Black Carbon Resource Packet

Black Carbon Symposium

November, 14, 2012

U.S. Environmental Protection Agency

Background

- Report to Congress on Black Carbon, *U.S. Environmental Protection Agency*
    - Executive Summary
    - Excerpt from Chapter 2 on metrics

- Climate Change – Characterization of Black Carbon and Organic Carbon Air Pollution Emissions and Evaluation of Measurement Methods Executive Summary, *Prepared by Chow et. al Desert Research Institute and submitted to the California Air Resources Board*

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

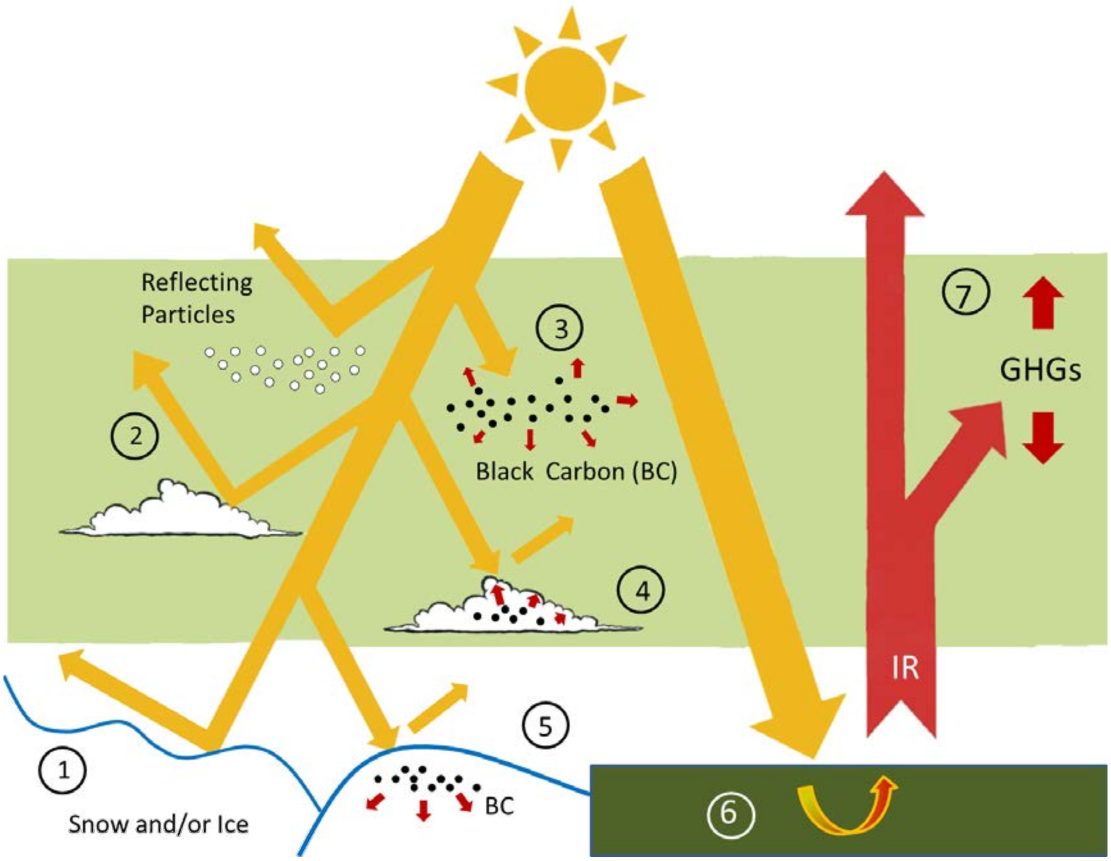

Figure 2-1. Effects of BC on Climate, as Compared to GHGs. (Source: U.S. EPA)

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.

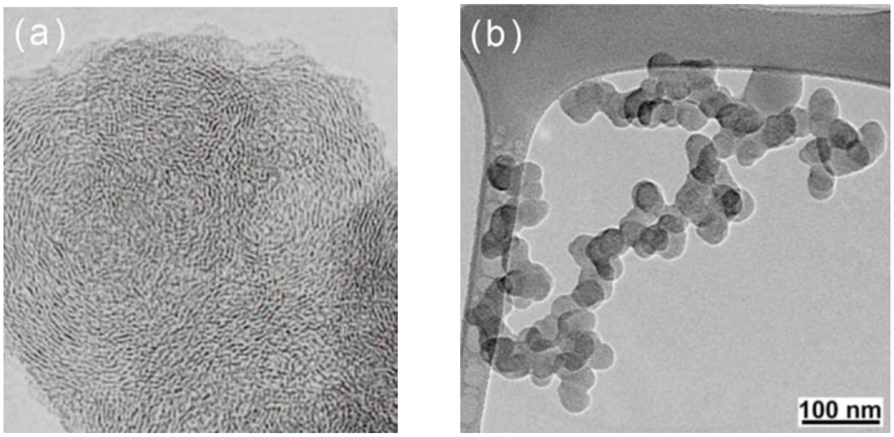

Figure 2-2. BC Images. (a) High resolution transmission electron microscopy (TEM) image of a BC spherule (Pósfai and Buseck, 2010). (b) TEM image of a representative soot particle. Freshly emitted soot particles are aggregates of soot spherules (Alexander et al., 2008).

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.

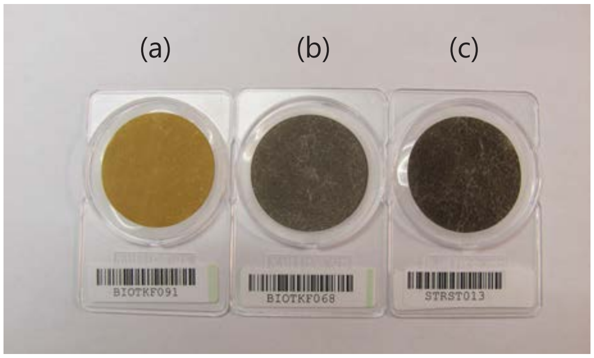

Figure 2-3. Representative Examples of Filter Samples Collected from Different Sources, including: (a) Smoldering Biomass, (b) Flaming Biomass, and (c) Diesel Exhaust. (Photo courtesy of Desert Research Institute)

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).

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.

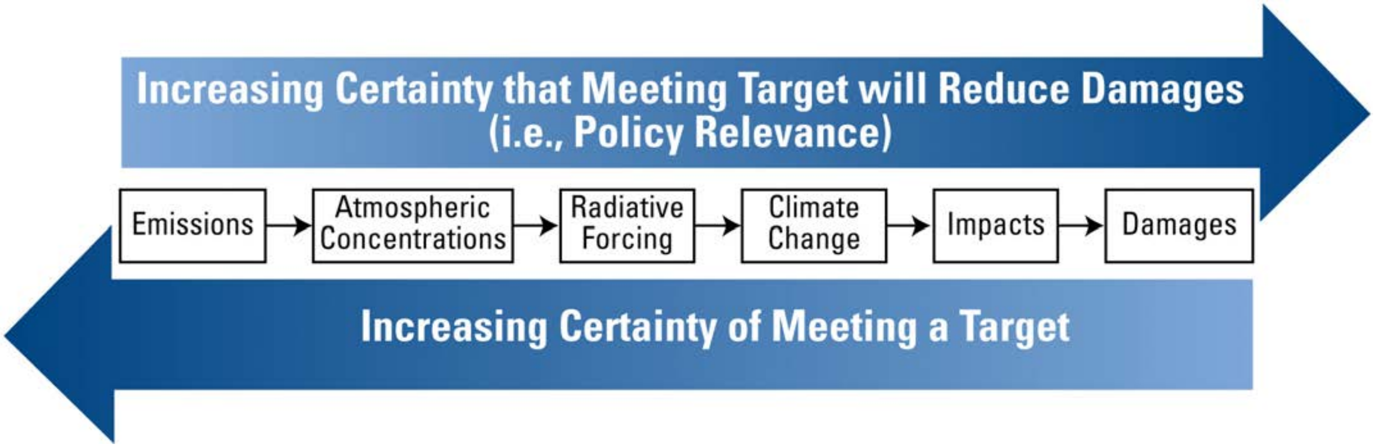

Figure 2-24. Cause and Effect Chain from Emissions to Climate Change, Impacts, and Damages. (Adapted from Fuglestvedt et al., 2003.) The arrows indicate that a policy could focus on different elements along the causal chain and, depending on whether the policy focuses on the emissions or damages end 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°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° 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

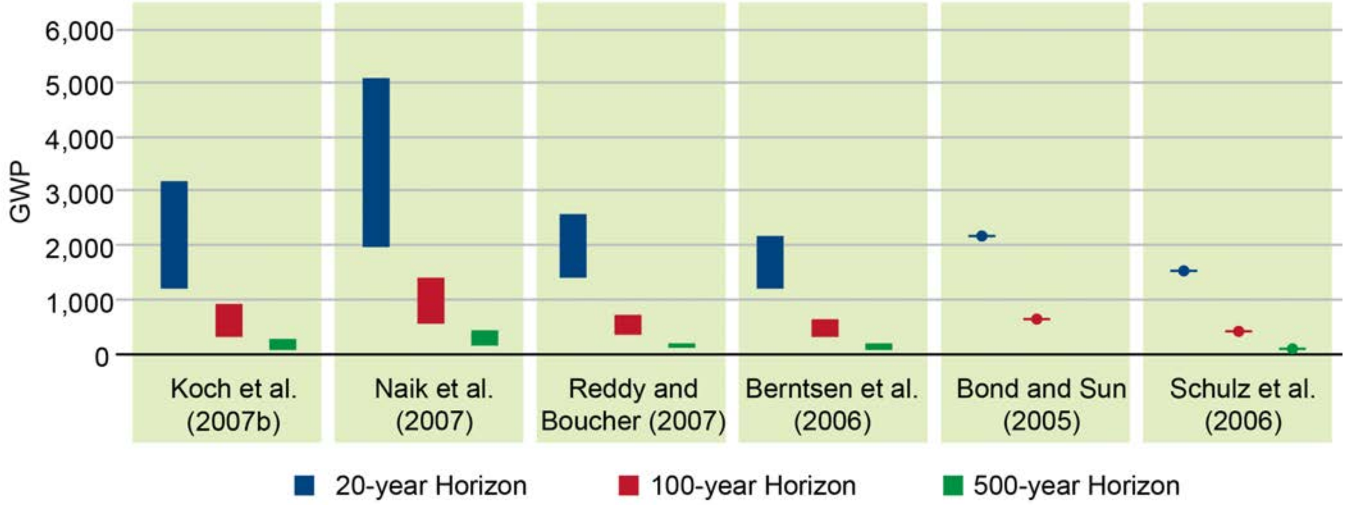

Figure 2-25. Ranges and Point Estimates for Regional Estimates of GWP Values for One-Year Pulse Emissions of BC for Different Time Horizons. The GWP values in the Y axis of the figure refer to the number of tons of CO_2 emissions which are calculated to be equivalent to one ton of BC emissions based 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.

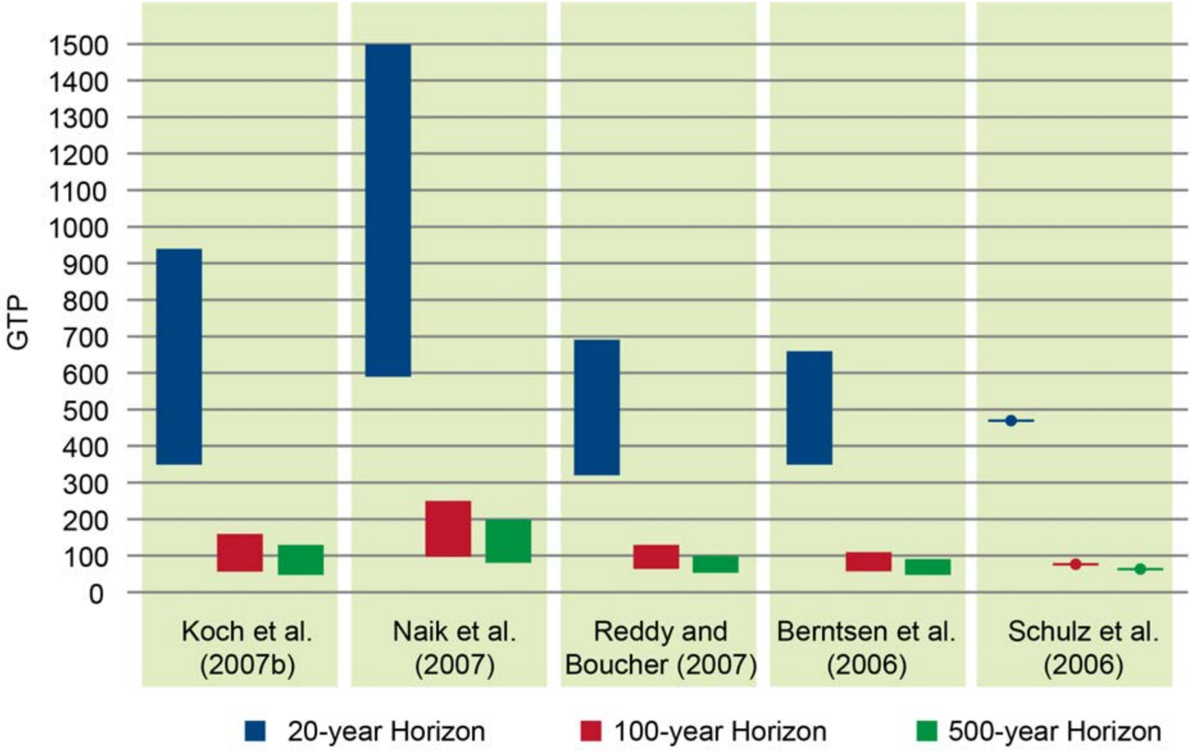

Figure 2-26. Ranges and Point Estimates for Regional Estimates of GTP Values for One-Year Pulse Emissions of BC for Different Time Horizons. The GTP values in the Y axis of the figure refer to the number of tons of CO_2 emissions which are calculated to be equivalent to one ton of BC emissions based 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.

Climate Change – Characterization of Black Carbon and Organic Carbon Air Pollution Emissions and Evaluation of Measurement Methods

Phase II: Characterization of Black Carbon and Organic Carbon Source Emissions

DRI Contract Number: 04-307

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Executive Summary

Background: Direct and indirect radiative effects of suspended particulate matter (PM) are major sources of uncertainty in current climate models. While aerosol organic carbon (OC) contributes to cooling through light scattering, black or elemental carbon (BC or EC) absorbs light, producing a forcing of +0.2 to +1 W/m^2 that leads to global warming. BC and OC nearly always accompany each other in PM emissions from incomplete combustion of carbon-containing fossil and biomass fuels. Including the direct and indirect effects of BC into the global- and regional-scale climate models requires accurate BC emission inventories and conversion factors (i.e., mass absorption efficiencies, σ_{abs} [λ]) that translate BC concentration into light absorption coefficients (b_{abs}) for different wavelengths. The overall objective of this study is to improve BC/EC and OC emission inventories by understanding what is currently available, by better characterizing BC and EC measurement methods, and by measuring emission rates and profiles from BC-emitting sources. One of the major issues is that there is no single, universally accepted standard for BC or EC measurement, and the available thermal and optical methods vary by more than two to three orders of magnitude. Neither are there widely accepted methods to connect BC or EC to b_{abs}, the relevant observable for radiative transfer. Simplified optical theory for calculating σ_{abs} (λ) and single scattering albedo of BC may not be applied to BC from various sources featuring different size, morphology, and internal mixing.

To meet the overall goal of the study, the Desert Research Institute has completed a comprehensive study on BC measurements and emissions. The first phase of this study evaluated methods for measuring BC and light absorption (b_{abs}). The goals of Phase I include: 1) critically review the literature on carbon analysis methods and comparisons; 2) create carbon analysis QA/QC methods and plans; 3) conduct laboratory inter-comparison experiments of organic carbon (OC), EC, BC, and light absorption (b_{abs}) measurement methods; and 4) perform a field comparison of different measurement methods for b_{abs}, BC, EC, and OC at the Fresno Supersite. The Phase I report was reviewed by the California Air Resources Board (ARB), accepted on October 6, 2006, and is included as an appendix to this report. The second phase of the study evaluated global and regional BC inventories and approaches for constructing a BC inventory for California. The goals of Phase II include: 1) review emission inventory methodology and current inventories for BC, EC, and OC; 2) review BC/EC and OC emission factors used in the ARB emission inventory and compare with emission factors for mobile and biomass burning sources from recent studies; 3) review models and source profiles used to convert estimated particulate matter (PM) emissions to BC/EC and OC emissions; 4) apply a model using the ARB $PM_{2.5}$ emission inventory to estimate BC/EC and OC emissions in California; 5) evaluate uncertainties in estimated BC/EC and OC emissions; 6) summarize database availability and quality assurance/quality control; and 7) develop recommendations for constructing BC/EC and OC emission inventories for California. Study outline and major findings of each Phase are summarized below.

Phase 1: Phase I of this study was carried out through four major tasks: 1) the first task is a critical review of literature on 19 different carbon-analysis methods and 80 carbon intercomparison studies published between 1981 and 2005 was conducted; 2) the second task focused on developing carbon analysis quality assurance and quality control (QA/QC) plans; 3) for the third task, pure and externally mixed (with sodium chloride [NaCl]) aerosols from diesel engine, acetylene flame, electric arc, and wood-combustion aerosols were generated and sampled in the laboratory under controlled conditions. Continuous b_{abs} and BC measurements were made using the photoacoustic analyzer (PA, 1047 nm) and a seven-color aethalometer (7-AE, 370, 470, 520, 590, 660, 880, 950 nm), along with sample collection on Teflon-membrane and quartz-fiber filters. In addition, carbon black and graphite powders were resuspended and collected on quartz-fiber filters for carbon analysis; and 4) the fourth task completed an intensive measurement campaign at the Fresno Supersite between 8/18/05 and 9/17/05, which included six continuous light absorption instruments (two wavelength [2-AE, 370, 880 nm] and 7-AE aethalometers, two PA [532 and 1047 nm], one particle soot absorption photometer [PSAP; 467, 530, 660 nm], and one multi-angle absorption photometer [MAAP; 670 nm]), along with 24-hr sample collection using integrated samplers. This complemented measurements taken during a winter intensive operating period (IOP, 12/1/03 to 12/22/03). Findings from the laboratory intercomparisons were applied in understanding the differences observed at Fresno.

The literature review identified possible biases in thermal and optical methods. For filter-based thermal/optical analyses, the charring correction followed by early EC evolution in an inert atmosphere (due to trace oxidants) represented the most important uncertainty in thermal/optical methods (Chow et al., 2004a), biasing the OC/EC split. For the DRI Model 2001 carbon analyzer, QA/QC procedures were developed including: 1) multi-point temperature calibrations;

2) characterization of analysis atmosphere; 3) carbon analyzer calibration; and 4) calibration of laser intensity using neutral density filters. These procedures have been shown to improve the precision of OC/EC and carbon fraction measurements. For instance, without temperature calibration, the sample temperature is typically biased high by 14 to 22 °C, causing up to 30% change in carbon fraction concentrations. This does not affect the OC/EC split, however. The review indicated that b_{abs} measurements by the PA compared well (within ±3%) with the difference between light extinction by optical extinction cell (OEC) and scattering by nephelometer for pure soot sample or soot mixed with salts (Sheridan et al., 2005). The studies also pointed out the need for correcting filter-based absorption methods for particle light scattering (b_{scat}), the uncertainty involved in σ_{abs} estimates and its effect on b_{abs} measurements, the influence of organic aerosols on b_{abs}, and its influence on the Angstrom absorption exponent (α).

In terms of total carbon (TC), diesel, acetylene flame, and electric arc samples were generated typically within 15% variability. Wood smoke samples showed as much as 50% variability. EC/TC ratios measured by thermal/optical methods showed consistency within each source type, as well as diversity between source types. The STN and French two-step protocols yielded EC/TC ratio similar to (within ±5%) those of the IMPROVE_A protocol for diesel soot (EC/TC ~60%), acetylene flame soot (~96%), and electric arc soot (~50%). The French two-step and STN protocols were lower for EC (86% and 46%, respectively) in wood smoke compared to the IMPROVE_A protocol. The presence of NaCl caused EC to be released at lower temperatures, and was limited by the presence of oxygen (O_2) and charring correction. While it affected the abundance in the EC fractions, it did not affect the OC/EC split in the IMPROVE_A and STN protocols. The French two-step protocol that operates in pure O_2, without charring corrections, reported >60 to 90% lower EC than IMPROVE_A_TOR for all 19 samples. When comparing the IMPROVE_A EC to PA (1047 nm) b_{abs}, the EC σ_{abs} (1047 nm) varied by ~50% in the range of 2.7 to 5.3 m^2/g among the different source types. There is no universal conversion factor that can be applied to convert b_{abs} to BC/EC concentrations. The ratio of AE b_{abs} to PA b_{abs} was influenced by BC concentrations; lower ratios were found to be associated with higher BC concentrations.

Using the IMPROVE_A protocol, the EC/TC ratios at the Fresno Supersite were 0.22 ± 0.04 and 0.26 ± 0.05 for summer and winter IOPs, respectively. The EC/TC ratio during winter was close to the EC fraction in wood smoke (0.26 ± 0.12). The σ_{abs} (1047 nm) of EC during the winter IOP (2.5 m^2/g) was also similar to that of wood smoke EC (2.7 m^2/g). The value of α in the Angstrom Power Law, determined by 7-AE during the summer IOP (0.95 ± 0.04) was 10–20% higher than that observed for diesel and acetylene flame soot (0.79 ± 0.09 to 0.86 ± 0.12), from both pure source aerosol and when mixed with NaCl. This indicates that the summer aerosol at Fresno, while being influenced by diesel emissions, might be mixed with aged or secondary aerosols. The α during the winter period (1.2 ± 0.11) was closer to that observed for emissions from wood combustion (1.2 ± 0.51). Despite the potential bias in the aethalometer, this study confirms a higher α for wood smoke than for diesel soot.

Results suggest that the IMPROVE_A and STN protocols estimate similar EC for the source samples (except wood smoke). The presence of a catalyst such as NaCl changes the abundances in EC fractions, but not the OC/EC split in IMPROVE_A and STN protocols. The French two-step protocol was influenced greatly by the aerosol matrix. A single value of σ_{abs} does not exist. Moreover, $\alpha = 1$ in the Angstrom Power Law that is commonly used to scale b_{abs} to different wavelengths varied from 0.5 to 1.4. These observations may be explained by more complex aerosol optical models that consider particle size distributions, morphology, and internal/external mixing characteristics.

Phase II: Phase II of this study describes and evaluates state-of-the-science BC/EC and OC emission inventories and provides a framework for creating inventories for California. Global BC and OC emission estimates range from 8 – 24 and 33 – 62 Tg/yr, respectively. North American BC emissions accounted for ~6% of the global total, and California BC emissions accounted for <~0.4% of global emissions. Global inventories are based on fuel use estimates and emission factors taken from published articles and reports. These emission factors vary regionally and depend on the degree of economic development. They do not represent California's special mixture of fuels, combustion technology, operating conditions, and aggressive emission controls.

The most accurate inventories use a bottom-up approach where emission factors and activities are specified for all stationary, area, and mobile sources. Examples include the California Air Resources Board (ARB) inventory for criteria pollutants in California and the U.S. National Emissions Inventory (NEI). Because such inventories estimate PM emissions, they provide a basis for estimating BC and OC emissions when the BC and OC PM fractions are measured in specific source types. EC and OC mass fractions are included in source profiles that are used to produce speciated PM inventories and for receptor-oriented source apportionment modeling.

ARB PM emission factors are based on emission models such as EMFAC2007 and OFFROAD for mobile sources and the Emission Estimation System (EES) model for biomass burning sources. In this study, EMFAC2007 produced reasonable agreement with recently-measured values for heavy-duty diesel-fueled vehicles, but it did not capture the large variability in measured gasoline-fueled vehicle emissions. EES provided reasonable estimates for dry litter burning, but it underestimated PM emissions from wet herb and shrub, regen, and wet needles from Ponderosa and Lodgepole Pine trees. ARB emission factors overestimated Chemise (Chaparral) but underestimated rice straw and grass (Grassland) burning. The EES emission model can be updated with more recent emission factor measurements, provide flexibility for estimating specific fire events, and estimate the uncertainty of the emission factor estimates.

Recently measured source profiles were compiled into a database to supplement the U.S. EPA SPECIATE version 4.0 and ARB source profile libraries. Many of the recent studies lack EC and OC measurements, or they applied EC and OC analysis methods that are not compatible with ambient data. A set of the assembled source profiles was applied to the ARB 2006 $PM_{2.5}$ emission inventory to estimate BC/EC and OC emissions in California. Total BC/EC emissions were 52,084 tons/yr. Major sources included biomass burning (wildfires, managed burning, and residential fuel combustion), and off-road and on-road engine exhaust. Statewide OC emissions (107,979 tons/yr) were twice BC/EC emissions (52,084 tons/yr). BC/EC emissions derived from the 1995 ARB inventory (38,781 tons/yr) were in reasonable agreement (33,281 tons/yr) with those extracted from California's grid squares from a 1996 global inventory. However, there were large differences for fuel categories (e.g., fossil fuels and biofuels) and source types, indicating that the overall agreement may have been fortuitous.

California BC/EC and OC emission estimates are sensitive to the choice of source profiles used to convert $PM_{2.5}$ to BC/EC and OC. Recently measured gasoline- and diesel-fueled vehicle exhaust source profiles from the U.S. EPA's SPECIATE version 4.0 source profile library resulted in twice the mobile on-road BC/EC emissions compared with the profiles drawn from the ARB source profile library. Using ARB gasoline- and diesel-fueled vehicle source profiles provided 17% lower statewide BC emissions. Source profile documentation in both libraries is limited, making selection of appropriate profiles difficult to justify.

ARB can build on its current $PM_{2.5}$ emission inventory effort by coupling relevant source profiles containing BC/EC and OC abundances with its inventory system. Existing profiles were assembled into a database and documented, and this can serve as a starting point. Examination of these existing profiles indicates that they are insufficient to represent all of the major source types, especially for biomass burning and non-road engine exhaust. More systematic testing of these emissions, using diluted plumes and a common carbon analysis method, are needed to fill in the gaps.

Further traceability is also needed for emission factors and activity databases, especially those used by the local air districts to construct the emissions they submit to the state inventory. Such data are currently not always available but would enable studies to evaluate the sensitivity of BC/EC and OC emissions to variability and uncertainty in these parameters. California has a wealth of speciated $PM_{2.5}$ measurements from the long-term IMPROVE network operated in its national parks and wilderness areas and numerous special studies conducted in central California, the San Francisco Bay Area, and the South Coast Air Basin. Estimated $PM_{2.5}$, EC, and OC emission inventories can be evaluated by comparing measured concentrations with those estimated in air quality models.

Example State and Local Policy/Initiatives

- Low Emission Vehicle III Technical Support Document, Appendix U: LEV III Climate Change Impacts of Black Carbon Particles, *California Air Resources Board*

- Understanding Particulate Matter, Section 1-C PM and Climate Change, *Bay Area Air Quality Management District*

- Multiple Air Toxics Exposure Study III Final Report Executive Summary, *South Coast Air Quality Management District*

- Forest Resource Sustainability in Placer County California, *Placer County Air Pollution Control District*

APPENDIX U

PROPOSED

LEV III CLIMATE CHANGE IMPACTS OF BLACK CARBON PARTICLES

TECHNICAL SUPPORT DOCUMENT

Date of Release: December 7, 2011
Scheduled for Consideration: January 26, 2012

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I. INTRODUCTION

Airborne particles in the ambient air play an important role in the overall energy balance of the atmosphere by scattering and absorbing incoming and outgoing solar and terrestrial radiation (the “direct effect”) and by modifying the microphysical properties of clouds (the “indirect effects”) through their role as cloud condensation nuclei and/or ice nuclei . Direct and indirect effects on the climate by atmospheric particles remain one of the principal uncertainties in estimates of total anthropogenic radiative forcing. But recent advances in the understanding of the role of particles in the global energy balance are closing the existing knowledge gaps. Changes to the Earth’s radiation balance can impact the climate at both global and regional scales.

The chemical composition and climate change impacts of particles vary with their sources; for example, particles emitted into the air as urban industrial pollution influence the climate vary differently from windblown desert dust or sea salt particles. To further complicate this picture, particles tend to remain in the air for only a few days to about a week, resulting in extreme spatial and temporal variability over the surface of the Earth. Major components of fine particles such as sulfate, nitrate, organic compounds, dust, and sea salts have reflective properties that scatter radiation (negative radiative forcing or cooling impact). Carbonaceous particles (those that contain organic and black carbon) are particularly important because of their abundance in the atmosphere, and the characteristics of the carbon vary significantly depending on their origin. Black carbon (BC) is the principal absorber of visible solar radiation in the atmosphere while organic carbon (OC) is often described as light-reflecting compounds. Recent studies show that certain fractions of organic carbon can also absorb solar radiation efficiently but differ from typical BC, and they are referred to as “brown carbon”[1]. Its sources are known to be low-temperature biomass and biofuel burning as well as heterogeneous or multiphase processes that are not clearly determined yet. In addition, optical and chemical properties of brown carbon have not been determined consistently[2]. Further work is necessary through both observations and improved model simulation of brown carbon particle for better assessing its effects on climate.

The major anthropogenic sources of BC are fossil fuels and biofuels (biomass burning for domestic energy). The atmospheric fate and climate impacts of BC from different regions differ considerably. Atmospheric processes that occur after BC is emitted, such as mixing, aging, and coating, also affect the net influence of BC on climate. Because the climate effects of BC aerosol depend strongly on its physical and chemical properties, as well as on its residence time and distribution in the

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[1] Lukacs, H., et al., (2007). Seasonal trends and possible sources of brown carbon based on 2-year aerosol measurements at six sites in Europe. Journal of Geophysical Research e Atmospheres 112. http://publik.tuwien.ac.at/files/pub-tch_7878.pdf

[2] Yang, M.; Howell, S.G.; Zhuang, J.; Huebert, B.J. (2009). Attribution of aerosol light absorption to black carbon, brown carbon, and dust in China -interpretations of atmospheric measurements during EAST-AIRE. Atmos. Chem. Phys., 9(6): 2035-2050.http://www.atmos-chem-phys.org/9/2035/2009/acp-9-2035-2009.pdf

atmosphere[3], a thorough understanding of these properties and accurate techniques for the determination of BC in the atmosphere and from sources are deemed essential.

In principle, the relatively strong light absorption properties of BC can be used to infer BC from an optical measurement and knowledge of the mass specific absorption of BC. Estimates of BC are made with a variety of instrumentation and measurement techniques. This has also resulted in a variety of definitions related to chemical and/or physical particle properties, intended applications, and the different measurement and estimation approaches, and has given rise to an array of descriptive terms such as "graphitic carbon", "elemental carbon", "black carbon", and "soot" which are used in the literature as interchangeable with BC. Light absorption, BC, and elemental carbon derived from different measurement methods and in different environments have been compared in more than 100 reports and publications[4]. Among commonly used methods, the results are highly correlated, but the absolute values can differ by factors of two or more. Nevertheless, a high correlation among values for different methods suggests that empirical relationships might be established that would allow some predictability of one type of measurement from another.

The large interest in airborne particles and their radiative impact is derived in part from the Intergovernmental Panel on Climate Change (IPCC)'s[5] conclusion that human-caused climate change has resulted primarily from changes in the amounts of greenhouse gases (GHGs) in the atmosphere, but also from changes in small particles. In recent years there has been increased attention in the particle research community about the potential of BC to cause global warming. The ability of BC to absorb light energy and its role in key atmospheric processes link it to a range of climate impacts, including increased temperatures, accelerated ice and snow melt, and disruptions to precipitation patterns. It has been proposed that light absorbing particles in the atmosphere act as a greenhouse pollutant whose net forcing is warming and is second only to carbon dioxide (CO_2). Ramanathan and Carmichael [6] estimate a BC forcing of 0.9 watts per square meter (W/m^2) or more than half of the 1.6 W/m^2 for CO_2. This estimate of the forcing due to BC is larger than most prior estimates including those of the IPCC 4th assessment report.

Numerous national and international reports highlight the critical role of BC in climate change. EPA, in consultation with other Federal agencies, prepared a comprehensive report[7] to Congress on the climate effects of BC. The report synthesized available information on sources of BC, its impacts on global and regional climate, and the

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[3] Jacobson, M. Z., (2001) Strong radiative heating due to the mixing state of black carbon in atmospheric aerosols *Nature, 409*, 695-697. http://www.stanford.edu/group/efmh/jacobson/Articles/VI/nature.pdf

[4] Watson, J.G., Chow, J.C., Chen, L.-W.A., (2005). Summary of Organic and Elemental Carbon/Black Carbon Analysis Methods and Inter-comparisons. AAQR 5, 65-102. http://aaqr.org/VOL5_No1_June2005/6_AAQR-05-06-OA-0006_65-102.pdf

[5] IPCC (2007), Working Group I: The Physical Science Basis, Chapter 2: Changes in Atmospheric Constituents and in Radiative Forcing, available at http://www.ipcc.ch/pdf/assessment-report/ar4/wg1/ar4-wg1-chapter2.pdf

6. Ramanathan, V. and Carmichael, G. (2008) Global and regional climate changes due to black carbon. Nature Geoscience 156, 221-227.

[7] EPA (2011). Report to Congress on Black Carbon External Peer Review Draft. EPA-450/D-11-001, March 2011.

potential utility and cost-effectiveness of mitigation options for reducing climate and public health impacts of BC. The United Nations Environment Program (UNEP)[8] recently released a report which summarizes findings and conclusions of an assessment of BC and tropospheric ozone. The assessment looks into all aspects of anthropogenic emissions of BC and tropospheric ozone precursors, such as methane. It examines a large number of potential measures to reduce harmful emissions, identifying a small set of specific measures that would likely produce the greatest benefits, and which could be implemented with currently available technology. In May 2011, a task force, convened by the Arctic Council, produced a comprehensive technical document [9] on assessment of emissions and mitigation options for black carbon. The task forces has compiled and compared national and global BC emissions inventories, examined emission trends and projections, synthesized existing policies and programs, and identified additional emission mitigation opportunities for BC.

The heightened interest in BC mitigation today is built on the well-recognized association of these emissions with localized air pollution and their severe negative health impacts while also achieving significant climate co-benefits. Understanding the role played by BC is therefore critical for three reasons: from the perspective of understanding climate change, distinguishing between those particles that are exacerbating the GHG impacts from those that are masking it (through their cooling effect) is important. That understanding will allow better characterization of the source impacts and identifying mitigation options. The second reason is that unlike carbon dioxide, BC's effects are immediate, lasting only a matter of weeks, and are regional in their effect. Thus, a mitigation strategy of reducing BC emissions in certain locations would create an immediate relative cooling. This means buying some time and avoiding going beyond some of the irreversible tipping points such as the melting of arctic ice and mountain snowpacks. Third, BC, as a key component of fine particulate matter (PM2.5), contributes to harmful health effects, including premature death. If BC can indeed provide a double win from both a global climate and local air pollution perspective, it makes the strong case for climate action that much more compelling.

The following is a summary of the current scientific knowledge on black carbon, including where it comes from, its atmospheric effects, the overall impact on environment, and the need for motor vehicle control.

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[8] UNEP (2011). Integrated Assessment of Black Carbon and Tropospheric Ozone: Summary for Decision Makers. http://www.unep.org/dewa/Portals/67/pdf/Black_Carbon.pdf

[9] Arctic Council (2011). Technical report of the Arctic Council Task Force on Short-Lived Climate Forcers. http://arctic-council.org/filearchive/ACTF_Report_22July2011.pdf).

I.

II. OVERVIEW OF RELEVANT SOURCES OF BC

To curb the warming caused by BC, two questions must be answered: where did the BC come from, and what created it. In general, sources of BC include either (a) activities or technologies that emit BC, and (b) geographic areas or regions from where BC is transported to elsewhere. Neither of these is susceptible to easy analysis, mostly because BC is an air pollutant that is regulated indirectly, if at all. For example, some nations and states regulate particulate matter, which includes BC, but other constituents as well. No nation, however, has yet adopted a requirement that directly and explicitly regulates BC, whether for the purpose of protecting human health or curbing global warming.

BC comes from both open biomass burning and from energy-related burning. 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. The use of increasingly clean technologies such as better cook stoves and emission controls on transportation and industrial emissions reduces the particle and BC emissions for a given activity. However, the increase in total energy use that accompanies development, as well as changing industrial development can offset some of the gains from these improvements.

On the global scale, fossil fuels and biofuels account for 66% and 34% of energy-related BC emissions, respectively. East and South Asia account for more than 50% of global energy related BC emissions. In their respective regions China and India account for most of the atmospheric burden. These emissions are from the transport sector and from biofuels which suffer from inefficient combustion. In addition, domestic coal combustion in East Asia accounts for a considerable fraction of BC emissions in this region. The contribution of different regions to the global burden follows the corresponding contributions to emissions. The largest contribution to the burden is from East Asia (37%) followed by South Asia (16%), Africa (14%), Europe (12%), North America (10%), South America (7%), west and central Asia (4%), Australia (<1%) and Oceanic regions (<1%). The relatively longer atmospheric residence time for African emissions results in a contribution of 14% to the global BC burden, compared to a contribution of 10% to emissions[10].

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 the one from Bond et al.[11] is the most widely used and referenced. Compiling a global BC inventory is difficult for several reasons: varying emissions among similar sources, varying measurement techniques, and different PM

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[10] Reddy, M. S., and O. Boucher (2007), Climate impact of black carbon emitted from energy consumption in the world's regions, *Geophys. Res. Lett.*, 34, L11802, doi:10.1029/2006GL028904.

[11] Bond, T.C.; Streets, D.G.; Yarber, K.F.; Nelson, S.M.; Woo, J.H.; Klimont, Z (2004). A Technology-Based Global Inventory of Black and Organic Carbon Emissions from Combustion; *J. Geophys. Res. Atmos 109*, D14203; doi: 10.1029/2003JD003697.

size cut points used in the measurements, and the definition of BC itself used in the inventories. Global BC and OC emission inventories are also complicated by a lack of detailed information on source types, emission factors, activities, and controls, especially in the developing world. Recent estimates of global emissions of BC and OC range from 8 to 24 terragrams (Tg) and 33 to 62 Tg per year, respectively. Thus, uncertainty is the hallmark of these studies, with emission estimates varying by a factor of 2 or more[12]. Bond et al estimated that the United States accounted for 5.6% (0.45 Tg/yr) of the global total.

In 2005, emissions of BC from U.S. sources total about 0.65 million tons (0.58 Tg), which represents about 8% of the global total. Mobile sources account for a little more than half (52%) of the domestic BC emissions. Nearly 90% of the mobile source total is from diesel sources. Open biomass burning is the next largest sources in the U.S., 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 (EPA, 2011). The degree of difference between the EPA inventory and Bond et al (2004) inventory for U.S. emissions is driven by EPA estimates for open burning and (to a lesser extent) for mobile sources in the U.S. that are higher than those from the global inventories. Wildfire emissions can vary greatly from year to year, and this may explain some of the difference between the estimates for open burning. Also, EPA estimates include all non-road and on-road emissions in the transportation source category, while global inventories group emissions from some of the smaller non-road 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.

California's emissions inventory for criteria pollutants and the U.S. National Emissions Inventory include PM emissions, which provide a basis for building a bottom-up BC and OC emissions inventory. BC and OC emissions can be estimated from source-specific PM2.5 emissions and the relative BC and OC fractions in the emitted PM. For an ARB-sponsored study by the Desert Research Institute[13], Chow et al. evaluated global and regional BC inventories and approaches for constructing a BC inventory for California. A black carbon inventory for California of 38,730 t/yr was comparable to the 33,280 t/yr estimated from a bottom-up global BC inventory. However, further examination showed substantial differences among subcategories. Most of the discrepancy was due to differences in open biomass burning (wildfires and agricultural waste) for which carbon emissions are highly variable. BC and OC emissions are sensitive to the availability and variability of existing source profiles, and profiles more specific to fuels and operating conditions are needed to increase emission accuracy.

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[12] Chow JC, Watson JG, Lowenthal DH, Chen LW, Motallebi N., (2010). Black and organic carbon emission inventories: review and application to California. J. Air Waite Manag Assoc. Apr;60(4):497-507.

[13] Chow, J.C., J.G. Watson, D.H. Lowenthal, L.W.A. Chen, (2009). Climate Change—Characterization of Black Carbon and Organic Carbon Air Pollution Emissions and Evaluation of Measurement Methods. Phase II: Characterization of Black Carbon and Organic Carbon Source Emissions. California Air Resources Board (Contract No.04-307). http://www.arb.ca.gov/research/apr/past/04-307_v2.pdf

Using a bottom-up approach, California emitted 52,084 t BC/yr during 2006. Among the sources, the largest single source was wildfires, 29% of total BC emission. The next two largest single sources were off-road mobile sources, 23% of the total BC emission, and on-road mobile sources, 20% of total BC emission (Table 1). Statewide OC emissions of about 107,979 tons/yr were twice BC emissions.

Table 1. BC and OC emissions (tons/year) for California in 2006 (Source: Chow et al.).

Source	BC	OC	%BC	%OC
Mobile (On-Road)				
Gasoline Powered	2,644	4,657	5%	4%
Diesel Powered	7,840	5,047	15%	5%
On-Road Total	10,484	9,704	20%	9%
Mobile (Off-Road)	12,158	13,890	23%	13%
Residential fuel combustion	4,004	17,422	8%	16%
Managed burning and disposal	7,374	19,652	14%	18%
Wildfires	15,161	29,530	29%	27%
Miscellaneous	2,903	17,781	6%	16%
Total	**52,084**	**107,979**	**100%**	**100%**

Note: percentage shows the variability of BC and OC fractions for major source types relative to the total BC and OC emissions, respectively.

California BC emission estimates were sensitive to the choice of source profiles. When a base-case diesel profile with a 50% BC abundance was replaced by one with an BC content of 26%, BC emissions from mobile on-road and mobile other (off-road) sources resulted in a 17% decrease in total BC emissions. The efficacy of the approach used to estimate BC and OC emissions in California depends on the accuracy and comprehensiveness of available PM emission factors, source activities, and BC and OC source profiles. Measurement techniques for BC and OC should also be standardized. These limitations need to be overcome to improve BC and OC inventories.

The starting point of all climate modeling studies of BC forcing is the emission inventory. However, Bond et al.[14] estimate that the global (and regional) emission inventory is subject to an uncertainty of about a factor of two, because of a lack of proper knowledge of emission factors, activity data, and technology splits. The regional contributions to atmospheric burden largely follow emissions from the respective regions. The emission of BC is quite dependent on the combustion process and emissions can vary significantly even among apparently similar sources. The need for a robust BC emission inventory that can capture some of these fine details is paramount. Studies are underway to test the emission estimates against currently available field observations; and we expect that iteration among emissions, atmospheric measurements, model results, and combustion tests will result in improved understanding of the magnitude of carbonaceous particle emissions.

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[14] Bond, T.C. et al. (2007). Historical emissions of black and organic carbon aerosol from energy-related combustion, 1850–2000, Global Biogeochem. Cycles, 21, GB2018, doi:2010.1029/2006GB002840. http://www.sage.wisc.edu/pubs/articles/M-Z/Trautmann/BondetalGBC07.pdf

III. CLIMATE IMPACT OF BLACK CARBON IN CALIFORNIA

Recent work on climate change impacts in the western USA has focused attention on the shift in snowmelt timing toward earlier dates (Stewart et al.[15]), the shift from snow to rain (Knowles et al.[16]), the earlier onset of spring, and the effect that these changes will have on water supply in California and throughout the western USA. Pierce et al.[17] showed that about half of the observed decline in western USA springtime snowpack (1950–1999) results from climate changes forced by anthropogenic GHGs, ozone and particles. Although these trends in climate are largely attributable to increasing atmospheric concentrations of GHGs, recent modeling work has drawn attention to the role of soot (which is mostly BC) in modifying climate by reducing snow albedo. Hansen and Nazarenk[18] showed that soot may reduce snow and ice albedo in Northern Hemisphere land areas by as much as 3%, resulting in a climate forcing of +0.3 W/m^2. They found that due to positive feedbacks, the "efficacy" (change in air temperature per unit forcing) of soot is about twice that of CO_2. This indirect soot forcing may have contributed to global warming of the past century, including the trend toward early springs in the Northern Hemisphere, thinning Arctic sea ice, and melting land ice and permafrost. However, Hansen and Nazarenk (2004) also states that the substantial role inferred for soot in global climate does not alter the fact that greenhouse gases are the primary cause of global warming in the past century and are expected to be the largest climate forcing for the rest of this century.

The snowpack in the Sierra Nevada region is important to California's water resources. The high elevation snowpack serves as a natural reservoir that stores fresh water during the wet, cold season and releases it gradually during the dry, warm season. About 60% of the water supply for Southern California comes from melting Sierra Nevada snowpack. Snowmelt also affects hydropower generation in California (Vicuña et al.[19]). Snow albedo is among the most important local parameters in shaping the spatio-temporal variations in snowpack. Surface insolation and more specifically the portion of insolation absorbed by the snowpack is the leading energy source in the evolution of snowpack, especially during the melting period. Thus, variations in snow albedo can exert significant impact on snowpack during the course of accumulation and ablation. The surface albedo of sufficiently deep snowpack, and in turn the amount of

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[15] Stewart IT, Cayan D.R, Dettinger.M.(2005) Changes toward earlier streamflow timing across western North America. J Clim 18:1136–1155. http://tenaya.ucsd.edu/~dettinge/stewart_timing.pdf

[16] Knowles N, Dettinger M, Cayan D (2006) Trends in snowfall versus rainfall in the western United States. J Clim 19:4545–4559. http://tenaya.ucsd.edu/~dettinge/jclim_rain_v_snow.pdf

[17] Pierce DW, Barnett TP, Hidalgo H, Das T, Bonfils C, Santer B, Bala G, Dettinger M, Cayan D, Mirin A, Wood A, Nozawa T (2008) Attribution of declining western U.S. snowpack to human effects. J Clim 21:6425–6444. http://tenaya.ucsd.edu/~dettinge/swe_over_p_attribution.pdf

[18] Hansen J, Nazarenk L (2004) Soot forcing via snow and ice albedos. Proc Natl Acad Sci 101:423–428. http://www.pnas.org/content/101/2/423.full.pdf+html

[19] Vicuna, S., R. Leonardson, M. Hanemann, L. Dale, and J. Dracup. 2008. "Climate change impact on high elevation hydropower generation in California's Sierra Nevada: A case study in the upper American River." *Climatic Change* 87:S123–S137.

the insolation absorbed by the snowpack, depends largely on the ice grain size and impurities within or at the surface of ice grains (Waliser et al.[20]). There exist only a limited number of studies on the alteration in snow albedo and its impact on surface hydrology due to dust and BC particles deposited on snowpack. This is an important concern because the amount of BC deposition on snowpack is closely related with anthropogenic emissions. Thus, anthropogenic emissions that have bearing on the causes and characteristics of global climate change include an influence on local snowpack by altering snow albedo.

Hadley et al.[21] examined the concentration of BC particles in snow in California and the potential of these particles to reduce albedo and increase melt. Samples of falling snow and rain were collected at three locations in California: Central Sierra Snow Laboratory in the Sierra Nevada, Lassen Volcano National Park in the Southern Cascades, and at Trindidad Head on the Northern California coast. This study provides one of the first direct measurements for the efficient removal of black carbon from the atmosphere by snow and its subsequent deposition to the snow packs of California. The data reveal that BC concentrations in the Sierra Nevada snowpack are sufficient to perturb both snow melt and surface temperatures. The concentration of BC measured in the snow is consistent with recent model predictions for BC concentration in California mountain snow.

The associated reduction in snow albedo and reduced snow packs in early spring snowpack has been shown by regional climate models to be significant. All three stations reveal large BC concentrations in precipitation, ranging from 1.7 ng/g to 12.9 ng/g. The BC concentrations in the air after the snowfall were negligible suggesting an extremely efficient removal of BC by snow. The data suggest that below cloud scavenging, rather than ice nuclei, was the dominant source of BC in the snow. A five-year comparison of BC, dust, and total fine particle mass concentrations at multiple sites reveals that the measurements made at the sampling sites were representative of large- scale deposition in the Sierra Nevada. The relative concentration of iron and calcium in the mountain particle indicates that one-quarter to one-third of the BC may have been transported from Asia.

Coats[22] quantified the decadal-scale time trends in air temperature, precipitation phase and intensity, spring snowmelt timing, and lake temperature in the Tahoe basin, and related the trends to large-scale regional climatic trends in the western USA. It states that the Tahoe basin has abundant winter-time emission sources of BC. Many homes are heated with wood-burning stoves, and traffic during the ski season is heavy

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at times. Air quality data are available for a station at South Lake Tahoe (SLT) and at Bliss State Park (BSP). SLT is the most urbanized area of the basin and had the highest elemental carbon concentrations, averaging 2.02 μg/m^3. BSP is less influenced by local sources, but like the rest of the basin is down-wind from major metropolitan areas in Sacramento Valley and Bay Area. The study indicates that atmospheric deposition of black carbon in the Tahoe basin may be implicated in the shift in snowmelt timing, increasing air temperature and the shift from snowfall to rain. Snowpack energy budget studies together with analysis of snowpack black carbon concentrations are needed to test this "snow albedo perturbation" hypothesis. Monitoring of black carbon in snow should be added to routine water quality monitoring in the Tahoe basin.

IV. GLOBAL WARMING POTENTIAL OF BC

Multi-gas mitigation strategies require metrics to compare the effect of emissions of different GHGs. Following its endorsement by the IPCC and its adoption within the Kyoto Protocol, the Global Warming Potential (GWP) has established itself as the metric of choice for the derivation of CO_2-equivalent emissions. GWP is a well-defined metric based on radiative forcing that continues to be useful in a multi-gas approach. Shortcomings have been identified; however, the scientific basis has not been fully established to address these shortcomings comprehensively in any currently discussed metric.

GWPs were meant to compare emissions of long-lived, well-mixed GHGs. Short-lived species, like BC, vary spatially and, consequently, it is very difficult to quantify their global warming forcing. 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) 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. The choice of a metric depends greatly on the policy goal. No single metric can be used to accurately address all the consequences of emissions of all the different climate forcers. The appropriate metric to use depends on factors such as: the time scale (20 years, 100 years, or longer), the nature of the impact (radiative forcing, temperature, or damages), concern over different processes (indirect effects, snow albedo changes, co-emissions), and whether sources and impacts should be calculated regionally or globally. The assessment of metrics will be included in the IPCC 5th assessment report (AR5; to be released around September 2013) process in an integrated manner with participation from all three working groups and the IPCC Task Force on GHG inventories. This process will likely include an assessment of numerical values for metrics that have been proposed in the literature.

It is important to note that different climate models could yield different results even if the same metric definition is chosen. Climate models are recently beginning to address the full effects of light-absorbing BC. However, estimates of the importance of carbonaceous particles as global warming agents vary greatly. Part of the variability between models is due to the inclusion or exclusion of certain physical effects, including the nature of the mixing of absorbing BC aerosols with other, scattering aerosols, the multiple effects of aerosols on clouds, and the effects of BC on surface albedo. Differences also arise between models based purely on physical theory and those that attempt to fit observational data. An additional challenge in comparing model results is that different authors often present their results in different terms: some use the net radiative forcing (RF) of carbonaceous particles, others the RF of BC alone, and still others the net temperature change due to carbonaceous aerosols.

However, within bounded limits, there is no question that BC alters the Earth's energy balance, and is a net warming species. What is clear is that as scientists are able to disaggregate the effects of different particles, BC dominates so clearly and with such magnitude that it can no longer be ignored. The balance has now tipped in favor of taking action despite the remaining uncertainties on the exact metric to use.

While the IPCC 4th assessment report did not publish a GWP estimate for BC in its most recent report, independent estimates have been published in the peer-reviewed literature, including estimates drawn from IPCC report itself. Several leading scientists have reported estimates of the GWP for BC emissions from different sources. Most of the regional differences in GWP are caused by differences in the lifetime of BC. In general, we find in the published literature there are significant variations in the GWP values for BC emissions assigned to different regions. This indicates that BC emissions cause warming primarily in the region they are emitted, and that the role of BC in warming requires close attention to the geography of emissions.

Table 2 shows values from the literature for 100-year and 20-year GWP values of BC. As the table shows, BC's climate-forcing effect is generally very high (compared to CO_2, which by definition has a GWP of 1), and it is much greater when considered on a shorter time horizon like 20 years instead of 100 years. The table shows that generally the 20-year time horizon for BC is about 2000, whereas the 100-year GWP for BC is generally found to be more like 500 to 800. Overall, Table 2 indicates that BC is capable of generating warming that is two orders of magnitude greater than carbon dioxide. On a 20-year horizon, which places greater emphasis on rapid, near-term climate impacts, this BC warming is three orders of magnitude greater than the CO_2 warming.

Hansen et al. (2007) has calculated a GWP for fossil-fuel derived BC of 500, which includes both positive forcing from soot particles as well as the negative forcing from co-emitted OC. This value compares well with other published values: Bond and Sun (2005) calculated a GWP for BC of 680 for the same time horizon, but they did not include the effect of co-emitted species and cloud effects. The direct forcing of BC is significantly dependent on assumptions about how BC particles are mixed with other components of the particle population, with internal mixing tending to accentuate the positive forcing. BC also causes a complex set of impacts on cloudiness, including the so-called semi-direct effect, whereby changes in temperature and humidity structure due to the absorption of solar radiation by BC, alter the structure of clouds.

Table 2. Black carbon global warming potential values from research literature.

Source	Black carbon global warming potential		Indirect (cloud change) forcing
	100-yr	20-yr	
Hansen et al, 2007[23]	~500	~2000	Yes
Bond and Sun, 2005[24]	680	2200	No
Jacobson, 2007[25]	840-2240	2530	Yes
Reddy and Boucher (2007)[26]	480	--	No
Rypdahl et al, 2009[27]	830	2900	No
Fuglestvedt et al, 2010[28]	460	1600	No

Reddy and Boucher (2007) simulated the atmospheric cycle of energy-related BC from different regions and estimated the regional contributions to the global BC burdens and direct radiative forcing. The regional contributions of BC to global mean forcings closely follow the respective contributions to atmospheric burden in the form of PM. The GWP of BC for different regions ranges from 374 to 677 with a global mean of 480. Another variable that plays a role in the overall estimation of BC impacts is the albedo effect. The global mean indirect GWP due to the BC effect on snow albedo is estimated at 281. The indirect GWP due to the BC effect on snow albedo is estimated to be largest for Europe, suggesting that BC emission reductions from this region are more efficient to mitigate climate change. Rypdal et al (2009) also examined the regional differences in BC climate impact. The regions considered are EU17, Rest of Europe, Russia, North America, Latin America, East Asia, Centrally Planned Asia, South Asia, Japan, the Pacific OECD, Africa and the Middle East. The regional GWP values range from 640 to 1130 for the direct effect of BC. Most of the regional differences in GWP are caused by differences in the lifetime of BC, although up to 20% can be explained by differences in the RF per unit mass BC.

In general, we find in the published literature there are significant variations in the GWP values for BC emissions assigned to different regions. This indicates that BC emissions cause warming primary in the region they are emitted, and that the role of BC in warming requires close attention to the geography of emissions. We find Hansen et al. (2007) 100-year GWP of 500, which has been estimated globally, provides a reasonable estimate for use in calculating CO_2 equivalent benefits. Hansen's work is

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widely recognized and used in IPCC. Although, this should be considered a conservative estimate for fossil fuel BC forcing, as discussed by Hansen et al. (2007), because it assumes a high OC/BC ratio for fossil fuel emissions. Also it assigns 50% of the particle indirect effect (which causes cooling) to soot (BC/OC).

Hansen et al (2007) and Bond and Sun (2005) provide a 20-year GWP value for BC of 2000. Because BC particles produce a rapid, short-term, localized impact on the climate, we should also examine the climate effects using GWPs for a shorter time horizon. The 20-year GWP can be considered a means to control the rate of warming, while the 100-year GWP can be considered a means to control committed warming. Hence, reductions in short-lived pollutants like BC will make a significant contribution toward reducing the rate of warming, which is the basis for arguments that such reductions can buy time for future climate mitigation measures to be effective.

V. LIGHT-DUTY VEHICLE BC CLIMATE IMPACT

In developed countries, the BC problem is primarily associated with the high volume of fossil fuel use in transportation, particularly diesel. Chow et al. (2010) assembled data from about 800 PM2.5 source profile libraries and recent studies, and found that the highest BC abundances were found for on-road diesel vehicles and off-road diesel engines. In a study by Strawa et al (2010)[29], particulate emissions from motor vehicles inside a San Francisco Bay Area roadway tunnel (Caldecott) were characterized in 2004 and 2006. The amount of absorbing aerosol emitted and the low values of single scattering albedo indicate that particulate matter from motor vehicles exerts a positive (i.e., warming) radiative forcing and that the impact of medium/heavy duty diesel trucks is greater than light-duty vehicles. Another study by Kirchstetter et al (2008)[30] indicates that annual average BC concentrations in the Bay Area decreased by a factor of 3 over the 1967–2003 period, while diesel fuel use, the main source of BC emissions, increased by a factor of 6. The study also states 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 technology and fuel changes to reduce BC emissions have been successful.

Using data from the Interagency Monitoring of Protected Visual Environments (IMPROVE) program, Bahadur et al. (2011)[31] examined the temporal and the spatial trends in the concentrations of BC for the past 20 years in California. Annual average BC concentrations in California have decreased by about 50% from 0.46 μg m^{-3} in 1989

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[30] Kirchstetter, et al. (2008) Black carbon concentrations and diesel vehicle emission factors derived from coefficient of haze measurements in California, AE, 42(3): 480-491.

[31] Bahadur R., Feng Y., Russell L. M., 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.

to 0.24 μgm^{-3} in 2008 compared to a corresponding reductions in diesel BC emissions (also about 50%) from a peak of 0.013 Tg Yr^{-1} in 1990 to 0.006 Tg Yr^{-1} by 2008. Bahadur et al. attribute the observed negative trends to the reduction in vehicular emissions due to stringent statewide regulations. Their conclusion that the reduction in diesel emissions is a primary cause of the observed BC reduction is also substantiated by a significant decrease in the ratio of BC to non-BC aerosols. The absorption efficiency of aerosols at visible wavelengths - determined from the observed scattering coefficient and the observed BC – also decreased by about 50% leading to a model-inferred negative direct radiative forcing (a cooling effect) of -1.4 Wm^{-2} over California.

California along with national and international climate-change policies have embraced a multi-gas approach where a “basket” of GHG emissions are considered together. In such a plan, emissions of each gas are given a weight relative to CO_2 so that multiple gases can be considered together. Sufficient scientific consensus exists on BC to provide appropriately accurate metrics on how to assess the climate value of actions to reduce BC emissions. Table 3 presents the climate impact of BC as compared with the other GHGs, and shows the product of the GWP and GHG and BC emissions. 2020 BC emissions are based on 3 mg/mi PM emissions, 66% BC fraction (500 $GWP_{100\text{-}yr}$; 2000 $GWP_{20\text{-}yr}$).

Table 3. Approximate illustration of equivalent CO_2 emissions from different GHGs for 2009 and for future (2020 and beyond) year new vehicles.

GHG emission	Global warming potential[a]		100-year gCO_2e/mile		20-year gCO_2e/mile	
	100-yr	20-yr	2009	Future (2020+)	2009	Future (2020+)
CO_2	1	1	337	<200	337	<200
AC refrigerant	1430	3830	6	0	16	0
CH_4	25	72	1.8	0.5	1.8	0.5
N_2O	298	289	0.1	0.03	0.3	0.05
BC [b]	500	2000	0.77	0.5	3.08	1.98

- [a] From IPCC 2007 fourth assessment review (AR4), except BC global warming potential estimate is based on ARB review of scientific literature. Pavley I used IPCC 2001 TAR GWPs values (e.g. 1300 for HFC-134a, 23 for CH_4, 296 for N_2O).
- 100-yr GWPs are IPCC 2007 AR4 (1,430 for HFC-134a, 25 for CH_4, 298 for N_2O) as used by U.S. EPA. 20-yr GWPs are IPCC 2007 AR4 (3,830 for HFC-134a, 72 for CH_4, 289 for N_2O.
- [b] Based on ARB’s recent estimates: i.e., for the four vehicle types LDA, LT1, LT2, and LT3 the 2009 fleet average gasoline exhaust PM is 7.0 mg/mi; for 2020 it is 4.5 mg/mi assuming phase-in of the 3 mg/mi standard from 2017 to 2020 – with 0.22 BC/PM fraction for on-road gasoline from Chow et al[32]

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[32] Chow, J.C., Watson, J.G., Lowenthal, D.H., Antony Chen, L.-W., Motallebi, N. (2011) PM2.5 source profiles for black and organic carbon emission inventories. Atmospheric Environment 45, 5407-5414.

We use two time horizons to estimate the climate impact: the short-term scenario is based on GWP for 20 years, and the long-term scenario is based on the more conventional 100-year GWP. Because BC aerosol exerts a rapid warming effect in the vicinity of the source, we think a 100-year weighted GWP for BC is less appropriate than a GWP based on a 20-year time. This shorter time horizon is better reflective of the speed at which control of, for example, soot emissions can benefit the climate relative to controlling carbon dioxide emissions. The two time frames can lead to a four-fold difference in CO_2eq emissions. Hence, a 20-year time frame gives a better perspective of the speed at which BC controls could benefit the atmosphere relative to CO_2 emission controls.

There is a wide range of BC and OC abundances in PM2.5 source profiles representing the same source type. The median PM2.5-BC abundance of 22% was used in Table 3 for on-road gasoline emission. As discussed earlier, Chow et al (2011) examined BC and OC abundances in source profiles from the U.S. EPA SPECIATE data base along with additional profiles obtained by the authors to evaluate their variability within and between source-types and to assess the effect of this variability on BC and OC emission rates. For profiles compiled in their study, BC and OC ranged 6-38% and 24-75% for on-road gasoline vehicles, and 33-74% and 20-47% for on-road heavy-duty diesel vehicles, respectively.

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 which may change the warming profile of emissions from these vehicles. More testing is needed to determine the typical ratios of EC/PM in LDV exhaust, as these ratios seem to be changing over time. These data can then be used, in combination with other vehicle fleet information, to more accurately estimate the atmospheric contribution of BC by LDV in climate models. The influence of BC on climate and public health in the future, and the need to pinpoint more precisely the effectiveness of various mitigation strategies for reducing BC, depend in large part on the magnitude of future emissions.

While uncertainties remain, emerging research suggests that targeting emission reductions from key sectors can have measurable benefits for both climate and public health climate. Mitigation of BC thus offers a clear opportunity: carefully designed programs that consider the full air pollution mixture (including BC, OC, and other co-pollutants) can slow near-term climate change while simultaneously achieving lasting public health benefits. Furthermore, currently available control technologies and mitigation approaches have already been shown to be effective in reducing BC emissions, often at quite reasonable costs. These mitigation approaches could be utilized to achieve further BC reductions.

VI. SUMMARY

BC is the light-absorbing carbonaceous fraction of PM that results from incomplete combustion of fossil fuels and biomass. BC causes warming primarily in the regions where it is emitted, and therefore merits analysis and solutions at the local scale. The ability of BC to absorb light energy and its role in key atmospheric processes link it to a range of climate impacts, including increased temperatures, accelerated ice and snow melt, and disruptions to precipitation patterns. Numerous national and international reports highlight the critical role of BC in climate change. The heightened interest in BC mitigation today is built on the well-recognized association of these emissions with localized air pollution and their severe negative health impacts. Any climate strategy for reducing BC emissions offers these important co-benefits.

BC is different from long-lived GHGs like CO_2 both in the variety of mechanisms by which it affects climate and its short atmospheric lifetime of days to weeks. This short lifetime, combined with the strong warming potential of BC, means that the climate benefits of reductions in current emissions of BC will be nearly immediate. It also makes reductions in BC emissions a potential near-term opportunity to postpone the effects of rising GHG levels on the global climate. In contrast, long-lived GHGs persist in the atmosphere for centuries. Therefore, reductions in GHG emissions will take longer to influence atmospheric concentrations and will have less impact on climate on a short timescale.

Because BC particles produce a rapid, short-term, localized impact on the climate, we should also examine the climate effects using GWPs for a shorter time horizon. The 20-year GWP can be considered a means to control the rate of warming, while the 100-year GWP can be considered a means to control committed warming. Hence, reductions in short-lived pollutants like BC will make a significant contribution toward reducing the rate of warming, which is the basis for arguments that such reductions can buy time for future climate mitigation measures to be effective. However, since GHGs are by far the largest contributor to current and future climate change, BC reductions cannot substitute for reductions in long-lived GHGs and that deep reductions in these pollutants are essential for mitigating climate change in the long run.

In conclusion, although there remains considerable uncertainty as to the magnitude of the effect of BC on the climate, mounting scientific evidence suggests that reducing current emissions of BC can provide near-term climate benefits, particularly for sensitive regions such as the Arctic. Because of its strong warming potential and short atmospheric lifetime, BC mitigation offers an opportunity to address key climate effects and slow the rate of climate change (i.e., reducing the risk of crossing thresholds with dramatic climate changes). Furthermore, currently available control technologies and mitigation approaches have already been shown to be effective in reducing BC emissions, often at quite reasonable costs. These mitigation approaches could be utilized to achieve further BC reductions. LDVs are currently a minor source of BC emissions compared to heavy-duty diesel engines. CARB anticipates that the stringency of the proposed amendments to the existing PM mass standard will result in reduced BC emissions that can yield significant local and regional climate and health benefit.

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The spatial distribution and extent of roadway emissions may vary based upon temporal factors, such as time of day and season of the year. A study (Zhu et al. 2006) which compared ultrafine particle numbers for daytime and nighttime conditions near a major freeway (I-405) in Los Angeles found that the rate of decrease in ultrafine particles downwind of the freeway was much less at night than during the day. Although traffic volume on I-405 at night was only 25% of the daytime volume, the particle count 30 meters downwind of the freeway was about 80% of the daytime value. The authors attribute the higher ratio of particles to traffic volume at night to a combination of lower wind speed and weaker atmospheric dilution, as well as cooler temperatures which cause increased particle formation in the vehicle exhaust. The study also found that particle counts near the freeway were higher in winter than in summer, for similar reasons to the factors that lead to higher particle counts at night.

Dispersion is key to reducing ambient concentrations and exposure to PM. However, it is important to note that some urban environments, such as tunnels and "urban street canyons", are not conducive to dispersion of air pollutants. When emissions are trapped in enclosed areas, this can lead to much higher local concentrations, and thus much higher population exposure. One study (Morwaska et al. 2008) found that ultrafine particle numbers in the near-roadway environment were roughly 18 times higher than in a non-urban background environment, while measured concentrations in street canyons and tunnels were 27 and 64 times higher, respectively, than background. Another study (Zhou et al. 2008) found that, due to high population density, combined with the lack of dispersion, the intake fraction of emissions in urban street canyons is very high, similar in magnitude to the intake fraction associated with indoor tobacco smoke.

In-Vehicle Exposure

Concerns about elevated exposure to PM near major roadways also apply to drivers and passengers traveling in vehicles on high-volume roads. In fact, the evidence suggests that in-vehicle exposure may be a leading source of exposure to PM and other air pollutants for people who drive on freeways or major arterials on a regular basis. In-vehicle exposure depends on the volume and mix of vehicles on a given road, as well as the type of ventilation system used in the vehicle. Moving vehicles typically have high air exchange rates, allowing emissions from the stream of traffic to penetrate into vehicles. One study (Fruin et al. 2008) found that 36% of total daily exposure to ultrafine particles occurred during a daily commute of 1.5 hours round trip (6% of the day) in Los Angeles, and that 22% of total exposure occurred during 0.5 hours (just 2% of the day) that was spend on freeways. This indicates that exposure rates may be 5 to 10 times higher than average when driving on busy roadways. Thus, even limited time on a freeway can account for a significant portion of total daily exposure to ultrafine particles.

Freeways are also where people are most likely to experience higher exposure to diesel PM, which has been classified by the Air Resources Board as a toxic air contaminant. The 2008 Fruin study found that on freeways in Los Angeles, concentrations of ultrafine PM, black carbon, nitric oxide, and polycyclic aromatic hydrocarbons (PAH) bound to small particles are generated primarily by diesel-powered vehicles, even though diesel vehicles account for only a small fraction (6%) of the traffic on LA freeways. This study also found, however, that on arterial roads concentrations of ultrafine particles appear to be emitted primarily by gasoline-powered vehicle undergoing hard accelerations. Concentrations of ultrafine particles on arterials were roughly one-third those of freeways.

“Self-pollution”, which occurs when the exhaust from a vehicle infiltrates its own passenger cabin, may also contribute to in-vehicle exposure. This has raised concern about risks to children who ride to school on diesel-powered buses. One study (Adar et al. 2008) found that PM2.5 on school buses was double the on-road levels, and that 35% of PM2.5 measured in school buses came from self-pollution. (See description of the Lower-Emission School Bus Program in Section 4 regarding actions to address this issue.)

Aircraft and Airports

Studies conducted by the South Coast AQMD suggest that jet aircraft may be major emitters of ultrafine particles. Typical ultrafine particle concentrations are on the order of 50,000-200,000 particles per cm^3 near freeways; by contrast, ultrafine particle concentrations near jet exhaust can reach 6,000,000 particles per cm^3. As shown in Table 1-3.

Table 1-3 Comparison of Ultrafine Particle Concentrations[4]

Environment	Ultrafine Particle Concentration
Clean background	500 – 2,000 particles per cubic centimeter
Typical urban air	5,000 – 30,000 particles per cc
Freeway	50,000 – 200,000 particles per cc
Jet exhaust	Up to 6,000,000 particles per cc

A study (Hu et al. 2009) that measured ultrafine particles near the Santa Monica Airport, at the residence closest to the airport, and at a nearby school showed correlations of ultrafine particle concentrations from jet exhaust at all three locations. Aircraft operations resulted in average ultrafine particle concentrations elevated by a factor of 10 at 100 meters downwind and by a factor of 2.5 at 660 meters downwind. In fact, the area impacted by elevated UFPM concentrations was found to extend beyond 660m downwind and 250m perpendicular to the wind on the downwind side of the

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[4] Presentation by Dr. Philip Fine of South Coast AQMD to BAAQMD Advisory Council Meeting: Ultrafine Particles 2012 Atmospheric Monitoring of Ultrafine Particles, February 2012.

Santa Monica Airport. This study demonstrated that there may be health implications for residences living in proximity to jetports, especially in the downwind direction.

A study by Carnegie Mellon University researchers (Miracolo et al. 2011) evaluated the effects of photo-oxidation on ultrafine PM emissions from a gas turbine engine designed to mimic a jet aircraft engine. The study found that photo-oxidation created substantial secondary PM, suggesting that it is also important to consider potential secondary PM formation when assessing the influence of jet aircraft emissions.

Back-up Generators

Back-up generators (BUGs), also known as stationary engines and emergency generators, are used frequently by hospitals, office buildings, schools, grocery stories, and government facilities to supply power to a building during a power failure. While power failures are generally rare, BUGs are operated several times a year for testing. Diesel BUGs emit diesel particulate matter and other toxic air contaminants and may contribute significantly to people's exposure to toxics and health risks. In addition, BUGs tend to be concentrated in populated areas, where high numbers of people may be already exposed to high levels of pollution.

A new BUG installed today in the Bay Area poses little health risk during its operating testing hours due to the Air District's and ARB's regulations. However, old BUGs that were installed prior to regulations and continue to be in use today generate high levels of toxics and pose a serious health risk challenge. Even though these BUGs may be used as little as 100-50 hours a year, they can emit enormous amounts of diesel PM since their engines do not comply with any emission standards or contain retrofit technologies. In the Air District's general screening of health risks for BUGs in the Bay Area, the cancer risk for grandfathered BUGs ranges from 20 to 200 in a million in some cases. There are close to 3,000 BUGs in the Bay Area, approximately 1,500 of which may have cancer risks over 10 in a million. The majority of these BUGs are located in Bay Area urban centers. These BUGs contribute heavily to health risks already experienced by people living near roadways and other mobile emissions of diesel PM. The Air District's general health risk screening for stationary sources indicates that addressing emissions from grandfathered back-up generators could significantly reduce exposure to diesel PM, especially in urban areas with already high exposure rates.

Indoor Exposure to PM

Studies have found that most people experience a major portion of their total PM exposure when they are indoors. This is not surprising, since people spend the majority of their time indoors, in the home, office, school, stores, restaurants, etc. According to one study (Qing Yu Meng et al. 2005), adults typically spent 87% of their time indoors, 7% in vehicles, and just 6% outside. The PM that we breathe indoors is a combination of ambient (outdoor) PM that penetrates to the indoor environment, as well as PM emissions produced by indoor sources.

Most people experience a major portion of their exposure to PM when they are indoors.

Studies to date to measure indoor PM levels and population exposure have generally been limited to small numbers of sites, because gaining access to suitable sites (private homes, schools, etc.), installing monitors, and analyzing data requires substantial time and resources. Analyzing indoor concentrations and exposures in multi-unit buildings, such as apartment buildings, is complicated by the fact that PM created indoors can move between units, as well as the fact that heating and ventilation systems, if not properly designed and maintained, can transfer pollutants between units. Nonetheless, the findings of existing studies suggest that indoor exposure to PM is a serious issue that merits more attention.

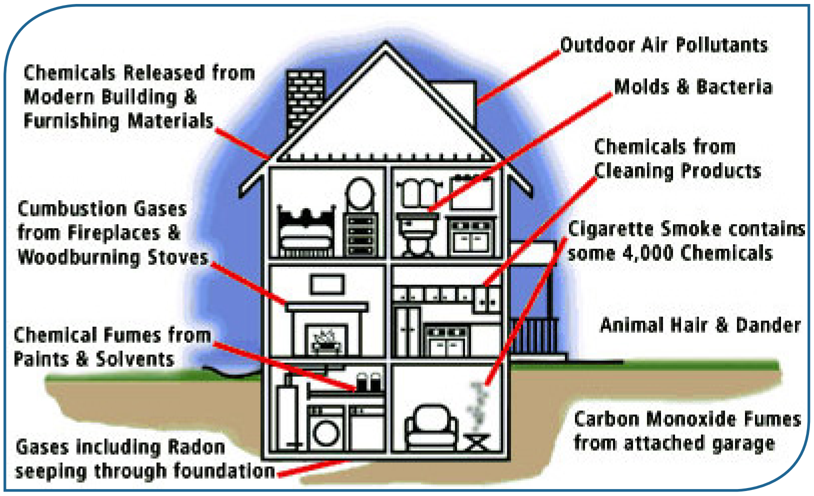
Factors that determine indoor exposure to PM include (1) the ambient (outdoor) PM concentration in the vicinity of the building, 2) the **infiltration rate**: i.e., how much of the ambient outdoor PM penetrates indoors, 3) the **air exchange rate**: how quickly indoor air is replaced by outdoor air, and (4) the amount of primary PM emissions and PM precursors produced in the indoor environment from sources such as cooking, wood-burning, and cigarette smoking. These factors can vary considerably depending upon building type and location, the type of heating and ventilation system, and meteorological conditions.

The infiltration rate of ambient (outdoor) PM to the indoor environment depends upon building materials, characteristics, and design, such as the type of ventilation system, the location of air intake units, whether windows are open or closed, and whether a building has air conditioning or an air filtration system. The PM infiltration rate also varies upon the size and composition of the particles present in the ambient PM. Because different sizes and types of particles have different infiltration rates, the composition of PM in the indoor environment generally differs from the ambient outdoor PM. Ammonium nitrate levels, for example, are generally higher outside than indoors. Ammonium nitrate can exist in either particle or gaseous form in the atmosphere, depending upon temperature. In colder weather, ammonium nitrate particles account for a sizable portion of total ambient PM2.5 in the Bay Area. However, when they encounter warmer air in the indoor environment, ammonium nitrate particles generally volatize (convert to the gaseous form), such that they no longer exist in particle form.

Ultrafine particles are less likely to penetrate through a building envelope because they deposit more rapidly on building surfaces due to Brownian motion at the molecular level. Whereas typical infiltration factors for PM10 and PM2.5 are in the range of 50%, (Ott et al. 2000), infiltration factors for ultrafine particles are on the order of 30% (Wallace & Howard-Reed, 2002). Since ultrafine particles do not easily penetrate to the indoors, this suggests that indoor sources of ultrafine particles play an important role in determining total personal exposure to UFPM.

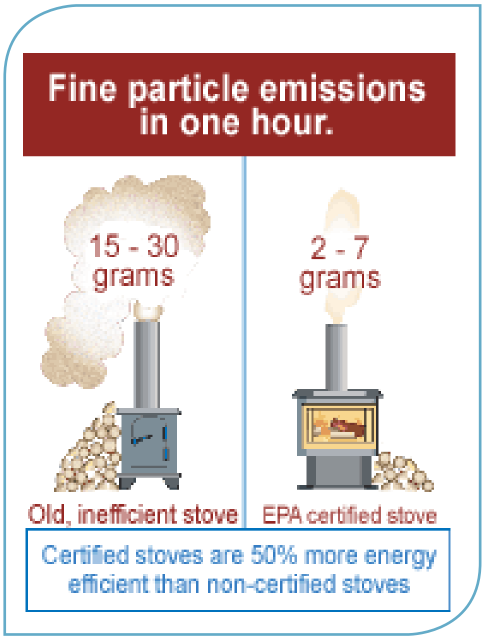
Sources of PM in the Home Environment

Although PM in outdoor air does penetrate to the indoor environment, particles generated within the home often account for a substantial share of indoor PM levels and exposure. Indoor sources of PM include fireplaces and wood stoves, cooking, gas stoves, cleaning products, cigarette smoking, candles, and incense, laser printers, as well as human activities that may re-suspend PM2.5. Indoor PM may also include a mixture of dander from pets, other types of allergens, chemical substances, mineral particulate, mold spores, viruses and bacteria. The RIOPA study (Polidari, A. et al. 2006) found that fine organic particles dominates indoor-generated PM2.5 in the homes that were studied. Indoor sources of PM can cause PM levels to spike, especially because the emissions are often retained within a confined area. Several of the key sources of PM generated in the home environment are briefly described below.



Cooking: Studies have found that cooking is a leading source of ultrafine particles in many homes. Indoor monitors show that ultrafine particle counts spike whenever cooking occurs. Studies suggest that emissions of UFPM are higher from natural gas stoves than from electric stoves, but the particle emission rates are high in both cases. Ultrafine particle levels tend to be significantly higher in homes with gas stoves that use a pilot light (compared to pilot-less stoves). Emission rates when the oven is in use may be greater than for stove-top cooking. One study found that the indoor concentration of ultrafine particles jumped from 5,000 particles per cubic centimeter to 1 million particles – a 200-fold increase – within a few minutes after the oven in a residential kitchen was turned on. [5]

Wood-burning devices: People are exposed to wood smoke in both indoor and outdoor environments. In addition to its negative impact on outdoor air quality, residential wood-burning can be a major source of indoor PM, especially if the chimney or stovepipe does not vent smoke to the outdoors effectively. This problem occurs most commonly when a fire is first ignited and the fireplace flue is not warmed up, thus failing to draw smoke efficiently. One



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[5] Presentation by Susanne Hering, Ph. D., of Aerosol Dynamics to BAAQMD Advisory Council on March 9, 2011.

study (Pierson et al. 1989) found that 70% of smoke from chimneys can reenter the home where it originated and/or neighboring dwellings.

Appliances: Common household appliances, such as clothes dryers, toaster ovens, irons, and laser printers can also produce ultrafine particles, especially appliances that operate by heating metal surfaces.

Cleaning products: Household cleaning products can also produce ultrafine and fine particles in the indoor environment. Scented cleaning products contain terpenes such as pinene (pine scent) and limonene (citrus scent); these terpenes can react with ozone to form ultrafine particles.

Contribution of Indoor Exposure to Total PM Exposure

Lance Wallace and Wayne Ott have done pioneering work using portable particle counters to measure personal exposure to ultrafine particles. In one of their recent studies (Wallace & Ott, 2010) using personal monitors to measure exposures in environments such as homes, cars, and restaurants, they estimated that, on average, 47% of daily personal exposure to ultrafine particles for the participants in the study can be attributed to indoor sources, 36% to outdoor sources, and 17% to in-vehicle exposure. Consistent with the SHEDS-PM estimates for PM2.5 described below, cooking and cigarette smoking were the dominant sources of indoor emission of UFPM. In households with one or more smokers, the cigarette smoke more than doubled the exposure from all other sources. By measuring the particle count per cubic centimeter (cm^3) and multiplying this by the size of the impacted indoor area, this study estimates that smoking a single cigarette emits approximately 2 trillion (2×10^{12}) ultra-fine particles.

Lynn Hildebrand at Stanford University and William Nazaroff at UC Berkeley have also done important research to advance our understanding of exposure to PM in various micro-environments. A recent study directed by Professor Nazaroff (Bhangar at al. 2011) monitored ultrafine particle concentrations and exposures in seven residences (with non-smoking inhabitants) in urban and suburban Alameda County. This study provides several findings of interest:

- Ultrafine particle concentrations in the home environment are heavily impacted by episodic **indoor source events** that cause sharp spikes in particle counts. These events are triggered by activities such as cooking on the stove; uses of appliances such as toaster ovens, steam irons, or clothes dryers; burning candles; and use of the furnace.
- Frequency of use of the cooking range (either gas or electric) is the single most important determinant of exposure from episodic indoor sources.
- Gas stoves with pilot lights are a key source of indoor emissions and exposures to ultrafine particles.
- Indoor particle counts are much higher when occupants are at home and active (thus generating particles via indoor source events), compared to when they are away from home, or at home but asleep.

- Emissions from indoor sources of ultrafine particles accounted for roughly 60% of the indoor particles; the remaining 40% represent particles that infiltrated from outdoor air.
- Active particle removal systems can reduce indoor particle levels (of both particles generated indoors, as well as particles that infiltrate from outdoors) by a factor of 2 to 4.

The papers cited above analyzed personal exposure to PM at the individual level. Efforts have also been made to estimate the major sources of aggregate population exposure to PM in various urban areas. Many of these studies have employed the *Stochastic Human Exposure & Dosage Simulation for PM* (SHEDS-PM) model developed by the US EPA National Exposure Research Laboratory. Synthesizing data from many sources, including personal activity logs, ambient PM2.5 concentrations for outdoor air, and results from studies of indoor PM, the SHEDS-PM model has been used to estimate the contribution of outdoor exposure and indoor exposure to total population exposure, and to examine the role of key indoor sources of PM2.5 such as cigarette smoking and cooking.

An analysis (Burke et al. 2001) using SHEDS-PM for Philadelphia found that, on average, ambient (outdoor) PM2.5 accounted for only 37.5% of total exposure; however, this percentage varied greatly within the population. The study found relatively low variation in personal exposure to ambient (outdoor) PM2.5; however, exposure to PM in the indoor environment varied greatly, with high levels of indoor exposure caused primarily by emissions from cigarette smoking and/or cooking. Another study (Cao & Frey, 2011) had similar findings, using SHEDS-PM to analyze and compare PM exposures in three different areas and climate zones (New York City; Harris County, Texas; and six counties along the I-40 corridor in North Carolina). This study found that ambient exposure accounted for approximately 40% of the estimated total daily average PM2.5 exposure in each of the three areas. As in the case of the Burke study of Philadelphia, the Cao study also found that some individuals have extremely high PM exposures, primarily due to indoor emissions from cigarette smoking and/or cooking.

The *Relationship of Indoor, Outdoor and Personal Air* (RIOPA) study (Polidari et al. 2006) investigated residential indoor, outdoor and personal exposures to PM2.5 in three cities with different climates: Houston, TX; Los Angeles, CA; and Elizabeth, NJ. The study found that the median contribution of ambient (outdoor) sources to indoor PM2.5 concentrations was 56% for all study homes (63%, 52% and 33% for California, New Jersey and Texas study homes, respectively).

Exposure to PM in Schools

Another recent study directed by Professor Nazaroff (Mullen et al. 2011) measured PM concentrations in six elementary school classrooms in Alameda County; measurements were performed for a total of 18 days (from 2-4 days in each classroom). None of the schools was in close proximity to a major freeway; distance from the nearest freeway was 0.5 km or greater in all cases. Two of the classrooms were equipped with mechanical ventilation systems; the other four used natural ventilation (windows and doors that open). The study found that (1) indoor particle counts were typically about half of the outdoor concentrations, and (2) roughly 90% of the ultrafine particles measured in the classrooms originated outdoors. The authors compared exposure in the classrooms to exposure in the homes (per Bhangar 2011), noting that the results suggest that elementary school students are subject to much greater overall exposure to ultrafine particles in the home environment, because in-home particle counts are higher and because the students spend more time at home than at school. The authors attribute the difference in concentrations to the fact that fewer ultrafine particles are emitted in classrooms than in homes. In particular, indoor source events, such as cooking, that lead to sharp spikes in UF particle levels, are common in the home, but much less prevalent in the school setting.

Summary of Indoor Population Exposure to PM

Key findings regarding indoor exposure to PM can be summarized as follows:

- Ambient contribution to indoor PM exposure depends on outdoor concentrations in combination with the infiltration rate.
- When indoor sources are present, indoor PM concentrations can be substantially higher than outdoor PM concentrations.
- Indoor PM emissions are generated primarily by specific activities and sources: cooking, cleaning, ironing clothes, burning candles, use of forced-air furnaces, fireplaces, etc.
- PM levels in the home are characterized by sharp spikes triggered by the types of activities mentioned above.
- Ventilation to control PM spikes can greatly reduce indoor concentrations and population exposure.
- PM concentrations in the home are generally much lower at night (when people are sleeping, and PM-generating activities are not occurring) than when people are at home and active.

Occupational Exposure

Exposure to PM and other pollutants on the job is regulated by the Occupational Safety & Health Administration (OSHA). Occupational exposure to PM may differ from ambient exposure in terms of particle type and composition, as well as the intensity, frequency, and duration of exposure. Certain job types may expose workers to significant occupational exposures. For example, truck drivers and other people who drive a lot on the job may be exposed to higher levels of PM from both diesel and gasoline vehicles. Restaurant workers may be exposed to PM from cooking and wood smoke from charbroilers. Construction workers and quarry workers may be exposed to diesel PM, as well as to geologic dust particles from mechanical processes. Firefighters, especially those who combat wildfires, may be subject to extremely high acute exposures to PM. Janitorial workers may be exposed to high levels of PM in the indoor environment when they use cleaning products that contain chemicals which react with ambient ozone to form PM. Researchers (Morawska et al. 2007) have founds that people who work in office buildings may be exposed to PM (as well as VOCs) from printers.

Brigham and Woman's Health Hospital conducted a study (Laden et al. 2007) of mortality patterns associated with job-specific exposure to fine particulate and especially particulate matter from vehicle exhaust. They examined rates of cause-specific mortality and compared this to the general population. This study concluded that in the U.S. trucking industry there was an excess of mortality due to lung cancer and heart disease particularly among drivers.

Summary

Population exposure to PM is heavily dependent on individual activity patterns and the types of PM emissions sources that people are exposed to in the course of their day-to-day activities. PM levels, and population exposure to PM, may be greatly elevated in certain micro-environments, such as in-vehicle, near-roadway, and in the home.

The key to avoiding negative health impacts from PM is to reduce population exposure to PM among Bay Area residents. Recognizing the importance of reducing population exposure to air pollutants, the Air District has been working to identify areas that are disproportionately impacted and implementing policies and programs to protect these communities, as described in Section 4.

But to better protect public health, we need to improve our understanding of population exposure to PM in the Bay Area. Future steps to enhance our understanding of population exposure to PM are discussed in Section 5.

Simple steps that Bay Area residents can take to reduce their exposure to PM in the course of their day-to-day activities are also described in Section 5.

SECTION 1-C: PM AND CLIMATE CHANGE

This section describes the complex interplay between particulate matter and climate change, including how PM affects climate, as well as how higher temperatures due to climate change may impact local PM levels.

Although more work is needed to fully discern the connections, research reveals a two-way relationship in which air pollutants impact the climate at both the local and global scale, while changes in climate impact air quality. Most discussion has focused on the need to reduce emissions of carbon dioxide and other greenhouse gases, but researchers have found that particulate matter also affects the climate, especially the type of PM known as black carbon.

How PM Affects Climate Change

The thin atmosphere that surrounds the Earth enables our planet to support life and the complex ecosystems that sustain us. There is irrefutable scientific evidence that the Earth's atmosphere is getting hotter, and that a wide range of human activities, such as combustion of fossil fuels, emit carbon dioxide (CO2) and other **greenhouse gases** (GHG) that are building up in the atmosphere and changing the climate at the global scale. The effects of this man-made global heating are already being experienced in California and on a global basis in terms of temperature trends, extreme weather events (e.g., drought, frequency and intensity of hurricanes and cyclones), sea-level rise, changes in precipitation patterns, the frequency and intensity of wildfires, changes in habitat for flora and fauna, etc.

Certain types of PM, especially black carbon, can have a potent effect in heating the climate.

Efforts to date to protect the climate have focused primarily on reducing man-made emissions of GHGs that trap solar radiation (heat) that would normally escape back into space. Reducing emissions of CO2 has been the main focus of climate protection efforts to date, because on a mass basis emissions of CO2 dwarf the other GHGs, and because CO2 remains in the atmosphere for a very long time.

However, in recent years researchers have discovered that other short-lived air pollutants, including particulate matter and tropospheric ozone, also affect the climate. Although the effects are complex, there is evidence that certain types of particulate matter, especially **black carbon**, can have a potent effect in heating the climate at both the local scale (in the area where PM is emitted) and the global scale. In response to this research, there is a growing recognition that we need to incorporate strategies to reduce emissions of black carbon into climate protections efforts. Reducing black carbon can help to slow the rate of atmospheric heating in the near-term, while also protecting air quality and public health. Emission control opportunities that provide co-benefits in terms of protecting both air quality and the climate are highly desirable from the policy perspective.

Climate Forcing

Climate change is primarily caused by man-made activities that impact the Earth's energy balance (Denman et al. IPCC, 2007). Energy constantly flows to the Earth in the form of sunlight and other forms of solar radiation. Some of this solar energy is reflected back into space, and the rest is absorbed by the planet and stored in the atmosphere, as well as in oceans, forests, etc. Factors external to the natural energy system – so-called **external forcings** - can disturb the Earth's energy balance. These external forcings can be positive or negative. Positive forcers, such as carbon dioxide, methane, and other greenhouse gases, cause more of the sun's energy to be retained by the planet. In contrast, negative forcers, such as volcanic dust that reflects sunlight back into space, cause less of the sun's energy to be retained by the planet. The overall impact of human activities on the climate depends upon the net sum of positive and negative forcings caused by a wide spectrum of man-made activities, including emissions of GHGs and other air pollutants, agriculture and forestry practices, land development and road-paving that affect the reflectivity (albedo) of the Earth's surface.

Climate Forcing Effects of Particulate Matter (PM)

Particulate matter is composed of solid or liquid particles that are suspended in the air; these particles are sometimes referred to as **atmospheric aerosols.** Fine particles affect the climate by means of several direct and indirect processes, some of which heat, and others of which cool, the climate. All PM in the atmosphere can affect the Earth's climate either by absorbing light or by scattering light. Particles that absorb sunlight add energy to the earth's system; they act as positive forcers that lead to climate heating. Particles that scatter light increase the reflection of incoming sunlight back to space; they serve as negative forcers that cool the climate. In addition to the direct effect caused by absorbing or scattering incoming sunlight, fine particles may also have indirect effects on the climate by altering the properties of clouds in various ways. [6] More analysis is needed to fully define the impacts of particles on clouds, but researchers have noted various different processes by which aerosols can affect the reflectivity and lifespan of clouds, in ways that can have both heating and cooling effects, as further describe below. (The 2007 IPCC report discusses five processes; Jacobson 2002 lists 12 processes.)

For purposes of analyzing the impacts of PM on climate, scientists have identified several types of carbon: black carbon, brown carbon, and organic carbon. The effect of primary (directly-emitted) PM on sunlight spans a continuum from light-absorbing to light-scattering, with **black carbon** at the light-absorbing end of the spectrum, most **organic carbon** at the opposite, light-scattering end of the spectrum, and **brown carbon** (a subset of organic carbon) somewhere in the middle. The ratio of black carbon, brown carbon and organic carbon produced by fuel combustion depends upon the specific fuel being burned and the type of combustion conditions. PM emitted by diesel engines is primarily black carbon, whereas the PM emitted by gasoline engines is mostly organic carbon.

Table 1-4 lists the most significant types of anthropogenic (man-made) aerosol particles in terms of impact on the climate, and their most common sources. At the global scale, the dominant negative

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[6] "Atmospheric Aerosol Properties and Climate Impacts" U.S. Climate Change Science Program Synthesis and Assessment Product 2.3; January 2009.

forcing agent is sulfate,[7] whereas the dominant particle as a positive forcing agent is black carbon. But organic carbon, brown carbon, and ammonium nitrate also affect the climate in various ways that can have both heating and cooling effects. In the Bay Area, ammonium nitrate levels are greater than sulfates.

Table 1-4 Climate-Forcing Properties of PM Components

Negative Forcer (Cooling Agent)	Positive Forcer (Heating Agent)	Direct Effect	Indirect Effect	Source
Sulfates				
X		Reflects sunlight	Increases reflectivity of clouds	Secondary PM formed by SO2 emissions from fossil fuel-burning
Ammonium nitrate				
X		Reflects sunlight	Increase reflectivity of clouds	Secondary PM formed by combination of NOx and ammonia emissions.
Black carbon				
	X	Absorbs sunlight	1) Reduces reflectivity of clouds; impacts cloud formation. 2) Heats snow & ice by reducing their reflectivity in polar regions.	Incomplete combustion of fossil fuels, biofuels, and biomass (wood-burning)
Brown carbon				
	X	Absorbs some wavelengths of sunlight		Incomplete combustion of fossil fuels, biofuels, and biomass (wood-burning)
Organic carbon				
?		Mildly absorbs sunlight		Incomplete combustion of fossil fuels, biofuels, and biomass (wood-burning)

The various particle types are never emitted into the atmosphere in isolation. The emissions produced by a given combustion process or event contain a mixture of black carbon, brown carbon,

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[7] Text from NASA Fact Sheet: “While a large fraction of human-made aerosols come in the form of smoke from burning tropical forests, the major component comes in the form of sulfate aerosols created by the burning of coal and oil. The concentration of human-made sulfate aerosols in the atmosphere has grown rapidly since the start of the industrial revolution. At current production levels, human-made sulfate aerosols are thought to outweigh the naturally produced sulfate aerosols.” http://www.nasa.gov/centers/langley/news/factsheets/Aerosols.html

San Francisco Health Code Article 38

Model Ordinance for Cities: San Francisco is the first jurisdiction in the country to create a law, known as Article 38, to protect future residents from exposure to roadway air pollution. The law will prevent avoidable lung disease and premature death in residents living near busy roadways, as well as prevent avoidable health care spending, for example, on hospital charges for prevented asthma attacks.

San Francisco Health Code Article 38, adopted in 2008, requires residential projects with more than 10 units located in "Potential Roadway Exposure Zones" (as defined according to maps provided by the San Francisco Department of Public Health) to prepare an air quality assessment, using modeling tools, to determine whether residents would be exposed to unhealthy levels of PM2.5. The Department of Public Health has defined "unhealthy" levels of PM2.5 as roadway concentrations greater than 0.2 µg/m^3. If the air quality assessment indicates that the roadway-attributable PM2.5 would be less than 0.2 µg/m^3, then no further action is required. If the air quality assessment for the residential project indicates that concentrations would be unhealthy, then the project is required to mitigate the traffic-related PM2.5 pollutants, using available technology and design features, to reduce or remove at least 80% of the ambient PM2.5 from indoor spaces.

Meeting the performance standard can be accomplished in several ways, including:

1. Designating lower floors for commercial use and upper for residential use;
2. Setback of buildings from roadway air pollution sources;
3. Locating the intake for fresh air ventilation sources at a non-polluted site;
4. Filtration of fresh air ventilation sources; and/or
5. Recirculation and filtration of indoor air.

Economic Impacts: The City/County of San Francisco's Office of the Controller has determined that the economic impacts of Article 38 on the San Francisco economy, the development community, and future residents of the City are neutral to positive. Although there is a cost associated with implementation of the mitigation measures described above, Article 38 will also prevent avoidable health care spending (for example, hospital charges for emergency room visits for asthma attack) and help to prevent premature mortality associated with exposure to PM. If using a filtration system, the City estimates that costs to install and maintain the system will range from approximately $50-700 per year per unit, while the monetary benefit of the reduction of premature death is estimated to be approximately $2,100 per unit per year. On the basis of this analysis, if installation of a filtration system is required in order to comply with the requirements of Article 38, then the Controller has determined that the net economic benefit of Article 38 would be approximately $1,400 per unit per year.

The Air District has also been providing technical assistance to help the City of San Jose develop a CRRP. The Air District is preparing city-wide emissions inventory for on-road mobile sources on freeways and surface streets, permitted stationary sources, and railroads, airports, and construction projects. Initial air dispersion modeling is underway. The City has also engaged in public outreach, in partnership with the Air District. As a first step on the policy side, the City included several policies in its 2011 General Plan update to analyze and mitigate population exposure from major emissions sources. For example, the air quality section of the General Plan includes policies which (1) require completion of air quality modeling for sensitive land uses such as new residential developments located near emission sources such as freeways and industrial uses; (2) require new residential development projects and projects characterized as sensitive receptors to incorporate effective mitigation into project designs or to be located an adequate distance from sources of toxic air contaminants to avoid significant health risks; and (3) require projects that would emit toxic air contaminants to prepare health risk assessments as part of environmental review and employ effective mitigation to reduce possible health risks to a less-than-significant level. In addition, the General Plan policies mentioned above encourage the use of air filtration devices in existing schools, houses and other sensitive land uses; re-designation of truck routes; and the use of vegetative buffers between emission sources and sensitive receptors.

Promoting Healthy Focused Development

Continued growth in motor vehicle travel could erode the air quality benefits from the ARB and Air District programs described above. We need to better integrate land use, transportation, and air quality planning in order to constrain future increases in vehicle travel and emissions. Therefore, the Air District supports the effort to focus future development in the Bay Area in areas where public transit, biking and walking are viable transportation options. At the same time, however, many of the areas identified as good sites for focused growth already experience high concentrations of air pollutants due to emissions from existing local sources. In fact, a comparison of areas that have been designated as *Priority Development Areas* (PDAs) to date and the impacted communities identified by the Air District's CARE program shows that there is considerable overlap. This emphasizes that we need to plan for focused growth in a way that protects people from exposure to air pollutants, especially local pollutants such as PM and air toxics. To address this issue, the Air District is committing its resources to help planning agencies (cities, counties, MTC, and ABAG) identify, evaluate and mitigate these impacts through the planning and design processes.

The Air District is working actively with partners at both regional and local agencies to support focused development to reduce motor vehicle emissions, while ensuring that development is planned and designed so as to minimize public exposure to air pollutants and protect public health.

At the regional scale, the Air District is engaged with its regional agency partners in the effort to develop *Plan Bay Area*. Plan Bay Area, scheduled for adoption in 2013, will update the Regional Transportation Plan (RTP) and incorporate a **Sustainable Communities Strategy** to better integrate land use and transportation planning, in response to the requirements of Senate Bill (SB) 375. Although SB 375 requirements focus on the need to reduce emissions of greenhouse gases, the Air District worked with its regional agency partners to make sure that the performance targets for Plan

EXECUTIVE SUMMARY

Executive Summary

The Multiple Air Toxics Exposure Study III (MATES III) is a monitoring and evaluation study conducted in the South Coast Air Basin (Basin). The study is a follow on to previous air toxics studies in the Basin and is part of the South Coast Air Quality Management District (SCAQMD) Governing Board Environmental Justice Initiative.

The MATES III Study consists of several elements. These include a monitoring program, an updated emissions inventory of toxic air contaminants, and a modeling effort to characterize risk across the Basin. The study focuses on the carcinogenic risk from exposure to air toxics. It does not estimate mortality or other health effects from particulate exposures. The latter analysis was conducted as part of the 2007 Air Quality Management Plan and is not included here.

A network of ten fixed sites was used to monitor toxic air contaminants once every three days for two years. The location of the sites was the same as in the previous MATES II Study to provide comparisons over time. The one exception is the West Long Beach site, which was about 2.5 miles east of the Wilmington location used in MATES II. The locations of the sites are shown in Figure ES-1.

The initial scope of the monitoring was for a one-year period from April 2004 through March 2005. Due to the heavy rains in the Basin in the fall and winter of this period, there was concern that the measurements may not be reflective of typical meteorology. The study was thus extended for a second year from April 2005 through March 2006.

In addition to the fixed sites, five additional locations were monitored for periods of several months using moveable monitoring platforms. These microscale sites were chosen to determine if there were gradients between communities that would not be picked up by the fixed locations.

The study also included an update of the toxics emissions inventories for the Basin and computer modeling to estimate toxics levels throughout the Basin. This allows estimates of air toxics risks in all areas of the Basin, as it is not feasible to conduct monitoring in all areas.

To provide technical guidance in the design of the study, a Technical Advisory Group was formed. The panel of experts from academia, environmental groups, industry, and public agencies provided valuable insights on the study design. Components of the study recommended by the Advisory Group included monitoring for longer periods at the microscale sites, including naphthalene in the monitoring program, and including more up-to-date methods to estimate the contribution of diesel exhaust to ambient particulate levels. In the monitoring program, over 30 air pollutants were measured. These are listed in Table ES-1. These included both gaseous and particulate air toxics.

The monitored and modeled concentrations of air toxics were then used to estimate the carcinogenic risks from ambient levels. Annual average concentrations were used to estimate a lifetime risk from exposure to these levels, consistent with guidelines established by the Office of Environmental Health Hazard Assessment (OEHHA) of the California Environmental Protection Agency (EPA).

Table ES-1 Substances Measured in MATES III

Benzene	1,3-Butadiene	Carbon Tetrachloride
Chloroform	Dichlorobenzene	Methylene Chloride
MTBE	Perchloroethylene (Tetrachloroethylene)	Dichloroethane
Dibromoethane	Ethyl Benzene	Toluene
Trichloroethylene	Xylene	Styrene
Vinyl Chloride	Acetaldehyde	Formaldehyde
Acetone	Methyl ethyl ketone	
Arsenic	Cadmium	Hexavalent Chromium
Copper	Lead	Manganese
Nickel	Selenium	Zinc
Elemental Carbon	Organic Carbon	Naphthalene
PAHs	PM_{10}	$PM_{2.5}$

To assess the potential carcinogenic risk, at least one full year of data is preferred to represent exposure potential. Thus, the fixed site data was used to calculate risk estimates and the microscale sites used solely to determine any gradients compared to the nearest fixed monitoring site. To estimate the risks from the fixed sites, the concentrations measured over each of the two years were averaged to estimate exposure. The Huntington Park and Pico Rivera sites did not have a full year of data for the second year of the study; thus, only the first year of data was used for these two sites.

In the MATES II Study, elemental carbon (EC) was used as a surrogate for diesel particulate levels, as staff determined that this was the best method available during the MATES II Study. For the present study, staff used the Chemical Mass Balance (CMB) source apportionment technique to estimate the contribution from diesel, as well as from other major source categories, to the measured particulate levels.

Key results of the study are presented below.

Fixed Site Monitoring

The carcinogenic risk from air toxics in the Basin, based on the average concentrations at the fixed monitoring sites, is about 1,200 per million. This risk refers to the expected number of additional cancers in a population of one million individuals that are exposed over a 70-year lifetime. Using the MATES III methodology, about 94% of the risk is attributed to emissions associated with mobile sources, and about 6% of the risk is attributed to toxics emitted from stationary sources, which include industries, and businesses such as dry cleaners and chrome plating operations. The average risks from the annual average levels of air toxics calculated from the fixed monitoring sites data are shown in Figure ES-2.

The air toxics risk at the fixed sites ranged from 870 to 1,400 per million. The risk by site averaged over the two study years is depicted in Figure ES-3. For the second year of the study, a full year of data was not collected at two of the sites (the Huntington Park site access was not extended for the second year; and the Pico Rivera site was moved during the second year resulting in several months without data). The second year data include results for only eight sites. Sites with higher levels of risk include Burbank, Central Los Angeles, Inland Valley San Bernardino, Huntington Park, and West Long Beach. The site with the lowest risk is Anaheim.

The results indicate that diesel exhaust is the major contributor to air toxics risk, accounting on average for about 84% of the total.

To compare different methods used to estimate diesel particulate levels, the method used in MATES II, which was based on the emissions ratios of diesel particulate and elemental carbon from a study conducted in the South Coast in the 1980's, and a method based on the ratio of $PM_{2.5}$ emissions from the 2005 emissions inventory were both calculated. For MATES II, the PM_{10} elemental carbon levels were multiplied by 1.04 to estimate diesel particulate. The 2005 $PM_{2.5}$ inventory finds a ratio of diesel particulate to elemental carbon emissions of 1.95. Multiplying the $PM_{2.5}$ elemental carbon levels by the 1.95 ratio gives another estimate of diesel particulate. The estimates using these methods compared to using the CMB model are shown in Table ES-2. Should one use the same diesel particulate estimation methodology as MATES II, there is about a 30% reduction in ambient levels between the two studies. Based on comparisons of the three methods to estimate diesel particulate, the method used for MATES II gives the lowest estimates of ambient diesel particulate.

For the CMB model, the estimates were sensitive to the species profile used for gasoline vehicles. Table ES-2 shows the range of values using two different gasoline profiles. The estimates used for the risk calculations were the midpoint of the range. As shown in the table, both the CMB model and the $PM_{2.5}$ emissions ratio from the 2005 emissions inventory method give similar estimates, and both are higher than the MATES II method. Thus the MATES II Study method is likely underestimating the levels of diesel particulate.

Table ES-2 CMB Estimate of Diesel Particulate Compared to Emissions Inventory Ratio Methods.

Estimation Method	**MATES III Diesel PM μg/m3**
MATES II Method: PM_{10} EC x 1.04	2.16
2005 Inventory Method: $PM_{2.5}$ EC x 1.95	3.5
CMB Method	3.20 – 3.49

Note: Year 2 includes data for eight sites only. The MATES II diesel particulate was estimated at 3.4 $\mu g/m^3$.

Modeling

Several updates to the modeling platform were included in this study compared to MATES II. The model used was the Comprehensive Air Quality Model with Extensions (CAMx). This model is consistent with that used in the 2007 Air Quality Management Plan. A grid size of 2 kilometers was used.

In addition to using an updated air toxics emissions inventory, an improved geographical allocation of diesel emissions was employed.

The modeling results are shown in Figure ES-4. The grid cell with the highest air toxics risk was at the ports. The grid cells near the ports ranged from about 1,100 to 3,700 in a million. In addition to the ports, an area of elevated risk is shown near the Central Los Angeles area with grid cells ranging from about 1,400 to 1,900 per million. There are also higher levels of risk that track transportation corridors and freeways.

Since the modeling platform and emissions inventory methods are different in MATES III than those used in MATES II, the CAMx model was applied to the MATES II time frame for a more ”apples to apples” assessment. The MATES III methodology was also used to back-cast the estimates of air toxics emissions for the MATES II timeframe. Comparing the results, a lesser level of carcinogenic risk was estimated across the Basin for MATES III compared to the MATES II time period. The model also shows the dominant contribution from mobile sources and diesel emissions to air toxics risk in the MATES II timeframe as well.

For comparison purposes, Table ES-3 shows the estimated population weighted risk across the Basin for the MATES III and MATES II periods. The population weighted risk was about 8% lower compared to the MATES II period.

The MATES III modeling analysis represents several improvements over that used in MATES II and represents the state-of-science application of regional modeling tools and chemistry applied to an updated set of meteorological and emissions data input.

Table ES-3 Modeled Air Toxics Risk Comparisons Using the CAMx Model

	MATES III	MATES II	Change
Population weighted risk (per million)	853	931	-8%

Figure ES-5 depicts the 1998-99 to 2005 change in air toxics risk for each model grid cell estimated from the CAMx simulations. Overall, air toxics risk improves to varying levels in most of the Basin with the exceptions of the areas directly downwind of the ports and those areas heavily impacted by activities associated with goods movement. The model comparison shows an increase in air toxics risk occurred in the immediate areas encompassing the ports of more than 800 in a million between the two periods. This increase correlates with the increased container cargo moving through the ports and increases in goods movement that occurred between the MATES II and MATES III time periods.

Noncancer Assessment

To assess the potential for noncancer health risks, the monitored average levels were compared to the Chronic Reference Exposure Levels (RELs) established by OEHHA. The chronic REL is the air concentration at or below which adverse noncancer health effects would not be expected in the general population with exposure for at least a significant fraction of a lifetime. In general, the measured concentrations of air toxics were below the RELs.

The exception is formaldehyde. The chronic REL is 3 μg/m3 (2ppb). All of the fixed site annual averages were above this concentration, ranging from 2.9 ppb for Anaheim to 4.5 ppb at Los Angeles. Formaldehyde effects include eye irritation, injury to nasal tissue, and respiratory discomfort. OEHHA, however, is proposing revisions to the RELs for several toxic air contaminants. For formaldehyde, the proposed chronic REL is 9 μg/m3 (7 ppb). If the proposed level is promulgated, then all sites would be under the chronic REL.

Caveats and Uncertainty

One source of uncertainty is that currently there is no technique to directly measure diesel particulates, the major contributor to risk in this study, so indirect estimates based on components of diesel exhaust must be used. The method chosen to estimate diesel particulate is the CMB source apportionment model. This method is a weighted multiple linear regression model based on mass balance of each chemical species applied to apportion contributions to ambient particulates using measured source profiles. The CMB method accounts for major source categories and geographic differences in source contributions and was recommended by the Technical Advisory Group. It is staff’s judgment that this is the most appropriate method to estimate the ambient levels of diesel particulate matter.

The MATES II Study used elemental carbon as a surrogate for diesel particulate. Elemental carbon, however, is not a unique tracer for diesel, as there are additional emission sources of elemental carbon. Using the CMB model takes advantage of the specific profile of chemical species emitted from different particulate matter sources. Twenty-three species were used in the CMB model to reconcile source contributions to observed ambient concentrations. This results in a more robust apportionment of source contributions to ambient particulate matter levels, since all major sources of particulate matter and elemental carbon are considered.

The CMB model uses the profile of chemical tracer chemical species from different source categories to estimate the contribution to ambient particulates. Some tracers are unique to a given source, such as levoglucosan from biomass burning, whereas other sources show specific chemical profiles that can be used to apportion these sources, such as gasoline and diesel combustion. The advantage of the CMB model is that it can apportion several sources to ambient levels. Additional discussion is provided in Chapter 2 and Appendix VII on the CMB methodology.

The Positive Matrix Factorization (PMF) model was also evaluated for estimation of diesel particulate. The PMF model is an alternating least squares method that estimates source profiles and source contributions from the ambient data. Since possible solutions to this model can be negative, the procedure uses restrictive functions so that no sample can have a negative source contribution and no species can have a negative fraction in any source profile. Estimated source profiles are then attributed to specific sources using experienced judgment. However, using the MATES III data, the initial attempts at source apportionment found that some source profiles could not be interpreted, and some profiles could not be confirmed with confidence.

Additionally, the statistical parameters of the PMF model performance were outside of the bounds used to determine adequate performance of the model. Also, in perusing the literature of applications of PMF approach, it was found that substantial amounts of measured data were sometimes excluded from the analyses, and an uncertainty parameter for some variables was altered to improve the model performance. Staff did not censor any data or alter certain parameters in the model in an attempt to improve the model performance statistics. The uncertainties used for the ambient measurements were those provided by the laboratory analyses. Thus, the PMF method was not pursued.

When compared to the MATES II method, the CMB model available from the U.S. EPA gives higher estimates of diesel particulates. The CMB model estimate for diesel particulate was found to be sensitive to the gasoline emissions profile used. To account for this, the midpoint of a range of estimates using two different gasoline profiles was used.

There are also uncertainties in the risk potency values used to estimate lifetime risk of cancer. This study used the unit risks for cancer potency established by OEHHA and the annual average concentration measured or modeled to calculate risk. This methodology has long been used to estimate the relative risks from exposure to air toxics in California and is useful as a yardstick to compare potential risks from varied sources and emissions and to assess any changes in risks over time that may be associated with changing air quality.

The estimates of health risks are based on the state of current knowledge, and the process has undergone extensive scientific and public review. However, there is uncertainty associated with the processes of risk assessment. This uncertainty stems from the lack of data in many areas necessitating the use of assumptions. The assumptions are consistent with current scientific knowledge, but are often designed to be conservative and on the side of health protection in order to avoid underestimation of public health risks.

As noted in the OEHHA risk assessment guidelines, sources of uncertainty, which may either overestimate or underestimate risk, include: (1) extrapolation of toxicity data in animals to humans, (2) uncertainty in the estimation of emissions, (3) uncertainty in the air dispersion models, and (4) uncertainty in the exposure estimates. Uncertainty may be defined as what is not known and may be reduced with further scientific studies. In addition to uncertainty, there is a natural range or variability in the human population in such properties as height, weight, and susceptibility to chemical toxicants.

Thus, the risk estimates should not be interpreted as actual rates of disease in the exposed population, but rather as estimates of potential risk, based on current knowledge and a number of assumptions. However, a consistent approach to risk assessment is useful to compare different sources and different substances to prioritize public health concerns.

Conclusion

Compared to previous studies of air toxics in the Basin, this study found a decreasing risk for air toxics exposure, with the estimated Basin-wide population-weighted risk down by 8% from the analysis done for the MATES II time period. The ambient air toxics data from the ten fixed monitoring locations also demonstrated a reduction in air toxic levels and risks.

Policy Implications

While there has been improvement in air quality regarding air toxics, the risks are still unacceptable and are higher near sources of emissions such as ports and transportation corridors. Diesel particulate continues to dominate the risk from air toxics, and the portion of air toxic risk attributable to diesel exhaust is increased compared to the MATES II Study.

The highest air toxics risks are found near the port area, an area near Central Los Angeles, and near transportation corridors. The results from this study underscore that a continued focus on reduction of toxic emissions, particularly from diesel engines, is needed to reduce air toxics exposure.

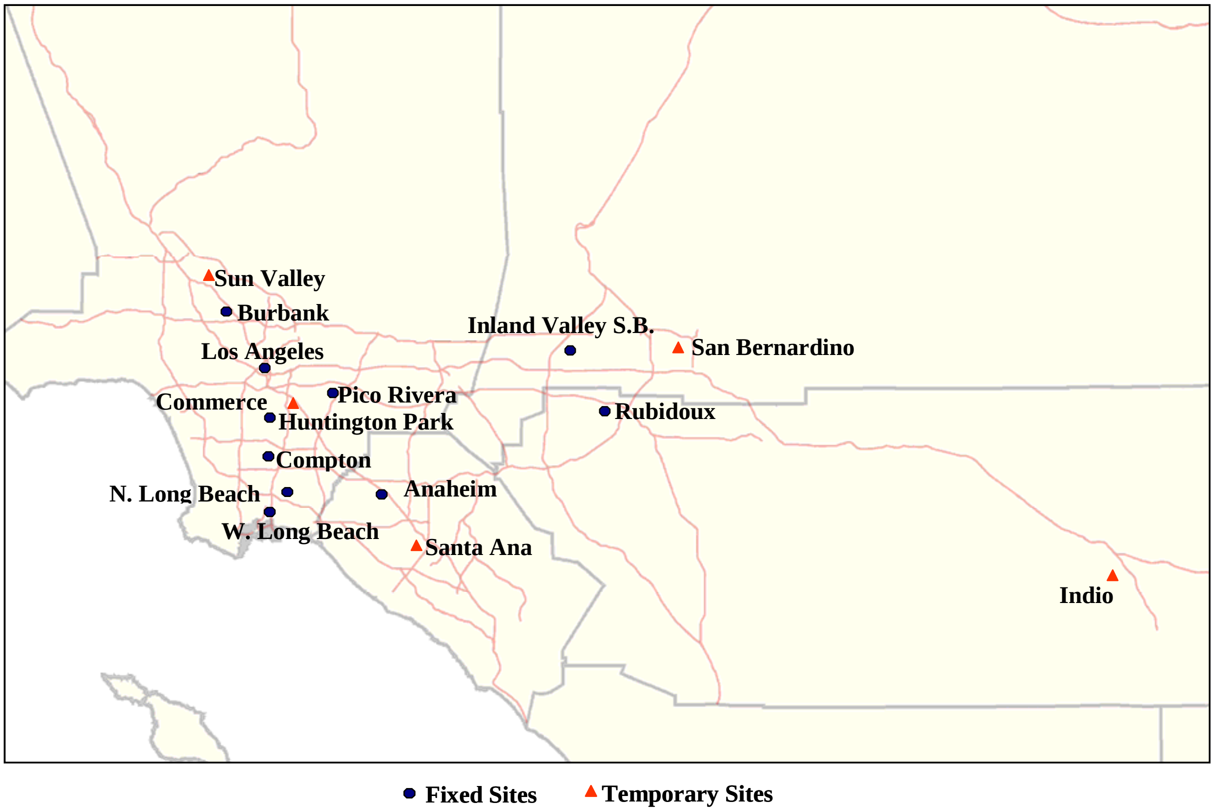


Figure ES-1 Map of MATES III Monitoring Sites

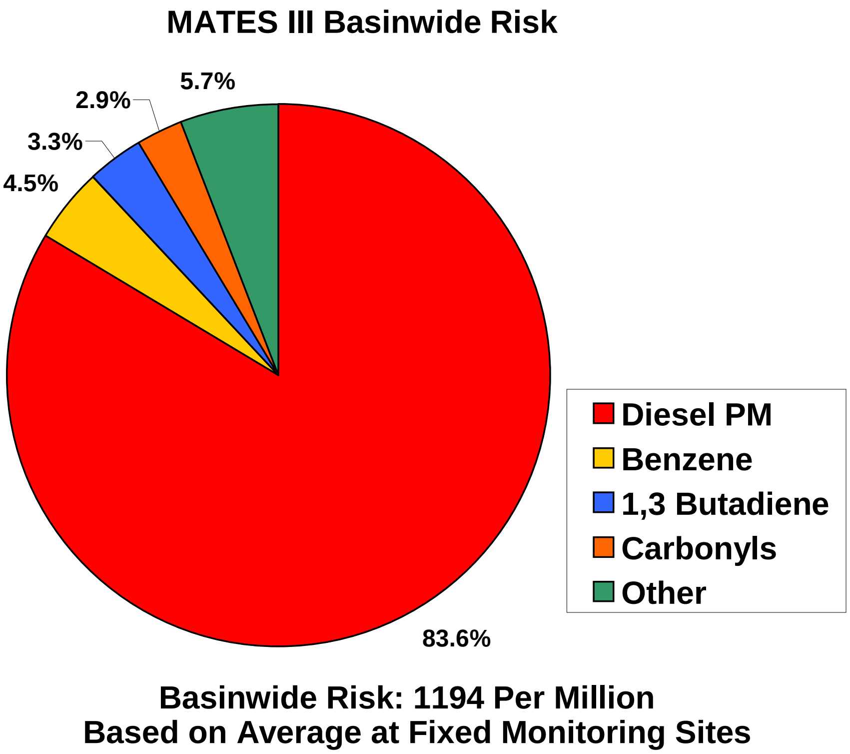


Figure ES-2

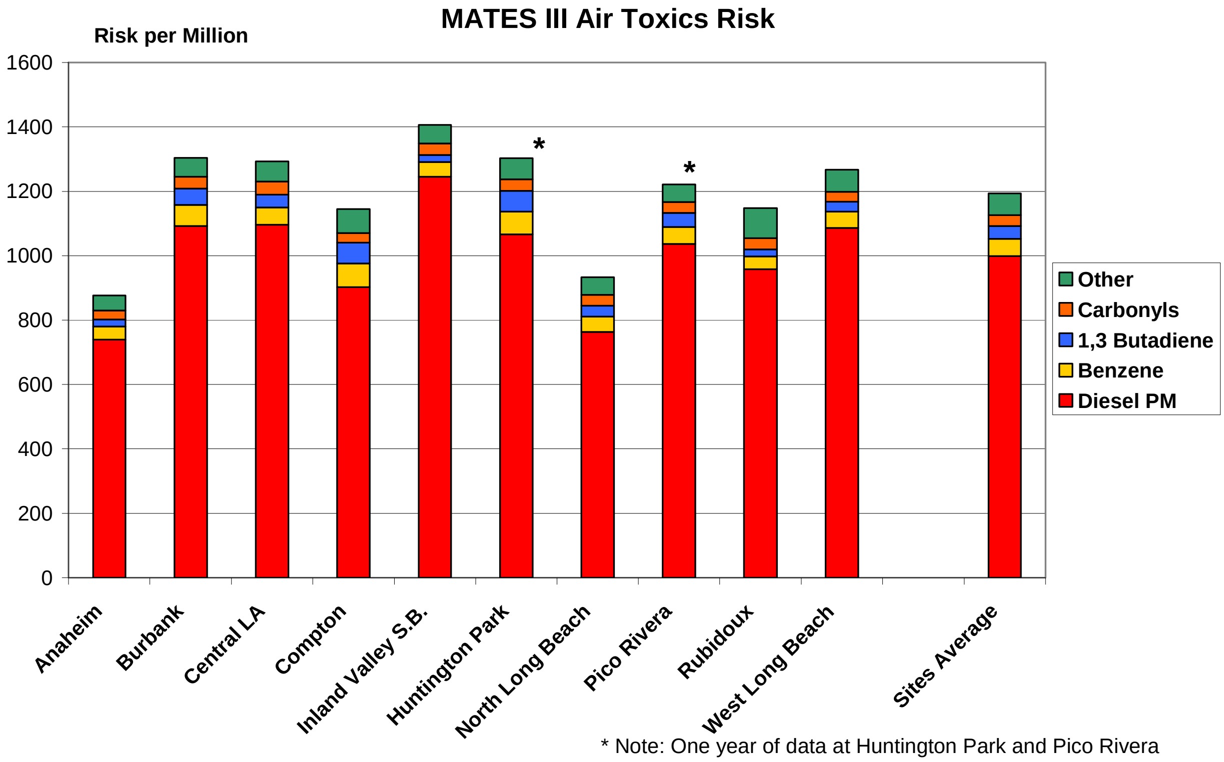


Figure ES-3

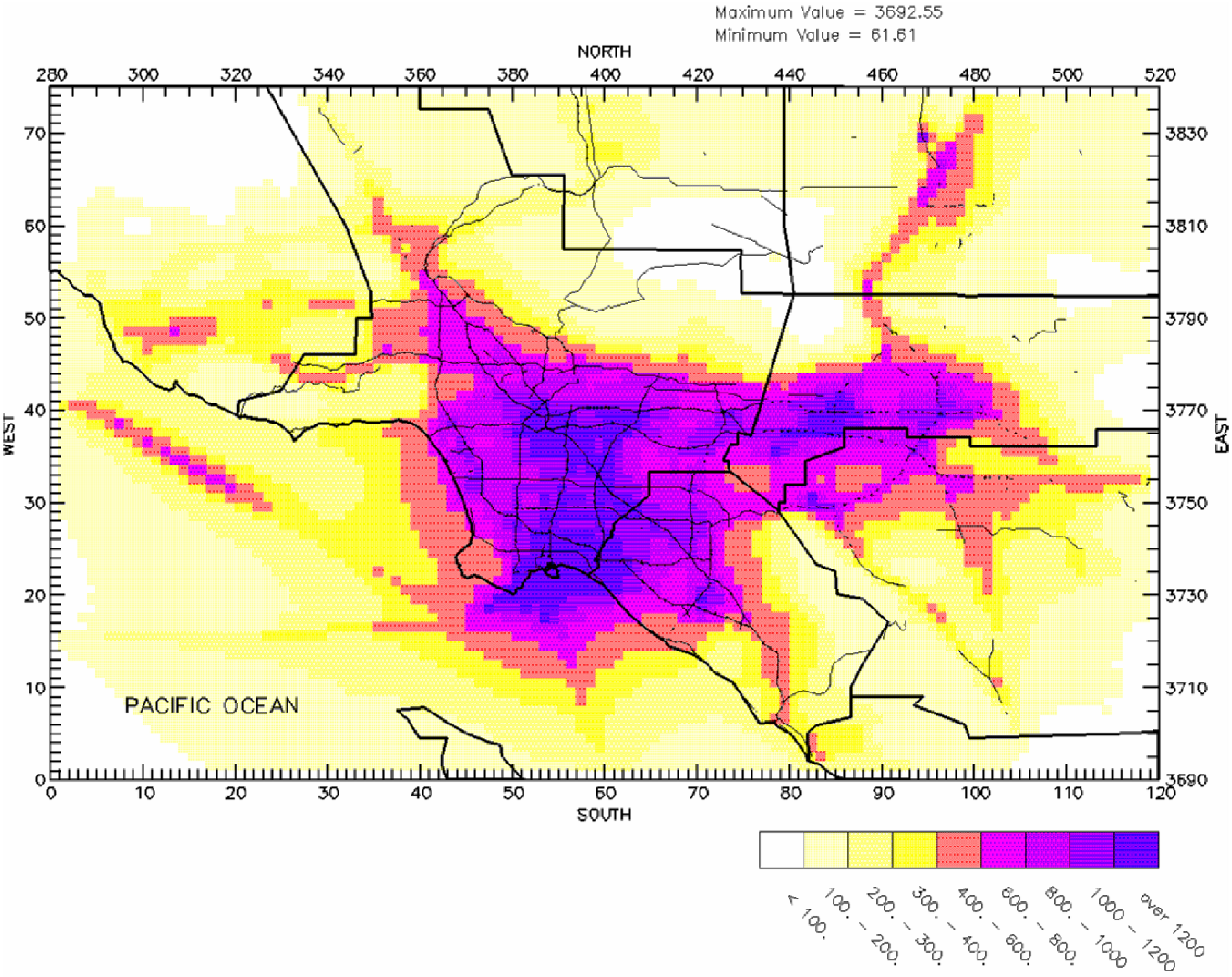


Figure ES-4
MATES III Model Estimated Risk

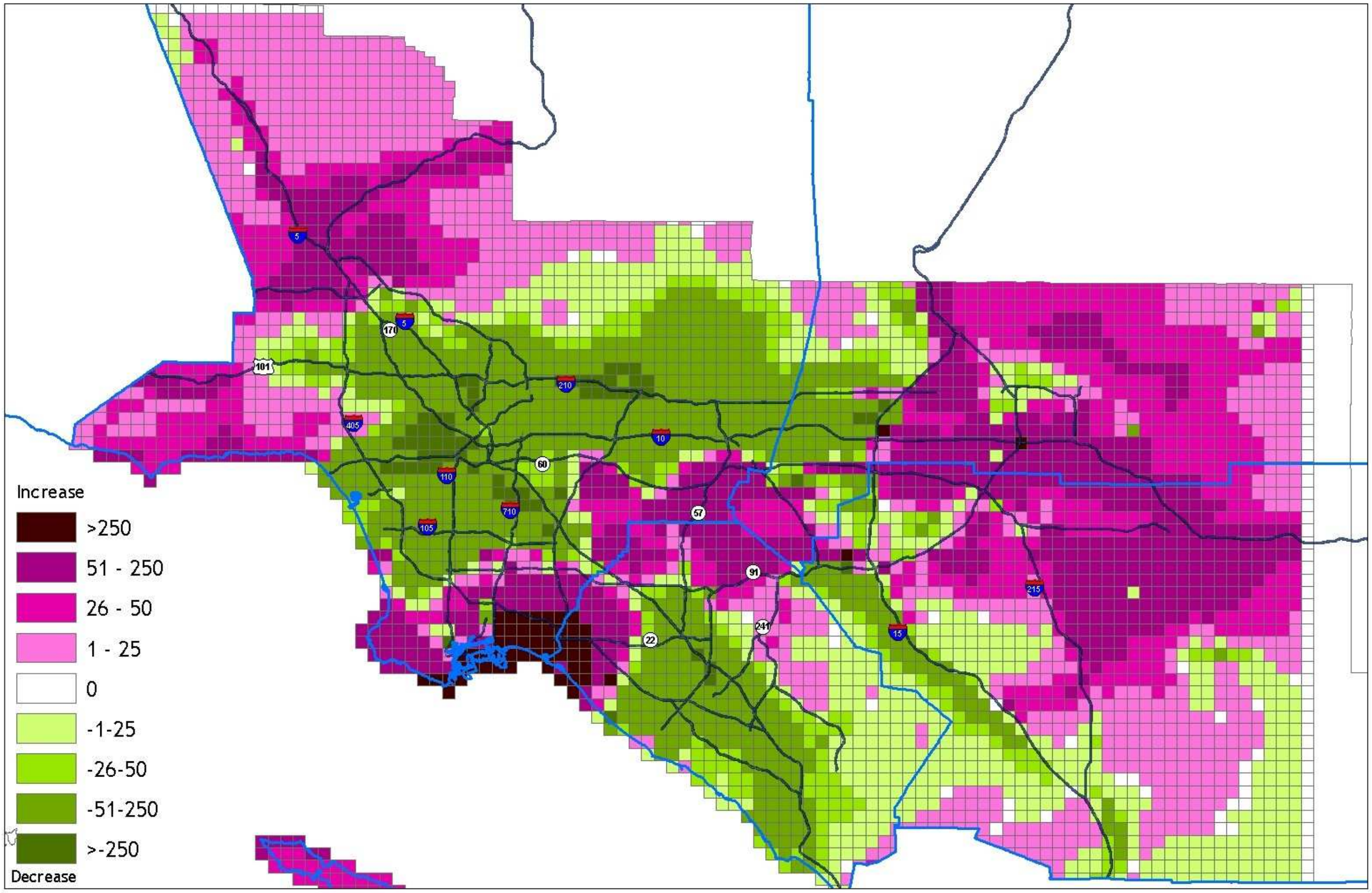


Figure ES-5
Change in CAMx RTRAC Air Toxics Simulated Risk (per million) from 1998-99 to 2005 Using Back-Cast 1998 Emissions and 1998-99 MM5 Generated Meteorological Data Fields

Forest Resource Sustainability
in Placer County California

Placer County

AIR POLLUTION CONTROL DISTRICT

Forest Resource Sustainability in Placer County, California

Millions of acres of forested land in the Sierra Nevada foothills and mountains are at significant risk for wildfire. Placer County alone encompasses approximately 550,000 acres of forested land stretching from Auburn to Lake Tahoe, including parts of two national forests and 60 percent of Lake Tahoe's west shore.

After decades of successful fire suppression, unnaturally dense vegetation presents a significant wildfire hazard. Just since 2001, Placer County has experienced six major wildfires that burned more than 55,000 acres, including important upland watersheds. Also in the last few decades, many homes and business have been developed within these forests in high fire hazard areas. Between 2006 and 2010, state and federal fire agencies spent an average of $1.2 billion annually fighting wildfires in California. Fire restoration costs average in the tens of millions, but more importantly, fire jeopardizes ecological integrity by creating unacceptable impacts to forest resources like water and wildlife habitat, and generate additional tons of harmful emissions into the air.

In response, the Placer County Biomass Partners – including the Placer County Air Pollution Control District, Placer County, and U.S. Forest Service – are implementing cost effective projects that help promote the ecological, economic and social sustainability of forests and forest-dependent communities. These projects focus on improving forest health, reducing the negative effects of catastrophic wildfires by utilizing excess biomass from forest management projects to produce renewable power and create family wage jobs.

The partners specifically are focused on projects that support hazardous fuels reduction and processing and transporting excess biomass material – limbs, tops and brush – for energy production as an alternative to open pile burning. These efforts improve air and watershed quality, protect soil productivity, lower fire suppression costs, and produce renewable energy that reduces regional reliance on fossil fuels.

The partners specifically are focused on projects that support hazardous fuels reduction for mitigating catastrophic wildfire behavior, and processing and transporting excess biomass material – limbs, tops and brush – for energy production as an alternative to open pile burning.

Sustainable Forestry Practices in Support of Wildfire Mitigation

Stewardship Projects on Public Lands

We are implementing innovative agreements and contracts with the U.S. Forest Service and other partners aimed at facilitating cost-effective removal and utilization of forest biomass material for energy. Placer County has also entered into a master stewardship agreement for fuels treatment and biomass material recovery with the U.S. Forest Service's Lake Tahoe Basin Management Unit.

These forest fuels reduction projects to reduce the potential for catastrophic wildfire events include selective thinning and removal of trees and brush to return forest ecosystems to more natural fuel stocking levels, resulting in more fire-resilient and healthy forests.

These forest fuels reduction projects return forest ecosystems to more natural fuel stocking levels, resulting in more fire-resilient and healthy forests.

Left: Biomass collection box. Right: Biomass chipping and transport operations.

Regional Biomass Collection

We are currently implementing a program to cost effectively collect biomass material from forestland management and defensible space clearing. Centralized collection yards have been located and advertised for convenient biomass material drop-off. Biomass material processing and transport for utilization at energy recovery facilities then becomes cost effective.

Left: Roadside biomass pile. Right: Biomass pile loading operation.

Forest Management Benefits

We are sponsoring research efforts to determine the benefits of forest management projects. Models are being developed to quantify the effects of various levels of thinning and hazard reduction treatments on wildfire size and intensity. Initial results indicate three types of carbon benefits that are realized from forest management in the Sierra Nevada:

- Fuels treatments in the study area were shown to reduce the GHG and criteria air pollutant emissions from wildfires by decreasing the probability, extent, and severity of fires and the corresponding loss in forest carbon stocks.
- Utilizing biomass from forest management projects for energy production was shown to reduce GHG and criteria air pollutant emissions over the duration of the fuels treatment project compared to fossil fuel energy.
- Specific management and thinning of forests stimulates growth, resulting in more rapid uptake of atmospheric carbon – including reductions in air pollutant and greenhouse gas emissions, tree mortality, and improved forest growth and vigor.

Far Left: Effect of typical high intensity wildfire.

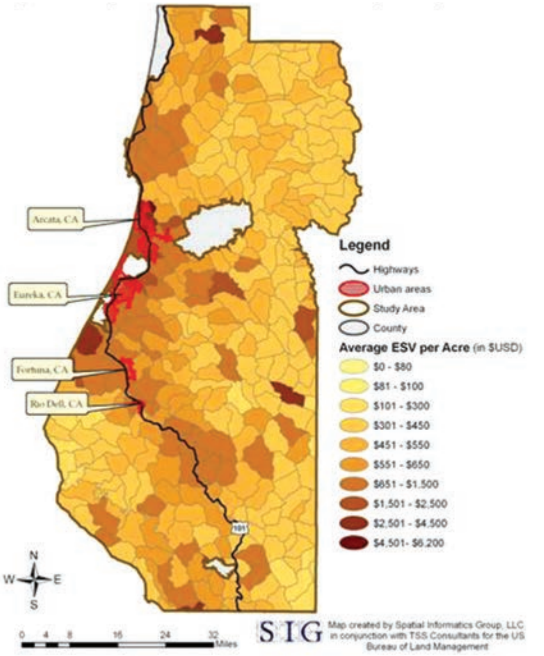
Left: Thinning and hazardous fuel treatments can reduce the effect of wildfires.

Current projects are exploring the valuation of the benefits of sustainable forest management strategies on key related resources such as carbon, water quantity and quality, and wildlife habitat.

Ecosystem Services

Fire, like development, can dramatically alter forested landscapes and significantly reduce or eliminate the flow of ecosystem services. Project partners are exploring the valuation of the benefits of sustainable forest management strategies on key resources such as carbon, water quantity and quality, and wildlife habitat.

Average Estimated Ecosystem Service Value per acre by Tributary Basin for Humboldt County Study Area



Source: Spatial Informatics Group (www.sig-gis.com)

Air Quality

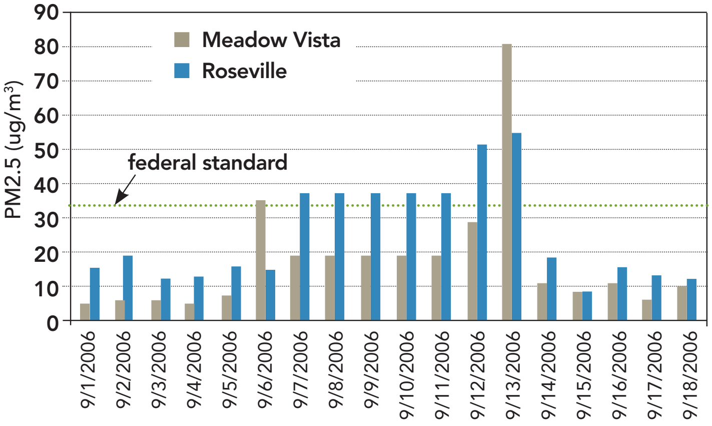
Wildfires and Open Burning of Biomass

Catastrophic wildfires with high burn intensity and landscape-scale footprints in the densely forested Sierra Nevada mountain range have significant adverse effects on air quality. The September 2006 Ralston Fire in the Tahoe National Forest east of Foresthill, Calif., for example, burned a total of 8,423 acres and generated massive amounts of harmful emissions.

Forest management thinning and defensible space clearing generate biomass material in the form of limbs, tops, and brush from a wide variety of land ownerships, including federal, state, large and small private forestlands, conservation organizations and residences.

These forest biomass materials are frequently disposed of through open pile burning, due to economics – the cost to process and transport biomass is usually higher than the value paid by renewable power generation facilities.

Particulate Matter During Ralston Wildfire



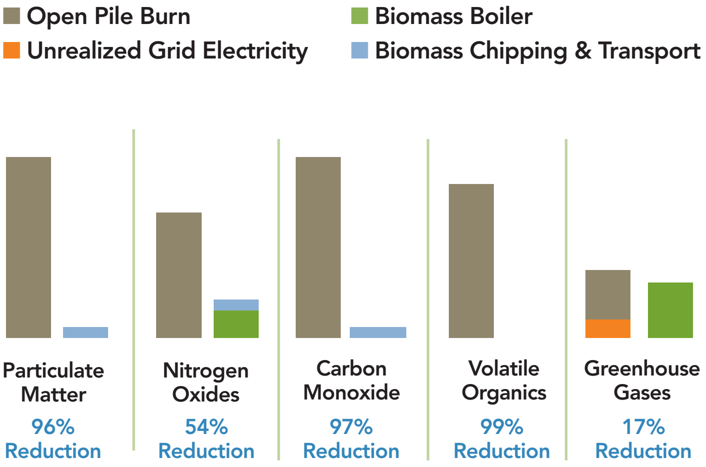
Disposing forest biomass materials through open pile burning.

Utilization of Biomass

We are sponsoring projects which collect, process, transport, and utilize woody biomass material for renewable energy generation in controlled conversion units as an alternative to open pile burning. Placer County has worked with the Forest Service's Lake Tahoe Basin Management Unit and the Tahoe National Forest to implement pilot projects that analyzed processes for collection, processing and transportation of excess woody biomass from forest management and hazardous fuels reduction projects. To date the projects have processed over 15,000 tons of biomass, produced 15,000 MWh of electricity and provided insights into improving the overall economics of forest management and biomass utilization.

Life cycle analysis has shown significant reductions in air pollutants by diverting biomass material away from open burning to renewable energy generation in biomass conversion facilities that are equipped with Best Available Control Technology. There are significant benefits from offsetting fossil fuel use and few emissions from the processing and transportation of the biomass – even if transporting for long distances.

Emission Benefits of Biomass Energy Project

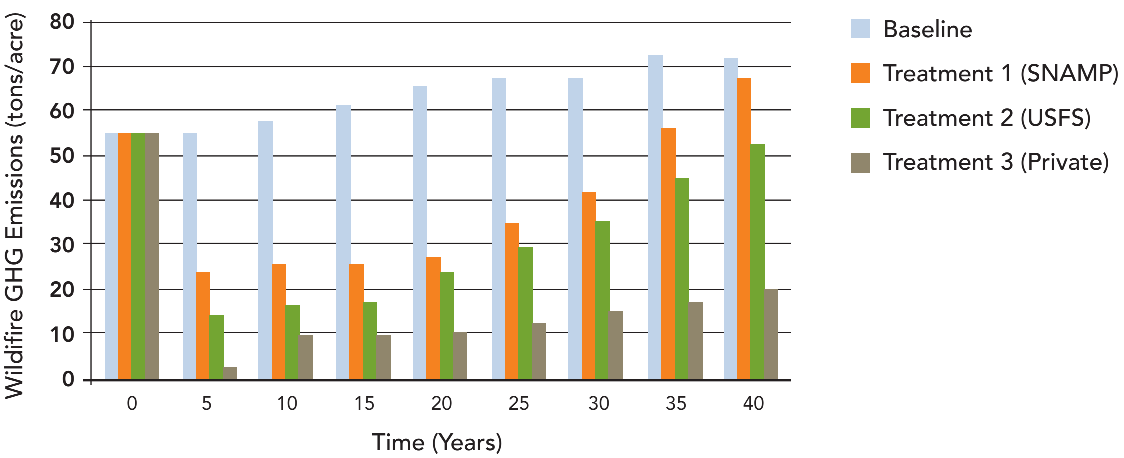


Results based on work by TSS Consultants (www.tssconsultants.com)

Benefits of Hazardous Fuels Reduction Treatments

Hazardous fuels treatment activities can reduce air pollutant emissions through the reduction of wildfire intensity and size. We are supporting ongoing research efforts to develop models to quantify air benefits. Initial results from these research efforts confirm that proactive treatment of hazardous forest fuels mitigates both wildfire behavior and air emissions, including greenhouse gas.

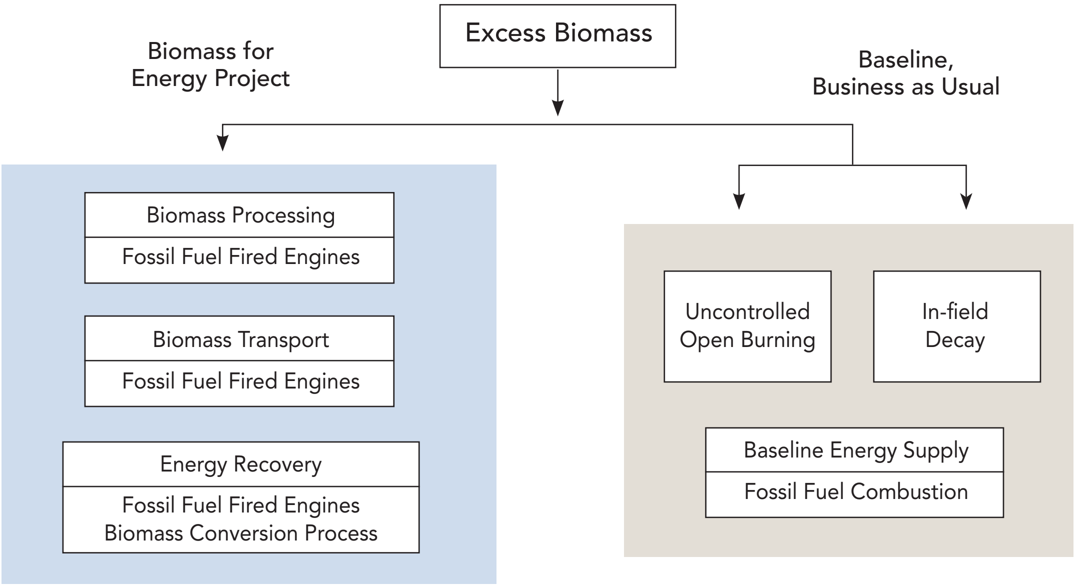
Forest Wildfire Carbon Benefits of Fuel Treatment



The monetary value secured from the sale of GHG offset credits is proposed to be re-invested in forest management projects like reforestation and thinning that provide carbon benefits.

Emissions Accounting

We have developed a greenhouse gas (GHG) offset protocol. This protocol provides a rigorous accounting framework for obtaining monetary value for the emissions benefits (in the form of emission offset credits) from the diversion of biomass material away from pile and burn activities and into renewable energy facilities. The monetary value secured from the sale of emission offset credits will be re-invested in forest management projects like reforestation and thinning that provide GHG benefits. Thinning helps reduce the effects of wildfires that emit large quantities of GHG's and negatively impact key forest resources like water and wildlife habitat.



GHG Reduction = GHG Open Burn + GHG Decay + GHG Baseline Energy − GHG Biomass Energy − GHG Biomass Processing − GHG Biomass Transport

Open pile burn.

Cogeneration facility.

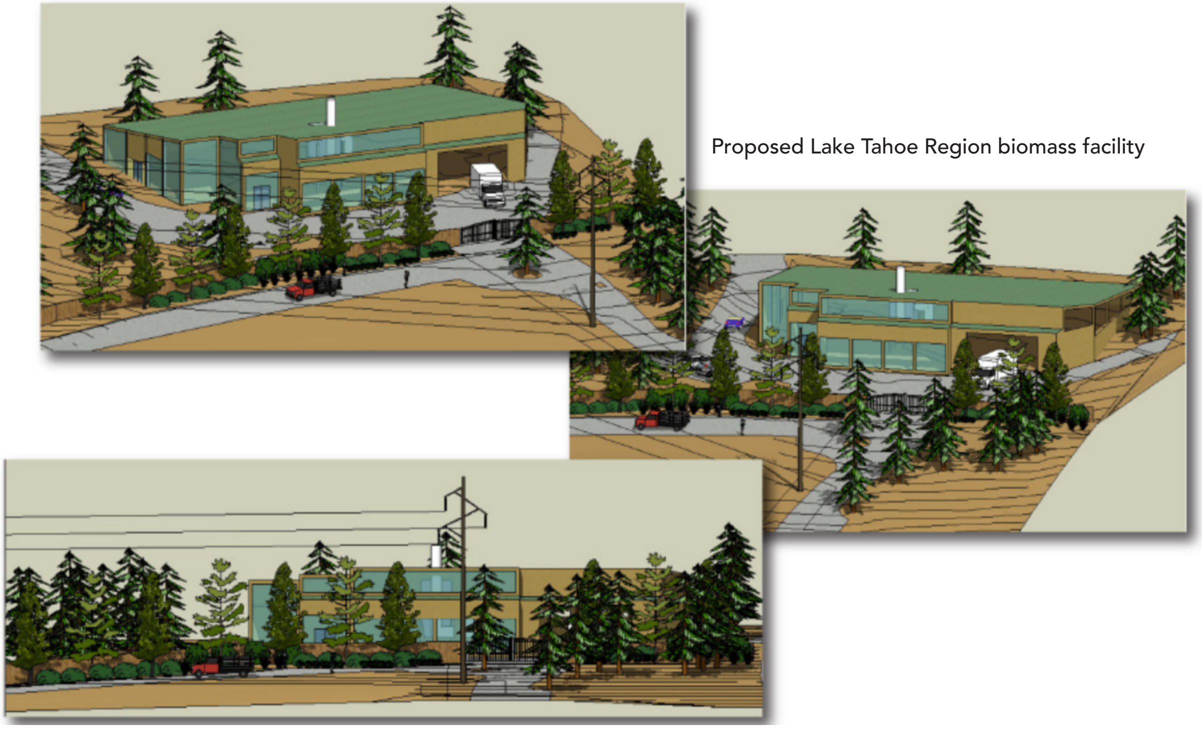
Renewable Bioenergy

Sustainable Distributed Bioenergy in the Lake Tahoe Area

Placer County is proposing a Lake Tahoe Region biomass facility through a public-private partnership. The distributed generation renewable energy facility is scheduled to commence operation in FY 2014. Innovative features of the proposed facility include:

- Conservatively sized to produce 2 MW of electricity through use of excess woody biomass materials produced from nearby forest thinning and community defensible space fuels reduction projects.
- Lower energy facility fuel cost by developing innovative woody biomass supply agreements with local land managers. Funds normally used to dispose of excess biomass through open burning or mastication can be used instead to help fund the processing and transport of forest biomass to an energy production facility.
- Lower fuel cost by efficiently integrating forest management with biomass production.
- Utilize state-of-the art gasification energy conversion technology that provides high electricity production efficiency and low air emissions that meet stringent local and federal limits.
- Fully realize the economic value of renewable energy and air emission benefits in support of community protection, healthy forests and local employment.


Loading chipped biomass for transport


Proposed Lake Tahoe Region biomass facility

We are also developing a guidebook that will provide guidance, and lessons learned, in support of small distributed-generation biomass power facility development. The guidebook will assist communities that are interested in the proactive treatment of hazardous forest fuels and utilization of resulting biomass material for the production of power and/or thermal energy.

Advanced Biofuels for Transportation

Placer County is a participant in a pilot study grant from the California Energy Commission to demonstrate and evaluate the production of biomethane (also known as renewable natural gas) from forest-sourced woody biomass materials for use as an alternative transportation fuel.

Placer County and the Placer County Air Pollution Control District have been strong and visible advocates and participants, at both the state and national level, for development and adoption of policies that favor protection of communities, wise forest management, utilization of excess biomass for energy, and long-term sustainability of forests, forest resources and forest-dependent communities.

Policy Recommendations

We have been strong and visible advocates and participants, at both the state and national level, for development and adoption of policies that favor protection of communities, wise forest management, utilization of excess biomass for renewable energy, and long-term sustainability of forests, forest resources and forest-dependent communities. These policies could include:

- Feed-in tariff renewable energy pricing that provides market-based incentives that facilitate private sector investment in community-scale for bioenergy facilities
- Greenhouse gases from excess biomass materials need to be recognized as at least carbon neutral, and beyond carbon neutral if appropriate
- Streamlining of interconnection, siting requirements for community-scale bioenergy facilities
- Electricity contract pricing that is not based on short run avoided costs or market price referent
- Federal tax credits for biomass energy generation that is commensurate with other forms of renewable energy (wind, solar)
- Continued funding for proactive forest fuels treatment on public lands
- Continued use of long-term stewardship contracts and agreements to facilitate landscape level fuels treatment activities
- Federal tax credit for private forest landowners that treat hazardous forest fuels
- Development and implementation of ecosystem services/payments

Select Project Publications

Bruce Springsteen, Tom Christofk, Steve Eubanks, Tad Mason, Chris Clavin, and Brett Storey, "Emission Reductions from Woody Biomass Waste for Energy as an Alternative to Open Burning", *Journal of the Air and Waste Management Association*, Volume 61, pages 63 - 68, January 2011.

"Forest Biomass Removal on National Forest Lands", Prepared by the Placer County Executive Office and TSS Consultant for the Sierra Nevada Conservancy, November 17, 2008, available at http://www.tssconsultants.com/presentations.php

"Assessment of Small-Scale Biomass Combined Heat and Power Technologies For Deployment in The Lake Tahoe Basin", Prepared by TSS Consultants for the Placer County Executive Office, High Sierra Resource Conservation and Development Council, and U.S. Forest Service, December 2008, available at http://www.placer.ca.gov/Departments/CommunityDevelopment/Planning/Biomass/Grants.aspx

Placer County
AIR POLLUTION CONTROL DISTRICT

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