mitra A.P. (2005) Carbonaceous aerosol emissions from India. *Atmospheric Environment,* **39**, 40, 7861-7871.

Park S.K., O'Neill M.S., Vokonas P.S., Sparrow D., and Schwartz J. (2005) Effects of air pollution on heart rate variability: The VA normative aging study. *Environmental Health Perspectives,* **113**, 304-309.

Park S.K., O'Neill M.S., Vokonas P.S., Sparrow D., Spiro A., Tucker K.L., Suh H., Hu H., and Schwartz J. (2008) Traffic-related Particles are Associated with Elevated Homocysteine. *American Journal of Respiratory and Critical Care Medicine,* **178**, 283-289.

Park S.S., Kim Y.J., and Fung K. (2001) Characteristics of $PM_{2.5}$ carbonaceous aerosol in the Sihwa industrial area, South Korea. *Atmospheric Environment,* **35**, 657- 665.

Park S.S., Hansen A.D.A., and Cho S.Y. (2010) Measurement of real time black carbon for investigating spot loading effects of Aethalometer data. *Atmospheric Environment,* **44**, 1449-1455.

Parrish D.D. (2006) Critical evaluation of US on-road vehicle emission inventories. *Atmospheric Environment,* **40**, 2288-2300.

Partanen W.E. and Allen C. (2005) Green energy from fats, oil & grease (FOG). Prepared by Power-Gen International, Las Vegas, NV.

Partnership for Clean Fuels and Vehicles (2009) Cleaning up urban bus fleets: With a focus on developing and transition countries. Available on the Internet at *http://www.unep.org/transport/pcfv/PDF/Retrofit.pdf*.

Partnership for Clean Indoor Air (2005) Household energy, indoor air pollution and health: overview of experiences and lessons in India. By Winrock International, Little Rock, AR.

Patel M.M., Chillrud S.N., Correa J.C., Hazi Y., Feinberg M., Kc D., Prakash S., Ross J.M., Levy D., and Kinney P.L. (2010) Traffic-related particulate matter and acute respiratory symptoms among New York City area adolescents. *Environmental Health Perspectives,* **118**, 9, 1338-1343, doi: 10.1289/ehp.0901499.

Paxian A., Eyring V., Beer W., Sausen R., and Wright C. (2010) Present-day and future global bottom-up ship emission inventories including polar routes. *Environmental Science & Technology,* **44**, 4, 1333-1339, doi: 10.1021/es9022859.

Peck M.D., Kruger G.E., Merwe A.E.v.d., Godakumbura W., and Ahuja R.B. (2008) Burns and fires from non-electric domestic appliances in low and middle income countries. *Burns,* **34**, 303-311, doi: 10.1016/j.burns.2007.08.014

Peng R., Bell M., Geyh A., McDermott A., Zeger S., Samet J., and Dominici F. (2009) Emergency admissions for cardiovascular and respiratory diseases and the chemical composition of fine particle air pollution. *Environmental Health Perspectives,* **117**, 957-963.

Penner J.E., Eddleman H., and Novakov T. (1993) Towards the development of a global inventory for black carbon emissions. *Atmospheric Environment,* **27A**, 8, 1277-1295.

Penner J.E., Chen Y., Wang M., and Liu X. (2009) Possible influence of anthropogenic aerosols on cirrus clouds and anthropogenic forcing. *Atmospheric Chemistry and Physics,* **8**, 879-896.

Perovich D.K., Grenfell T.C., Light B., Elder B.C., Harbeck J., Polashenski C., Tucker III W.B., and Stelmach C. (2009) Transpolar observations of the morphological properties of Arctic sea ice. *Journal of Geophysical Research,* **114**, C00A04, doi: 10.1029/2008JC004892.

Peters A., Dockery D.W., Muller J.E., and Mittleman M.A. (2001) Increased particulate air pollution and the triggering of myocardial infarction. *Circulation,* **103**, 2810-2815.

Peters G.P., Nilssen T.B., Lindholt L., Eide M.S., Glomsrød S., Eide L.I., and Fuglestvedt J.S. (2011) Future emissions from shipping and petroleum activities in the Arctic. *Atmospheric Chemistry and Physics,* **11**, 5305-5320.

Peterson M.R. and Richards M.H. (2002) Thermal-optical-transmittance analysis for organic, elemental, carbonate, total carbon, and OCX2 in $PM_{2.5}$ by the EPA/NIOSH method. In *Proceedings, Symposium on Air Quality Measurement Methods and Technology-2002*, E.D. Winegar and R.J. Tropp eds., Air & Waste Management Association, Pittsburgh, PA (83-1-83-19).

Petzold A., Kopp C., and Niessner R. (1997) The dependence of the specific attenuation cross-section on black carbon mass fraction and particle size. *Atmospheric Environment,* **31**, 5, 661-672, doi: 10.1016/S1352-2310(96)00245-2.

Petzold A., Dopelheuer A., Brock C.A., and Schroder F. (1999a) *In situ* observations and model calculations of black carbon emission by aircraft at cruise altitude. *Journal of Geophysical Research,* **104**, D18, 22171-22181.

Petzold A., Strom F.P., Schroder F.P., and Karcher B. (1999b) Carbonaceous aerosol in jet engine exhaust: emission characteristics and implications for heterogeneous chemical reactions. *Atmospheric Environment,* **33**, 2689-2698.

Petzold A., Weingartner E., Hasselbach J., Lauer P., Kurok C., and Fleischer F. (2010) Physical properties, chemical composition, and cloud forming potential of particulate emissions from a marine diesel engine at various locations. *Environmental Science & Technology,* **44**, 10, 3800-3805, doi: 10.1021/es903681z.

Pitchford M., Malm W., Schichtel B., Kumar N., Lowenthal D., and Hand J.L. (2007) Revised algorithm for estimating light extinction from IMPROVE particle speciation data. *Journal of Air and Waste Management,* **57**, 1326-1336, doi: 10.3155/1047-3289.57.11.1326.

Polenske K. and McMichael F. (2002) A Chinese cokemaking process-flow model for energy and environmental analyses. *Energy Policy,* **30**, 10, 865–883, doi: 10.1016/S0301-4215(01)00147-1.

Pope A.C. (2011) Review of the Draft Report to Congress on Black Carbon. Letter from Dr. Arden C. Pope III, Chair, Advisory Council on Clean Air Compliance Analysis, to the Honorable Lisa P. Jackson, Administrator, U.S. EPA, August 11. Available on the Internet at *http://yosemite.epa.gov/sab/sabproduct.nsf/fedrgstr_activites/38059D3EA6FE3A19852578EA004A7469/$File/EPA-COUNCIL-11-002-unsigned.pdf*.

Pope C.A., Burnett R.T., III, and Thun M.J. (2002) Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *Journal of the American Medical Association* **287**, 1132-1141, doi: 10.1001/jama.287.9.1132.

Pope C.A., Ezzati M., and Dockery D.W. (2009) Fine-particulate air pollution and life expectancy in the United States. *New England Journal of Medicine,* **360**, 4, 376-386, doi: 10.1056/NEJMsa0805646.

Pope D.P., Mishra V., Thompson L., Siddiqui A.R., Rehfuess E.A., Weber M., and Bruce N.G. (2010) Risk of low birth weight and stillbirth associated with indoor air pollution from solid fuel use in developing countries. *Epidemiology Reviews*, doi: 10.1093/epirev/mxq005.

Pósfai M., Gelencsér A., Simonics R., Arató K., Li J., Hobbs P.V., and Buseck P.R. (2004) Atmospheric tar balls: Particles from biomass and biofuel burning. *Journal of Geophysical Research,* **109**, D06213, doi: 10.1029/2003JD004169.

Pósfai M. and Buseck P.R. (2010) Nature and climate effects of individual tropospheric aerosol particles. *Annual Review of Earth and Planetary Sciences,* **38**, 17-43.

Prestemon J.P., Butry D.T., Abt K.L., and Sutphen R. (2010) Net benefits of wildfire prevention education efforts. *Forest Science,* **56**, 2, 181-192.

Prospero J.M., Landing W.M., and Schulz M. (2010) African dust deposition to Florida: Temporal and spatial variability and comparisons to models. *Journal of Geophysical Research,* **115**, D13304, doi: 10.1029/2009JD012773.

Pueschel R.F., Boering K.A., Verma S., Howard S.D., Ferry G.V., Goodman J., Allen D.A., and Hamill P. (1997) Soot aerosol in the lower stratosphere: pole-to-pole variability and contributions by aircraft. *Journal of Geophysical Research,* **102**, D11, 13113-13118.

Qian Y., Gustafson W.I., Leung L.R., and Ghan S.J. (2009) Effects of soot-induced snow albedo change on snowpack and hydrological cycle in western United States based on Weather Research and Forecasting chemistry and regional climate simulations. *Journal of Geophysical Research,* **114**, D03108, doi: 10.1029/2008JD011039.

Quincey P. (2007) A relationship between black smoke index and black carbon concentration. *Atmospheric Environment,* **41**, 7964-7968.

Quinn P.K., Shaw G., Andrews E., Dutton E.G., Ruoho-Airola T., and Gong S.L. (2007) Arctic haze: current trends and knowledge gaps. *Tellus,* **59B**, 99-114, doi: 10.1111/j.1600-0889.2006.00238.x.

Quinn P.K., Bates T.S., Baum E., Doubleday N., Fiore A.M., Flanner M., Fridlind A., Garrett T.J., Koch D., Menon S., Shindell D., Stohl A., and Warren S.G. (2008) Short-lived pollutants in the Arctic: their climate impact and possible mitigation strategies. *Atmospheric Chemistry and Physics,* **8**, 1723-1735, doi:10.5194/acp-8-1723-2008.

Quinn P.K., Stohl A., Arneth A., Berntsen T., Burkhart J.F., Christensen J., Flanner M., Kupiainen K., Lihavainen H., Shepherd M., Shevchenko V., Skov H., and Vestreng V. (2011) The impact of black carbon on Arctic climate. Arctic Monitoring and Assessment Program (AMAP), Oslo, Technical report no. 4(2011). Available on the Internet at *www.amap.no*.

Quiroz R., Grimalt J.O., and Fernandez P. (2010) Toxicity assessment of polycyclic aromatic hydrocarbons in sediments from European high mountain lakes. *Ecotoxicology and Environmental Safety*, **73**, 559-564.

Raaschou-Nielsen O. and Reynolds P. (2006) Air pollution and childhood cancer: a review of the epidemiological literature. *International Journal of Cancer,* **118**, 12, 2920-2929.

Ram K. and Sarin M.M. (2009) Absorption coefficient and site-specific mass absorption efficiency of elemental carbon in aerosols over urban, rural, and high-altitude sites in India. *Environmental Science & Technology,* **43**, 21, 8233-8239, doi: 10.1021/es9011542.

Ram K., Sarin M.M., and Tripathi S.N. (2010) Inter-comparison of thermal and optical methods for determination of atmospheric black carbon and attenuation coefficient from an urban location in northern India. *Atmospheric Research,* **97**, 335-342.

Ramana M.V., Ramanathan V., Feng Y., Yoon S.-C., Kim S.-W., Carmichael G.R., and Schauer J.J. (2010) Warming influenced by the ratio of black carbon to sulphate and the black-carbon source. *Nature Geoscience,* **3**, 542-545, doi: 10.1038/NGEO918.

Ramanathan V., Crutzen P.J., Kiehl J.T., and Rosenfeld D. (2001) Aerosols, climate, and the hydrological cycle. *Science,* **294**, 2119-2124.

Ramanathan V., Chung C., Kim D., Bettge T., Buja L., Kiehl J.T., Washington W.M., Fu Q., Sikka D.R., and Wild M. (2005) Atmospheric brown clouds: impacts on South Asian climate and hydrological cycle. *Proceedings of the National Academy of Sciences,* **102**, 15, 5326-5333.

Ramanathan V., Li F., Ramana M.V., Praveen P.S., Kim D., Corrigan C.E., and al. e. (2007) Atmospheric brown clouds: hemispherical and regional variations in long-range transport, absorption, and radiative forcing. *Journal of Geophysical Research,* **112**, D22821, doi: 10.1029/2006JD008124.

Ramanathan V. and Carmichael G. (2008) Global and regional climate changes due to black carbon. *Nature Geoscience,* **1**, 221-227.

Ramanathan V. and Feng W.Y. (2009) Air pollution, greenhouse gases and climate change: Global and regional perspectives. *Atmospheric Environment,* **43**, 37-50, doi: 10.1016/j.atmosenv.2008.09.063.

Ramanathan V. (2010) Interactive comment on "Quantifying immediate radiative forcing by black carbon and organic matter with the Specific Forcing Pulse" by T.C. Bond, et al. *Atmospheric Chemistry and Physics Discussions,* **10**, C6227-C6241.

Ramirez-Venegas A., Sansores R.H., Perez-Padilla R., Regalado J., Velazquez A., Sanchez C., and Mayar M.E. (2006) Survival of patients with chronic obstructive pulmonary disease due to biomass smoke and tobacco. *American Journal of Respiratory and Critical Care Medicine,* **173**, 393-397.

Ramos L., Hernandez L.M., and Gonzalez M.J. (1994) Sequential fractionation of copper, lead, cadmium and zinc in soils from or near Donana National Park. *Journal of Environmental Quality,* **23**, 50-57.

Reddy M.S. and Venkataraman C. (2002a) Inventory of aerosol and sulphur dioxide emissions from India. Part II--biomass combustion. *Atmospheric Environment,* **36**, 4, 699-712.

Reddy M.S. and Venkataraman C. (2002b) Inventory of aerosol and sulphur dioxide emissions from India: Part I--fossil fuel combustion. *Atmospheric Environment,* **36**, 4, 677-697.

Reddy M.S. and Boucher O. (2007) Climate impact of black carbon emitted from energy consumption in the world's regions. *Geophysical Research Letters,* **34**, L11802.

Reff A., Bhave P.V., Simon H., Pace T.G., Pouliot G.A., Mobley J.D., and Houyoux M. (2009) Emissions inventory of $PM_{2.5}$ trace elements across the United States. *Environmental Science & Technology,* **43**, 5790-5796.

Regalado J., Perez-Padilla R., Sansores R.H., Páramo Ramirez Jl, Brauer M., Paré P., and Vedal S. (2006) The effect of biomass burning on respiratory symptoms and lung function in rural Mexican women. *American Journal of Respiratory and Critical Care Medicine,* **174**, 901-905.

Regional Environmental Centre for the Caucasus (2008) Fuel quality and vehicle emission standards overview for the Azerbaijan Republic, Georgia, the Kyrgyz Republic, the Republic of Armenia, the Republic of Kazakhstan, the Republic of Moldova, the Republic of Turkmenistan, the Republic of Uzbekistan and the Russian Federation. Available on the Internet at *http://www.unep.org/transport/pcfv/PDF/FuelQuality_en.pdf*.

Regional Planning Organization (2004a) Research and development of planned burning emission inventories for the Central States Regional Air Planning Association. Final report prepared for the Central States Air Resource Agencies and the Central Regional Air Planning Association, Oklahoma City, OK by Sonoma Technology, Inc., Petaluma, CA, STI-902514-2516-FR, July. Available on the Internet at *http://www.dnr.mo.gov/env/apcp/docs/d2-cenrap-plannedburn.pdf*.

Regional Planning Organization (2004b) Fire emissions inventory development for the Midwest Regional Planning Organization. Prepared for the Lake Michigan Air Director's Consortium by EC/R Incorporated, September.

Regional Planning Organization (2005) 2002 fire emission inventory for the WRAP Region – Phase II. Prepared for Western Governors Association/Western Regional Air Partnership by Air Sciences, Inc., Project No. 178-6, July. Available on the Internet at *http://www.wrapair.org/forums/fejf/documents/WRAP_2002_PhII_EI_Report_20050722.pdf*.

Regional Planning Organization (2006) Technical support document on agricultural and forestry smoke management in the MANE-VU Region. Draft. Available upon request, March.

Regional Planning Organization (2008) Documentation of the base G2 and best & final 2002 base year, 2009 and 2018 emission inventories for VISTAS. Prepared by Mactec, Inc., Revision 1, April. Available on the Internet at *http://www.dep.state.fl.us/air/rules/regulatory/regional_haze_imp/app_d.pdf*.

Regoli F., Gorbi S., Fattorini D., Tedesco S., Notti A., Machella N., Bocchetti R., Benedetti M., and Piva F. (2006) Use of the land snail Helix aspersa sentinel organism for monitoring ecotoxicologic effects of urban pollution: an integrated approach. *Environmental Health Perspectives,* **114**, 63-69.

Rehfuess E., Mehta S., and Pruss-Ustun A. (2006) Assessing household solid fuel use: multiple implications for the Millennium Development Goals. *Environmental Health Perspectives,* **114**, 3, A178.

Rehman I.H., Ahmed T., Praveen P.S., Kar A., and Ramanathan V. (2011) Black carbon emissions from biomass and fossil fuels in rural India. *Atmospheric Chemistry and Physics,* **11**, 7289-7299, doi: 10.5194/acp-11-7289-2011.

Reisinger P., Wonaschütz A., Hitzenberger R., Petzold A., Bauer H., Jankowski N., Puxbaum H., Chi X., and Maenhaut W. (2008) Intercomparison of measurement techniques for black or elemental carbon under urban background conditions in wintertime: Influence of biomass combustion. *Environmental Science & Technology,* **42**, 3, 884-889, doi: 10.1021/es0715041.

Remer L.A., Tanre D., Kaufman Y.J., Ichoku C., Mattoo S., Levy R., Chu D.A., Holben B.N., Dubovik O., Smirnov A., Martins J.V., Li R.R., and Ahmad Z. (2002). Validation of MODIS aerosol retrieval over ocean. *Geophysical Research Letters*, **29**, 12, 1618.

Ren C., Park S.K., Vokonas P.S., Sparrow D., Wilker E., Baccarelli A., Suh H.H., Tucker K.L., Wright R.O., and Schwartz J. (2010) Air pollution and homocysteine: More evidence that oxidative stress-related genes modify effects of particulate air pollution. *Epidemiology,* **21**, 2, 198-206.

Rennermalm A., Smith L., Stroeve J., and Chu V. (2009) Does sea ice influence Greenland ice sheet surface-melt? *Environmental Research Letters,* **4**, 024011 (2).

Riahi K., Gruebler A., and Nakicenovic N. (2007) Scenarios of long-term socio-economic and environmental development under climate stabilization. *Technological Forecasting and Social Change,* **74**, 7, 887-935.

Rich D.Q., Schwartz J., Mittleman M.A., Link M., Luttmann-Gibson H., Catalano P.J., Speizer F.E., and Dockery D.W. (2005) Association of short-term ambient air pollution concentrations and ventricular arrhythmias. *American Journal of Epidemiology,* **161**, 1123-1132.

Rich D.Q., Mittleman M.A., Link M.S., Schwartz J., Luttmann-Gibson H., Catalano P.J., Speizer F.E., Gold D.R., and Dockery D.W. (2006) Increased risk of paroxysmal atrial fibrillation episodes associated with acute increases in ambient air pollution. *Environmental Health Perspectives,* **114**, 1, 120-123, doi: 10.1289/ehp.8371.

Rich K.E., Petkau J., Vedal S., and Brauer M. (2004) A case-crossover analysis of particulate air pollution and cardiac arrhythmia in patients with implantable cardioverter defibrillators. *Inhalation Toxicology,* **16**, 363-372.

Rinne S.T., Rodas E.J., Bender B.S., Rinne M.L., Simpson J.M., R. G.-U., and Glickman L.T. (2006) Relationship of pulmonary function among women and children to indoor air pollution from biomass use in rural Ecuador. *Respiratory Medicine,* **100**, 7, 1208-1215.

Roberts D.L. and Jones A. (2004) Climate sensitivity to black carbon aerosol from fossil fuel combustion. *Journal of Geophysical Research,* **109**, D16202, doi: 10.1029/2004JD004676.

Robinson A.L., Donahue N.M., Shrivastava M.K., Weitkamp E.A., Sage A.M., Grieshop A.P., Lane T.E., Pierce J.R., and Pandis S.N. (2007) Rethinking organic aerosols: semivolatile emissions and photochemical aging. *Science,* **315**, 1259-1262.

Roden C.A., Bond T.C., Conway S., Pinel A.B.O., MacCarty N., and Still D. (2009) Laboratory and field investigations of particulate and carbon monoxide emissions from traditional and improved cookstoves. *Atmospheric Environment,* **43**, 1170-1181.

Rojas-Bracho L. (2009) An international strategy for black carbon controls in the transport sector: the case of Mexico. Presented at the *International Workshop on Black Carbon, Nobel House, London, UK, January 2009*. Available on the Internet at *http://theicct.org/sites/default/files/Rojas_2009.pdf*.

Rosen H. and Novakov T. (1983) Optical transmission through aerosol deposits on diffusely reflective filters: a method for measuring the absorbing component of aerosol particles. *Applied Optics,* **22**, 9,1265-1267.

Roth C. (2011) *Micro-gasification: cooking with gas from biomass; an introduction to the concept and the applications of wood-gas burning technologies for cooking*, Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH, Eschborn, Germany.

RTI International (2006) Evaluation of $PM_{2.5}$ emissions and controls at two Michigan steel mills and a coke oven battery: final report. Work Assignment 4-12 under EPA Contract No. 68-D-01-073.

RTI International (2008) Emission factor documentation for AP-42, Section 12.2 - coke production. Final Report, RTI Project 021426.001.001, May.

Rückerl R., Ibald-Mulli A., Koenig W., Schneider A., Woelke G., Cyrys J., Heinrich J., Marder V., Frampton M., Wichmann H.E., and Peters A. (2006) Air pollution and markers of inflammation and coagulation in patients with coronary heart disease. *American Journal of Respiratory and Critical Care Medicine,* **173**, 432-441, doi: 10.1164/rccm.200507-1123OC .

Ruckstuhl C., Philipona R., Behrens K., Coen M.C., Duerr B., Heimo A., Mätzler C., Nyeki S., Ohmura A., Vuilleumier L., Weller M., Behrens K., Wehrli C., and Zelenka A. (2008) Aerosol and cloud effects on solar brightening and the recent rapid warming. *Geophysical Research Letters,* **35**, L12708, doi: 10.1029/2008GL034228.

Russell P.B., Bergstrom R.W., Shinozuka Y., Clarke A.D., DeCarlo P.F., Jimenez J.L., Livingston J.M., Redemann J., Dubovik O., and Strawa A. (2010) Absorption angstrom exponent in AERONET and related data as an indicator of aerosol composition. *Atmospheric Chemistry and Physics,* **10**, 1155-1169, doi: 10.5194/acp-10-1155-2010.

Ryan M., Archer S., Birdsey R., Dahm C., Heath L., Hicke J., Hollinger D., Huxman T., Okin G., Oren R., Randerson J., and Schlesinger W. (2008) Land resources. In *The Effects of Climate Change on Agriculture, Land Resources, Water Resources, and Biodiversity*, U.S. Climate Change Science Program and the Subcommittee on Global Change Research, Washington, DC.

Rypdal K., Berntsen T., Fuglestvedt J.S., Aunan K., Torvanger A., Stordal F., Pacyna J.M., and Nygaard L.P. (2005) Tropospheric ozone and aerosols in climate agreements: scientific and political challenges. *Environmental Science & Policy,* **8**, 1, 29-43, doi: 10.1016/j.envsci.2004.09.003.

Rypdal K., Rive N., Berntsen T., Klimont Z., Mideksa T.K., Myhre G., and Skeie R.B. (2009) Costs and global impacts of BC abatement strategies. *Tellus B,* **61**, 4, 625-641, doi: 10.1111/j.1600-0889.2009.00430.x.

Sabbioni C., Ghedini N., and Bonazza A. (2003) Organic anions in damage layers on monuments and buildings. *Atmospheric Environment,* **37**, 9-10, 1261-1269, doi: 10.1016/s1352-2310(02)01025-7.

Sahu L.K., Kondo Y., Miyazaki Y., Kuwata M., Koike M., Takegawa N., Tanimoto H., Matsueda H., Yoon S.C., and Kim Y.J. (2009) Anthropogenic aerosols observed in Asian continental outflow at Jeju Island, Korea, in spring 2005. *Journal of Geophysical Research,* **114**, D03301, doi: 10.1029/2008JD010306.

Sahu S.K., Beig G., and Sharma C. (2008) Decadal growth of black carbon emissions in India. *Geophysical Research Letters,* **35**, 2, 1-5, doi: 10.1029/2007GL032333.

Saikawa E., Naik V., Horowitz L.W., Liu J., and Mauzerall D.L. (2009) Present and potential future contributions of sulfate, black and organic carbon aerosols from China to global air quality, premature mortality and radiative forcing. *Atmospheric Environment,* **43**, 17, 2814-2822, doi: 10.1016/j.atmosenv.2009.02.017.

Salam M.T., Islam T., and Gilliland F.D. (2008) Recent evidence for adverse effects of residential proximity to traffic sources on asthma. *Current Opinion in Pulmonary Medicine,* **14**, 1, 3-8, doi: 10.1097/MCP.0b013e3282f1987a.

Saldiva P.H.N., Clarke R.W., Coull B.A., Stearns R.C., Lawrence J., Krishna-Murthy G.G., Diaz E., Koutrakis P., Suh H., Tsuda A., and Godleski J.J. (2002) Lung inflammation induced by concentrated ambient air particles is related to particle composition. *American Journal of Respiratory and Critical Care Medicine,* **165**, 1610-1617.

Sarnat J.A., Marmur A., Klein M., Kim E., Russell A.G., Sarnat S.E., Mulholland J.A., Hopke P.K., and Tolbert P.E. (2008) Fine particle sources and cardiorespiratory morbidity: An application of chemical mass balance and factor analytical source-apportionment methods. *Environmental Health Perspectives,* **116**, 459-466.

Sarofim M. (2010) Using black carbon metrics in climate policy. J*ournal of Integrative Environmental Sciences*, **7**, Supplement 1, 135-144, doi: 10.1080/19438151003621409.

Sato M., Hansen J., Koch D., Lacis A., Ruedy R., Dubovik O., Holben B., Chin M., and Novakov T. (2003) Global atmospheric black carbon inferred from AERONET. *Proceedings of the National Academy of Sciences,* **100**, 11.

Saylor R.D., Edgerton E.S., and Hartsell B.E. (2006) Linear regression techniques for use in the EC tracer method of secondary organic aerosol estimation. *Atmospheric Environment,* **40**, 39, 7546-7556.

Schimel D., Alves D., Enting I., Heimann M., Joos F., Raynaud D., Wigley T., Prather M., Derwent R., Ehhalt D., Fraser P., Sanhueza E., Zhou X., Jonas P., Charlson R., Rodhe H., Sadasivan S., Shine K.P., Fouquart Y., Ramaswamy V., Solomon S., Srinivasan J., Albritton D., Isaksen I., Lal M., and Wuebbles D. (1996) Radiative forcing of climate change. In *Climate Change 1995: the Science of Climate Change. Contribution of Working Group I to the Second Assessment Report of the Intergovernmental Panel on Climate Change*, J.T. Houghton, L.G.M. Filho, B.A. Callander, N. Harris, A. Kattenberg, and K. Maskell, eds., Cambridge University Press, Cambridge, United Kingdom and New York, NY, 65-131.

Schmidt G., Ruedy R., Hansen J., Aleinov I., Bell N., Bauer M., Bauer S., Cairns B., Canuto V., Cheng Y., A. Del Genio, Faluvegi G., Friend A., Hall T., Hu Y., Kelley M., Kiang N., Koch D., Lacis A., Lerner J., Lo K., Miller R., Nazarenko L., Oinas V., Perlwitz J., Perlwitz J., Rind D., Romanou A., Russel G., Sato M., Shindell D., Stone P., Sun S., Tausnev N., Thresher D., and Yao M. (2006) Present-day atmospheric simulation using GISS Model: Comparison to in situ, satellite, and reanalysis data. *Journal of Climate,* **19**, 153-192, doi: 10.1175/JCLI3612.1.

Schmidt M.W.I. and Noack A.G. (2000) Black carbon in soils and sediments: Analysis, distribution, implications, and current challenges. *Global Biogeochemical Cycles,* **14**, 3, 777-793, doi: 10.1029/1999gb001208.

Schneider S.H., Semenov S., Patwardhan A., Burton I., Magadza C.H.D., Oppenheimer M., Pittock A.B., Rahman A., Smith J.B., Suarez A., and Yamin F. (2007) Assessing key vulnerabilities and the risk from climate change. In *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden, and C.E. Hanson, eds., Cambridge University Press, Cambridge, UK, 779-810.

Schulz M., Textor C., Kinne S., Balkanski Y., Bauer S., Berntsen T., Berglen T., Boucher O., Dentener F., Guibert S., Isaksen I.S.A., Iversen T., Koch D., Kirkevåg, Liu X., Montanaro V., Myhre G., Penner J.E., Pitari G., Reddy S., Seland Ø., Stier P., and Takemura T. (2006) Radiative forcing by aerosols as derived from the aerocom present-day and pre-industrial simulations. *Atmospheric Chemistry and Physics,* **6**, 5225-5246, doi:10.5194/acp-6-5225-2006.

Schuster G.L., Dubovik O., Holben B.N., and Clothiaux E.E. (2005) Inferring black carbon content and specific absorption from Aerosol Robotic Network (AERONET) aerosol retrievals. *Journal of Geophysical Research,* **110**, D10S17, doi: 10.1029/2004jd004548.

Schwartz J., Litonjua A., Suh H., Verrier M., Zanobetti A., Syring M., Nearing B.D., Verrier R.L., Stone P., MacCallum G., Speizer F.E., and Gold D.R. (2005) Traffic related pollution and heart rate variability in a panel of elderly subjects. *Thorax,* **60**, 6, 455-461, doi: 10.1136/thx.2004.024836.

Seinfeld J. and Pandis S. (2006) *Atmospheric chemistry and physics: From air pollution to climate change (second edition)*, John Wiley & Sons, Inc., Hoboken, New Jersey, 1203 (ISBN 0-471-72018-6).

Sharma S., Brook J.R., Cachier H., Chow J., Gaudenzi A., and Lu G. (2002) Light absorption and thermal measurements of black carbon in different regions of Canada. *Journal of Geophysical Research,* **107**, D24, 4771, doi: 10.1029/2002JD002496.

Shaw G.E. and Stamnes K. (1980) Arctic haze: perturbation of the polar radiation budget. *Annals of the New York Academy of Sciences,* **338**, 533-539.

Shih S.-I., Lee W.-J., Lin L.-F., Huang J.-Y., Su J.-W., and Chang-Chien G.-P. (2008) Significance of biomass open burning on the levels of polychlorinated dibenzo-p-dioxins and dibenzofurans in the ambient air. *Journal of Hazardous Materials,* **153**, 276-284.

Shindell D., Lamarque J.-F., Unger N., Koch D., Faluvegi G., Bauer S., Ammann M., Cofala J., and Teich H. (2008a) Climate forcing and air quality change due to regional emissions reductions by economic sector. *Atmospheric Chemistry and Physics,* **8**, 7101-7113, doi: 10.5194/acp-8-7101-2008.

Shindell D.T., Chin M., Dentener F., Doherty R.M., Faluvegi G., Fiore A.M., Hess P., Koch D.M., MacKenzie I.A., Sanderson M.G., Schultz M.G., Schulz M., Stevenson D.S., Teich H., Textor C., Wild O., Bergmann D.J., Bey I., Bian H., Cuvelier C., Duncan B.N., Folberth G., Horowitz L.W., Jonson J., Kaminski J.W., Marmer E., Park R., Pringle K.J., Schroeder S., Szopa S., Takemura T., Zeng G., Keating T.J., and Zuber A. (2008b) A multi-model assessment of pollution transport to the Arctic. *Atmospheric Chemistry and Physics,* **8**, 5353-5372, doi: 10.5194/acp-8-5353-2008.

Shindell D. and Faluvegi G. (2009) Climate response to regional radiative forcing during the twentieth century. *Nature Geoscience,* **2**, 4, 294-300, doi: 10.1038/ngeo473.

Shindell D.T., Faluvegi G., Koch D.M., Schmidt G.A., Unger N., and Bauer S.E. (2009) Improved attribution of climate forcing to emissions. *Science,* **326**, 5953, 716-718, doi: 10.1126/science.1174760.

Shindell D., Faluvegi G., Walsh M., Anenberg S.C., Van Dingenen R., Muller N.Z., Austin J., Koch D., and Milly G. (2011) Climate, health, agricultural and economic impacts of tighter vehicle-emission standards. *Nature Climate Change,* **1**, 1, 59-66, doi: 10.1038/nclimate1066.

Shindell D., Kuylenstierna J.C.I., Vignati E., van Dingenen R., Amann M., Klimont Z., Anenberg S.C., Muller N., Janssens-Maenhout G., Raes F., Schwartz J., Faluvegi G., Pozzoli L., Kupiainen K., Höglund-Isaksson L., Emberson L., Streets D., Ramanathan V., Hicks K., Oanh N.T.K., Milly G., Williams M., Demkine V., and Fowler D. (2012) Simultaneously Mitigating Near-Term Climate Change and Improving Human Health and Food Security. *Science*, **335**, 6065, 183-189, doi: 10.1126/science.1210026.

Shine K.P., Fuglestvedt J.S., Hailemariam K., and Stuber N. (2005) Alternatives to the global warming potential for comparing climate impacts of emissions of greenhouse gases. *Climatic Change,* **96**.

Shine K.P., Berntsen T.K., Fuglestvedt J.S., Skeie R.B., and Stuber N. (2007) Comparing the climate effect of emissions of short- and long-lived climate agents. *Philosophical Transactions of the Royal Society A,* **365** (1856), 1903-1914, doi: 10.1098/rsta.2007.2050.

Shine K.P. (2009) The global warming potential--the need for an interdisciplinary retrial. An editorial comment. *Climatic Change,* **96**, 467-472, doi: 10.1007/s10584-009-9647-6.

Shiraiwa M., Kondo Y., Iwamoto T., and Kita K. (2009) Amplification of light absorption of black carbon by organic coating. *Aerosol Science and Technology,* **44**, 1, 46 - 54.

Simon H., L. B., Bhave P.V., Divita F., Hsu Y., Luecken D., Mobley J.D., Pouliot G.A., Reff A., Sarwar G., and Strum M. (2010) The development and uses of EPA's SPECIATE database. *Atmospheric Pollution Research,* **1**, 196-296.

Sinha B. (2002) The Indian stove programme: an insider's view – the role of society, politics, economics and education. *Boiling Point,* **48**, 23-26.

Sinton J.E., Smith K.R., Peabody J.W., Yaping L., Xiliang Z., Edwards R., and Quan G. (2004) An assessment of programs to promote improved household stoves in China. *Energy for Sustainable Development,* **8**, 3, 33-52, doi: 10.1016/S0973-0826(08)60465-2.

Skeie R.B., Berntsen T., Myhre G., Pedersen C.A., Ström J., Gerland S., and Ogren J.A. (2011) Black carbon in the atmosphere and snow, from pre-industrial times until present. *Atmospheric Chemistry and Physics,* **11**, 6809-6836, doi: 10.5194/acp-11-6809-2011.

Slater J.F., Currie L.A., Dibb J.E., and Benner B.A., Jr. (2002) Distinguishing the relative contribution of fossil fuel and biomass combustion aerosols deposited at Summit, Greenland through isotopic and molecular characterization of insoluble carbon. *Atmospheric Environment,* **36**, 4463-4477.

Slowik J.G., E. C., Han J.-H., Davidovits P., Onasch T.B., Jayne J.T., Williams L.R., Canagaratna M.R., Worsnop D.R., Chakrabarty R.K., Moosmuller H., Arnott W.P., Schwarz J.P., Gao R.-S., Fahey D.W., Kok G.L., and Petzold A. (2007) An inter-comparison of instruments measuring black carbon content of soot particles. *Aerosol Science and Technology,* **41**, 3, 295-314.

Smallwood G.J., Snelling D.R., Gülder Ö.L., Clavel D., Gareau D., Sawchuck R.A., and Graham L. (2001) Transient particulate matter measurements from the exhaust of a direct injection spark ignition automobile. SAE paper 2001-01-3581.

SMAQMD (2009) Rule 421, mandatory episodic curtailment of wood and other solid fuel burning. Staff report by the Sacramento Metropolitan Air Quality Management District. Available on the Internet at *http://www.airquality.org/bod/2009/SepRule421HearingAttD-StaffReport.pdf*.

Smith D.M., Griffin J.J., and Goldberg E.D. (1973) Elemental Carbon in Marine Sediments: A Baseline for Burning. *Nature of Climate Change,* **241**, 268-270.

Smith K.R., Samet J.M., Romieu I., and Bruce N. (2000a) Indoor air pollution in developing countries and acute lower respiratory infections in children. *Thorax,* **55**, 6, 518-532.

Smith K.R., Uma R., Zhang J., Rasmussen R.A., and Khalil M.A.K. (2000b) Household stoves in India, greenhouse gases from small-scale combustion devices in developing countries, phase IIa. EPA/600/R-00-052, June. Available on the Internet at *http://www.epa.gov/nrmrl/pubs/600r00052/600R00052.pdf*.

Smith K.R., Mehta S., and Maeusezahl-Feuz M. (2004) Indoor air pollution from household use of solid fuels. In *Comparative Quantification of Health Risks: Global and Regional Burden of Disease Due to Selected Major Risk Factors*, M. Ezzati, A.D. Lopez, A. Rodgers, and C.J.L. Murray, eds., World Health Organization, Geneva, Switzerland, 1435-1493.

Smith K.R. and Haigler E. (2008) Co-benefits of climate mitigation and health protection in energy systems: scoping methods. *Annual Review Public Health,* **29**, 11-25.

Smith K.R., McCracken J.P., Weber M.W., Hubbard A., Jenny A., Thompson L.M., Balmes J., Diaz A., Arana B., and Bruce N. (2011) Effect of reduction in household air pollution on childhood pneumonia in Guatemala (RESPIRE): a randomised controlled trial. *The Lancet,* **378**, 9804, 1717-1726, doi: 10.1016/S0140-6736(11)60921-5.

Smith S.J. and Wigley T.M.L. (2006) Multi-gas forcing stabilization with the MiniCAM. *The Energy Journal,* Special Issue Number 3: Multigas Mitigation and Climate Policy, 373-391.

Snyder D.C. and Schauer J.J. (2007) An inter-comparison of two black carbon aerosol instruments and a semi-continuous elemental carbon instrument in the urban environment. *Aerosol Science and Technology,* **41**, 463-474.

Sorensen M., Daneshvar B., Hansen M., Dragsted L., Hertel O., Knudsen L., and Loft S. (2003) Personal $PM_{2.5}$ exposure and markers of oxidative stress in blood. *Environmental Health Perspectives,* **111**, 161-165.

Spira-Cohen A., Chen L.C., Kendall M., Lall R., and Thurston G.D. (2011) Personal exposures to traffic-related air pollution and acute respiratory health among Bronx schoolchildren with asthma. *Environmental Health Perspectives,* **119**, 4, 559-565, doi: 10.1289/ehp.1002653.

Sram R.J., Binkova B., Dejmek J., and Bobak M. (2005) Ambient air pollution and pregnancy outcomes: a review of the literature. *Environmental Health Perspectives,* **113**, 375-382.

Stanek L.W., Sacks J.D., Dutton S.J., and Dubois J.-J.B. (2011) Attributing health effects to apportioned components and sources of particulate matter: An evaluation of collective results. *Atmospheric Environment,* **45**, 32, 5655-5663, doi: 10.1016/j.atmosenv.2011.07.023.

Stanhill G. and Cohen S. (2001) Global dimming: a review of the evidence for a widespread and significant reduction in global radiation with discussion of its probable causes and possible agricultural consequences. *Agricultural and Forest Meteorology,* **107**, 255-278.

STAPPA (2006) Controlling fine particulate matter under the Clean Air Act: a menu of options. Report by the State and Territorial Air Pollution Program Administrators/Association of Local Air Pollution Control Officers (now National Association of Clean Air Agencies, NACAA), March.

Stier P., Seinfeld J.H., Kinne S., Feichter J., and Boucher O. (2006) Impact of nonabsorbing anthropogenic aerosols on clear-sky atmospheric absorption. *Journal of Geophysical Research,* **111**, D18201, doi: 10.1029/2006JD007147.

Stohl A., Andrews E., Burkhart J.F., Forster C., Herber A., Hoch S.W., Kowal D., Lunder C., Mefford T., Ogren J.A., Sharma S., Spichtinger N., Stebel K., Stone R., Strom J., Torseth K., Wehrli C., and Yttri K.E. (2006) Pan-arctic enhancements of light absorbing aerosol concentrations due to North American boreal forest fires during summer 2004. *Journal Geophysical Research,* **111**, D22214: 10.1029/2006JD007216.

Stohl A., Berg T., Burkhart J.F., Fjǽraa A.M., Forster C., Herber A., Hov Ø., Lunder C., McMillan W.W., Oltmans S., Shiobara M., Simpson D., Solberg S., Stebel K., Ström J., Tørseth K., Treffeisen R., Virkkunen K., and Yttri K.E. (2007) Arctic smoke – record high air pollution levels in the European Arctic due to agricultural fires in Eastern Europe in spring 2006. *Atmospheric Chemistry and Physics,* **7**, 511-534, doi: 10.5194/acp-7-511-2007.

Strack J.E., Pielke R.A., Sr., and Liston G. (2007) Arctic tundra shrub invasion and soot deposition: Consequences for spring snowmelt and near-surface air temperatures. *Journal of Geophysical Research,* **112**, G04S44, doi:10.1029/2006JG000297.

Strawa A.W., Kirchstetter T.W., Hallar A.G., Ban-Weiss G.A., McLaughlin J.P., Harley R.A., and Lunden M.M. (2010) Optical and physical properties of primary on-road vehicle particle emissions and their implications for climate change. *Journal of Aerosol Science,* **41**, 36-50.

Streets D., Bond T., Lee T., and Jang C. (2004) On the future of carbonaceous aerosol emissions. *Journal of Geophysical Research,* **109**, doi: 10.1029/2004JD004902.

Streets D.G., Gupta S., Waldhoff S.T., Wang M.Q., Bond T.C., and Yiyun B. (2001) Black carbon emissions in China. *Atmospheric Environment,* **35**, 4281-4296.

Streets D.G., Bond T.C., Carmichael G.R., Fernandes S.D., Fu Q., He D., Klimont K., Nelson S.M., Tsai N.Y., and Wang M.Q. (2003a) An inventory of gaseous and primary aerosol emissions in Asia in the year 2000. *Journal of Geophysical Research,* **108**, D21, Art. No. 8809.

Streets D.G., Yarber K.F., Woo J.H., and Carmichael G.R. (2003b) Biomass burning in Asia: Annual and seasonal estimates and atmospheric emissions. *Global Biogeochemical Cycles* **17**, 4, 1099.

Streets D.G., Wu Y., and Chin M. (2006) Two-decadal aerosol trends as a likely explanation of the global dimming/brightening transition. *Geophysical Research Letters,* **33**, L15806, doi: 10.1029/2006GL026471.

Strydom C., Robinson C., Pretorius E., Whitcutt J.M., Marx J., and Bornman M.S. (2006) The effect of selected metals on the central metabolic pathways in biology: a review. *WaterSA,* **32**, 543-554.

Subramanian R., Kok G.L., Baumgardner D., Clarke A., Shinozuka Y., Campos T.L., Heizer C.G., Stephens B.B., de Foy B., Voss P.B., and Zaveri R.A. (2010) Black carbon over Mexico: The effect of atmospheric transport on mixing state, mass absorption cross-section, and BC/CO ratios. *Atmospheric Chemistry and Physics,* **10**, 1, 219-237.

Suglia S.F., Gryparis A., Schwartz J., and Wright R.J. (2008) Association between traffic-related black carbon exposure and lung function among urban women. *Environmental Health Perspectives,* **116**, 1333-1337.

Tagaris E., Liao K.-J., DeLucia A.J., Deck L., Amar P., and Russell A.G. (2009) Potential impact of climate change on air pollution-related human health effects. *Environmental Science & Technology,* **43**, 4979-4988.

Tankersley C.G., Campen M., Bierman A., Flanders S.E., Broman K.W., and Rabold R. (2004) Particle effects on heart-rate regulation in senescent mice. *Inhalation Toxicology,* **16**, 381-390.

Tankersley C.G., Bierman A., and Rabold R. (2007) Variation in heart rate regulation and the effects of particle exposure in inbred mice. *Inhalation Toxicology,* **19**, 621-629.

Tankersley C.G., Champion H.C., Takimoto E., Gabrielson K., Bedja D., Misra V., El-Haddad H., Rabold R., and Mitzner W. (2008) Exposure to inhaled particulate matter impairs cardiac function in senescent mice. *American Journal of Physiology – Regulatory, Integrative and Comparative Physiology,* **295**, R252-R263.

Ten Hoeve J.E., Remer L.A., and Jacobson M.Z. (2011) Microphysical and radiative effects of aerosols on warm clouds during the Amazon biomass burning season as observed by MODIS: impacts of water vapor and land cover. *Atmospheric Chemistry and Physics,* **11**, 3021-3036, doi: 10.5194/acp-11-3021-2011.

TF HTAP (2010) Hemispheric Transport of Air Pollution 2010, Part A: Ozone and Particulate Matter, F. Dentener, T. Keating, and H. Akimoto, eds., Prepared by the Task Force on Hemispheric Transport of Air Pollution, United Nations Economic Commission for Europe, Geneva, Switzerland, Air pollution studies No. 17 (ECE/EB.AIR/100). Available on the Internet at *http://www.htap.org/activities/2010_Final_Report/HTAP%202010%20Part%20A%20110407.pdf*.

Thevenon F., Anselmetti F.S., Bernasconi S.M., and Schwikowski M. (2009) Mineral dust and elemental black carbon records from an Alpine ice core (Colle Gnifetti glacier) over the last millennium. *Journal of Geophysical Research,* **114**, D17102, doi: 10.1029/2008JD011490.

Thomas R., Frederick E., Krabill W., Manizade S., and Martin C. (2006) Progressive increase in ice loss from Greenland. *Geophysical Research Letters,* **33**, L10503, doi: 10.1029/2006GL026075.

Thompson L. (2010) Understanding global climate change: paleoclimate perspective from the world's highest mountains. *Proceedings of the American Philosophical Society,* **154**, 133-157.

Thurston G.D., Ito K., Mar T., Christensen W.F., Eatough D.J., Henry R.C., and Kim E. (2005) Workgroup report: workshop on source apportionment of particulate matter health effects: an intercomparison of results and implications. *Environmental Health Perspectives,* **113**, 1768-1774.

Tolbert P.E., Klein M., Peel J.L., Sarnat S.E., and Sarnat J.A. (2007) Multipollutant modeling issues in a study of ambient air quality and emergency department visits in Atlanta. *Journal of Exposure Science and Environmental Epidemiology,* **17**, S2, S29-S35, doi: 10.1038/sj.jes.7500625.

Torres O., Bhartia P.K., Herman J.R., Sinyuk A., Ginoux P., and Holben B. (2002) A long-term record of aerosol optical depth from TOMS observations and comparison to AERONET measurements. *Journal of the Atmospheric Sciences,* **59**, 398-413.

Torres O., Tanskanen A., Veihelmann B., Ahn C., Braak R., Bhartia P.K., Veefkind P., and Levelt P. (2007) Aerosols and surface UV products from Ozone Monitoring Instrument observations: An overview. *Journal of Geophysical Research,* **112**, D24S47, doi: 10.1029/2007JD008809.

Trenberth K.E., Johnes P.D., Ambenje P., and al. E. (2007) Observations: surface and atmospheric climate change. In *Climate Change 2007: the Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller, eds., Cambridge University Press, Cambridge, United Kingdom, and New York, NY, USA.

Truce R. and Wilkison L. (2008) Enhanced Fine particle and mercury emission control using the Indigo Agglomerator. In the proceedings from the *11th International Conference on Electrostatic Precipitation, Hangzhou, China,*, K. Yan, ed., Zhejiang University Press.

Turpin B.J. and Lim H.-J. (2001) Species contributions to $PM_{2.5}$ mass concentrations: Revisiting common assumptions for estimating organic mass. *Aerosol Science and Technology,* **35**, 602- 610.

Twomey S. (1977) The influence of pollution on the shortwave albedo of clouds. Journal of the Atmospheric Sciences, 34, 7, 1149–52.

U.N. Foundation (2011) Igniting change: a strategy for universal adoption of clean cookstoves and fuels. Available on the Internet at *http://cleancookstoves.org/resources/key-documents/*.

U.S. Census Bureau (2004) 2002 Economic census: vehicle inventory and use survey. Census Bureau report EC02TV-US.

U.S. EPA (1992) Prescribed burning background document and technical information document for best available control measures.

U.S. EPA (1998) Interim air quality policy on wildland and prescribed fires. Report.

U.S. EPA (1999) Draft regulatory impact analysis. Report, EPA 420D-99-001, April.

U.S. EPA (2001) On-highway heavy-duty engine and vehicle standards and highway diesel fuel sulfur control requirements. *Federal Register,* **66**, 12, 5002-5193.

U.S. EPA (2002a) National Emissions Inventory data and documentation, CHIEF website. Available on the Internet at *http://www.epa.gov/ttn/chief/net/2002inventory.html*.

U.S. EPA (2002b) Health assessment document for diesel engine exhaust. Report by the National Center for Environmental Assessment Office of Research and Development, U.S. Environmental Protection Agency, Washington, D.C., EPA/600/8-90/057F. Available on the Internet at *http://oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=36319*.

U.S. EPA (2003) EPA MOBILE6.2 model. Available on the Internet at *http://www.epa.gov/otaq/m6.htm*.

U.S. EPA (2004a) Final regulatory impact analysis: control of emissions from nonroad diesel engines. Report, EPA420-%-04-007. Available on the Internet at *http://www.epa.gov/ttn/ecas/ria.html*.

U.S. EPA (2004b) The particle pollution report: current understanding of air quality and emissions through 2003. EPA 454-R-04-002, December. Available on the Internet at *http://www.epa.gov/airtrends/aqtrnd04/pm.html*.

U.S. EPA (2004c) EPA's designations for $PM_{2.5}$ nonattainment areas in New England questions and answers. Available on the Internet at *http://www.epa.gov/ne/airquality/pdfs/pm25_qa.pdf*.

U.S. EPA (2005a) National Emissions Inventory data and documentation, CHIEF website. Available on the Internet at *http://www.epa.gov/ttn/chief/net/2005inventory.html*.

U.S. EPA (2005b) AirControlNET version 4.1 documentation report. Prepared for the U.S. Environmental Protection Agency by E.H. Pechan & Associates, Inc.

U.S. EPA (2006a) Diesel retrofit technology: an analysis of the cost-effectiveness of reducing particulate matter emissions from heavy-duty diesel engines through retrofits. Report, EPA420-S-06-002.

U.S. EPA (2006b) National ambient air quality standards for particulate matter; final rule. Federal Register, **71,** 61144. Available on the Internet at *http://www.epa.gov/ttn/naaqs/standards/pm/data/fr20061017.pdf*

U.S. EPA (2006c) 2006 regulatory impact analysis for the national ambient air quality standards for particle pollution. Report. Available on the Internet at *http://www.epa.gov/ttn/ecas/ria.html*.

U.S. EPA (2007) The cost-effectiveness of heavy-duty diesel retrofits and other mobile source emission reduction projects and programs. EPA420-B-07-006, May. Available on the Internet at *http://epa.gov/cleandiesel/documents/420b07006.pdf*.

U.S. EPA (2008a) Technology transfer network clearinghouse for inventories and emission factors, SPECIATE Version 4.2. Available on the Internet at *http://cfpub.epa.gov/si/speciate/ehpa_speciate_browse_details.cfm?ptype=P&pnumber=4737*.

U.S. EPA (2008b) NONROAD model (nonroad engines, equipment, and vehicles). Available on the Internet at *http://www.epa.gov/otaq/nonrdmdl.htm*.

U.S. EPA (2008c) Regulatory impact analysis: Control of emissions of air pollution from locomotive engines and marine compression-ignition engines less than 30 liters per cylinder; republication. *Federal Register,* **73**, 126 (EPA420-R-08-001a). Available on the Internet at *http://www.epa.gov/oms/regs/nonroad/420r08001a.pdf*.

U.S. EPA (2009a) Report to Congress: highlights of the Diesel Emissions Reduction Program. Report, EPA-420-R-09-006, August.

U.S. EPA (2009b) Integrated science assessment for particulate matter. Report by the U.S. Environmental Protection Agency, Washington, DC, EPA/600/R-08/139. Available on the Internet at *http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546*.

U.S. EPA (2010a) Technical bulletin: diesel particulate filter general information. EPA420-F-10-029.

U.S. EPA (2010b) Regulatory impact analysis: Amendments to the national emission standards for hazardous air pollutants and new source performance standards (NSPS) for the portland cement manufacturing industry: Final report. Available on the Internet at *http://www.epa.gov/ttn/ecas/regdata/RIAs/portlandcementfinalria.pdf*.

U.S. EPA (2010c) Final rulemaking to establish light-duty vehicle greenhouse gas emission standards and corporate average fuel economy standards regulatory impact analysis. (EPA-420-R-10-009). Available on the Internet at *http://www.epa.gov/otaq/climate/regulations/420r10009.pdf*.

U.S. EPA (2010d) WebFIRE information. Available on the Internet at *http://cfpub.epa.gov/webfire/*.

U.S. EPA (2010e) MOVES2010 model. Available on the Internet at *http://www.epa.gov/otaq/models/moves*.

U.S. EPA (2010f) Methods for measurement of filterable PM_{10} and $PM_{2.5}$ and measurement of condensable PM emissions from stationary sources. *Federal Register,* **75**, 244. Available on the Internet at *http://origin.www.gpo.gov/fdsys/pkg/FR-2010-12-21/pdf/2010-30847.pdf*.

U.S. EPA (2010g) Regulatory impact analysis: control of emissions of air pollution from category 3 marine diesel engines. Report.

U.S. EPA (2010h) Final regulatory impact analysis (RIA) for the SO_2 national ambient air quality standards (NAAQS). Available on the Internet at *http://www.epa.gov/ttn/ecas/regdata/RIAs/fso2ria100602full.pdf*.

U.S. EPA (2010i) Our nation's air: status and trends through 2008. Report prepared by the Office of Air Quality Planning and Standards, Air Quality Assessment Division, Research Triangle Park, NC, February. Available on the Internet at *http://www.epa.gov/airtrends/2010/report/fullreport.pdf*.

U.S. EPA (2011a) Regulatory impact analysis: national emission standards for hazardous air pollutants for industrial, commercial, and institutional boilers and process heaters. Available on the Internet at *http://www.epa.gov/ttn/ecas/regdata/RIAs/boilersriafinal110221_psg.pdf*.

U.S. EPA (2011b) Reducing black carbon emissions from the industrial, transportation, and residential sectors in South Asia. Draft report prepared for the U.S. Environmental Protection Agency, Research Triangle Park, NC, by Stratus Consulting, Boulder, CO, EPA Contract GS-10F-0229K.

U.S. EPA (2011c) The benefits and costs of the clean air act from 1990 to 2020. Final report, revision A by the U.S. Environmental Protection Agency, Office of Air and Radiation, Washington, DC, April. Available on the Internet at *http://www.epa.gov/oar/sect812/feb11/fullreport.pdf*.

U.S. EPA (2011d) Policy assessment for the review of the PM NAAQS. Final report by the U.S. Environmental Protection Agency Office of Air Quality Planning and Standards, Health and Environmental Impacts Division, Research Triangle Park, NC, April. Available on the Internet at *http://www.epa.gov/ttn/naaqs/standards/pm/s_pm_2007_pa.html*.

U.S. EPA (2011e) Regulatory impact analysis for the federal implementation plans to reduce interstate transport of fine particulate matter and ozone in 27 states; correction of SIP approvals for 22 states. Available on the Internet at *http://www.epa.gov/airtransport/pdfs/FinalRIA.pdf*.

UNDP (2007) Improving kiln efficiency for the brick making industry in Bangladesh - PDF B Phase. Report by the United Nations Development Programme Global Environment Facility, BDG/04/014.

UNDP (2010) Technical and financial fact sheet. By the United Nations Development Programme Bangladesh. Available on the Internet at *http://www.undp.org.bd/projects/prodocs/IKEBMI/Technical%20Fact%20Sheet.pdf*.

UNEP (2008a) Atmospheric brown clouds: regional assessment report with focus on Asia. Report by the United Nations Environment Programme, Nairobi, Kenya. Available on the Internet at *http://www.unep.org/pdf/ABCSummaryFinal.pdf*.

UNEP (2008b) Final report of the sixteenth meeting of the forum of ministers of the environment of Latin America and the Caribbean. Santo Domingo, Dominican Republic, UNEP/LAC-IGWG.XVI/9, January 27-February 1.

UNEP (2011) Near-term protection and clean air benefits: actions for controlling short-lived climate forcers. By the United Nations Environment Programme, ISBN: 978-92-807-3232-0, November. Available on the Internet at *http://www.unep.org/publications/ebooks/slcf/*.

UNEP and WMO (2011a) Integrated assessment of black carbon and tropospheric ozone. By the United Nations Environment Programme and World Meteorological Organization. Available on the Internet at *http://www.unep.org/dewa/Portals/67/pdf/BlackCarbon_report.pdf*.

UNEP and WMO (2011b) Integrated assessment of black carbon and tropospheric ozone: summary for decision makers. By the United Nations Environment Programme and World Meteorological Organization, UNEP/GC/26/INF/20. Available on the Internet at *http://www.unep.org/dewa/Portals/67/pdf/BlackCarbon_SDM.pdf*.

Unger N., Shindell D.T., and Wang J.S. (2009) Climate forcing by the on-road transportation and power generation sectors. *Atmospheric Environment,* **43**, 3077-3085.

Unger N., Bond T.C., Wang J.S., Koch D.M., Menon S., Shindell D.T., and Bauer S. (2010) Attribution of climate forcing to economic sectors. *Proceedings of the National Academy of Sciences,* **107**, 8.

USAID (2007) Fuel-efficient stove programs in IDP settings - summary evaluation report, Uganda. By the U.S. Agency for International Development. Available on the Internet at *http://www.usaid.gov/our_work/humanitarian_assistance/disaster_assistance/sectors/files/uganda_final_summary.pdf*.

USAID (2010a) Black carbon emissions in Asia: sources, impacts, and abatement options. By the U.S. Agency for International Development, April.

USAID (2010b) Evaluation of manufactured wood-burning stoves in Dadaab refugee camps, Kenya. By the U.S. Agency for International Development.

USAID (2011) In-home emissions of greenhouse pollutants from rocket and traditional biomass cooking stoves in Uganda. By the U.S. Agency for International Development. Available on the Internet at *http://iapnews.wordpress.com/2011/05/05/usaid-emissions-of-greenhouse-pollutants-from-rocket-and-traditional-biomass-cooking-stoves-in-uganda/*.

USDA (2010) USDA Arctic black carbon initiative. Fact sheet by the U.S. Department of Agriculture (USDA), Washington, D.C., September.

USEIA (2010) Electric power annual (with data for 2008). By the U.S. Energy Information Administration, January 21.

USEIA (2011) International energy outlook. By the U.S. Energy Information Administration, DOE/EIA-0484(2011), September.

Val Martin M., Logan J.A., Kahn R.A., Leung F.-Y., Nelson D.L., and Diner D.J. (2010) Smoke injection heights from fires in North America: analysis of 5 years of satellite observations. *Atmospheric Chemistry and Physics*, **10**, 1491-1510, doi: 10.5194/acp-10-1491-2010.

van Setten B.A.A.L., Makkee M., and Moulijn J.A. (2001) Science and technology of catalytic diesel particulate filters. *Catalysis Reviews,* **43**, 4, 489–564.

van Vuuren D., den Elzen M., Lucas P., Eickhout B., Strengers B., van Ruijven B., Wonink S., and van Houdt R. (2007) Stabilizing greenhouse gas concentrations at low levels: an assessment of reduction strategies and costs. *Climatic Change,* **81**, 119-159, doi: 10.1007/s10584-006-9172-9.

Venkatachari P., Zhou L., Hopke P.K., Schwab J.J., Demerjian K.L., Weimer S., Hogrefe O., Felton D., and Rattigan O. (2006) An intercomparison of measurement methods for carbonaceous aerosol in the ambient air in New York City. *Aerosol Science and Technology,* **40**, 788-795.

Venkataraman C., Sagar A.D., Habib G., Lam N., and Smith K.R. (2010) The Indian National Initiative for Advanced Biomass Cookstoves: the benefits of clean combustion. *Energy for Sustainable Development,* **14**, 63-72, doi: 10.1016/j.esd.2010.04.005.

Vignati E., Karl M., Krol M., Wilson J., Stier P., and Cavalli F. (2010) Sources of uncertainties in modeling black carbon at the global scale. *Atmospheric Chemistry and Physics,* **10**, 2595-2611, doi: 10.5194/acp-10-2595-2010.

Virkkula A., Mäkelä T., Hillamo R., Yli-Tuomi T., Hirsikko A., Hämeri K., and Koponen I.K. (2007) A simple procedure for correcting loading effects of Aethalometer data. *Journal of the Air & Waste Management Association,* **57**, 10, 1214-1222.

Walgraeve C., Demeestere K., Dewulf J., Zimmermann R., and Van Langenhove H. (2010) Oxygenated polycyclic aromatic hydrocarbons in atmospheric particulate matter: Molecular characterization and occurrence. *Atmospheric Environment,* **44**, 1831-1846.

Wang C. (2004) A modeling study on the climate impacts of black carbon aerosols. *Journal of Geophysical Research,* **109**, D03106 (D3), doi: 10.1029/2003JD004084.

Wang C. (2007) Impact of direct radiative forcing of black carbon aerosols on tropical convective precipitation. *Geophysical Research Letters,* **34**, L05709 (5), doi: 10.1029/2006GL028416.

Ward T.J., Palmer C.P., Bergauff M., Jayanty R.K.M., and Noonan C.W. (2011) Organic/elemental carbon and woodsmoke tracer concentrations following a community-wide woodstove changeout program. *Atmospheric Environment,* **45**, 5554-5560 (31), doi: 10.1016/j.atmosenv.2011.05.005.

Warneke C., Bahreini R., Brioude J., Brock C.A., de Gouw J.A., Fahey D.W., Froyd K.D., Holloway J.S., Middlebrook A., Miller L., Montzka S., Murphy D.M., Peischl J., Ryerson T.B., Schwarz J.P., Spackman J.R., and Veres P. (2009) Biomass burning in Siberia and Kazakhstan as an important source for haze over the Alaskan Arctic in April 2008. *Geophysical Research Letters,* **36**, L02813 (2), doi: 10.1029/2008GL036194.

Warneke C., Froyd K.D., Brioude J., Bahreini R., Brock C.A., Cozic J., de Gouw J.A., Fahey D.W., Ferrare R., Holloway J.S., Middlebrook A.M., Miller L., Montzka S., Schwarz J.P., Sodemann H., Spackman J.R., and Stohl A. (2010) An important contribution to springtime Arctic aerosol from biomass burning in Russia. *Geophysical Research Letters,* **37**, L01801 (1), doi: 10.1029/2009GL041816.

Warren S.G. and Clarke A.D. (1990) Soot in the atmosphere and snow surface of Antarctica. *Journal of Geophysical Research*, **95** (D2), 1811–1816, doi: 10.1029/JD095iD02p01811.

Warren S.G. and Wiscombe W.J. (1980) A model for the spectral albedo of snow, II: snow containing atmospheric aerosols. *Journal of the Atmospheric Sciences,* **37**, 2734-2745.

Warren S.G., Brandt R.E., and Grenfell T.C. (2006) Visible and near-ultraviolet absorption spectrum of ice from transmission of solar radiation into snow. *Applied Optics,* **45**, 5320-5334 (21), doi: 10.1364/AO.45.005320.

Watmough S.A., Hutchinson T.C., and Dillon P.J. (2004) Lead dynamics in the forest floor and mineral soil in south-central Ontario. *Biogeochemistry,* **71**, 43-68.

Watson J.G., Thurston G.D., Frank N.H., Lodge J.P., Wiener R.W., McElroy F.F., Kleinman M.T., Mueller P.K., and Chow J.C. (1995) Critical review discussion: measurement methods to determine compliance with ambient air quality standards for suspended particles. *Journal of the Air & Waste Management Association,* **45**, 666-684.

Watson J.G., Fujita E.M., Chow J.C., Zielinska B., Richards L.W., Neff W., and Dietrich D. (1998) Northern front range air quality study. Final report prepared for Colorado State University, Cooperative Institute for Research in the Atmosphere, Fort Collins, CO, by Desert Research Institute, Reno, NV, STI-996410-1772-FR, June. Available on the Internet at *http://www.dri.edu/images/stories/editors/eafeditor/Fujitaetal1998NFRAQSCMB.pdf*.

Watson J.G., Chow J.C., and Chen L.-W.A. (2005) Summary of organic and elemental carbon/black carbon analysis methods and intercomparisons. *Aerosol and Air Quality Research,* **5**, 1, 65-102.

Weingartner E., Saathoff H., Schnaiter M., Streit N., Bitnar B., and Baltensperger U. (2003) Absorption of light by soot particles: determination of the absorption coefficient by means of Aethalometers. *Journal of Aerosol Science,* **34**, 1445-1463.

Wellenius G.A., Yeh G.Y., Coull B.A., Suh H.H., Phillips R.S., and Mittleman M.A. (2007) Effects of ambient air pollution on functional status in patients with chronic congestive heart failure: a repeated-measures study. *Environmental Health,* **6**, 1-7.

Wesson K., Fann N., Morris M., Fox T., and Hubbell B. (2010) A multi–pollutant, risk–based approach to air quality management: case study for Detroit. *Atmospheric Pollution Research,* **1**, 4, 296-304.

White W.H. (2007) Shift in EC/OC split with 1 January 2005 TOR hardware upgrade. Available on the Internet at *http://vista.cira.colostate.edu/improve/Data/QA_QC/Advisory/da0016/da0016_TOR2005.pdf*

Widmann J.F., Duchez J., Yang J.C., Conny J.M., and Mulholland G.W. (2005) Measurement of the optical extinction coefficient of combustion-generated aerosol. *Journal of Aerosol Science,* **36**, 2, 283-289, doi: 10.1016/j.jaerosci.2004.09.005.

Wiedinmyer C. and Hurteau M.D. (2010) Prescribed fire as a means of reducing forest carbon emissions in the western United States. *Environmental Science & Technology,* ***44***, 1926-1932.

Wild M., Gilgen H., Roesch A., Ohmura A., Long C., Dutton E., Forgan B., Kallis A., Russak V., and Tsvetkov A. (2005) From dimming to brightening: decadal changes in solar radiation at the Earth's surface. *Science,* **308**, 847-850.

Wild M. (2009) Global dimming and brightening: a review. *Journal of Geophysical Research,* **114**, D00D16, doi: 10.1029/2008JD011470.

Wilker E.H., Baccarelli A., Suh H., Vokonas P., Wright R.O., and Schwartz J. (2010) Black carbon exposures, blood pressure, and interactions with single nucleotide polymorphisms in microRNA processing genes. *Environmental Health Perspectives,* **118**, 7, 943-948, doi: 10.1289/ehp.0901440.

Wilkerson J.T., Jacobson M.Z., Malwitz A., Balasubramanian S., Wayson R., Fleming G., Naiman A.D., and Lele S.K. (2010) Analysis of emission data from global commercial aviation: 2004 and 2006. *Atmospheric Chemistry and Physics,* **10**, 6391-6408.

Wilkinson P., Smith K.R., Davies M., Adair H., Armstrong B.G., Barrett M., Bruce N., Haines A., Hamilton I., Oreszczyn T., Ridley I., Tonne C., and Chalabi Z. (2009) Public health benefits of strategies to reduce greenhouse-gas emissions: household energy. *The Lancet,* **374**, 9705, 1917-1929, doi: 10.1016/S0140-6736(09)61713-X.

Winebrake J.J., Corbett J.J., Falzarano A., Hawker J.S., Korfmacher K., Ketha S., and Zilora S. (2008) Assessing energy, environmental, and economic tradeoffs in intermodal freight transportation. *Journal of the Air & Waste Management Association,* **58**, 1004-1013.

Winkler D., Hunt W., and McGill M. (2007) Initial performance assessment of CALIOP. *Geophysical Research Letters,* **34**, L19803, doi: 10.1029/2007GL030135.

Wiscombe W. and Warren S. (1980) A model for the spectral albedo of snow II. Snow containing atmospheric aerosols. *Journal of the Atmospheric Sciences,* **37**, 2734-2274.

Wise M.A., Calvin K.V., Thomson A.M., Clarke L.E., Bond-Lamberty B., Sands R.D., Smith S.J., Janetos A.C., and Edmonds J.A. (2009) Implications of limiting CO_2 concentrations for land use and energy. *Science,* **324**, 1183-1186.

Wolff G.T., Groblicki P.J., Cadle S.H., and Countiss (1982) Particulate carbon at various locations in the United States. In *Particulate Carbon: Atmospheric Life Cycle*, G.T. Wolff and R.L. Klimisch, eds., Plenum Press, New York, NY, 79-88.

World Bank (2010) Household Cookstoves, environment, health, and climate change: A new look at an old problem. Available on the Internet at *http://climatechange.worldbank.org/sites/default/files/documents/Household%20 Cookstoves-web.pdf*

World Health Organization (2009) *Global health risks: mortality and burden of disease attributable to selected major risks*, WHO Press, Geneva, Switzerland Available on the Internet at *http://www.who.int/healthinfo/global_burden_disease/ GlobalHealthRisks_report_full.pdf*.

WRAP (2002) Non-burning management alternatives on agricultural lands in the western United States, Volume II: non-burning management alternatives and implementation plan strategies. Prepared for the Western Regional Air Partnership by Eastern Research Group, Inc.

Wu J., Fu C., Xu Y., Tang J.P., Wang W., and Wang Z. (2008) Simulation of direct effects of black carbon aerosol on temperature and hydrological cycle in Asia by a Regional Climate Model. *Meteorology and Atmospheric Physics,* **100**, 179-193, doi: 10.1007/s00703-008-0302-y.

Xu B.-Q., Wang M., Joswiak D.R., Cao J.-J., Yao T.-D., Wu G.-J., Yang W., and Zhao H.-B. (2009a) Deposition of anthropogenic aerosols in a southeastern Tibetan glacier. *Journal of Geophysical Research,* **114**, D17209 (D17), doi: 10.1029/2008JD011510.

Xu B., Yao T., Liu X., and Wang N. (2006) Elemental and organic carbon measurements with a two-step heatinggas chromatography system in snow samples from the Tibetan Plateau. *Annals of Glaciology,* **43**, 1, 257-262, doi: 10.3189/172756406781812122.

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. (2009b) Black soot and the survival of Tibetan glaciers. *Proceedings of the National Academy of Sciences,* **106**, 22114-22118, doi: 10.1073/pnas.0910444106.

Yadav V.K., Prasad S., Patel D.K., Khan A.H., Tripathi M., and Shukla Y. (2010) Identification of polycyclic aromatic hydrocarbons in unleaded petrol and diesel exhaust emission. *Environmental Monitoring and Assessment,* **168**, 173-178.

Yang L., Xiaoye Z., Sunling G., Huizheng C., Dan W., Wenjun Q., and Junying U. (2006) Comparison of EC and BC and evaluation of dust aerosol contribution to light absorption in Xi'an, China. *Environmental Monitoring and Assessment,* **120**, 301-312, doi: 10.1007/s10661-005-9062-z.

Yttri K.E., Aas W., Bjerke A., Cape J.N., Cavalli F., Ceburnis D., Dye C., Emblico L., Facchini M.C., Forster C., Hanssen J.E., Hansson H.C., Jennings S.G., Maenhaut W., Putaud J.P., and Tørseth K. (2007) Elemental and organic carbon in PM_{10}: a one year measurement campaign within the European Monitoring and Evaluation Programme EMEP. *Atmospheric Chemistry and Physics,* **7**, 5711-5725, doi:10.5194/acp-7-5711-2007.

Yu H., Liu S.C., and Dickinson R.E. (2002a) Radiative effects of aerosols on the evolution of the atmospheric boundary layer. *Journal of Geophysical Research,* **107**, 4142 (D12), doi: 10.1029/2001JD000754.

Yu J.Z., Xu J., and Yang H. (2002b) Charring characteristics of atmospheric organic particulate matter in thermal analysis. *Environmental Science & Technology,* **36**, 4, 754-761.

Zanobetti A. and Schwartz J. (2006) Air pollution and emergency admissions in Boston, MA. *Journal of Epidemiology & Community Health,* **60**, 890-895.

Zanobetti A., Stone P.H., Speizer F.E., Schwartz J.D., Coull B.A., Suh H.H., Nearing B.D., Mittleman M.A., Verrier R.L., and D.R. G. (2009) T-wave alternans, air pollution and traffic in high-risk subjects. *American Journal of Cardiology,* **104**, 665-670.

Zeka A., Sullivan J.R., Vokonas P.S., Sparrow D., and Schwartz J. (2006) Inflammatory markers and particulate air pollution: characterizing the pathway to disease. *International Journal of Epidemiology,* **35**, 1347-1354.

Zhang Q., Streets D.G., Carmichael G.R., He K.B., Huo H., Kannari A., Klimont Z., Park I.S., Reddy S., Fu J.S., Chen D., Duan L., Lei Y., Wang L.T., and Yao Z.L. (2009) Asian emissions in 2006 for the NASA INTEX-B mission. *Atmospheric Chemistry and Physics*, **9**, 5131-5153, doi: 10.5194/acp-9-5131-2009.

Zhang Q., Streets D.G., He K., and Klimont Z. (2007) Major components of China's anthropogenic primary particulate emissions. *Environmental Research Letters,* **2**, 045027, doi: 10.1088/1748-9326/2/4/045027.

Zhang X.Y., Wang Y.Q., Zhang X.C., Guo W., and Gong S.L. (2008) Carbonaceous aerosol composition over various regions of China during 2006. *Journal of Geophysical Research,* **113**, D14111, doi: 10.1029/2007JD009525.

Zheng M., Cass G.R., Ke L., Wang F., Schauer J.J., Edgerton E.S., and Russell A.G. (2007) Source apportionment of daily fine particulate matter at Jefferson Street, Atlanta, GA, during summer and winter. *Journal of the Air & Waste Management Association,* **57**, 2, 228-242.

Zhi G., Peng C., Chen Y., Liu D., Sheng G., and Fu J. (2009) Deployment of coal briquettes and improved stoves: possibly an option for both environment and climate. *Environmental Science & Technology,* **43**, 15, 5586-5591.

Zhou J., Ito K., Lall R., Lippmann M., and Thurston G. (2011) Time-series analysis of mortality effects of fine particulate matter components in Detroit and Seattle. *Environmental Health Perspectives,* **119**, 4, 461-466.

Zhou Y. and Levy J.I. (2007) Factors influencing the spatial extent of mobile source air pollution impacts: a meta-analysis. *BMC Public Health,* **7**, 1, 89, doi: 10.1186/1471-2458-7-89.

Zhu Q. (2003) *Developments in particulate control*, IEA Clean Coal Centre (ISBN 92-9029-388-8).

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