Title

Our Breaths We Take: Outdoor Air Quality, Health, and Climate Change Consequences of Household Heating and Cooking with Solid Fuels

Author
Chafe, Zoe Anna

Publication Date
2016

Peer reviewed|Thesis/dissertation

Our Breaths We Take:
Outdoor Air Quality, Health, and Climate Change Consequences of Household Heating and Cooking with Solid Fuels

By

Zoë Anna Chafe

A dissertation submitted in partial satisfaction of the

requirements for the degree of

Doctor of Philosophy

in

Energy and Resources

in the

Graduate Division

of the

University of California, Berkeley

Committee in charge:

Professor Kirk R. Smith, Chair
Professor Daniel M. Kammen
Professor John Balmes

Spring 2016

Abstract

Our Breaths We Take: Outdoor Air Quality, Health, and Climate Change Consequences of Household Heating and Cooking with Solid Fuels

by

Zoë Anna Chafe

Doctor of Philosophy in Energy and Resources

University of California, Berkeley

Professor Kirk R. Smith, Chair

Worldwide, nearly 3 billion people—40% of the global population—burn wood, coal, and other solid fuels every day to cook their food; this number is even larger when including those who heat their homes with solid fuels as well. Exposure to pollution from heating and cooking fires causes about 3 million deaths each year, making it one of the biggest environmental health problems the world faces. The harm from this smoke is not restricted to those who breathe it, however: it contains gases and particles that contribute to global climate change as well.

Chapter 2 shows that household cooking with solid fuels caused an estimated 12% of population-weighted ambient $PM_{2.5}$ worldwide in 2010. Exposure to this air pollution caused the loss of 370,000 lives and 9.9 million disability-adjusted life years (DALYs) globally in the same year.

In Chapter 3 I demonstrate that household heating with solid fuels caused an estimated 21% of population-weighted ambient $PM_{2.5}$ in 2010 in Central Europe, 13% in Eastern Europe, 12% in Western Europe, and 8% in North America. Exposure to this air pollution results caused approximately 60,000 premature deaths in Europe, and nearly 10,000 deaths in North America, as well as an estimated 1.0 million disability-adjusted life years (DALYs) in Europe and 160,000 DALYs in North America.

Chapter 4 addresses drivers of household wood combustion pollution in the San Francisco Bay Area, where the sector is the largest source of $PM_{2.5}$ and regulators recently introduced amendments to wood burning rules for the airshed. Fireplaces are the source of the vast majority (84%) of $PM_{2.5}$ from residential wood combustion in the San Francisco Bay Area, despite their use primarily as an aesthetic or recreational combustion activity. By evaluating hypothetical fuel and combustion device changeouts, I find that replacing fireplaces with gas would yield significant health and economic benefits.

Specifically, retrofitting frequently used fireplaces (300,000 units) to gas inserts in the Bay Area's nine counties would reduce sector emissions by about 90%, avoiding approximately 140-310 premature deaths and 19,000 lost days of work each year, and creating upwards of $1 billion in annual financial benefits from improved public health.

Chapter 5 explains methodological overlaps and differences between the previous chapters. In Chapter 6, I explore the current regulatory and policy mechanisms specific to household heating with solid fuels, and relate these to the climate change implications associated with the sector. In Chapter 7, I highlight the relative dearth of data on household heating with biomass and its nuanced climate implications. This leads to a series of recommendations for future research, including collection of better household heating data in China and further work to understand how household combustion of biomass interfaces with both local air quality policy and climate change mitigation, outlining areas where this topic is currently visible in California.

To Mary Elizabeth (Mez) Chafe-Powles

Table of contents

Contents

List of Figures

List of Tables

Preface

Material in Chapter 2 and some material in Chapter 5 was previously published in Environmental Health Perspectives [Chafe, Z. A., et al. (2014), "Household Cooking with Solid Fuels Contributes to Ambient $PM_{2.5}$ Air Pollution and the Burden of Disease." Environmental Health Perspectives 122(12): 1314-1320]. I was the lead author, responsible for the assembly of relevant data sets, analysis, and main manuscript drafting. Kirk Smith, Michael Brauer, and Sumi Mehta provided invaluable advice on the project idea and analysis. The supporting data sets were assembled by IIASA (Zbigniew Klimont, Shilpa Rao, Keywan Riahi) and the European Community Joint Research Center (Rita Van Dingenen and Frank Dentener). Other authors contributed to editing and revision of the manuscript draft.

Some material in Chapter 3 and Chapter 6 was published by the World Health Organization [Chafe, Z., et al. (2015). "Residential heating with wood and coal: health impacts and policy options in Europe and North America." World Health Organization. Bonn, Germany.], by request of the Convention on Long-Range Transboundary Air Pollution (LRTAP) Task Force on Health. I was the lead author of the document, conceived of the analysis to isolate the outdoor air pollution and health effects from household space heating with solid fuels, assembled the necessary data, and drafted the manuscript. Kirk Smith and Michael Brauer advised the analysis, suggested alternative techniques, and edited the findings. I worked with all co-authors of the WHO document to construct and edit the manuscript.

Chapters 1, 4, and 7 consist of my original unpublished work.

Acknowledgments

I have been truly privileged to work with and learn from Kirk Smith. Kirk has been an invaluable mentor throughout my time at Berkeley, creating countless opportunities for me to stretch my knowledge and skills in new ways while also providing unwavering support and guiding this study forward. Kirk's infectious enthusiasm for research on household energy, climate change mitigation, and co-benefits for public health was instrumental in my decision to purse this dissertation topic.

I benefited from the inspiration and feedback provided by members of the Household Energy, Climate and Health Research Group, including Ajay Pillarisetti, Amanda Northcross, Anna Zimmermann Jin, David Holstius, Drew Hill, Heather Adair-Rohani, Ilse Ruiz Mercado, Jiawen Liao, Manish Desai, María Teresa Hernández, Nick Lam, and Seth Shonkoff, as well as Jennifer Mann, Justin Girard, Luis Alvarado, Michael Murphy, Nancy Smith, and Terry Jackson. I am appreciative of the advice and expertise offered by John Balmes, whose unique perspectives have improved this project.

Whenever someone asks me what makes ERG stand out from other programs and departments, I answer without hesitation that it is the collaborative, extremely inquisitive, and intellectually diverse group of people that ERG somehow continuously manages to attract and retain. I thank Dan Kammen for simultaneously encouraging and challenging me to further expand both my masters and dissertation research on household energy; to Dick Norgaard and Gene Rochlin for their advising and interest in my research; and to Isha Ray for her support throughout my years at ERG. I am deeply grateful to the ERG staff, including Kay Burns, Bette Evans, and Sandra Dovali, who go to extraordinary lengths to ensure that students thrive; and to the many students from whom I learned tremendously while at ERG.

I value the external collaborations that developed over the course of this research, including with Michael Brauer (UBC); Fabian Wagner, Keywan Riahi, Shilpa Rao, Shonali Pachauri, and Zbigniew Klimont (IIASA); Sumi Mehta (Global Alliance for Clean Cookstoves); David Fairley (Bay Area Air Quality Management District); Marie-Eve Héroux (World Health Organization); Shan Ming (Tsinghua University) and Alistair Woodward (University of Auckland).

This work was supported in part by the National Science Foundation (NSF) Graduate Research Fellowship (Grant DGE-1106400), the National Academy of Sciences Board on International Scientific Organizations (Grant OISE-0738129), and the World Health Organization Regional Office for Europe.

More than anything, I am grateful for the incredible support of my immediate and extended family. Evan Lovett-Harris and Zia Lovett are my constant inspirations. My parents, Chris and June, gave me the incredible gift of freedom: to explore, to learn, and to follow the paths that seemed most interesting and important. Only recently have I come to truly understand what a momentous gift that is.

Let us use no euphemisms, no glossing words, to cover our own misdemeanours in the vain attempt to blame Dame Nature.

The citizens themselves, along with some manufacturers, are alone to blame;

and the dire effect, death—this excessive death—is due to one thing and one alone, and that is smoke!

And the pity of it all is that it is mostly our children that are slain.

—Peter Fyfe, Glasgow Chief Sanitary Inspector, 1909
Public lecture at Glasgow's Technical College. Quoted in Mosley (2001).

Chapter 1 : Introduction

1.1 Aims and objectives

Worldwide, nearly 3 billion people—40% of the global population—burn wood, coal, and other solid fuels every day to accomplish the basic task of cooking their food (Bonjour et al., 2013; Pachauri et al., 2012); this number is even larger when families who heat their homes with solid fuels are included well. Household solid fuel combustion occurs either because families are too poor to afford other forms of energy, such as electricity or liquefied petroleum gas (LPG); because other fuels are not available; or, in a minority of cases, because they prefer solid fuels over others. Exposure to pollution from cooking fires causes about 3 million deaths each year, making it one of the biggest environmental health problems the world faces (Forouzanfar et al., 2015; Lim et al., 2012; Smith et al., 2014). Pollution from household space heating emissions adds an additional health burden that has not yet been well-estimated. The harm from this smoke is not restricted to those who breathe it, however: it contains gases and particles that contribute to global climate change as well (Anenberg et al., 2013; Smith et al., 2009).

While many of the families who rely on solid fuels live in developing countries and use the fuel for cooking, household heating with wood and coal remains a widespread practice in Europe and North America (Chafe et al., 2015; Rogalsky et al., 2014). Household use of coal is now widely discouraged—by the World Health Organization (WHO) and local governments from Dublin to Beijing—because it is so damaging both to human health and the environment (Clancy et al., 2002; World Health Organization, 2014b). However, use of biomass for home heating is increasing in some countries, especially those with climate change mitigation goals, because of its attractiveness as an alternative to fossil fuels (European Commission, 2014).
The emissions associated with household combustion of solid fuels are of particular concern because of their contribution to indoor air pollution, their contribution to outdoor (ambient) air pollution, their negative environmental effects, and their effects on individuals' health, communities' health, and public health more broadly. The many health effects of indoor air pollution from household solid fuel combustion are increasingly well understood and have been well-documented by hundreds of researchers over the past three decades (Balmes, 2010; Baumgartner et al., 2011; Chowdhury et al., 2012; Dherani et al., 2008; Ezzati and Kammen, 2001; Ezzati and Kammen, 2002; Kurmi et al., 2010; Lim et al., 2012; Mondal et al., 2010; Mortimer et al., 2012; Pope et al., 2010; Smith et al., 2011; Smith, 1993; Smith et al., 2014; Smith and Mehta, 2003). (See Figure 1.1.)

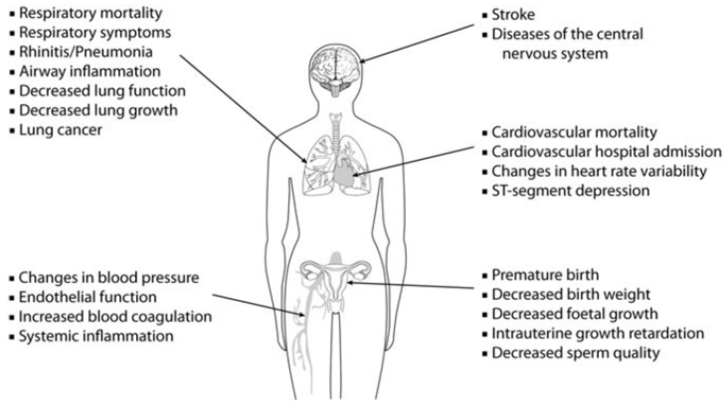

Figure 1.1: Organs of the human body affected by fine particulate air pollution. From: Peters et al. (2011)

For decades, researchers have suspected that household cooking with solid fuels causes not only harmful air pollution levels indoors and around the household, but also contributes to ambient (outdoor) air pollution (AAP) to which a broader population (in the neighborhood and beyond) is exposed (Smith et al., 1994). When households cook with solid fuels, the cooks and their families are exposed to harmful household air pollution; but this pollution does not stay in kitchens or courtyards, because the smoke exits through windows, gaps in walls and roofs, and/or ventilation systems such as chimneys, becoming a source of outdoor air pollution as well. This is particularly concerning in areas where most households cook with solid fuels, such as rural villages in many developing countries. Although rural air quality is often assumed to be better than air quality in cities, new analysis done by the GBD AAP risk factor group shows that fine particulate air pollution ($PM_{2.5}$) concentrations are actually quite high in many rural areas (Brauer et al., 2012). (See Figure 5.1.)

Beyond the scope of the immediate neighborhood or village where the smoke is released, emissions from household combustion have implications for regional air quality, as $PM_{2.5}$ can move via atmospheric transport, and for climate change, since many constituents of the smoke that is released from incomplete combustion of solid fuels is associated with radiative warming. (See Chapter 6.)

This dissertation explores the outdoor air quality and health consequences of residential combustion of solid fuels, focusing first on household cooking patterns, second on household heating patterns, and thirdly on the air quality, health, and economic impacts of household combustion policymaking in a specific place (the San Francisco Bay Area in California).

In this dissertation I address three goals:

1. The first goal is to estimate the contribution to outdoor (ambient) fine particulate air pollution from household combustion of solid fuels, filling a crucial knowledge gap in our understanding of the environmental consequences of cooking and heating with solid fuels.

2. The second goal is to estimate the health effects of the outdoor air pollution generated by household combustion of solid fuels, to better understand the extent to which household cooking and heating with solid fuels contributes to ill-health and premature deaths.

3. The third goal is to assess the policy interventions and voluntary actions available to reduce emissions from household combustion of solid fuels for heating, and their relevance to climate change mitigation strategies.

1.2. Redefining Household Air Pollution from Cooking and Heating with Solid Fuels

Whether regarding appliance or stove use, most efforts to understand household fuel use have centered, understandably, on households. Taking cues from other development-related research fields, such as water and sanitation, the household fuel use community is beginning to focus attention on neighborhood-level measurements and interventions. This is particularly important for air pollution-related research, because the fuel and appliance choices made by a single household can affect ambient air quality for neighbors. Past air pollution research efforts, such as the 2004 Global Burden of Disease risk factor report on outdoor air pollution, have often focused exclusively on urban air quality, since that is where most data are collected (Cohen et al., 2004). Relatively new satellite data collection and advanced modeling techniques have very recently made more geographically-comprehensive analysis possible (Brauer et al., 2012), something that has been reflected in recent GBD publications (Brauer et al., 2016; Forouzanfar et al., 2015; Lim et al., 2012).

1.2.1 Residential Cooking

An estimated 2.8 billion people cook with solid fuels worldwide (Bonjour et al., 2013). Though the issue of the contribution of household cooking to AAP has been mentioned in the past, as in Smith's critique of so-called "smokeless" stoves that simply divert smoke out of the kitchen via a chimney (Smith, 1989), few publications have quantified the proportion of outdoor particulate pollution attributable to cooking with solid fuels on a global scale, despite its often-prominent role in determining outdoor air quality. The Global Burden of Disease (GBD) and associated Comparative Risk Assessment (CRA) reported in 2004 the burden of morbidity and mortality associated with indoor air pollution from household cooking and, separately, outdoor air pollution, but only in cities due to data limitations at the time (Ezzati et al., 2004). As interest in understanding not only the individual and family-level effects of household solid fuel combustion have grown, so have efforts to better understand 'improved' cookstoves and their impact on both indoor and ambient air pollution.

Smith describes three phases of interest in, research on, and dissemination and evaluation of 'improved' stoves designed for biomass fuels (also called biofuels): the classic phase, which sought to lower smoke exposures; the energy phase, which focused on fuel savings; and the 'phoenix' phase, which can benefit from lessons learned in the earlier phases (Smith, 1989). Similar phases are identified by US Congress Office of Technology Assessment, from a 1953 pamphlet promising women "five freedoms—from smoke, from soot, from heat, from waste, and from fire risk" to a technical assistance project funded by USAID and IBM-Europe in the 1980s

(U.S. Congress Office of Technology Assessment, 1992); and by the World Bank (World Bank, 2011). In general, cookstove designers often attempt to either maximize energy efficiency or reduce human exposure to harmful emissions; doing both things at once has proved difficult. To add to the challenge, many stoves are never rigorously lab- or field-tested before dissemination; although many stoves perform quite differently in the field than in the lab, Jetter recently made a significant contribution by testing 22 stoves in controlled conditions (Jetter et al., 2012).

Even after hundreds of studies to assess household air pollution levels in developing countries, and the associated health outcomes, major questions remain about what constitutes an "improved" cookstove, the efficacy of various stoves when used in households (rather than lab conditions), and whether high levels of "improved" stove adoption can ever be achieved. These large questions lead to myriad other questions, such as whether stoves should be sold, partially subsidized, or given away for free; whether it is ethical to distribute stoves that increase energy efficiency but are known not to result in significant pollution reductions, or those that cannot be fixed easily at the local level if they break; what adoption rates should be used in calculating benefits associated with stove adoption, including carbon credits, and for how long stoves should be expected to last once distributed.

The study described in Chapter 2 filled a gap in the existing literature by specifically examining and attempting to estimate the impact of household cooking on population-wide exposure to AAP. Since its initial publication (Chafe et al., 2014), at least one other study has estimated the deaths attributable to population exposure to AAP from household combustion, though it did not distinguish between heating and cooking activities (Lelieveld et al., 2015). However, this paper does not report concentrations of $PM_{2.5}$ attributable to household combustion; it focuses purely on premature mortality.

I anticipate that the findings from Chapter 2 will help policymakers understand the role of household cooking in achieving better urban and rural air quality, while also helping health researchers to better understand the complete burden of disease from household cooking—through both indoor and outdoor exposure routes. More specifically, results show that, in many regions, the fraction of outdoor air pollution attributable to cooking with solid fuels is so high that it will be difficult or impossible to reach international air quality standards without addressing cooking fuels and appliances. It also highlights the need for better emissions inventories hosted at international institutions, as well as better air pollution data collection in many countries where household cooking accounts for a large fraction of total $PM_{2.5}$ emissions.

1.2.2 Residential Heating

Residential heating is an essential energy service required by many people worldwide. Even with widespread availability of electricity and natural gas, the use of solid fuels for residential heating continues to be common practice in many places, including within European and North American countries. Although there have been efforts in recent years to better understand and track the number of people using solid fuels for cooking, to my knowledge no systematic study of the number of people using solid fuels for heating exists.

Solid heating fuels consist primarily of wood and coal but can also include forestry and agricultural residues and even garbage. Most fuels are burned in small-scale combustion devices, such as household heating stoves or small boilers for single houses, apartment buildings or district heating. Open fireplaces are popular in many parts of the developed world but do not actually provide net heating in most circumstances; they are therefore often characterized as for "recreational" or "aesthetic" use rather than for space heating.

Currently, most burning of solid fuels for space heating is done in devices that incompletely combust the fuel owing to their low combustion temperature and other limitations. This results in relatively high emissions per unit of fuel, including many products of incomplete combustion such as particulate matter with an aerodynamic diameter of less than 2.5 micrometers ($PM_{2.5}$) and carbon monoxide (CO) – two major health-damaging air pollutants. Small-scale solid fuel combustion is also an important source of black carbon (BC) emissions. BC is a component of $PM_{2.5}$ that warms the climate. When coal is used for residential heating it can also result in emissions of sulfur and other toxic contaminants found in some types of coal; even with good combustion these contaminants are not destroyed.

The amount of heating fuel needed in a particular climate is dependent on the fuel efficiency of the stove, as well as the characteristics of the housing in which it is used (such as insulation infiltration – infiltration through the building envelope), an issue this publication does not address further. In developed countries nearly all space heating devices have chimneys; in some developing countries much space heating is done with open stoves inside the house. In both cases most of the emissions end up in the atmosphere and contribute to outdoor air pollution. However, to date, no systematic international study of the effects of household heating with solid fuels on outdoor air pollution has been published.

Chapter 3 begins to fill this gap by drawing on energy use and emissions databases and models to estimate the impact of household combustion of solid fuels (wood and coal) for space heating on population-weighted ambient particulate air pollution ($APM_{2.5}$). Due to data constraints, the chapter focuses primarily on Europe and North America. However, there is strong evidence that household heating contributes to high levels of seasonal air pollution in many Central Asian and East Asian countries, including China (Chen et al., 2013) and Mongolia (Ochir et al., 2014), as well as in Latin America (Sanhueza et al., 2009; Smith and Pillarisetti, 2012), and Australia and New Zealand (Ancelet et al., 2013; Johnston et al., 2013).

The concept for the analysis in Chapter 3 was developed in response to research needs expressed by the Convention on Long-Range Transboundary Air Pollution Joint Task Force on Health, an advisory group convened by the WHO European Centre for Environment and Health and the Executive Body for the Convention on Long-range Transboundary Air Pollution, to assess the health effects of such pollution and to provide supporting documentation. A summary of the results from Chapter 3 were distributed to the Task Force and presented as a document to the United Nations Economic and Social Council Economic Commission for Europe (UNECE) (Executive Body for the Convention on Long-range Transboundary Air Pollution Joint Task Force on the Health Aspects of Air Pollution, 2014).

Chapter 4 focuses on heating but at a much narrower geographic scale: the San Francisco Bay Area. This part of Northern California is a temperate region with a large airshed that spans nine counties. The air quality in the counties is regulated by the Bay Area Air Quality Management District, which faces the challenge of reducing wintertime $PM_{2.5}$ levels to bring the airshed into attainment for the daily average standards (35 $\mu g/m^3$) set by the US Environmental Protection Agency (EPA). Household combustion of wood during winter months is the top source of $PM_{2.5}$ in the Bay Area. This chapter presents a model for estimating the relative influences of three wood burning device types (fireplaces, wood stoves, and pellet stoves) on mass emissions of $PM_{2.5}$, and analyzes the projected health and economic effects of five hypothetical scenarios designed to reduce $PM_{2.5}$ emissions from the residential wood heating sector. The findings presented in Chapter 4 form the basis for new work that will done in conjunction with the Bay Area Air Quality Management District (BAAQMD) to better understand potential trade-offs or synergies between efforts to reduce local $PM_{2.5}$ air pollution and to mitigate greenhouse gases (GHG) and short-lived climate pollutants.

1.3. Hypotheses and Questions

In this dissertation, I explore to what extent household combustion of solid fuel for cooking and heating affects outdoor air pollution and human health. I focus on a specific type of air pollution: $PM_{2.5}$.

1.3.1 Chapter 2: Ambient Air Pollution from Household Cooking with Solid Fuels

In Chapter 2, I hypothesize that household combustion of solid fuels for cooking is an important source of fine particulate air pollution in regions where cooking with solid fuels continues to be a common practice. I also hypothesize that exposure to this air pollution results in a population-wide burden of disease. This chapter describes the methodology for isolating the emissions from household cooking that I developed for this dissertation. I apply this methodology internationally and report regional and global estimates of the concentration of $APM_{2.5}$ attributable to household cooking, as well as the proportion of AAP attributable to this source, and the human health effects of exposure to this outdoor air pollution.

1.3.2 Chapter 3: Ambient Air Pollution from Household Heating with Solid Fuels

In Chapter 3, I test the hypothesis that household combustion of solid fuels for space heating has a notable impact on levels of fine particulate air pollution in temperate regions where space heating is needed. I also consider whether this energy use for heating results in population-wide burden of disease. In the chapter, I apply the same methodology to isolate the particulate emissions from household heating with solid fuels, though my analysis is restricted to regions for which robust data about solid fuel use for heating exists. I find that estimates of solid fuel use for household heating are not complete or comparable for many regions worldwide, so I focus my analysis on Europe and North America. I report the concentrations of $APM_{2.5}$ attributable to household heating with solid fuels, the proportion of $APM_{2.5}$ attributable to this energy source, and the effects of exposure to this AAP on human health.

1.3.3 Chapter 4: Household Wood Combustion in the San Francisco Bay Area

In Chapter 4, I ask which wood burning devices contribute most to household wood combustion emissions in the San Francisco Bay Area. I present a model that explores the relative influences of wood burning device use, behavioral patterns, and $PM_{2.5}$ emission rates specific to the San Francisco Bay Area. I also design and test the impacts of five hypothetical scenarios on public health and economic outcomes linked to avoided emissions from the residential wood burning sector. I then relate these scenarios to current policy and regulatory measures available to the San Francisco Bay Region, including emission limits on specific devices set by the US EPA, sector regulations enforced and recently amended by the Bay Area Air Quality Management District, relevant local ordinances, and educational campaigns designed to influence wood burning behavior.

1.4. Exclusions

This dissertation focuses specifically on residential combustion of solid fuels for cooking and home heating, meaning that I consider emissions from burning wood, coal, and other fuels in stoves, fireplaces, and other technologies that are located directly in homes. In this analysis, I do not characterize or include emissions from district heating, which is centralized combustion of fuels for production of heat that is piped throughout an apartment building, complex, neighborhood or city.

I also do not include the fuel used for—or resulting emissions from—commercial cooking or heating. However, the distinctions between household and commercial uses of energy are not always clear, and the two uses are often mixed in energy use and emission inventories, particularly in developing countries. I separate residential and commercial uses of energy where possible, here, but recognize that there is some overlap between the sectors. Especially in the case of cooking with solid fuels, commercial solid fuel use and related emissions are dwarfed by residential uses, so there is no major concern with regards to data usability and interpretation of results.

Finally, I focus here on the AAP from household combustion of solid fuels. I do not make new estimates of health effects from exposure to the indoor and near household air pollution that is produced by household combustion of solid fuels My focus in this dissertation is on population-wide exposure to this pollution, and the resulting population-wide health impacts, rather than on the exposures that occur among the family members who are exposed in the houses where the combustion is occurring. Chapters 2 and 3 explain the contribution of household cooking and heating emissions to regional $APM_{2.5}$, and Chapter 4 examines the impact of lessening AAP associated with household wood burning in the San Francisco Bay Area.

1.5 Methods developed and employed

In the following chapters, I examine the relative impacts of household combustion of solid fuels for cooking (Chapter 2) and space heating (Chapter 3) on AAP. I also estimate the ill-health associated with this combustion. In Chapter 4, I model the air pollution attributable to residential combustion of wood for space heating in the San Francisco Bay Area, as well as the avoided ill-health and economic effects of five hypothetical scenarios to reduce emissions from this sector.

1.5.1 Basic Approaches

The basic approach for the first two studies is to combine particulate air pollution data, modeled emissions inventory data, and population data to estimate the proportion of population-exposure weighted annual average $PM_{2.5}$ air pollution attributable to household cooking with solid fuels. The years of analysis are 1990, 2005, and 2010, following the CRA 2010 methodology (Institute for Health Metrics and Evaluation, 2010b).

The 2010 GBD/CRA update incorporates satellite measurements of $APM_{2.5}$ to better estimate the complete burden, rather than just that in urban populations (Brauer et al., 2012). I use data from that study, as well as estimates of country-level cooking emissions from the International Institute for Applied Systems Analysis (IIASA) Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) model (Amann et al., 2011; IIASA, 2014), which references the Model for Energy Supply Strategy Alternatives and their General Environmental Impact (MESSAGE) for energy use estimates. TM5-FASST provided estimates of the contribution of household emissions to total emissions, importantly including secondary $PM_{2.5}$ formation. The data used from each source are described below.

For the San Francisco Bay Area case, I use survey data commissioned by the Bay Area Air Quality Management District (Fairley, 2014; True North, 2014), emission factors published in the literature (Gullett et al., 2003) and by the US government (Macdonald, 2009), emission inventories published by BAAQMD (Bay Area Air Quality Management District, 2014) and the California Air Resources Board (CARB) (California Air Resources Board, 2015), and a health and economic outcome methodology published by EPA (US Environmental Protection Agency, 2013).

1.5.2 Estimating the Burden of Disease and Health Benefits of Avoided Pollution

Given the different geographic scales, model structures, and health-related questions asked in Chapter 4, as compared to Chapters 2 and 3, the methodology used to determine the relationships between household solid fuel combustion and public health also differed. In brief, the approach used in Chapters 2 and 3 followed that of the Global Burden of Disease, in which ill-health is attributed to a given cause through a rigorous analysis of risk factors and health outcomes that are reconciled across international regions, age groups, and sexes. Chapter 4, which focuses on a sub-state region in the United States—the San Francisco Bay Area—employs a series of factors that relate avoided mass of particulate emissions in a specific sector—residential wood combustion—to various health and economic outcomes. The differences between these approaches are described in detail in Chapter 5.

Chapters 2 and 3 report the burden of disease associated with population-wide exposure to AAP from household solid fuel combustion, based on the contribution of this source to $APM_{2.5}$. The metrics used to express the health findings are premature deaths and disability-adjusted life years (DALYs). DALYs combine years of life lost (mortality) and years of life lived with disability (morbidity) into a single indicator. Chapter 4 reports health effects of avoided mass emissions of $PM_{2.5}$ by describing premature deaths and specific morbidity-related outcomes (hospitalization, respiratory symptoms, asthma exacerbation, heart attacks). It also describes economic impacts (monetized benefits per ton of $PM_{2.5}$ avoided), whereas Chapters 2 and 3 do not.

1.6 Structure of Dissertation

In Chapters 2 and 3, I describe original analysis on AAP and human health impacts of household cooking and heating with solid fuels. In Chapter 4, I describe construction and results of a model I created to evaluate the influence of various household wood combustion factors and hypothetical scenarios on projected health and economic impacts of avoided emissions from solid fuels in the San Francisco Bay Area.

Appendix A lists relevant abbreviations used in this manuscript. Appendix B describes the regional definitions used to calculate and report results in Chapters 2 and 3; these were also used in the 2010 Global Burden of Disease Project coordinated by the Institute for Health Metrics and Evaluation (IHME). They include the two major energy use, emissions, and air pollution models that I rely on for my analysis: the GAINS model and TM5-FASST. Appendix C lists input data used in calculating population-weighted averages by region, such as population and emissions data. Appendix D contains a graphic that describes in detail the emissions and particles covered by various models used in the analysis in Chapter 2 and Chapter 3.

Chapter 2 : Household Cooking with Solid Fuels Contributes to Ambient $PM_{2.5}$ Air Pollution and the Burden of Disease*

*Some material in this chapter was previously published in the following article:

Chafe, Z. A., Brauer, M., Klimont, Z., Van Dingenen, R., Mehta, S., Rao, S., Riahi, K., Dentener, F., Smith, K.R. (2014). "Household Cooking with Solid Fuels Contributes to Ambient $PM_{2.5}$ Air Pollution and the Burden of Disease." *Environmental Health Perspectives* 122(12): 1314-1320.

2.1 Abstract

Background: Approximately 2.8 billion people cook with solid fuels. Research has focused on the health impacts of indoor exposure to fine particulate pollution. Here, for the 2010 Global Burden of Disease project (GBD 2010), I evaluated the impact of household cooking with solid fuels on regional population-weighted ambient $PM_{2.5}$ (particulate matter with aerodynamic diameter ≤ 2.5 μm) pollution ($APM_{2.5}$).

Objectives: I estimated the proportion and concentrations of $APM_{2.5}$ attributable to household cooking with solid fuels ($PM_{2.5\text{-cook}}$) for the years 1990, 2005, and 2010 in 170 countries, and associated ill health.

Methods: I used an energy supply–driven emissions model (GAINS: Greenhouse Gas and Air Pollution Interactions and Synergies) and source-receptor model (TM5-FASST) to estimate the proportion of $APM_{2.5}$ produced by households and the proportion of household $PM_{2.5}$ emissions from cooking with solid fuels. I estimated health effects using GBD 2010 data on ill health from $APM_{2.5}$ exposure.

Results: In 2010, household cooking with solid fuels accounted for 12% of $APM_{2.5}$ globally, varying from 0% of $APM_{2.5}$ in five higher-income regions to 37% (2.8 μg/m^3 of 6.9 μg/m^3 total) in southern sub-Saharan Africa. $PM_{2.5\text{-cook}}$ constituted > 10% of $APM_{2.5}$ in seven regions housing 4.4 billion people. South Asia showed the highest regional concentration of $APM_{2.5}$ from household cooking (8.6 μg/m^3). On the basis of GBD 2010, I estimate that exposure to $APM_{2.5}$ from cooking with solid fuels caused 370,000 premature deaths globally in 2010, about the same number as that caused by exposure to unimproved water and sanitation systems (340,000). The total burden of disease from exposure to $APM_{2.5}$ from cooking with solid fuels was 9.9 million disability-adjusted life years globally in 2010, about four times the burden associated with exposure to ambient ozone air pollution (2.5 million disability-adjusted life years) and half of the burden associated with exposure to second-hand smoke (20 million disability-adjust life years) in 2010.

Conclusions: $PM_{2.5}$ emissions from household cooking constitute an important portion of $APM_{2.5}$ concentrations in many places, including India and China. Efforts to improve ambient air quality will be hindered if household cooking conditions are not addressed.

2.2 Introduction

Approximately 2.8 billion people, more than ever before in human history, use solid fuels, including wood, coal, charcoal, and agricultural residues, as their primary fuel for cooking (Bonjour et al., 2013). Solid fuel is usually combusted in inefficient cookstoves, producing a variety of health-damaging gases and particles (Smith et al., 2009), such as BC, organic carbon (OC), methane, and carbon monoxide. The 2010 Global Burden of Disease/Comparative Risk Assessment Project (GBD 2010) estimated that exposure to household air pollution from cooking with solid fuels caused 3.5 million premature deaths in 2010 (Lim et al., 2012).

The potential for harm does not stop when this smoke exits house windows or chimneys, however: in areas where solid fuels are the primary source of household cooking, particulate emissions from household cooking with solid fuels contribute significantly to AAP (Smith, 2006). Indeed, the AAP exposure assessment prepared for GBD 2010 shows substantial exposures occurring in rural areas (Brauer et al., 2012), as do others (Anenberg et al., 2010; Rao et al., 2012).

The important contribution of household fuel use (for heating and cooking) to particulate matter emissions has been established in previous emission inventory research. Residential coal and biomass combustion remains a key source of fine particulate matter ($PM_{2.5}$) in China, accounting for 47% (4.3 teragrams (Tg) of 9.3 Tg total) and 34% (4.4 Tg of 13.0 Tg total) of China's $PM_{2.5}$ emissions in 1990 and 2005 (Lei et al., 2011); the drop in relative contribution was attributable primarily to growth in industrial emissions. Besides industrial processes, power production and ground transportation are other sectors that contribute substantially to $PM_{2.5}$ pollution.

Recent studies have found that 50-70% of the BC (Cao et al., 2006; Klimont et al., 2009; Lei et al., 2011) and 60-90% of OC emissions in China can be attributed to residential coal and biomass use; Klimont et al. (2009) found similar proportions in India. Even higher contributions were estimated by Ohara (2007): in 2000, 86% of BC emissions in both India and China—together home to more than a third of the world's population—could be attributed to residential coal and biomass use; for OC, the proportion was 96% in India and 97% in China.

Source apportionment studies in India and China have shown that biomass combustion can be a major source of ambient particulate air pollution across the urban-rural spectrum (Chowdhury et al., 2012; Wang et al., 2005), despite the observation that household energy use patterns—and associated emissions—tend to differ by population density, economic status, and geographic location (van Ruijven et al., 2011; Zhang et al., 2010). In many countries, solid fuel use is more prevalent in rural areas (Barnes and Floor, 1996). However, solid fuels are still used by households in many cities for heating and cooking, as evidenced by the major contributions of biomass burning to urban particulate pollution found in previous source apportionment studies (Health Effects Institute, 2010; Pant and Harrison, 2012). In China, for example, although coal use has been banned in most urban areas, many urban households reported in a recent energy use survey that they use coal and other solid fuels for cooking (20.3%, equal to approximately 144.5 million people) (Duan et al., 2014). For the analysis presented here, which focuses on the relative contributions of emission source categories, the exact location of the emission sources is not as significant as it would be for research on individual-level human exposures.

My objective was to systematically estimate the contribution of household air pollution from cooking with solid fuels ($PM_{2.5\text{-cook}}$) to outdoor ambient population-weighted $PM_{2.5}$ air pollution ($APM_{2.5}$), by region, in 1990, 2005, and 2010. My estimates are based on the fraction of ambient primary combustion-derived household particulate emissions ($PPM_{2.5\text{-hh}}$) attributable to cooking and the fraction of $APM_{2.5}$ attributable to household activities ($PM_{2.5\text{-hh}}$). These calculations enable us to estimate the burden of disease from AAP that can be attributed to household cooking ($PM_{2.5\text{-cook}}$), and to better understand the degree to which attainment of outdoor air quality goals depends on control of household air pollution.

I focused specifically on household cooking with solid fuels, as this is one of the air pollution risk factors included in GBD 2010. Other household sources of combustion air pollution, including household space heating, were not considered in this analysis. (See Chapter 3.) I explored $PM_{2.5\text{-cook}}$ at the national level in 170 countries, for the years 1990, 2005, and 2010, and report the results at the regional level in concordance with GBD 2010 (Brauer et al., 2012; Institute for Health Metrics and Evaluation, 2010a).

The main data sources used in this analysis were 1) emissions estimates from the Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) models hosted by IIASA (http://gains.iiasa.ac.at/models/) (Amann et al., 2011; Cofala et al., 2012) and 2) atmospheric concentration estimates from the TM5-FASST (Fast Scenario Screening Tool for Global Air Quality and Instantaneous Radiative Forcing, paired with TM-5, a global chemical transport model) screening tool hosted by the European Commission Joint Research Center (JRC) based on emissions estimates from MESSAGE (Rao et al., 2012).

This chapter details the methods for calculating the ill-health associated with population-wide exposure to just the AAP caused by household cooking with solid fuels. Together, household and ambient exposure to fine particulate air pollution from household cooking with solid fuels caused an estimated 3.9 million premature deaths in 2010 (Smith et al., 2014), including adjustment for overlaps between the two routes of exposure.

2.3. Methods

Because most emission inventories report total residential emissions (Bond et al., 2004; Lamarque et al., 2010; Shen et al., 2012; Streets et al., 2003), with no distinction between cooking and heating, I took the approach of calculating 1) the proportion of $PM_{2.5\text{-hh}}$ emissions attributable to cooking (rather than heating), and then 2) the proportion of $APM_{2.5}$ attributable to $PM_{2.5\text{-hh}}$.

2.3.1 Residential emissions

To focus specifically on the residential sector, I used GAINS and Equation 1 to determine the fraction of $PPM_{2.5\text{-}hh}$ from cooking with solid fuels such as hard coal, agricultural residues, fuelwood, and dung, for each country or sub-national jurisdiction (IIASA, 2012):

$$(\text{PIT} + \text{STOVE}) / \Sigma\ \text{DOM} = PPM_{2.5\text{-}hh} \text{ from cooking} \qquad [1]$$

where PIT indicates emissions from open fire cooking with solid fuels (Tg of $PPM_{2.5}$ per country), STOVE represents emissions from combusting solid fuels in residential cooking stoves (Tg of $PPM_{2.5}$ per country), and DOM indicates total emissions from all residential sources, including boilers and heating stoves (Tg of $PPM_{2.5}$ per country). Non-fuel emissions associated with cooking (such as volatile organic compounds (VOC) created by frying) are not included.

Within GAINS, I used a scenario that draws on data from the International Energy Agency (International Energy Agency, 2011a). GAINS estimates current and future $PPM_{2.5}$ emissions using activity data, fuel-specific uncontrolled emission factors, the removal efficiency of emission control measures and the extent to which such measures are applied (Amann et al., 2011; Kupiainen and Klimont, 2007). For household cooking with solid fuels from 1990 through 2010, no technical control measures were applied in the model. I multiplied the fraction of residential $PPM_{2.5}$ attributable to household cooking by the proportion of total ambient population-weighted $PM_{2.5}$ attributable to household combustion ($PM_{2.5\text{-}hh}$) (Equation 2). The latter proportion (% $PM_{2.5\text{-}hh}$) was generated using TM5-FASST.

$$\%\ PPM_{2.5\text{-}hh} \text{ from cooking} \times \%\ PM_{2.5\text{-}hh} = \%\ PM_{2.5\text{-}cook} \qquad [2]$$

All analysis in Equation 2 is at the country level, % $PPM_{2.5\text{-}hh}$ from cooking is the quantity derived in Equation 1, and % $PM_{2.5\text{-}hh}$ = $\mu g/m^3$ $PM_{2.5\text{-}hh}$ / $\mu g/m^3$ $PM_{2.5}$.

2.3.2 Regional population-weighted estimates

Equation 3 shows the method by which country-level results were combined to produce regional population-weighted estimates.

$$\frac{\sum_{i=1}^{\substack{countries \\ in\ region}} \left[\frac{PPM2.5\ from\ cooking_i}{PPM2.5{-}hh_i} \times \frac{PM2.5{-}hh_i}{Total\ APM2.5_i} \times Population_i\right]}{\sum_{i=1}^{\substack{countries \\ in\ region}} Population_i} = PM2.5{-}cook_{region} \qquad [3]$$

I used global estimates of annual average ambient population-weighted $PM_{2.5}$ concentrations, which were developed for the GBD 2010 study (Brauer et al., 2012) as well as the Global Energy

Assessment (Riahi et al., 2012), to estimate the proportions and absolute concentrations of $PM_{2.5-cook}$, on a regional basis. The underlying methodology for deriving $PM_{2.5}$ concentrations is described in Rao et al. (2012) and combines the global integrated assessment model MESSAGE (Rao and Riahi, 2006; Strubegger et al., 2004) with TM5 (see Chapter 5.) MESSAGE covers all greenhouse gas-emitting sectors; in the residential sector, MESSAGE includes an explicit representation of the energy use of rural and urban households with different income levels. Fuel choices at the household level consider the full portfolio of commercial fuels as well as traditional biomass for cooking, heating and specific use of electricity of household appliances (Ekholm et al., 2011).

TM5-FASST was used to determine $PM_{2.5-hh}$. Secondary organic aerosol formation was included in TM5-FASST estimates of annual average population-weighted $PM_{2.5}$ concentrations (see Appendix D for more information on the emission and source categories included in this analysis.) Dust and sea salt increments were estimated by comparing concentrations generated by TM5-FASST to those developed with TM5-FASST, satellite data and ground measurements for GBD 2010 and published in Brauer et al. (2012). Positive differences between GBD 2010 and TM5-FASST were assumed to be representative of dust and sea salt increments and were included in estimates of $APM_{2.5}$ to better approximate the proportional role of household solid fuel use for cooking in creating $APM_{2.5}$.

Data sources and models used in my analysis are summarized in Table 2.1. Regional population and household emissions estimates are shown in Appendix B.

Table 2.1. Sources of input data for household cooking analysis

Data source and model	Purpose in this analysis	Data attributes	Spatial resolution	References
GAINS	Calculate proportion of household $PM_{2.5}$ emissions that comes from cooking	Includes household cooking stoves and open-pit cooking emissions. Does not include nonfuel cooking emissions. Units: mass emissions of primary $PM_{2.5}$, by sector and technology used.	Country or subcountry	IIASA 2012; IEA 2011; Purohit et al. 2010
TM5-FASST (MESSAGE)	Calculate proportion of ambient $PM_{2.5}$ that comes from household combustion	Uses MESSAGE to calculate particulate matter emissions by sector and TM5 atmospheric chemical transport model to calculate secondary organic aerosol formation. Units: concentrations (μg/m^3) of annual average population-weighted $PM_{2.5}$. Includes secondary organic aerosol formation. Dust and sea salt estimated by comparing combustion-derived $PM_{2.5}$ to total ambient $PM_{2.5}$ reported by Brauer et al. (2012).	Country or region (derived from gridded 1° × 1° concentration results)	Brauer et al. 2012
Global burden of disease	Calculate ill health resulting from exposure to outdoor $PM_{2.5}$ air pollution	Uses estimates of average annual population-weighted $PM_{2.5}$ concentrations to calculate ill health from outdoor air pollution. Units: annual deaths and DALYs, by region.	Deaths and DALYs: region $PM_{2.5}$. Concentrations: 0.1° × 0.1° gridded	Brauer et al. 2012; Lim et al. 2012

2.3.3 Years and Countries Included

Following GBD 2010 (Institute for Health Metrics and Evaluation, 2010a), this analysis considers $PM_{2.5}$ emissions for three time points: 1990, 2005, and 2010. The data cover 170 countries (see Appendix C) in 20 of the 21 GBD 2010 regions; the majority of missing countries are small (population <1 million each) and together they account for 34 million people in 2010, that is <1% of the world population.

2.3.4 Burden of Disease

I estimated the burden of disease associated with exposure to outdoor $PM_{2.5}$ air pollution that can be attributed to household cooking by applying the derived proportions of $APM_{2.5}$ due to household cooking with solid fuels to the GBD 2010 burden of disease estimates for AAP (Lim et al., 2012). GBD 2010 burden of disease estimates were calculated using an integrated exposure response function, as described in Burnett et al. (2014). Results were scaled by applying the proportion of $APM_{2.5}$ due to household cooking with solid fuels (the risk factor) to the burden estimates while preserving the exposure-response relationships used to determine the overall burden of disease attributable to AAP.

2.4. Results

Based on estimates of household energy use for cooking, and associated emissions, reported in GAINS and TM5-FASST model results, I estimate that globally about 12% of population-exposure weighted average ambient $PM_{2.5}$ is attributable to household use of solid cooking fuels (Table 2.2, Figure 2.1). In 7 of the 20 regions analyzed, at least 10% of ambient $PM_{2.5}$ was attributed to household cooking in 2010. These 7 regions encompass 41 countries and are home to > 4 billion people. In contrast, seven of the regions analyzed (representing 56 countries with 1.4 billion people) had negligible levels (<2% $PM_{2.5\text{-cook}}$) throughout the 1990–2010 study period. By region, estimated proportions of $APM_{2.5}$ attributable to $PM_{2.5\text{-cook}}$ in 2010 ranged from 0 to 37% (Figure 2.1). In general, I observed that an increase in country-level economic status was accompanied by a decrease in the contribution of household cooking to $APM_{2.5}$.

Table 2.2. Population-weighted contribution of cooking to ambient particulate matter pollution by region.

	$PM_{2.5\text{-cook}}$ (%)[b]			$PM_{2.5\text{-cook}}$ (μg/m^3)[c]			$APM_{2.5}$[d]		
GBD 2010 region[a]	1990	2005	2010	1990	2005	2010	1990	2005	2010
Southern sub-Saharan Africa	13.0	32.0	37.0	0.8	2.2	2.8	6.4	6.6	6.9
South Asia	15.0	30.0	26.0	4.4	9.4	8.6	30.0	32.0	33.0
Southern Latin America	11.0	13.0	15.0	0.8	0.8	1.0	6.4	6.0	5.9
Eastern sub-Saharan Africa	4.9	12.0	13.0	0.5	1.1	1.2	11.0	12.0	12.0
Southeast Asia	22.0	13.0	11.0	3.9	2.5	2.0	16.0	17.0	17.0
East Asia	23.0	14.0	10.0	11.0	9.1	7.3	49.0	63.0	72.0
Western sub-Saharan Africa	3.4	9.0	10.0	0.9	2.2	2.4	27.0	27.0	27.0
Central sub-Saharan Africa	3.7	9.4	9.8	0.6	1.3	1.4	16.0	14.0	14.0
Tropical Latin America	3.9	6.2	7.1	0.2	0.3	0.4	5.2	5.2	5.1
Andean Latin America	5.7	5.2	5.7	0.4	0.4	0.4	7.8	8.2	8.0
Central Latin America	5.5	5.0	5.3	0.7	0.5	0.5	14.0	11.0	12.0
Caribbean	7.1	4.7	5.3	0.6	0.4	0.4	8.6	9.3	9.1
North Africa and Middle East	3.3	3.8	3.3	0.9	1.0	0.9	30.0	29.0	29.0
High-income Asia Pacific	1.1	0.6	0.6	0.3	0.2	0.2	31.0	27.0	26.0
Central Asia	0.9	0.1	0.2	0.1	0.0	0.0	24.0	21.0	20.0
Australasia	0.0	0.1	0.0	0.0	0.0	0.0	5.0	5.0	5.7
Western Europe	0.0	0.0	0.0	0.0	0.0	0.0	25.0	17.0	15.0
Central Europe	0.0	0.0	0.0	0.0	0.0	0.0	31.0	19.0	16.0
Eastern Europe	0.0	0.0	0.0	0.0	0.0	0.0	19.0	10.0	10.0
High-income North America	0.0	0.0	0.0	0.0	0.0	0.0	18.0	13.0	13.0
Global	11.0	13.0	12.0	4.0	4.5	4.0	29.0	30.0	31.0

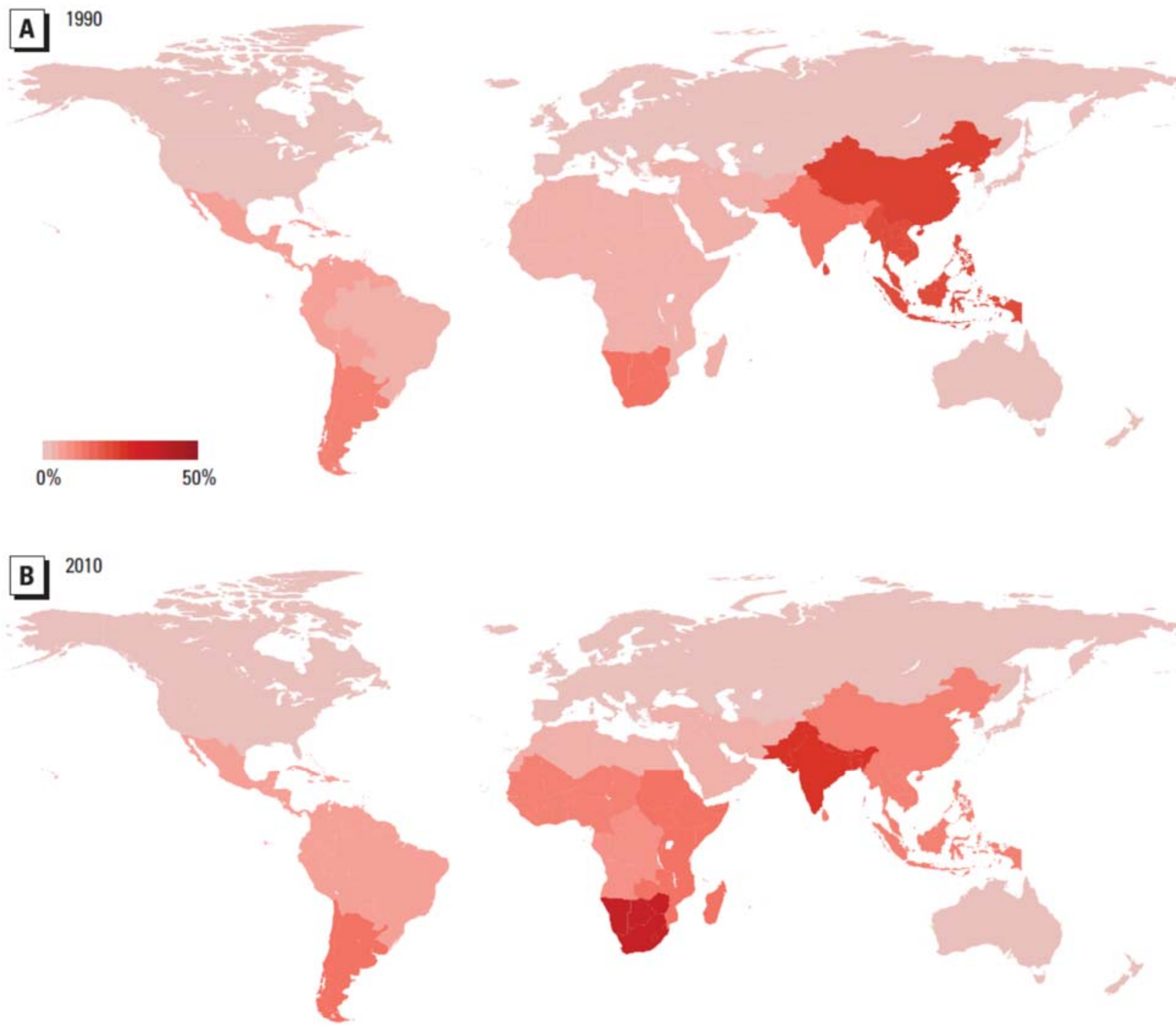

Figure 2.1. Percentage of population-weighted ambient $PM_{2.5}$ attributable to household cooking with solid fuels, 1990 (A) and 2010 (B).

Between 1990 and 2010, East Asia (including China) experienced a decline in absolute levels of $PM_{2.5\text{-cook}}$ (from 11 to 7 μg/m^3) (Figure 2.2) as well as a decline in the percent of $PM_{2.5}$ from cooking (from 23% to 10% in 2010) (Figure 2.1). This occurred alongside a global increase in ambient $PM_{2.5}$ concentrations: Brauer et al. (2012) reported that population-weighted regional annual average $PM_{2.5}$ concentrations rose between 1990 and 2010 in most parts of Asia, including East Asia (from 49 μg/m^3 in 1990 to 72 μg/m^3 in 2010), while falling in North America and Europe, including Central Europe (31 μg/m^3 in 1990, 16 μg/m^3 in 2010).

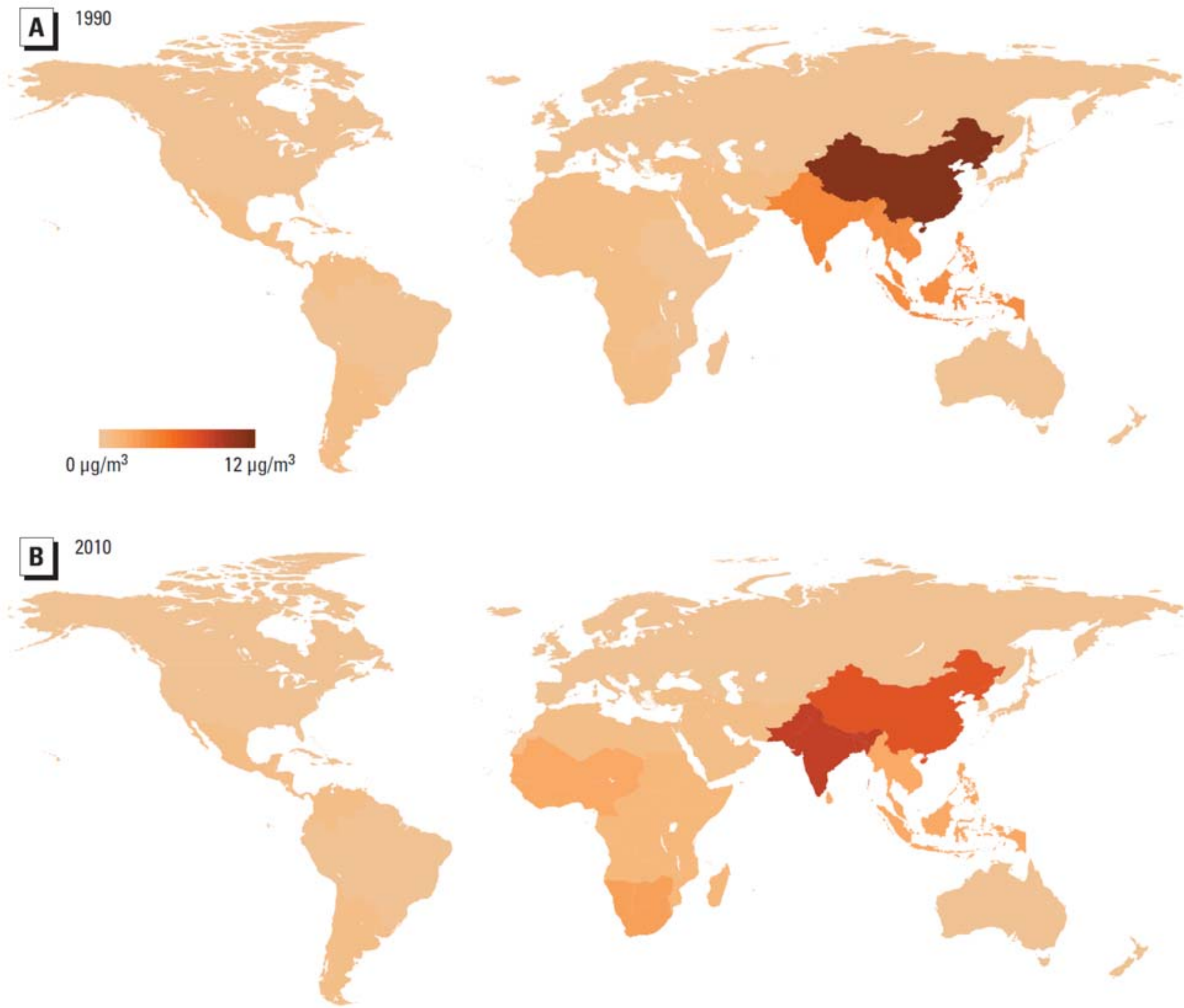

Figure 2.2. Population-exposure weighted concentration of ambient $PM_{2.5}$ attributable to household cooking with solid fuels, 1990 (A) and 2010 (B).

Overall, the estimated population-weighted global annual average $PM_{2.5}$ concentration rose slightly from 29-31 μg/m^3 over this period. This was driven partly by increases in household cooking emissions in South Asia, which includes India: although the percentage of $PPM_{2.5\text{-hh}}$ attributable to cooking remained steady around 82% between 1990 and 2010, $PM_{2.5\text{-cook}}$ rose from 15% to 26%, or 4 μg/m^3 to 9 μg/m^3 (Table 2.2), while $APM_{2.5}$ rose from 30 μg/m^3 to 33 μg/m^3.

The $APM_{2.5}$ formed by household cooking emissions has major implications for human health, as well as outdoor and indoor air quality. Worldwide, the use of solid fuels for household cooking is estimated to have resulted in 370,000 deaths and 9.9 million disability-adjusted life years (DALYs) in 2010 (Table 2.3). The vast majority of these deaths were in South Asia (200,000), which includes India, and East Asia (130,000), which includes China. The relative decrease in $PM_{2.5\text{-cook}}$ in East Asia from 1990 through 2010 (Table 2.2), which was estimated to result in

90,000 fewer deaths per year (Table 2.3), was more than offset by an estimated increase of 121,000 deaths per year from exposure to $PM_{2.5\text{-cook}}$ in South Asia over the same time period. The total burden of disease from exposure to $APM_{2.5}$ from cooking with solid fuels was 9.9 million disability-adjusted life years globally in 2010, about four times the burden associated with exposure to ambient ozone air pollution (2.5 million disability-adjusted life years) and half of the burden associated with exposure to second-hand smoke (20 million disability-adjust life years) in 2010.

Table 2.3. Estimated burden of disease from exposure to ambient $PM_{2.5}$ attributable to household cooking with solid fuels.

	Deaths			DALYs		
GBD 2010 region[a]	1990	2005	2010	1990	2005	2010
South Asia	79,000	210,000	200,000	3,100,000	6,700,000	6,000,000
East Asia	220,000	170,000	130,000	5,700,000	3,700,000	2,600,000
Southeast Asia	24,000	20,000	18,000	800,000	510,000	450,000
Western sub-Saharan Africa	2,400	6,300	7,800	140,000	320,000	380,000
North Africa and Middle East	4,500	6,200	5,800	150,000	170,000	160,000
Eastern sub-Saharan Africa	1,400	3,200	3,500	74,000	150,000	140,000
Central sub-Saharan Africa	480	1,300	1,600	24,000	53,000	65,000
Central Latin America	1,200	1,100	1,400	37,000	26,000	33,000
Southern sub-Saharan Africa	330	1,000	1,400	11,000	36,000	41,000
Tropical Latin America	240	480	540	6,800	12,000	13,000
High-income Asia Pacific	840	470	530	17,000	7,800	8,200
Southern Latin America	440	440	500	9,800	9,000	9,900
Caribbean	390	330	380	9,900	7,500	8,700
Andean Latin America	140	140	160	5,500	3,900	4,200
Central Asia	490	51	78	16,000	1,400	2,000
Western Europe	150	4	2	2,400	64	24
Australasia	0	1	1	4	9	9
Central Europe	0	0	0	0	0	0
Eastern Europe	0	0	0	0	0	0
High-income North America	0	0	0	0	0	0
Global	330,000	420,000	370,000	10,000,000	12,000,000	9,900,000

[a]Regional groupings, defined by IHME for the GBD 2010 project, are described in Supplemental Material, Table S1.

Despite the high proportion of $APM_{2.5}$ attributable to household cooking in Southern sub-Saharan Africa, the estimated health impacts from resulting AAP exposures were relatively modest (41,000 DALYs in 2010) (Table 2.3). However, across the four sub-Saharan African regions, estimated annual deaths due to exposure to $APM_{2.5}$ from cooking more than doubled (Eastern sub-Saharan Africa), tripled (Central and Western sub-Saharan Africa), or quadrupled (Southern sub-Saharan Africa) between 1990 and 2010.

2.5. Sensitivity analysis on spatial resolution of data

I implemented a sensitivity analysis to test the effects of spatial misalignment between the data sources used in this research project. In particular, I sought to address the concern that not differentiating data by urban and rural designations would lead to an overestimate of health effects from AAP attributable to household cooking with solid fuels. Rural populations have high rates of household solid fuel use but relatively few inhabitants; conversely, cities have high population density but low rates of household solid fuel combustion for cooking. Using this methodology I find that, although variations in spatial resolution do probably contribute to (relatively minor) biases in my results, the merits of including urban/rural analyses by country are far outweighed by the substantial uncertainties introduced by multiple inconsistencies in the data sets used to perform analysis at the subnational (urban/rural) level. A summary of the sensitivity analysis follows.

2.5.1 Assumptions and new data sources needed

One of the reasons that I devised the methods featured in this manuscript is that I was able to integrate data sets (GAINS and TM5-FASST) that allowed me to report results in total ambient $PM_{2.5}$ (rather than primary $PM_{2.5}$) terms, at a global scale (rather than for select countries), using data sources that are complimentary and consistent with each other. Unfortunately, not all of my original data sources report emissions estimates or concentrations in urban/rural terms, so I must make assumptions to cobble together urban/rural analyses using other databases (described below). This results in a breakdown of internal consistency. (GAINS and TM5-FASST do not report urban/rural breakdowns.)

Because not all of the data sources used in my initial analysis are available at the sub-national (urban/rural) level, I needed to introduce other sources of data (from WHO, and new analysis from the Brauer et al. (2012) gridded data) that would allow me to answer my original research question. My approach for this sensitivity analysis is shown in the chart below. (See Figure 2.3.)

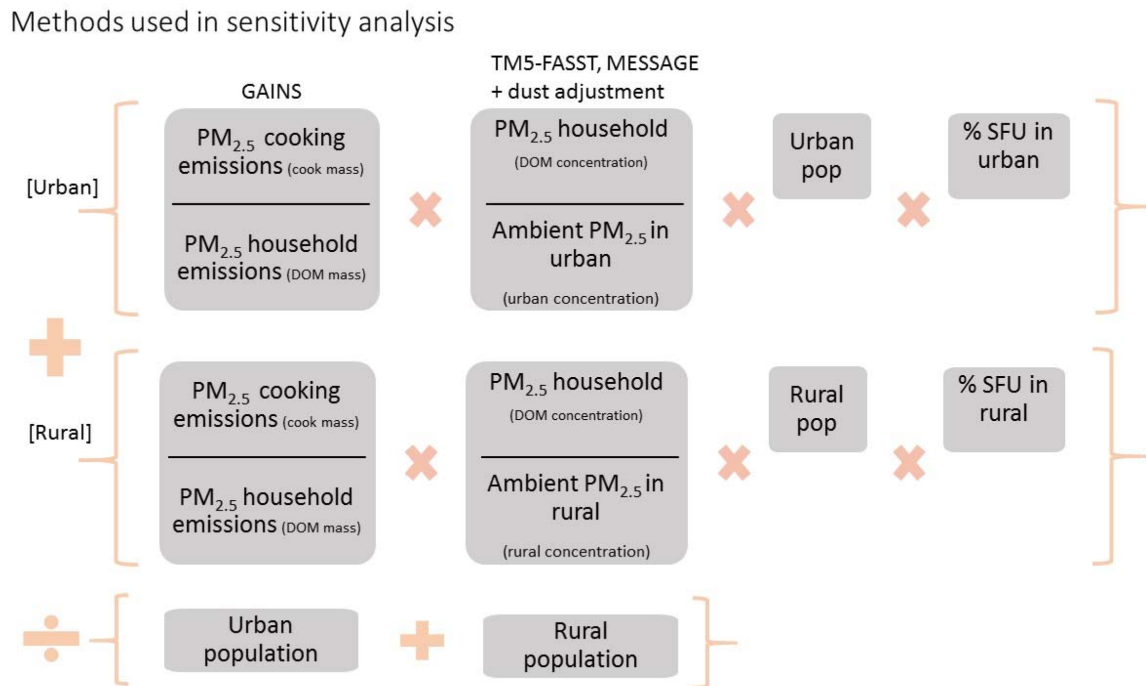

Figure 2.3. Methods used in sensitivity analysis.

2.5.2 Urban/rural population, fuel use, and $PM_{2.5}$ data

The urban and rural population data are derived from an air pollution concentration dataset used in the 2010 Global Burden of Disease Project and published in Brauer et al. (2012). The underlying population data in that data set are drawn from Global Urban Gridded Database (http://sedac.ciesin.columbia.edu/data/collection/grump-v1).

The proportions of solid fuel use occurring in urban and rural areas were derived from household energy use surveys that are described in a WHO data set (available at http://apps.who.int/gho/data/node.main.134?lang=en). Survey data were available for 108 countries. Survey data were *not* available for approximately 60 countries.

Ambient $PM_{2.5}$ in rural and urban areas was calculated from the Brauer et al. (2012) data set described above, which contains data for approximately 1.5 million gridded cells (0.1° x 0.1°) that are each designated as urban or rural, based upon the Global Urban Gridded Database (link above).

2.5.3 GBD vs TM-5 FASST

Importantly, in the second term of the equations used (concentrations from household sources divided by concentrations from all sources i.e. total ambient $PM_{2.5}$), the sensitivity analysis done here uses gridded results from GBD, which incorporate satellite data as well as TM5 results to estimate total ambient $PM_{2.5}$. The reason that I use GBD here is that the emissions were coded, by grid cell, as urban or rural, allowing me to do an urban/rural-scale analysis. In the paper, however, I use TM5-FASST results because the model allows users to isolate sector-specific emissions (including secondary particulate matter formed through atmospheric processes) and I have the benefit of keeping the numerator (sector-specific emissions) and denominator (estimate of total $PM_{2.5}$) internally consistent.

Using the data and methods outlined above, I ran several different sensitivity analyses at the global scale. In addition, I analyzed the five countries with the highest total population (China, India, US, Brazil and Pakistan) in depth; as well as one country in which most household solid fuel use is reported (by WHO) to be in rural areas (Egypt) and one in which solid fuel use is mostly in urban areas (Malaysia).

2.5.4 Sensitivity analysis results

Analysis of the six most populous countries shows that using urban/rural breakdowns results in estimates of $PM_{2.5}$ concentrations from household cooking ranging from 9 $\mu g/m^3$ higher (in China, and 5.2 $\mu g/m^3$ higher in India) to 0.2 $\mu g/m^3$ lower, by country, than the country-wide (not urban/rural) estimates used as inputs to the regional results reported in this paper. When all 108 countries for which household fuel use survey data were available are considered, the results from using urban/rural breakdowns are 1 $\mu g/m^3$ lower than the values from the manuscript (country-wide) analysis (3.9 $\mu g/m^3$ vs 4.9 $\mu g/m^3$).

I also compared these urban/rural analyses, at the country-level, to new country-wide (not urban/rural) estimates using the same modified data inputs used in these sensitivity analyses (and described above). For the 108 countries for which household fuel use survey data were available, the urban/rural estimates were 1.5 $\mu g/m^3$ lower than the modified country-wide results, which used data from the WHO and reanalysis of GBD inputs (3.9 $\mu g/m^3$ vs 5.4 $\mu g/m^3$). This indicates that the introduction of different data sources results in larger differences than does the incorporation of urban/rural differences into the analysis.

I also note that for the 63 countries (out of 170 in my original analysis) for which GAINS does not report any emissions from household cooking, this urban/rural analysis does not change results (0% $PM_{2.5}$ from cooking; 0 $\mu g/m^3$ from cooking). These countries represent 1.4 billion people.

2.5.5 Discussion of sensitivity analysis

Incorporating different data sources while doing analysis at the country level leads to greater uncertainty and differences in results, at the country-level. I believe that introducing this new uncertainty overwhelms any benefit achieved from addressing potential spatial misalignment between sources by performing urban/rural-level analysis.

Results calculated at the urban/rural level, for the 108 countries that have solid fuel use for household cooking, are lower than those calculated at the country-level. However, they are not lower by a consistent concentration ($\mu g/m^3$) or percent of $PM_{2.5}$. Also, the urban/rural results are closer to the (country-wide) results presented in the manuscript than to the country-wide results from this sensitivity analysis. In China and India, two countries with large populations and high proportions of households using solid fuels, the urban/rural analysis generates attributable μg/m^3 estimates that are higher than those used in the manuscript itself.

This leaves the challenge of interpreting these results. The urban/rural results are sometimes higher and sometimes lower, on a country-by-country basis, than those calculated for the paper, while the country-level results calculated as part of the sensitivity analysis are always higher than those reported in the paper.

I believe that this analysis represents the best possible treatment of the important urban/rural spatial misalignment issue, given the data currently available. The weaknesses in these analyses point to the pressing need for more research on source sector contributions to exposure and burden (rather than emissions or concentration) in general.

2.5.6 Sensitivity analysis conclusions

In sum, I decided against directly including the numbers generated in the sensitivity analysis in the general results section because of the following factors:

1) Lack of consistency in urban/rural data definitions: Most data sets that distinguish between urban and rural areas use (in some cases radically) different definitions of “urban” and “rural.” This became obvious when comparing urban and rural population numbers for a given country between data sets, for example. This is a problem in UN data sets as well, as urban and rural designations are defined at the country level rather than at the international level, and so are inconsistent between countries.

2) Potential lack of comparability across household solid fuel use surveys: Household solid fuel use is defined in different ways across countries and survey instruments, making it difficult to be sure that household survey results are comparable for the information I are deriving here (proportion of country-wide households using solid fuels who live in either urban or rural areas).

Because of the lack of direct comparability between this sensitivity analysis and the analysis presented in the results section above, and the disintegration of internal consistency introduced by the new data sources, I used them solely to inform my understanding of the potential bias introduced by spatial variation in data sets and its relationship to the additional biases introduced by adding variability around urban/rural definitions (or lack thereof).

2.6. Discussion

Although all household cooking contributes to AAP, either directly at the household level, through production and transport of fuel, or indirectly through the manufacture of cooking technologies, I estimated only particulate emissions from the combustion of solid fuels in the household. Kerosene, for example, creates BC and other particulate matter at the point of use (Lam et al., 2012b), and even electric cooking contributes indirectly to air pollution through emissions at power plants, but these emissions were not counted in the present analysis.

2.6.1 Assumptions made in the analysis

In addition, I made the following important assumptions in my analysis:

2.6.1.1 Isolating household cooking emissions

I assumed that household cooking emissions are correctly split from commercial cooking emissions, although I realize that there is often an overlap between these two categories. I also assumed that energy use and emissions databases (GAINS and MESSAGE), and their underlying data sources, correctly characterize the split between fuels used for household cooking and those used for household heating, though I realize that cooking and heating energy use may overlap. IIASA collaborates with partners in China, India, and Pakistan and uses published sources of information (local reports and peer reviewed research), as well as regional GAINS studies (Amann et al., 2008a; Purohit, 2010) to distinguish household fuel use for heating from that for cooking, especially in northern China. In a number of countries in Asia, GAINS allocates activities also at the subnational level, for example, provinces in China or India, where information from. The split between cooking and heating in Europe was developed using data from European Commission consultations under the Convention for Long Range Transboundary Air Pollution.

2.6.1.2 Escape fraction

I assumed that the particle escape fraction is 100%; that is, all particles generated by combustion inside a home or cooking structure are eventually incorporated into ambient air, and there is no significant mass loss due to particle deposition on indoor surfaces. Although little work has been done to characterize the fate of indoor combustion particles and their flow out of enclosed spaces, modeling estimates show that approximately 90% of fine particles are likely to reach the outdoor environment, a figure that probably rises to nearly 100% in houses with high air exchange rates (Lam et al., 2012a). In addition, many households cook outdoors for at least part of the year.

2.6.1.3 Atmospheric transformation

GAINS data are presented in units of mass of $PPM_{2.5}$. I assumed that all primary particulate household emissions contribute in the same way to total $PM_{2.5}$; that is, each gram of $PPM_{2.5\text{-cook}}$ will eventually create the same mass of $PM_{2.5}$ (after atmospheric interactions) as will any other gram of $PPM_{2.5\text{-hh}}$.

2.6.1.4 Atmospheric transport

I assumed that $PM_{2.5}$ concentrations attributed to household emissions result solely from particles emitted from households inside the country/region in question, without notable contribution (via atmospheric transport) from neighboring regions.

2.6.1.5 Spatial misalignment

I assumed that the proportion of ambient $PM_{2.5}$ attributable to $PM_{2.5\text{-cook}}$ is uniform across a given country. Although I recognize that there can be much local variation in the degree to which household fuels contribute to ambient $PM_{2.5}$, I made this assumption based on the spatial scale at which emissions are reported, which, in the case of this globally-consistent analysis, is at the country- or regional-level.

The analysis reported here was performed at the country-level (and is reported at the regional level). I were not able to systematically account for urban/rural differences in population density, household solid fuel use, or exposure to AAP within countries because of data limitations. I attempted to generate sensitivity analysis estimates at the urban/rural level (see following section titled "Sensitivity analysis on spatial resolution of data"), but inconsistencies among available international databases at this spatial scale introduced substantial unexplained variation. Currently, the definition of urban/rural areas is not consistent across countries or data sources. I concluded that the consequent loss of comparability, and difficulty of explaining the variations, obviated any improvement in estimated values that might have occurred in some countries.

2.6.2 Emissions estimates

This analysis used multiple emissions information sources with different system boundaries (see Appendices B, C, and D). The GAINS model provides estimates of $PPM_{2.5\text{-hh}}$; TM5-FASST provides estimates of $APM_{2.5}$ by source category, including primary combustion-derived emissions and secondary particulate formation. Neither model includes salt or dust emissions, though dust and sea salt were estimated by comparing combustion-derived $PM_{2.5}$ from TM5-FASST with $APM_{2.5}$ estimates developed for GBD 2010 (Brauer et al., 2012) and used in the burden estimates (Lim et al., 2012).

Insufficient input data made it challenging to conduct this analysis for some parts of the world, notably the eight sub-Saharan African and Latin American GBD regions. Regional assumptions about emissions patterns were made when country-level data were not available, and emission factors were often estimated within one country and applied to other countries when country-specific emissions data were not available.

Many countries, including India and China, lack the detailed national emission inventories that are available in the United States, Canada, and most European countries (Lei et al., 2011). Household cooking data remain scarce and relatively poor in quality, owing to the difficulties of measuring household fuel use in developing countries and emerging economies. From household survey questions that are too general to generate accurate projections, to emission factors that are sensitive to local meteorological or fuel conditions (such as wood moisture content), to poor data on emerging control strategies (such as advanced biomass cookstoves), the data used to create the results presented here have weaknesses. Furthermore, as noted above, the lack of urban and rural disaggregation of energy use and sectoral emissions data make it difficult to account for demographic trends that may influence exposure.

In addition to improving household energy use and emission estimates, there is a need to work toward more comprehensive data harmonization and sharing in this specific issue area. Major emissions inventories and models continue to use different household fuel use inputs (Fernandes et al., 2007; Klimont et al., 2009; Pachauri, 2011), so results are not directly comparable across models, although efforts to improve this issue are underway (Bonjour et al., 2013). This methodology represents a first attempt to generate globally-commensurate estimates of the contribution of household cooking to AAP but there is a need to improve upon this analysis as better data sources become available.

2.6.3 Uncertainty of emissions estimates and atmospheric chemistry models

Even when well-supported energy use information exists, there is a great deal of uncertainty associated with particulate emissions estimates, partly because emission factors vary with specific fuel type, fuel quality, and combustion conditions (UNEP, 2011). Household fuel use emissions estimates, especially from coal combustion, are more uncertain than estimates of emissions from other sectors, because of the range of combustion conditions and fuels used; one of the many reasons for this uncertainty is that laboratory experiments designed to understand household stove emissions often produce different results than those measured in the field (Jetter et al., 2012). Uncertainties around estimates of BC and OC emissions are notoriously high: in an analysis of the INTEX-B (Intercontinental Chemical Transport Experiment–Phase B) Asian emissions inventory, which used a similar modeling technique to the GAINS model used here, uncertainty around BC and OC emissions (± 208–364%, ± 258–450%) was found to be an order of magnitude greater than for some other air pollutants (SO_2, NO_x) (UNEP, 2011; Zhang et al., 2009). The uncertainty around undifferentiated $PM_{2.5}$ was somewhat smaller (± 130%) (Zhang et al., 2009).

Atmospheric chemistry transport models have their own uncertainties, related to chemistry, dispersion and removal of aerosol. For instance, intercomparisons of global models have shown that even when the same emission inventories were used, a large range of aerosol global properties were seen (Huneeus et al., 2011; Textor et al., 2006). However, the specific combination used in this analysis, of GAINS emissions and chemical transport model TM5, was tested and compared with a global dataset of $PM_{2.5}$ observations, as well as an independent study that combined MISR/MODIS (Multi-angle Imaging SpectroRadiometer/Moderate Resolution Imaging Spectroradiometer) satellite columns with assumed vertical aerosol distributions from the global GEOS-Chem (Goddard Earth Observing System) model (Brauer et al., 2012). Both studies showed a rather favorable comparison to outdoor $PM_{2.5}$ measurements, with relative errors in the order of ±10 % in the range of 10-200 $\mu g/m^3$.

Because I examined household emissions rather than human exposures, I probably underestimated the magnitude of associated health effects, for two reasons: First, household emissions vary seasonally (as do overall $PM_{2.5}$ emission levels and the specific composition of $PM_{2.5}$) and often peak in the winter in much of Asia and probably many other regions (Chowdhury et al., 2007; Stone et al., 2010). During the heating season, a particularly pronounced increase in mortality risk associated with exposure to secondary aerosols and combustion species has been documented in China (Huang et al., 2012). Second, household emissions probably have a higher average intake fraction than most sources of AAP, because people spend long hours in very close proximity to cooking and heating stoves; the intake fraction may, in urban areas, be on par with that of electric generators, construction equipment, and vehicles (Apte et al., 2011; Bennett et al., 2002; Health Effects Institute, 2010), though vehicles produce less primary $PM_{2.5}$ than households, in many countries, as noted below. In general, there is a pressing need for more research on sector-specific contributions to exposure and disease burden, rather than emissions or concentrations of air pollutants.

2.6.5 Technology and Policy Implications

Solid fuels are expected to remain an important source of energy for household cooking for decades to come (GEA, 2012; Pachauri, 2011). Although the demand for wood as a cooking fuel generally decreases with economic growth (Smith et al., 1994), and emissions can be partially controlled with the use of certain advanced cookstoves (Jetter et al., 2012), this decline may be offset by a trend toward smaller families, which tends to raise per capita solid fuel consumption (Knight and Rosa, 2011).

More than half of the world's population lives in areas where household cooking significantly affects air quality. My results indicate that it will be difficult to reduce ambient $PM_{2.5}$ to meet air quality standards unless household emissions are addressed, along with other sources (Balakrishnan et al., 2011). On-road cars, trucks, and other transport vehicles are more widely recognized as sources of AAP, compared with household cooking emissions, especially in industrialized countries (Bond et al., 2013; Bond et al., 2004; Kupiainen and Klimont, 2007; UNEP, 2011). However, direct $PM_{2.5}$ emissions associated with on-road transport are often much lower than the less well known and more dispersed problem of $PM_{2.5\text{-cook}}$, something that has been noted in other analyses as well (Lei et al., 2011); however, vehicles do contribute higher levels of other air pollutants, such as NO_x. Similarly, although not addressed here, in many temperate developed and developing countries, smoke from household heating with solid fuels is another consequential but generally overlooked and under-regulated problem (McGowan et al., 2002).

2.7. Conclusions

The combustion of solid fuels for household cooking is an important contributor to ambient fine particulate air pollution ($APM_{2.5}$) in many countries, accounting for > 10% of $APM_{2.5}$ pollution in seven regions housing > 50% of the global population in 2010. Regional proportions reach as high as 37% (sub-Saharan Africa), and the world as a whole, including many regions with no contribution from solid cooking fuel, averages about 12% of $APM_{2.5}$ from household cooking with coal, wood, and other solid fuels. Within countries, it can be expected that the proportion of $APM_{2.5}$ from household cooking is highest in rural areas where cooking with coal and biomass are most prevalent. The importance of this source of pollution extends to the regions with the two most populous countries (India in South Asia and China in East Asia) both with high ambient pollution levels; together these regions account for nearly 90% of the estimated global deaths from AAP that were attributed to household cooking with solid fuels. In terms of absolute concentrations, in two regions that face severe air pollution problems and are home to about 3 billion people, South Asia and East Asia, the estimated contribution of household cooking to $APM_{2.5}$ pollution ranged from 7 to 9 $\mu g/m^3$ in 2010.

AAP remains a significant health, environmental, and economic problem around the world. China, India, and many other countries with emerging economies, face daunting air pollution challenges. This problem is not confined to densely populated megacities, but is a feature of small cities and inter-urban areas as well (Brauer et al., 2012). My results indicate one important reason: the persistence of solid fuel use for cooking. Such fuels emit substantial amounts of

AAP, while being a risk in the household environment. Globally, more households use solid fuels for cooking today than at any time in human history, even as the fraction of the total population using solid fuels continues to slowly fall (Bonjour et al., 2013).
More collaboration and coordination will be needed between the household energy and general air pollution communities, both at the research and policy levels to deal with this issue.

Currently these communities act in essential isolation, as illustrated for example by the lack of ambient monitoring stations and reporting of pollution levels in rural areas in nearly all developing countries (Balakrishnan et al., 2011). In reality, both the household energy and air pollution communities have a stake in finding clean cooking fuels and clean cookstoves, which not only protect people in and around the households of the poor, who currently rely on polluting solid fuels, but also need to be part of national strategies to control ambient pollutions for the protection of all.

Chapter 3 : Ambient Air Pollution ($PM_{2.5}$), Premature Deaths, and Ill-Health from Household Heating with Solid Fuels in Europe and North America*

*Some material in this chapter was previously published by the World Health Organization in the following report:

Chafe, Z., Brauer, M., Heroux, M., Klimont, Z., Lanki, T., Salonen, R., Smith, K.R. (2015). Residential heating with wood and coal: health impacts and policy options in Europe and North America. World Health Organization. Bonn, Germany.

3.1 Abstract

Background: Incomplete combustion of solid fuels for household heating is a significant source of health-damaging fine particulate matter ($PM_{2.5}$), particularly in temperate areas during colder parts of the year. Recent work has shown that globally about 12% of population-weighted ambient $PM_{2.5}$ ($APM_{2.5}$) is due to household cooking with solid fuels; this proportion is much higher in some individual countries. No prior assessments of the contribution of household heating to $APM_{2.5}$ worldwide were identified.

Objectives: In this chapter, I estimate the proportion of $APM_{2.5}$ attributable to emissions from household combustion of wood and coal for space heating in Europe and North America. I also estimate the premature deaths and ill-health associated with population-wide exposure to $APM_{2.5}$ attributable to household combustion of solid fuels for space heating. Effects on ambient air pollution and human health are reported at the regional level, for Europe and North America (Canada and United States), from 1990-2010.

Methods: I use an energy supply-driven emissions model (GAINS) to calculate the fraction of total household $PM_{2.5}$ emissions attributable to heating with solid fuels, by country. I apply this fraction to global estimates of average ambient population-weighted $PM_{2.5}$ concentrations, calculated with source-receptor model TM5-FASST, to obtain the proportion of total $APM_{2.5}$ from household heating ($PM_{2.5\text{-heat}}$). Using data from the 2010 Global Burden of Disease Project (GBD-2010), I estimate the proportion of premature deaths and ill-health associated with population-wide exposure to $APM_{2.5}$ that is attributable to emissions from household combustion of solid fuels for space heating. My estimates do not include district heating systems.

Results: In 2010, the proportion of $APM_{2.5}$ from household solid fuel heating emissions was 21% in Central Europe, 13% in Eastern Europe, 12% in Western Europe, and 8% in North America. In Europe, the absolute concentration of $APM_{2.5}$ attributable to household heating with solid fuels was highest in Central Europe, where the contribution to absolute $APM_{2.5}$ (3.4 µg/m^3) was twice as large as in the next highest part of Europe (1.7 µg/m^3 in Western Europe). I estimate that, in 2010, population-wide exposure to $APM_{2.5}$ from household heating emissions caused approximately 60,000 premature deaths in Europe, and nearly 10,000 deaths in North America,

as well as an estimated 1.0 million disability-adjusted life years (DALYs) in Europe and 160,000 DALYs in North America.

Conclusions: Particulate air pollution from household combustion of wood and coal constitutes an important portion of $APM_{2.5}$ pollution in many regions, including in industrialized countries where most of the population has access to electricity, gas, and other heating fuels. Reducing emissions from household fuel combustion may have significant health advantages for populations beyond those using solid fuels themselves. Efforts to improve ambient air quality will be hindered if incomplete household combustion of solid fuels for residential space heating is not addressed. With 3-4 million deaths per year attributable to $APM_{2.5}$, a better understanding is needed of the contribution of specific sources, including household heating, to $APM_{2.5}$.

3.2. Introduction

Despite the availability of electricity and piped natural gas in many temperate regions of the world, including most of Europe and North America, household combustion of solid fuels (wood, coal, agricultural waste, and occasionally garbage) for residential space heating persists. This combustion creates outdoor air pollution that causes an important public health burden, both in terms of premature deaths and in healthy life-years lost, across many regions of the world.

Incomplete combustion of solid fuels for household heating is a significant source of health-damaging fine particulate matter ($PM_{2.5}$) and climate forcing BC, particularly in temperate areas during winter. With 3.2 million deaths per year attributable to ambient $PM_{2.5}$ in 2010 (Lim et al., 2012) and 3.7 million deaths in 2012 (World Health Organization, 2014a), including 482,000 deaths in Europe and 94,000 in Canada and the USA, a better understanding of the contribution of specific sources, including household heating, is needed.

Three main reasons for concern about the continued use of solid fuels for heating are that 1) the practice of combusting wood and coal for home space heating negatively impacts AAP and health; 2) there is evidence that the use of solid fuels for space heating is gaining popularity and may continue to increase in the near future; and 3) there is a possible trade-off between climate change mitigation and local/regional air quality concerns, an issue that is becoming increasingly important as climate change mitigation strategies are created and implemented worldwide. (See Chapter 6.)

In recent years, research has focused on the contribution to outdoor air pollution from household cooking with solid fuels (Chafe et al., 2014). (See Chapter 2.) This work has shown that globally about 12% of population-weighted ambient $PM_{2.5}$ (particulate matter with aerodynamic diameter ≤ 2.5 μm) pollution ($APM_{2.5}$) is due to household cooking with solid fuels; the proportion is much higher in some countries. Recent assessment of the ground transportation sector estimates that in 2010, about 184,000 premature deaths were attributable to $PM_{2.5}$ emissions from vehicles (World Bank Group and Institute for Health Metrics and Evaluation, 2014).

The ambient (outdoor) air pollution implications of household heating with solid fuels have not been well-explored. Though there are localized estimates of the contribution of household heating to ambient $PM_{2.5}$ (e.g. Ward and Lange (2010)), no systematic assessment has been made on a regional or global basis. Previous studies either overlook household heating as a contributor to ambient $PM_{2.5}$ (Leung, 2015), or note the need to estimate the AAP implications, and resulting health effects, of solid fuel combustion for household heating (Rogalsky et al., 2014).

In this chapter, I ask how much household heating with solid fuels (mainly wood and coal) contributes to outdoor air pollution, and explore the health implications of this pollution. I begin by summarizing the current household heating situation in several countries, presenting census results on the number of households using solid fuels for space heating. I then estimate 1) the $APM_{2.5}$, and 2) annual public health burden (premature deaths and lost healthy life years), associated with population-wide exposure to AAP that originates from the household combustion of solid fuels for space heating. Finally, I detail some policy options available to reduce the air pollution and health effects associated with household combustion of solid fuels for heating.

Throughout the analysis, the geographic focus is on North America and Europe. Seasonal space heating with wood is common in mountainous regions of many middle-income and low-income countries, such as Chile and Nepal, as well as in other industrialized countries such as Australia and New Zealand; and coal is used for space-heating in the parts of middle-income countries lying in temperate zones, such as Mongolia and China. However, due to data limitations, I restrict my analysis to Europe (Western Europe, Central Europe, and Eastern Europe) and North America (Canada and United States). While some energy use and air pollution models do include fuel use and emissions estimates for other parts of the world, data on heating fuel use (especially at the household level) is relatively incomplete, and results may be misleading when compared to regions with more complete fuel use information.

3.3. Background

Residential heating is an essential energy service required by billions of people worldwide (Isaac and van Vuuren, 2009). Even with widespread availability of electricity and natural gas, the use of solid fuels for residential heating continues to be a common practice in many places, including within European and North American countries (Ürge-Vorsatz et al., 2015). In the northern United States, about 10% of space heating in urbanized areas is accomplished with residential wood combustion, and up to 50% in some smaller towns (Larson and Koenig, 1994). Solid heating fuels consist primarily of wood and coal, but can also include forestry and agricultural residues and even garbage. Coal is more commonly used in Europe than in North America (Ürge-Vorsatz et al., 2015). Most fuels are burned in small-scale combustion devices, such as household heating stoves or small boilers for apartment buildings or district heating. Open fireplaces are popular in many parts of the developed world, but do not actually provide net heating in most circumstances; so they are often characterized as being for "recreational" or "aesthetic" use rather than for space-heating.

The majority of residential stoves and boilers used today are relatively inefficient, compared to the best models available. Under ideal burning conditions, all carbon in wood and other fuels (biomass, coal, etc.) would be completely converted to carbon dioxide (CO_2) while releasing energy. This is known as 100% combustion efficiency. Unfortunately, combustion efficiency of simple household stoves burning solid fuels is generally much lower than 100% (World Health Organization, 2014b). This results in relatively high emissions per unit fuel including many products of incomplete combustion such as fine particulate matter ($PM_{2.5}$) and carbon monoxide (CO), two major air pollutants. Small-scale solid fuel combustion is also an important source of BC emissions. BC is a component of $PM_{2.5}$ that warms the climate. When coal is used for residential heating, it can also result in emissions of sulfur and other toxic contaminants found in some types of coal; even with good combustion, these contaminants are not destroyed.

The less-than-ideal combustion conditions in most household fireplaces and stoves – including low combustion temperatures, suboptimal air circulation/oxygen availability, overloading of the firebox with wood, moist biomass fuel and heat loss – cause emissions of harmful particulate and gaseous compounds, together often referred to as "products of incomplete combustion." Unlike coal, wood contains few intrinsic contaminants that are emitted as air pollution during combustion, such as sulfur and ash. An exception is nitrogen in wood, which along with nitrogen fixed from the air during combustion, is emitted as NO_x.

The amount of heating fuel needed in a particular climate is a function of the fuel efficiency of the stove, as well as the characteristics of the housing (insulation infiltration, etc.), an issue I do not address further. In developed countries, nearly all space-heating devices have chimneys; in some developing countries, much space heating is done with open stoves inside the house. In both cases, most of the emissions end up in the atmosphere and contribute to outdoor air pollution.

3.3.1 Outdoor air pollution implications of household heating

Residential solid fuel combustion for heating is a major source of fine particulate air pollution ($PM_{2.5}$) in Europe and North America, generating an estimated 142 kilotonnes of $PM_{2.5}$ per year in Europe (11% of 1236 kilotonnes primary $PM_{2.5}$), 174 kilotonnes in the United States (4% of 4452 kilotonnes),[1] and 160 kilotonnes in Canada (9% of 1800 kilotonnes)[2], according to the most recent estimates available (Environment and Climate Change Canada, 2015; European Commission, 2015; European Environment Agency, 2015; US Environmental Protection Agency, 2013b, 2015a). This is equivalent to 0.28 kg per capita per year in Europe, 0.55 kg per capita per year in the United States, and 4.5 kg per capita per year in Canada.

In areas where wood combustion for residential heating is prevalent, previous studies have found relatively high short-term concentrations of $PM_{2.5}$, PM with an aerodynamic diameter of less than 10 micrometers (PM_{10}) and VOCs. In some places, wood combustion is the major source of ambient $PM_{2.5}$, especially during the heating season.

Source apportionment studies, which identify the emission source categories contributing to measured particulate air pollution levels, generally indicate that wood combustion accounts for

---

[1] Converted to metric tonnes from original values of 192 and 4908 1745 (short) tons. Does not include wildfire emissions. If "miscellaneous" emissions, which include prescribed burns and composting-related emissions, are not included, the proportion from residential wood burning rises to 11%.

[2] 1800 kilotonne $PM_{2.5}$ total includes estimates of open sources such as agriculture, waste treatment and prescribed burning. Without including emissions from these sources, residential solid fuel combustion accounts for 160 kilotonnes out of 300 kilotonnes, or 53%.

20–30% of local heating-season ambient $PM_{2.5}$ levels in areas where solid fuel is combusted for household heating, although this estimate varies greatly by location.

In the following sections, I summarize the results of previous studies that characterized the contribution of household solid fuel combustion emissions to ambient particulate air pollution in North America and Europe.

3.3.1.1 North America

Source apportionment studies from the US EPA show that residential wood combustion is a major source of ambient $PM_{2.5}$ in several parts of the country, accounting for more than 5 μg/m^3 annual mean $PM_{2.5}$ in California's San Francisco Bay Area and Central Valley, in the Pacific Northwest, and in parts of Colorado (see Figure 3.1). It is also a significant source of ambient particulate air pollution in the Northeast.

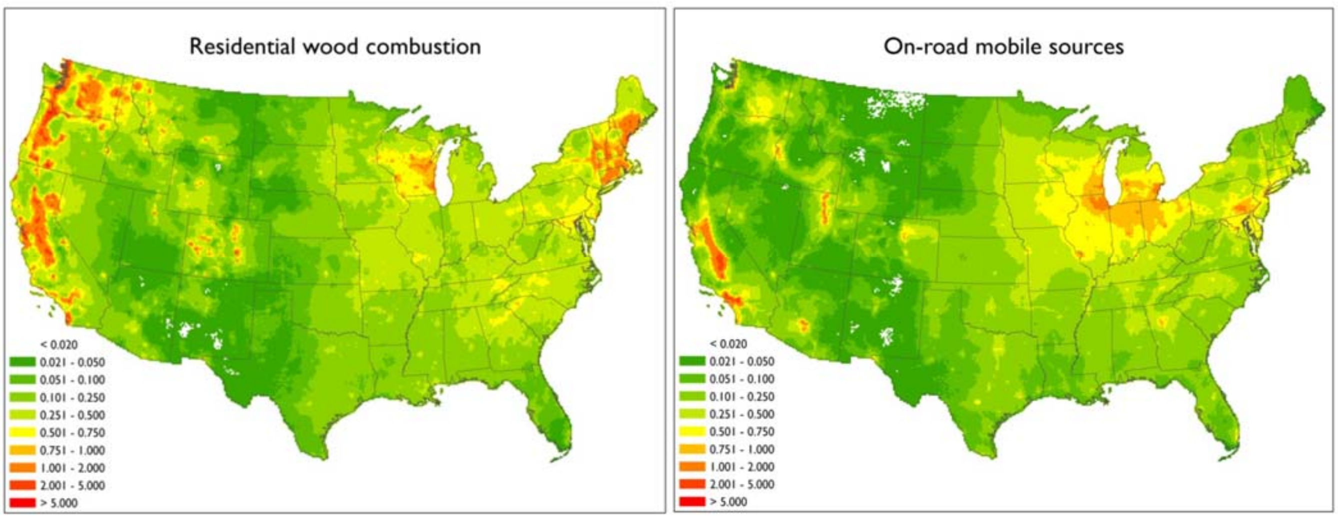

Figure 3.1. Modeled annual mean primary $PM_{2.5}$ levels attributable to residential wood combustion and on-road mobile sources in 2016 (US Environmental Protection Agency, 2013b).

Analysis of state-collected data shows that wood space heating was responsible for 7% of California's annual average $PM_{2.5}$ (50 tonnes per day), including wildfires (or 14% if wildfires are excluded) and 25% of the state's projected winter-time average $PM_{2.5}$ emissions (89 tonnes per day) (excluding wildfires) in 2015 (California Air Resources Board, 2013). (See Chapter 4 for more on California.) In Seattle, one study found that wood combustion is responsible for 60% of $PM_{2.5}$ in summer and 90% in winter months (Larson and Koenig, 1994). The same investigators later found that 62% of $PM_{2.5}$ in neighborhood measurement sites was from wood burning (Larson et al., 2004). Another study found that 31% of $PM_{2.5}$ measured at an outdoor monitoring site close to residential areas in Seattle was apportioned to wood combustion and other vegetative burning, higher than in the sites located closer to downtown (Kim and Hopke, 2008). There was awareness of the need to better understand emissions from wood heating as early as several decades ago, as evidenced by a 1982 study that found that 20-30% of wintertime $PM_{2.5}$ in Denver, Colorado was attributable to residential wood burning (Dasch, 1982).

3.3.1.2 Europe

A study in Italy found that in 2008 residential heating with wood caused 3% of PM_{10} in Milan, 18–76% in seven other urban areas and 40–85% in three rural areas (Gianelle et al., 2013). In the Helsinki Metropolitan Area, Finland, the contribution of wood heating to $PM_{2.5}$ emissions for the six-month cold season in 2005–2009 was 19–28% at urban and 31–66% at suburban monitoring sites (Saarnio et al., 2012). In Austria during the winter months of 2004 wood smoke caused about 10% of PM_{10} near Vienna and around 20% at rural sites in two densely forested regions (Salzburg and Styria) (Caseiro et al., 2009).

3.3.2 Trends in household combustion of solid fuels for space heating

In many countries, there is currently a general upward trend in the number of households using biomass as a fuel for space heating. In the US, the number of households (especially low- and middle-income households) heating with wood grew 34% between 2000 and 2010 – faster than any other heating fuel – and in two states the number of households heating with wood more than doubled during this period (Alliance for Green Heat, 2011; Fuller et al., 2013).

Reasons for increases in the number of households using solid fuels for space heating include:

- Household economics: Both the increasing costs of other energy sources, such as heating oil, and general economic hardship have the potential to increase dependence on solid fuels for household space heating. In response to economic hardship, some families revert to heating with solid fuels (such as discarded furniture, wood scrap and coal); this has happened recently in Greece and other European countries, such as Portugal, and has been documented in a small number of academic studies. For example, Saffari et al. (2013) found evidence of increased biomass burning and decreased heating oil use during the heating season in Greece during the economic downturn of 2013, and Gaidajis (2014) found evidence of increased particulate matter pollution from biomass heating in two Greek cities in 2013-2014.
- Government subsidies: Incentives and subsidies that encourage more use of biomass fuels or solid fuel stoves may increase emissions from this sector, as will the *lack* of subsidies to encourage exchange of existing inefficient stoves and boilers.
- Biomass as a “green” option: public perception that biomass is a “green” energy option, especially for household space heating, may increase household combustion, as will some climate change policies that consider biomass to be a renewable fuel. Note, however, that $PM_{2.5}$ emissions from household combustion of solid fuels include BC, which is a potent climate-warming substance. The net warming impact of BC-emitting sources depends on the concurrent emissions of cooling aerosols, such as OC (Bond et al., 2013). (See Chapter 6.)

The points above detail some causes of increases in the number of households combusting solid fuels for heating. In addition, the *proportion* of ambient $PM_{2.5}$ from household solid fuel combustion is rising in many areas, especially where emissions from other sources (such as ground transportation, industry and power plants) are already controlled or legislation is in place to reduce them; or where residential biomass combustion is expected to gain prominence as a source of $PM_{2.5}$, especially if no efforts are made to encourage (or incentivize) use of modern and efficient residential wood heating devices.

It should be noted, however, that the US EPA found that the absolute mass emissions of primary $PM_{2.5}$ attributable to residential wood combustion in the United States were expected to decrease from 201 kilotonnes in 2005 to 174 kilotonnes $PM_{2.5}$ in 2016 (US Environmental Protection Agency, 2011). Neither the assumptions made in the modeling exercise, related to the residential wood burning sector, nor the reasons for the downturn in emissions, were explained.

3.4. Methods

The analysis presented here uses much of the methodology outlined in Chapter 2, with the modification that emphasis is on household energy use and emissions associated with household combustion of solid fuels for space heating, rather than for cooking. General principles of the methodology are presented below, but for further detail, readers are encouraged to reference Chapter 2.

3.4.1 Data sources

To determine the effects of residential combustion of solid fuels for space heating on outdoor air pollution and public health, I used the following data sources: energy use and emissions estimates from the Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) model hosted by IIASA, secondary PM formation calculated with TM5-FASST software at the EC Joint Research Centre, and health impact data from the 2010 Global Burden of Disease (GBD-2010) Study (Amann et al., 2011; Centre, 2014; IIASA, 2014; Lim et al., 2012). Please see Chapter 2 for detailed information on the data used to determine $PM_{2.5}$ from household solid fuel use.

3.4.2 Definitions and system boundaries

I use the term "household heating with solid fuels" to refer to households that combust solid fuels, usually biomass or coal, in wood stoves, pellet stoves, or fireplaces (though fireplaces are actually not effective space heating technologies, since they often result in a net loss of heat from the building in which they are located). Importantly, I exclude district heating, in which solid fuels are combusted at a centralized location to provide heat (usually via production and transport of steam) to multiple households, usually apartment blocks or neighborhoods. District heating facilities often maintain combustion conditions that lead to less air pollution than household space-heating combustion technologies, such as wood stoves (Bowyer, 2012).

3.4.3 Calculating the proportion of ambient $PM_{2.5}$ attributable to residential solid fuel combustion for space heating

I estimate the proportion of ambient $PM_{2.5}$ attributable to residential wood and coal heating emissions in 1990-2010 by country. With these results, I then estimate ambient concentrations of $PM_{2.5}$ attributable to household heating with solid fuels ($PM_{2.5\text{-heat}}$). I use an energy supply-driven emissions model (GAINS) to calculate the fraction of household $PM_{2.5}$ emissions from heating with solid fuels, by country. I apply this fraction to global estimates of average ambient population-weighted $PM_{2.5}$ ($APM_{2.5}$) concentrations, calculated with source-receptor model TM5-FASST, to obtain the proportion of total $APM_{2.5}$ from $PM_{2.5\text{-heat}}$.

For additional information on this methodology, please see equations 1 and 2 in Chapter 2. This chapter follows the same methodology that was used for calculating the proportion of ambient $PM_{2.5}$ from cooking, but examines emissions from heating instead of cooking.

3.4.4 Population-weighting

The regional estimates of the proportion of $APM_{2.5}$ attributable to household space heating with solid fuels are population-weighted using Equation 1.

$$\frac{\sum_{i=1}^{\substack{countries\\ in\ region}}\left[\frac{PPM2.5\ from\ heating_i}{PPM2.5-hh_i}\times\frac{PM2.5-hh_i}{Total\ APM2.5_i}\times Population_i\right]}{\sum_{i=1}^{\substack{countries\\ in\ region}} Population_i} = PM2.5 - heat_{region} \qquad [1]$$

In addition, the $PM_{2.5}$ concentrations used to calculate $PM_{2.5\text{-heat}}$ are population-weighted, using a methodology described in Chapter 2 and in Brauer et al. (2012). As compared to simple average $PM_{2.5}$ indicators, using population-weighted $APM_{2.5}$ has the effect of better representing the effects of exposure to AAP among high density populations, when pollution sources exist close to those populations, and making estimates more conservative when the reverse is true (majority of population lives in areas with lower than average concentrations of $PM_{2.5}$).

3.4.5 Calculation of health effects of population-wide exposure to AAP attributable to household combustion of solid fuels for space heating

To estimate the burden of disease associated with population-wide exposure to $APM_{2.5}$ attributable to household combustion of solid fuels for space heating, I combined my estimates of the proportion of $PM_{2.5\text{-heat}}$ with burden of disease data from GBD-2010. Working at the regional level, with regions defined by GBD-2010, I apply $PM_{2.5\text{-heat}}$ (as a proportion) to regional estimates of premature deaths and ill-health (expressed in DALYs) caused by AAP.

Health impacts are therefore determined by scaling the total impacts from outdoor air pollution, based on the proportion of total air pollution attributable to residential solid fuel combustion for heating. This procedure is in line with the approach taken by the Global Energy Assessment (Riahi et al., 2012) and a World Bank report on the burden of disease from road transportation (World Bank Group and Institute for Health Metrics and Evaluation, 2014).

3.5. Results

3.5.1 $PM_{2.5}$ emissions from household heating

My analysis of the energy use data reported in the GAINS model reveals that the residential sector as a whole, which includes household combustion of fuels for cooking and space heating, causes about 40% of global anthropogenic primary $PM_{2.5}$ emissions. The fraction of household emissions that are associated with household space heating are a subsection of the total household emissions: I calculate that less than 10% of total $APM_{2.5}$ comes from residential heating stoves and small boilers, which warm water to distribute heat; about half of that comes from biomass heating, while most of the rest comes from household coal burning for heating. These figures do not include the $PM_{2.5}$ attributable to district heating facilities. Total $APM_{2.5}$ includes anthropogenic emissions, secondary $PM_{2.5}$ formed when $PM_{2.5}$ precursors undergo chemical transformation to produce additional $PM_{2.5}$, and natural sources of $PM_{2.5}$ such as dust and sea salt. In several specific regions of the world, however, residential combustion of solid fuels (biomass and coal) for heating makes a substantial contribution to $APM_{2.5}$, as detailed in Section 4.2, and results in a substantial burden of disease, as detailed in Section 4.3.

3.5.2 Air pollution

3.5.2.1 Europe

Globally, Europe has the highest proportion of outdoor $PM_{2.5}$ emissions attributable to household heating with solid fuels at 12% of total $PM_{2.5}$ in Western Europe, 21% in Central Europe and 13% in Eastern Europe in 2010. (See Table 3.1.) This corresponds to average population-weighted $PM_{2.5}$ concentrations of 1.7, 3.4 and 1.4 μg/m^3, respectively.

The proportion of $APM_{2.5}$ attributable to household heating with solid fuels increased in all three regions of Europe, between 1990 and 2010, notably more than doubling from 5.4% to 12% in Western Europe during that time period. (See Table 3.1.) However, Western Europe was the only one of the three regions that saw an absolute increase in $PM_{2.5\text{-heat}}$ concentrations (1.3 μg/m^3 in 1990 and 1.7 μg/m^3 in 2010). In both Central Europe and Eastern Europe, $PM_{2.5\text{-heat}}$ concentrations fell, by 0.1 μg/m^3 in Central Europe and (to 3.4 μg/m^3 in 2010) and by 0.6 μg/m^3 in Eastern Europe (to 1.4 μg/m^3 in 2010). These changes in the particulate air pollution from household space heating occurred at the same time as a general decrease in total $APM_{2.5}$ throughout Europe between 1990 and 2010. Annual average population-weighted $APM_{2.5}$ was cut nearly in half across the continent, from 31 μg/m^3 in 1990 to 16 μg/m^3 in 2010 in Central Europe, from 19 μg/m^3 in 1990 to 10 μg/m^3 in 2010 in Eastern Europe, and from 25 μg/m^3 to 15 μg/m^3 in Western Europe.

3.5.2.2 North America

In North America, defined here as Canada and the United States, in 2010, 8% of the region's $APM_{2.5}$ could be attributed to household heating with solid fuels for space heating. Regionally, this corresponds to a 1.1 μg/m^3 increment of annual average population-weighted $APM_{2.5}$. This 2010 result represents an increase over the corresponding figure for 1990 in terms of both proportion of $APM_{2.5}$ from heating (increase from 4.6% in 1990 to 8.3% in 2010) and in absolute concentration attributable to heating (0.9 μg/m^3 in 1990 and 1.1 μg/m^3 in 2010) and may reflect the growing popularity of residential space heating with biomass (see Section 5). Over the same time period, overall annual population-weighted $APM_{2.5}$ decreased in North America from 18 μg/m^3 in 1990 to 13 μg/m^3. Nearly all of the solid fuel use for heating in North America is biomass-based, rather than coal.

Table 3.1. Population-weighted contribution of space heating with solid fuels to ambient particulate matter pollution ($PM_{2.5\text{-heat}}$) by region.

	PM2.5-heat (%)			PM2.5-heat (μg/m3)			APM2.5		
GBD 2010 Region	1990	2005	2010	1990	2005	2010	1990	2005	2010
Central Europe	11.0	29.0	21.0	3.5	5.6	3.4	31	19	16
Eastern Europe	9.6	20.0	13.0	2.0	2.2	1.4	19	10	10
Western Europe	5.4	13.0	12.0	1.3	2.2	1.7	25	17	15
High-income North America	4.6	9.4	8.3	0.9	1.3	1.1	18	13	13

Table notes: Regional groupings, defined by IHME for the Global Burden of Disease 2010 project (Institute for Health Metrics and Evaluation, 2010b), are described in Appendix B. $PM_{2.5\text{-heat}}$ is the percent of population-weighted annual average ambient $PM_{2.5}$ attributable to household space heating and concentration of total population-weighted annual average ambient $PM_{2.5}$ (μg/m^3**)** attributable to household space heating.

3.5.3 Health implications

3.5.3.1 Europe

In 2010, an estimated 61,000 premature deaths in Europe were attributable to outdoor $PM_{2.5}$ pollution originating from residential heating with solid fuels, about the same number as in 1990. The number of attributable premature deaths rose in both Central and Western Europe between 1990 and 2010, by about 1,000 and 2,000 deaths respectively, or 7% in Central Europe and 14% in Western Europe. Over the same time period, premature deaths fell by about 3,000, or 14%, in Eastern Europe.

Outdoor air pollution from household heating with solid fuels also is estimated to be responsible for 1 million DALYs across Europe in 2010, a decrease of 7% from 1.1 million DALYs in 1990. As with premature deaths, there was a slight increase in attributable DALYs in Western Europe over this time period (3%), a slight decrease in Central Europe (9%), and a more sizeable decrease in Eastern Europe (17%).

Table 3.2. Estimated burden of disease from exposure to ambient $PM_{2.5}$ attributable to household cooking with solid fuels.

	Deaths			DALYs		
GBD 2010 Region	1990	2005	2010	1990	2005	2010
Eastern Europe	24,000	38,000	21,000	480,000	800,000	410,000
Central Europe	18,000	32,000	20,000	370,000	590,000	340,000
Western Europe	17,000	25,000	20,000	280,000	380,000	290,000
High-income North America	7,500	10,000	9,200	140,000	190,000	160,000

[a]Regional groupings, defined by IHME for the GBD 2010 project (Institute for Health Metrics and Evaluation, 2010b), are described in Appendix B.

3.5.3.2 North America
In North America, I estimate that exposure to outdoor $PM_{2.5}$ pollution from residential heating with solid fuels resulted in 9,200 deaths in 2010, an increase of 18% from 7,500 in 1990. This pollution also caused 160,000 DALYs in 2010, up slightly (13%) from 140,000 in 1990.

3.5.3.3 Other regions
Due to data limitations, I cannot make comparable estimates of the burden of disease associated with exposure to $PM_{2.5\text{-heat}}$ in other regions. However, preliminary work to estimate the emissions associated with household solid fuel use for heating indicate that exposure to $PM_{2.5\text{-heat}}$ was associated with at least 19,000 premature deaths and 390,000 DALYs in East Asia in 2010, and at least 10,000 premature deaths and 310,000 DALYs in South Asia in 2010. I believe that these calculations underestimate the emissions associated with household solid fuel use for space heating, as well as the associated burden of disease, based on more recent publications that carefully estimate household fuel use in China (Duan et al., 2014; Muye et al., 2015) and India (Ahmad et al., 2015; Ahmad and Puppim de Oliveira, 2015).

3.6. Discussion

3.6.1 Geographic data considerations

This analysis focuses on household heating emissions in Europe and North America, mainly because of data constraints. GAINS, one of the primary emission models used in this analysis, was originally designed to inform the development of air pollution and climate change policy in Europe (Amann et al., 2011). The energy use and emissions data information is most detailed in Europe for this reason. Some data is available for North America (US and Canada) but it is arguably not as detailed as for Europe. Efforts are underway to better understand household heating emissions in East Asia, South Asia, and Latin America, which are all regions with significant space heating needs. Data for much of Africa and the Middle East remains limited, but due to climate, there is less demand for space heating in those regions.

Health benefits of reducing exposure to AAP will also vary significantly by country due to background health and pollution conditions. Here I present results at the regional level because country level health impact data are not yet available for this risk factor. In future analyses, it may be possible to report country-specific and diseases-specific burdens associated with exposure to AAP. Although health impacts are presented by region here, the health benefits of reducing exposure to outdoor air pollution vary significantly by country as a result of background health and pollution conditions.

3.6.2 Rural versus urban data considerations

Because the modeled emission data used in this analysis are reported at the country level, there was a concern that this level of geographic analysis introduced a bias by ignoring differences in fuel use patterns between rural and urban areas (given that populations are not equally distributed between urban and rural areas). A sensitivity analysis for a related paper (see Chapter 2) concluded that sufficiently consistent data does not yet exist to fully analyze potential spatial bias. This is an area that needs further exploration, especially from the emissions inventory side.

3.6.3 Intake fraction

Indoor residential wood combustion sources are closer to the exposed population than most other outdoor combustion sources, except perhaps motor vehicles (Apte et al., 2011); as a result, the intake fraction (fraction of emitted particles that come into contact with exposed population) is higher. The composition of particles emitted by residential wood combustion sources and taken in by the exposed population is different than that emitted by other sources, because of the shorter mixing time for atmospheric reactions (between emission and human contact). Exactly how these factors modify exposure and subsequent health effects is unclear.

3.6.4 Secondary particulate matter formation assumptions

This analysis relies on the combination of two models (GAINS and TM5-FASST) to estimate the total $PM_{2.5}$ (primary and secondary particles) attributable to household space heating with solid fuels. The GAINS model estimates only primary emissions, but splits household emissions into cooking and heating emissions. The TM5-FASST model estimates total (primary and secondary) $PM_{2.5}$ from all household sources. When combining these two model outputs, we make the assumption that each mass unit of primary particulate matter from household solid fuel combustion (e.g. household cooking and heating emissions) contributes to formation of the secondary $PM_{2.5}$ in the same way and with the same resulting mass of $PM_{2.5}$. This assumption seems justified because households often use similar fuels (biomass, coal, etc.) for both heating and cooking tasks, and the combustion conditions are often (though not always) similar.

3.6.5 Non-linear relative risk considerations

Recent analysis of particle emission exposure across widely varying $PM_{2.5}$ concentrations (from AAP, household air pollution, and secondhand smoke) resulted in development of an integrated exposure-response (IER) function that is non-linear (Burnett et al., 2014). This non-linear function implies that the health implications (reduction of ill-health) associated with a given reduction of $PM_{2.5}$ exposure will vary depending on the given $PM_{2.5}$ concentrations. The IER indicates that more drastic health changes are associated with a given change in concentrations at lower levels of pollution than at high levels of pollution (see Figure 3.2).

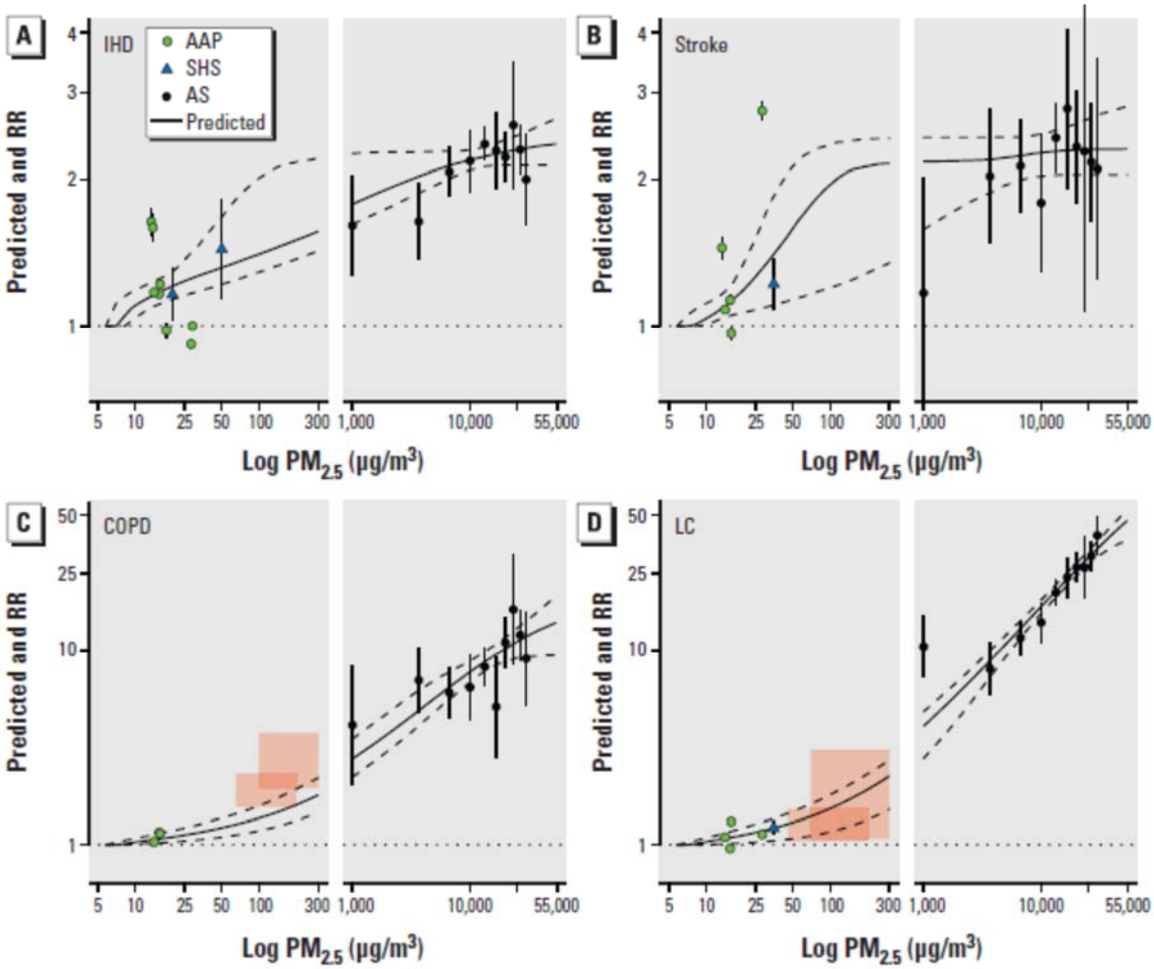

Figure 3.2. Examples of modeled integrated exposure-response functions for various health outcomes associated with exposure to $PM_{2.5}$ (ischemic heart disease, stroke, chronic obstructive pulmonary disease, lung cancer) (Burnett et al. 2014)

In this analysis, I begin with the total health burden attributable to exposure to ambient $PM_{2.5}$, as calculated for GBD-2010. This health burden was calculated using IERs, which use a non-linear curve to describe effects of ambient $PM_{2.5}$ on human health across a wide range of observed average ambient $PM_{2.5}$ levels. I then directly apply the proportion of emissions due to household heating to the total health burden from outdoor particle pollution, as calculated in the GBD-2010, by region. The result is a scaled approximation of the relationship between $PM_{2.5}$ concentration reductions and associated health effects. This health analysis procedure is in line with the approach taken by the Global Energy Assessment (Riahi et al., 2012).

This linear approximation for reductions was also used in the preparation of a publication prepared for the World Bank (World Bank Group and Institute for Health Metrics and Evaluation, 2014). The authors of that report compared three plausible approaches to estimating the health effects of a specific $PM_{2.5}$ source category (transportation) and found that the proportional approach produce higher health effects estimates (attributable mortality and

DALYs) than reducing emissions from the top of an IER curve, but lower estimates than an averaged IER approach (which assumes that we do not know exactly where on the curve the health effects will fall, within a given region, and therefore uses an average on the IER curve).

In reality, AAP concentrations vary greatly by region and reducing exposure to AAP at higher and lower concentrations will likely produce differing health benefits due to the non-linear character of the exposure-response relationship for many diseases (Burnett et al., 2014). Future studies may better characterize the nuances in health effects along various parts of the AAP concentration spectrum.

In cases where supralinear exposure-response relationships are observed, such as deaths from exposure to $PM_{2.5}$ at relatively low concentrations, the health effects of emission reductions in one sector will be dependent on emissions levels from other sectors as well (Marshall et al., 2015). Researchers suggest that, going forward, analyses may need to distinguish between studies that attempt to attribute health outcomes to emissions changes in a specific sector, and studies that estimate health outcomes associated with a policy change (because emissions effects of a given policy change will in turn change the relative impact of future policy changes) (Unosson et al., 2013).

This nascent discussion of non-linearity in $PM_{2.5}$ exposure-response curves implies that efforts to reduce emissions associated with household solid fuel combustion, in areas that have relatively clean ambient air, such as much of Europe and North America, may have even larger health benefits than originally expected, something that is explored by Marshall et al. in a recent publication (2015).

3.6.6 Health effects of differentiated PM

An important consideration is to what extent results from epidemiological studies on urban PM can be generalized to PM from residential wood combustion. In the WHO air quality guidelines (World Health Organization, 2006) it was concluded that there was little evidence that the toxicity of particles from biomass combustion would differ from the toxicity of more widely studied urban PM. This same approach was followed in this analysis and in the recent GBD Study (Lim et al., 2012), in which all combustion particles, regardless of source, were considered to be hazardous depending on the exposure level. This was based on the integrated exposure-response curves developed for the GBD Study, which linked exposures to combustion particles across four sources – AAP, secondhand tobacco smoke, household air pollution and active smoking – to the following health outcomes: ischemic heart disease, stroke, chronic obstructive pulmonary disease, lung cancer and child pneumonia (Burnett et al., 2014).

3.7. Conclusions

Household combustion of solid fuels, mainly wood and coal, is a significant source of outdoor air pollution in most regions where household heating is needed. This analysis shows that household space heating with biomass-based solid fuels (wood, charcoal, crop residues etc.) is creating outdoor air pollution that results in an important public health burden, both in terms of premature deaths and in terms of healthy life years lost, across many regions of the world. Reducing the use of biomass for space heating or reducing the emissions though better combustion, or pollution capture, would lessen this burden.

Technology and policy options are available to mitigate the pollution from household heating and prevent deaths that result from exposure to this pollution. Reducing the use of biomass for space heating or reducing emissions through better combustion or pollution capture would lessen the health burden. Given non-linearity of dose-response curves, there is reason to believe that health effects of reducing emissions from household heating with wood and coal are greater than what is presented here.

The results presented here indicate that it will be difficult to tackle outdoor air pollution problems in many parts of the world without addressing the combustion of biomass for heating at the household level along with other sources of air pollution (Ward and Lange, 2010). To protect health, policy-makers in regions that have relatively high levels of outdoor air pollution from household heating-related combustion need to provide incentives to switch from solid fuel combustion for heating to gas- or electricity-based heating.

Given that residential wood combustion for heating will continue in many parts of the world because of economic considerations and availability of other fuels, an urgent need exists to develop and promote the use of the lowest emission or best available combustion technologies. There is also a need for renewable energy or climate change-related policies that support combustion of wood for residential heating to consider the local and global AAP impacts and immediately promote only the use of lowest emission or best available combustion technologies.

Policy-makers in regions where the proportion of $PM_{2.5}$ emissions attributable to household space heating with biomass-based fuels is high might wish to consider incentives to assist with a transition to more efficient technologies that encourage more complete combustion, and thus reduce $PM_{2.5}$ and other health-relevant emissions. It may be preferable in many cases to focus on making biomass-based home heating more efficient and less polluting rather than transitioning away from biomass to fossil fuels, given the climate change implications of using fossil fuel for heating. A better understanding of the role of wood biomass heating as a major source of globally harmful outdoor air pollutants (especially fine particles) is needed among national, regional and local administrations, politicians and the public at large.

Chapter 4 : Residential Wood-Burning in the San Francisco Bay Area: Health and Economic Effects of Fuel- and Device-Switching Scenarios

4.1 Abstract

Background: Residential wood combustion is a leading wintertime source of fine particulate ($PM_{2.5}$) air pollution in the San Francisco Bay Area. The Bay Area Air Quality Management District (BAAQMD) has recently updated regulations designed to minimize $PM_{2.5}$ emissions from residential combustion for space heating and aesthetic purposes.

Objectives: I create a model to estimate the ambient air pollution, human health, and economic effects of five hypothetical interventions designed to reduce residential wood combustion emissions of $PM_{2.5}$ in the San Francisco Bay Area.

Methods: I model mass emissions of $PM_{2.5}$ from San Francisco Bay Area residential wood combustion using data on wood heating devices and use in the San Francisco Bay Area from surveys done for BAAQMD's Winter Spare the Air Day program, and emission rates and factors published in the literature and by the US EPA. I estimate mass emissions of $PM_{2.5}$ by county, device type, and fuel type for 2012. I assess the air pollution, health, and economic benefits of five hypothetical device and fuel modification scenarios, using a methodology published by the US EPA. I compare results with BAAQMD's criteria air pollutant emission inventory and CARB's emission inventory.

Results: Frequent fireplace users (those that use their fireplace at least once per week) were responsible for 84% of total $PM_{2.5}$ mass emissions from the residential wood combustion sector in BAAQMD in 2012, despite representing only 36% (230,000) of the estimated 620,000 wood burning devices in use in BAAQMD. Approximately 13% of emissions came from wood stove use and 1% from pellet stove use. The mass emissions calculated in this work were lower (3600 tons $PM_{2.5}$) than those reported in a BAAQMD emission inventory (4200 tons $PM_{2.5}$) and CARB emission inventory (5300 tons $PM_{2.5}$). Retrofitting frequently used fireplaces (300,000 units) to gas inserts would reduce sector emissions by 90%, annually avoiding 140-310 (120, 370) avoided premature deaths, 19,000 (16,000, 23,000) lost days of work, and creating upwards of $1 billion ($870 million, $3.2 billion) in financial benefits.

Conclusions: Modeled results based on survey data indicate that fireplaces are responsible for the vast majority of residential wood combustion emissions of $PM_{2.5}$ in the San Francisco Bay Area. There are opportunities to avoid premature deaths, serious morbidity outcomes, and financial losses by encouraging replacement of fireplaces with gas inserts, removing existing fireplaces, and upgrading existing wood stoves.

Note: Due to the geographic focus on California in this chapter, mass quantities are expressed in short tons (tons) rather than metric tons (tonnes).

4.2. Introduction

4.2.1 Background

Residential wood combustion is currently the largest source of wintertime fine particulate air pollution in the San Francisco Bay Area's airshed. This is despite the region's relatively temperate climate—with approximately 3000 heating degree days per year, far less than in the higher altitude Northern Sierra Air Quality Management District, which has 6000-8000 heating degree days per year for example (Western Regional Climate Center, 2015)—and widespread access to other forms of energy for home space heating, such as electricity and piped natural gas. This is also despite current limitations on wood burning on days when the air is considered unhealthy for residents. (Updates to the relevant regulations were finalized in 2015, as detailed below.)

The main motivation for the wood burning regulations is to reduce fine particulate pollution ($PM_{2.5}$) levels and thereby to protect public health (Bay Area Air Quality Management District 2012). Changes in National Ambient Air Quality Standards (NAAQS) for $PM_{2.5}$ levels in 2012 led the BAAQMD to expend more resources on understanding and abating $PM_{2.5}$ sources pertinent to the Bay Area (Environmental Protection Agency, 2013). The Bay Area is currently out of attainment for the 24-hour standard (which is measured with a three year average of the 98th percentile value) but not the annual standard (which is measured with an annual arithmetic mean averaged over three years). The non-attainment days fall in the winter months, when meteorology favors buildup of $PM_{2.5}$ and emissions in certain $PM_{2.5}$ source categories, including residential wood combustion, are higher (Fairley, 2012).

BAAQMD has taken measures to regulate emissions from residential wood burning in the Bay Area since 2008. (See Section 1.4.) Before the measures become mandatory, there was a voluntary emissions reduction program in place for two decades (Bay Area Air Quality Management District 2012). No matter the time of year, residents are instructed to "burn only clean, dry (seasoned) wood in short, hot fires with plenty of air to prevent excessive smoke. No visible smoke is allowed beyond the 20-minute start-up period" (Bay Area Air Quality Management District 2012).

CARB and the US EPA do not directly regulate residential wood burning. The board has depended on regional air districts, such as BAAQMD, to set policy regarding residential heating with wood (personal communication, CARB and EPA employees). CARB views residential wood burning as primarily a local air quality issue, and does not assign an employee to work specifically on residential wood burning regulation, although the board does fund research related to the health effects of wood smoke exposure (Balmes, 2016).

There is, however, evidence that CARB strongly discourages any form of residential wood combustion. In 2005, CARB and the California Environmental Protection Agency issued a handbook on residential wood burning, which states, "To be a good neighbor, eliminate wood burning" (California Environmental Protection Agency, 2009). The handbook outlines the reasons that most household wood burning appliances do not create heat efficiently and suggests alternative heat sources, such as electric heaters and pellet stoves. It also summarizes known health effects from exposure to wood smoke, and lists best practices for readers who will continue to burn wood, such as tips for identifying and buying properly seasoned wood. Readers are encouraged to contact their local air district for more information about burning regulations.

The purpose of this analysis is to create a model to better understand the main drivers of $PM_{2.5}$ emissions within the BAAQMD jurisdiction, to identify priority areas for possible intervention, to estimate health effects associated with the emissions, and to estimate economic benefits associated with mitigation or reduction of the emissions. Though $PM_{2.5}$ has been identified as a persistent air pollution problem in BAAQMD, and residential wood combustion is understood to be a main source of the pollution during the time of year when it is the biggest problem (winter months), relatively little is understood about the magnitude of health and economic effects of this emission source on the San Francisco Bay Area.

4.2.2 Residential wood combustion in the San Francisco Bay Area

In this section, I describe the general patterns of residential wood combustion for heating in California's San Francisco Bay Area, as well as the existing and proposed regulations that pertain to household wood combustion.

The Bay Area airshed is governed by the Bay Area Air Quality Management District (BAAQMD). Geographically, it extends from Napa Valley to Gilroy, and includes all or part of the following counties: Napa, Marin, San Francisco, Contra Costa, Alameda, San Mateo, Santa Clara, Sonoma, and Solano. This area is home to almost seven million residents (2.6 million occupied housing units) and an estimated 1.1 million (Fairley 2014) - 1.4 million (Lee 2015) fireplaces, pellet stoves, and wood stoves. (See Table 4.1.)

An estimated 43% of occupied housing units in the Bay Area (1.13 million homes) have a wood burning device, according to survey results from BAAQMD (Fairley, 2014). (See Table 4.1.) The majority of units with a wood burning device have fireplaces (38%), as opposed to pellet stoves (5.8%) or wood stoves (17.9%). By county, Alameda has the most pellet stoves (25,000) and the most wood stoves (36,000), though Sonoma County has the highest percent of occupied housing units with a pellet stove (5.8%) and Napa County has the highest percent with a wood stove (12.5%). Santa Clara County has the most wood burning fireplaces of any county in the district (260,000). San Francisco has dramatically less occupied units with a wood burning device (28%) than the other counties in the Air Quality Management District (43%-52%). A survey company hired by BAAQMD to assess wood burning behavior in 2013-2014 found that "among all households with at least one wood burning device, 25% expect to burn wood at least once per week this winter." (True North 2014)

Table 4.1. Households with wood burning devices in the Bay Area Air Quality Management District. From Fairley (2014).

	Number of occupied housing units*					Percentage of occupied housing units			
County	**Totals (2013)**	**wood burning fireplace**	**pellet stove**	**wood stove**	**any wood burning device**	**wood burning fireplace**	**pellet stove**	**wood stove**	**any wood burning device**
Alameda	545,000	190,000	25,000	36,000	230,000	36%	4.6%	6.5%	43%
Contra Costa	376,000	160,000	17,000	27,000	180,000	42%	4.4%	7.2%	48%
Marin	103,000	50,000	2,000	9,000	50,000	47%	2.3%	9.0%	52%
Napa	49,000	20,000	2,000	6,000	20,000	36%	3.4%	12.5%	45%
San Francisco	345,000	80,000	11,000	11,000	100,000	25%	3.3%	3.3%	28%
San Mateo	258,000	100,000	8,000	12,000	110,000	39%	3.0%	4.6%	44%
Santa Clara	609,000	260,000	7,000	33,000	280,000	43%	1.2%	5.4%	47%
Solano	141,000	60,000	7,000	6,000	70,000	40%	4.9%	4.3%	47%
Sonoma	186,000	60,000	11,000	33,000	90,000	34%	5.8%	17.9%	47%
Bay Area	**2,613,000**	**990,000**	**90,000**	**170,000**	**1,130,000**	**38%**	**3.4%**	**6.6%**	**43%**

An older survey of Bay Area households, conducted in 2001 (final report published in 2002), found that 70% of respondents had a wood burning device in their home (61% fireplaces, 10% wood stoves, 3.8% fireplace inserts), which is more than double the proportion reported in the survey results in Table 4.1. However, the predominant primary heat source reported was natural gas (83%), with <5% reporting reliance on wood burning devices for space heating. (Broderick and Houck 2003).

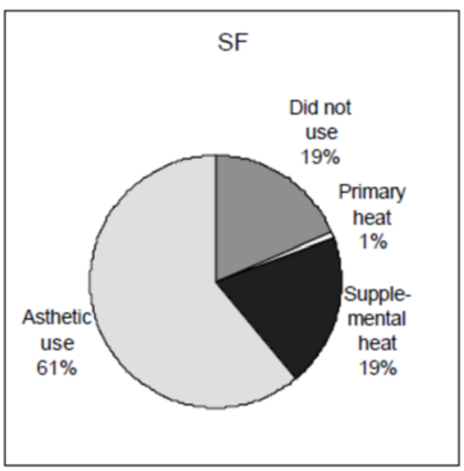

Figure 4.1. Wood-burning fireplace use in San Francisco Bay Area. Survey data collected in 2001 (Broderick and Houck, 2003).

4.2.2.1 Space heating vs. "aesthetic" wood combustion in the San Francisco Bay Area

In previous literature, residential wood burning in the Bay Area has been characterized as "primarily episodic burning for aesthetic reasons," (Gullett et al. 2003), meaning that most burning has been recreational rather than for space heating. Phone survey results from the early 2000s found that, among Bay Area fireplace users, 69% of fireplace usage was less than once per week in frequency, and that the average time of use per fireplace was only about 43.4 hours per year (Gullett et al., 2003).

Another survey, conducted in 2001, found similar results: 61% of those with wood burning fireplaces used fireplaces for "aesthetic" purposes, as opposed to those who used it as a heat source. (See Figure 4.1.) Of the 20% of respondents who used their fireplace for heat, 1% relied on it for their primary heat source, and 19% listed it as a supplementary heat source. About 19% of respondents with fireplaces did not use their fireplace at all (Broderick and Houck, 2003).

4.2.2.2 Trends in space heating with wood in Bay Area

Between 2000 and 2010, emissions from all types of wood combustion fell 14% (on an annual basis). In the winter months, residential wood combustion is the dominant source of emissions within this category; during the winter months in this time period, wood burning emissions fell about 40% (Fairley, 2012).

4.2.3. Air pollution attributable to residential wood combustion in the San Francisco Bay Area

Despite relatively low levels of reliance on fireplaces for space heating, BAAQMD has determined, using a chemical mass balance methodology, that wood burning is the greatest contributor (out of 15 source categories) to both annual and peak $PM_{2.5}$ air pollution in the Bay Area (Fairley, 2012). The wood burning source category includes residential wood combustion (in fireplaces, pellet stoves, and wood stoves), as well as wildfires, controlled burns, and secondary $PM_{2.5}$ that forms from precursor gases emitted by trees. In winter months, emissions in the wood burning category are almost exclusively from residential wood combustion for space heating; in summer months, wildfires and controlled burns are more significant sources in this category.

4.2.4 Residential wood smoke policies

4.2.4.1 BAAQMD rules

BAAQMD regulates residential wood smoke under Regulation 6, which regulates particulate matter and visible emissions, and specifically Rule 3, which applies to wood burning devices (Bay Area Air Quality Management District, 2015b). This rule was originally adopted by the district in 2008, and was substantially amended, after a public review process that included nine public workshops across the air district, in 2015.

Foundational aspects of the rule include (Bay Area Air Quality Management District, 2015b):

- Winter burn bans: the ability for the air district to call a mandatory burn ban (also known as a "Spare the Air" alert) during four winter months (November – February), when air quality is forecast to be unhealthy due to ambient $PM_{2.5}$ above 35 $\mu g/m^3$. During a burn ban, combusting or otherwise burning wood and other solid fuels is generally prohibited.

- Restrictions on types of fuels that can be burned: a prohibition on burning garbage, contaminated wood, non-seasoned wood, plastics, coal, and other materials besides wood in a wood burning device.
- Labeling of wood for sale as either seasoned or unseasoned, with instructions for seasoning if needed.

The updated rule includes the following changes, most of which take effect in mid-2016 (Bay Area Air Quality Management District, 2015a, b):

- New construction: Wood-burning devices may no longer be installed in new building construction. Cleaner and more efficient heating options, such as gas-fueled or electric heaters, must be installed instead.
- Rental properties: all real estate and rental properties must disclose the health hazards of $PM_{2.5}$ from burning wood or any solid fuel as a source of heat. If in an area with natural gas service, they must have a source of heat that does not require solid fuels.
- Remodels: all fireplace or chimney remodels costing >$15,000 and requiring a local building permit must replace an uncertified wood burning device or fireplace with an EPA certified wood burning device or a gas-fueled or electric heater.
- Sale of wood heaters: all wood heaters sold or manufactured in the Bay Area must comply with newly adopted EPA emission standards and compliance dates.
- Exemptions: Households with no permanently installed natural gas, propane or electric heating options may qualify for the "sole source of heat" exemption if the only source of heat is an EPA-certified wood burning device and is registered with the Air District.
- Smoke limits: After a 20-minute start-up allowance for new fires, visible emissions of greater than 20% opacity that last longer than 3 minutes in any hour are not allowed.
- Health warnings: when selling a new or used wood stove, seller must provide information on the health effects of burning wood.

BAAQMD estimated in 2012 that existing residential wood combustion regulations (the older version of Regulation 6 Rule 3 that was approved in 2008) resulted in reductions of 0.7 tons/day primary $PM_{2.5}$ from annual wood burning device changes and 6.0 tons/day primary $PM_{2.5}$ from changes in wood burning emissions during peak season (Bay Area Air Quality Management District, 2012b).

4.2.4.2 Local ordinances

As of 2012, 49 Bay Area cities and counties had adopted wood smoke or wood burning appliance ordinances, following the model ordinance developed by BAAQMD in the mid-1990s (Bay Area Air Quality Management District, 2012b). Cities and counties that currently have ordinances for regulation of residential wood smoke include Berkeley (2008), Contra Costa County (2000), Fremont (2002), Marin County (2003), Oakland (2005), Richmond (2006), City and County of San Francisco (2002), and Santa Rosa (2002). The ordinances include some or all of seven provision suggested in BAAQMD's model ordinance, including whether newly installed wood burning devices must be EPA-certified, whether there are restrictions on devices installed during a remodel, whether masonry fireplace construction is allowed or prohibited, whether burning is prohibited at the local level during Winter Spare the Air alerts, whether it is illegal to burn trash and inappropriate materials in a wood burning device, whether non-certified wood burning devices (such as fireplaces) must be removed from existing buildings/residences during remodel, sale or other event, and whether conversion of gas- to wood burning fireplaces is prohibited (Bay Area Air Quality Management District, 2012a).

4.3. Methodology

The following general approach was taken to model emissions from residential wood combustion in the San Francisco Bay Area. The equations described here form the core of my model and are used to calculate the "business as usual" (BAU) scenario. (See Section 4.2.4 for other residential solid fuel use scenarios.)

The mass of primary $PM_{2.5}$ emissions emitted annually from residential wood combustion for heating was calculated for each county (c) and device type (t) using the following equation:

$$M_{c,t} = S_{c,t}\ (\#\ devices) \times\ D_c \left(\frac{hr}{yr \cdot device}\right) \times\ E_t\ \left(\frac{g\ PM_{2.5}}{hr}\right) \qquad [1]$$

Where M is the mass of primary $PM_{2.5}$ emitted annually (g/yr) for a given county (c) and device type (t); S is the number of devices in active use by device type and county; D is the average duration of use (in hours/year per device) by county; and E is the emission rate (in g/hr of primary $PM_{2.5}$) by device type.

4.3.1 Number of wood burning devices in use in the San Francisco Bay Area

My goal was to characterize the number of residential wood burning devices that were used in the nine counties that fall within BAAQMD, in 2013, by type (fireplace, pellet stove, or wood stove) and by county. An estimated 43% of the 2.6 million occupied housing units in the BAAQMD's nine counties have at least one wood burning device, for a total of 1.1 million households with wood burning devices.

To construct this variable, I applied the proportion of survey respondents with wood burning devices who said that they would use a wood burning device in the survey year to the estimated number of wood burning devices per county. This produced an estimate of the number of fireplaces, pellet stoves, and wood stoves that are likely to be used in the survey year.

The number of actively used devices (*S*) of each type (*t*) in each county (*c*) is calculated in Equation 2:

$$S_{c,t} = O_c \times \left[\frac{W}{H}\right]_{c,t} \times \left[\frac{U}{W}\right]_c \qquad [2]$$

Where *O* is the number of occupied housing units in a county (c), *H* is the total number of households surveyed, *W* is the number of telephone survey respondents who reported having a wood burning device in their home, and *U* is number of telephone survey respondents who reported having at least one wood burning device in their home and said they would use it to burn wood during the winter in which they were surveyed. There are three types of devices (t) considered in this analysis: fireplaces, wood stoves, and pellet stoves. (The estimated number of wood burning devices, O_c * $(W/H)_c$, was reported by county in a memo prepared by BAAQMD (Fairley, 2014).)

One challenge is that households often have multiple wood burning devices and it is not apparent, at the county level, at what frequency various devices are being used. Survey data for BAAQMD as a whole revealed the proportion of households with specific devices (d) that expect to use them during the season surveyed. For the purposes of this analysis, I construct U, by applying the ratio "percent of households that will use device type (t): percent of households that will use any wood burning device" to the county-specific "percent of households that will use any wood burning device" (W/H) to obtain (U/H) by device type (t) and by county (c). This scaling factor, based on BAAQMD-wide self-reported expected annual usage data for all devices and for specific devices, was 0.93 for fireplaces, 1.2 for pellet stoves, and 0.96 for wood stoves (as compared to the BAAQMD-wide expected usage data for any wood burning device.

BAAQMD jurisdiction includes all of seven counties (Alameda, Contra Costa, Marin, Napa, San Francisco, San Mateo, and Santa Clara) and parts of two additional counties (Sonoma and Solano). The data I used as inputs to this model were county-specific. Because only portions of Solano and Sonoma counties fall within BAAQMD jurisdiction, I needed to scale the results for these counties to include only the relevant portions. Based on the population data provided by BAAQMD in their emission inventory, I scaled the estimated Solano and Sonoma county wood burning devices and relevant emissions to include only the proportions (75.8% for Solano County and 90.2% for Sonoma County, based on 2011 census data) of their populations that fall within BAAQMD jurisdiction. Data from 2011 were used for scaling because that was the base year provided in the available BAAQMD emission inventory (Bay Area Air Quality Management District, 2014).

4.3.2. Duration of devices use per year

To estimate the number of hours that each wood burning device is used per year, I took the general approach of estimating the number of hours per day the devices are used (based on survey results), and then the number of days per year they are used. (See Equation 3.) The results of this equation are applied to all device types, because more detailed information on differences in yearly duration of burning was not available in survey results.

$$D_c = \left(\left[\frac{Hours}{Day}\right]_{f,c} \times \left[\frac{Days}{Year}\right]_{f} \times \frac{H_{f,c}}{U_c} \right) + \left(\left[\frac{Hours}{Day}\right]_{s,c} \times \left[\frac{Days}{Year}\right]_{s} \times \frac{H_{s,c}}{U_c} \right) \qquad [3]$$

Where D, the duration of burning (hours/year), is calculated at the county level (c), by combining the weighted average of the yearly average duration of burning (hours/year) for two groups: frequent burners (f) and sporadic burners (s). The average duration of burning for each of these groups was calculated, at the county level, by combining the reported daily duration of burning (hours/day) at the county level, the average days per year that each group (f,s) is expected to burn wood, and the proportion of households in each county who expect to burn wood at least once during the survey season (U_c) and that fall into either the frequent ($H_{f,c}$) or sporadic ($H_{s,c}$) burning groups. Further information on these factors is provided in the sections that follow.

4.3.2.1 Frequency of combustion (days/year) among two groups of wood burners
I divided households that reported having at least one wood burning device in their home, and reported that they would use the device at least once during the survey season, into two groups: those that reported expecting to burn at least once (≥1) per week ("frequent" burners, *f*), and those that reported they expected to burn less than once (<1) per week ("sporadic" burners, *s*).

A small proportion of the frequent burning group (3.7% of those who said they would burn at least once per week) were not sure about how many times they would burn in a week, so I allocated them the mode value, which was 3 days per week.

As with the frequent burning group (*f*), a small proportion (3.1% of those who said they would burn at some point during the year) were not sure how many times they would burn throughout the year; I allocated these respondents to the "once per month" group.

4.3.2.2 Burn season
For these calculations, I assumed that the burning season in the Bay Area is six months long (October-March). This time period is 183 days, or just over 26 weeks.

4.3.2.3 Combustion time (hours/day) among two groups of wood burners
Data were available at the county level to describe the average duration of burn (hrs/day) among all households that expected to burn at least once during the season (True North, 2014). Data were not available at the county level to describe the average duration of each burn among the two different groups (*f*,*s*).

Duration of combustion event (hours/day) was calculated using two inputs: 1) average combustion time, as reported by respondents, by county; and 2) average combustion times among frequent burners and sporadic burners, as reported for BAAQMD. I estimated the average duration of each burn, at the county level, by creating a scaling factor based on the following BAAQMD-wide data: average burn duration (among all survey respondents who said they expected to burn at least once during the season), average burn duration among frequent burners, and average burn duration among sporadic burners. I applied the derived scaling factors (1.2 for frequent burners, as compared to all burners, and 0.78 for sporadic burners, as compared to all burners) to the average burn durations reported for each county to estimate the average burn duration (hours/day) among the frequent and sporadic burn groups in each county.

4.3.3 Emission rates of $PM_{2.5}$ (g/hr) (base case)

The emission rates are represented in Equation 1 by the factor E_t:

$$E_t = \left(\frac{g\ PM_{2.5}}{Hr}\right)_t \qquad [4]$$

I apply published emission rates specific to three types of residential wood burning devices. The emission rates for fireplaces and wood stoves are derived from tests that mimic Bay Area-specific residential wood burning conditions (Gullett et al., 2003). $PM_{2.5}$ emission factors, which estimate the mass of pollutant released per mass unit of fuel burned, were estimated at 9.58 g/kg for wood stoves and 2.8-16.6 g/kg for fireplaces, depending on the fuel used (oak, pine, or artificial logs) (Gullett et al., 2003). Though the emission rates measured total primary PM, rather than restricting to mass of $PM_{2.5}$, the authors note that “PM emitted from the wood stove/oak tests are found to be primarily in the submicrometer size,” with >99.5% of the particulate matter classified as $PM_{2.5}$ (Gullett et al., 2003). The emission rate for pellet stoves follows EPA certification guidelines for pellet stoves, since a Bay Area-specific value was not available (US Environmental Protection Agency, 2015c).

Table 4.2. Published emission rates (with standard deviation) specific to San Francisco Bay Area. From Gullett et al. (2003).

Device	**Fuel**	**g/hr** $PM_{2.5}$	**Notes**
Wood stove	Oak	20.7 (6.07)	Quadrafire 3100 (EPA certified non-catalytic); log 40 cm long and 18 cm wide
Fireplace	Oak	27.3 (2.76)	Majestic MRC42A with glass door; log 40 cm long and 18 cm wide
Fireplace	Pine	15.1	Majestic MRC42A with glass door; log 40 cm long and 18 cm wide
Fireplace	Artificial log	32.3	Majestic MRC42A with glass door; wax and sawdust log

For fireplaces, I use an emission rate of 27.3 g/hr (standard deviation of 2.76), which is the average value measured for oak logs by Gullett et al. (2003). I use this emission factor because survey results indicate that the majority (59%) of Bay Area households report burning oak; many fewer households (7%) report burning pine, for which Gullett et al. report a lower emission rate (15.1 g/hr), or manufactured logs (18%), which are associated with a higher emission rate (32.3 g/hr).

For wood stoves, I use an emission rate of 20.7 g/hr (standard deviation of 6.07), which is the average value calculated by Gullett et al. when burning oak logs for 300 minutes (Gullett et al., 2003). Gullet et al. did not test any other types of fuel when testing wood stove emissions.

For pellet stoves, I apply an emission rate of 1.0 g/hr, which is the standard set in the EPA NSPS regulations and also the value calculated in national laboratory testing of selected residential heating appliances (Macdonald, 2009; US Environmental Protection Agency, 2015c). No Bay Area-specific emission rate for pellet stoves was available; however, fuel and combustion conditions are more consistent in pellet stoves than in other wood burning devices, so using a more broadly geographically applicable rate seems justifiable. Given that 1.0 g/hr is an emission standard, and that combustion conditions are consistent, no uncertainty range was applied.

About 18% of BAAQMD residents who have wood burning devices and responded to a survey question about primary and secondary fuels indicated that they burn manufactured logs, rather than natural wood logs. This proportion varies by county from 5.4% (Napa) to 28.3% (San Francisco). However, the data provided do not contain enough information to estimate the number of households who both expect to burn at some point in 2013 and will use a manufactured log. Therefore, all fuel use was assumed to be natural logs. This produces a conservative emissions estimate, as manufactured logs produced higher $PM_{2.5}$ emission rates

(32.3 g/hr) than natural logs (15.1-27 g/hr), when burned in fireplaces, according to test results from Gullett et al. (2003).

4.3.4 Scenarios applied

To evaluate the potential for regulatory measures to assist with emission mitigation in the residential wood combustion sector, I first estimated emissions associated with a "business as usual" (BAU) case, based on data from BAAQMD's surveys and published emission rates, as detailed above. I then created and applied several hypothetical scenarios, which are described below. The estimated annual emission reduction, and resulting health and economic benefits, are described in the results section.

- Scenario 1 "EPA1": All existing wood stoves in the Bay Area are replaced with stoves meeting the EPA NSPS Step 1 emission rate criterion (4.5 g/hr $PM_{2.5}$).
- Scenario 2 "EPA2": All existing wood stoves in the Bay Area are replaced with stoves meeting the EPA NSPS Step 2 emission rate criterion (2.5 g/hr $PM_{2.5}$).
- Scenario 3 "15%GAS": 15% of frequently used fireplaces are either replaced with gas-burning inserts, removed, or no longer used (0 g/hr $PM_{2.5}$)
- Scenario 4 "GAS": All existing fireplaces in the Bay Area are either replaced with gas-burning inserts, removed, or no longer used (0 g/hr $PM_{2.5}$)
- Scenario 5 "PELLET": All frequently used fireplaces, and all wood stoves, are replaced with pellet stoves instead (1 g/hr $PM_{2.5}$)

4.3.5 Health and economic effects calculations

I use a methodology published by the US EPA to estimate the health benefits and economic benefits of avoided emissions, as estimated in the scenarios described above (US Environmental Protection Agency, 2013b). The benefits methodologies are specific to the residential wood combustion sector.

Health and economic benefits are estimated using a linear per-ton factor for avoided tons of $PM_{2.5}$. (See Table 4.3.) Two main papers (Krewski et al., 2009; Lepeule et al., 2012) are used in the calculation of premature mortality avoided per ton avoided $PM_{2.5}$ among adults over 25 years of age (Lepeule et al. 2012, analyzing the Harvard Six City Study prospective cohort) or 30 years of age (Krewski et al. 2009, analyzing the American Cancer Society prospective cohort), respectively. Avoided infant mortality, for babies under one year of age, is also included in the avoided premature mortality factor used here. The factors are specific to the residential wood combustion sector.

The methods used here to estimate the relationship between wood smoke emissions and health are different than those used in the previous two chapters. The differences between the approaches are described in Chapter 5.

Table 4.3. Factors used to estimate health and economic benefits per ton of avoided $PM_{2.5}$ in the residential solid fuel-heating sector. From US Environmental Protection Agency (2013b).[3]

Indicator	Factor/ton
Premature mortality-Krewski (2009)	0.042
Premature mortality-Lepeule (2012)	0.094
Respiratory emergency room visits	0.021
Acute bronchitis	0.064
Lower respiratory symptoms	0.82
Upper respiratory symptoms	1.2
Minor Restricted Activity Days	34
Work loss days	5.7
Asthma exacerbation	2.9
Cardiovascular hospital admissions	0.013
Respiratory hospital admissions	0.01
Non-fatal heart attacks (Peters)	0.043
Non-fatal heart attacks (All others)	0.0047
$2010 (Krewski 2009, 3% discount rate)	360,000
$2010 (Lepeule 2012, 3% discount rate)	810,000
$2010 (Krewski 2009, 7% discount rate)	320,000
$2010 (Lepeule 2012, 7% discount rate)	730,000

4.3.6 Sensitivity analysis

4.3.6.1 Model inputs

Uncertainties for each variable in Equation 1 were included in a Monte Carlo analysis to understand relative influences on resulting mass emissions of $PM_{2.5}$. Using Oracle Crystal Ball Release 11.1.2.4 in conjunction with Microsoft Excel 2013, I ran a Monte Carlo simulation with 1,000 trials, assuming normal distributions for the variables in question, and evaluating results at 90% certainty.

For *S,* the number of fireplaces, wood stoves, and pellet stoves in active use, I assumed a 1% standard deviation to account for potential sampling error.

For *D,* the average number of hours each device is used per year, I assumed a 10% standard deviation in the number of hours burned per week for all device types, given the difficulty involved in estimating device use and the possibility of recall bias.

For *E,* the emission rates: For wood stoves and fireplaces, I used standard deviations derived from percent precision values published alongside mean values in Gullett et al. (2003) (where percent precision values were defined as 100*S/average). For pellet stoves, I assumed no significant deviation from the standard of 1 g/hr $PM_{2.5}$ occurs, given the relatively controlled

[3] $2010 refers to monetary value expressed in US dollars as valued in 2010.

nature of both the combustion environment and pellet fuel. When implementing the scenarios, I used the following: for Scenario 1 and Scenario 2, which assume that all wood stoves are transitioned to EPA NSPS Step 1 stoves or Step 2 stoves, respectively, I used the standards of 4.5 g/hr for Step 1 and 2.5 g/hr for Step 2, each with a 20% standard deviation, to account for user error in stove operation. For Scenario 3 and Scenario 4, I assumed no uncertainty around the 0 g/hr $PM_{2.5}$ emission factor that would result from transition to gas or removal of a wood burning device. For Scenario 5, I assumed 1 g/hr $PM_{2.5}$ for pellet stoves, with no uncertainty, given the reasons detailed above.

4.3.6.2 Health and economic effects

There are no uncertainty factors available for the benefit per ton factors published by US EPA (2013b) and applied here to estimate health impacts and economic savings associated with hypothetical emission reduction policies. Using the uncertainty bounds generated by analysis of expected emission reductions from the BAU case, I note the corresponding bounds of health and economic impacts in the Results section below.

Authors of the EPA report state that, "When using these benefit per ton estimates in analyses, care should be taken to not overstate the accuracy of the total benefits estimates or estimates of avoided incidence. For this reason, it is EPA practice to round total benefits estimates to two significant digits and to round estimates of avoided incidence to the nearest whole number." The same practice is followed in the presentation of results below, with the convention of reporting two significant digits applied to avoided incidence estimates as well as economic benefits estimates.

4.4. Results

4.4.1 Mass of primary $PM_{2.5}$ emissions from residential wood burning in the Bay Area

Using the factors outlined in Equation 1, I estimate that residential combustion of wood fuels across the nine counties in BAAQMD resulted in 3600 tons (3000, 4300 90% uncertainty bounds) of primary $PM_{2.5}$ in 2013. (See Figure 4.2.) Approximately 87% of these emissions come from residential wood combustion in fireplaces and 13% come from combustion in wood stoves; pellet stove use accounts for less than 1% of the $PM_{2.5}$ from residential wood combustion in the BAAQMD jurisdiction.

Frequent fireplace users (those that use their fireplace at least once per week) were responsible for 84% of total $PM_{2.5}$ mass emissions from the residential wood combustion sector in BAAQMD in 2012, despite representing only 36% (230,000) of the estimated 620,000 wood burning devices in use in BAAQMD. Approximately 13% of emissions came from wood stove use and 1% from pellet stove use. Table 4.4 shows the distribution of these emissions (in tons of $PM_{2.5}$ in 2012) for BAAQMD by device type, with 90% uncertainty bounds.

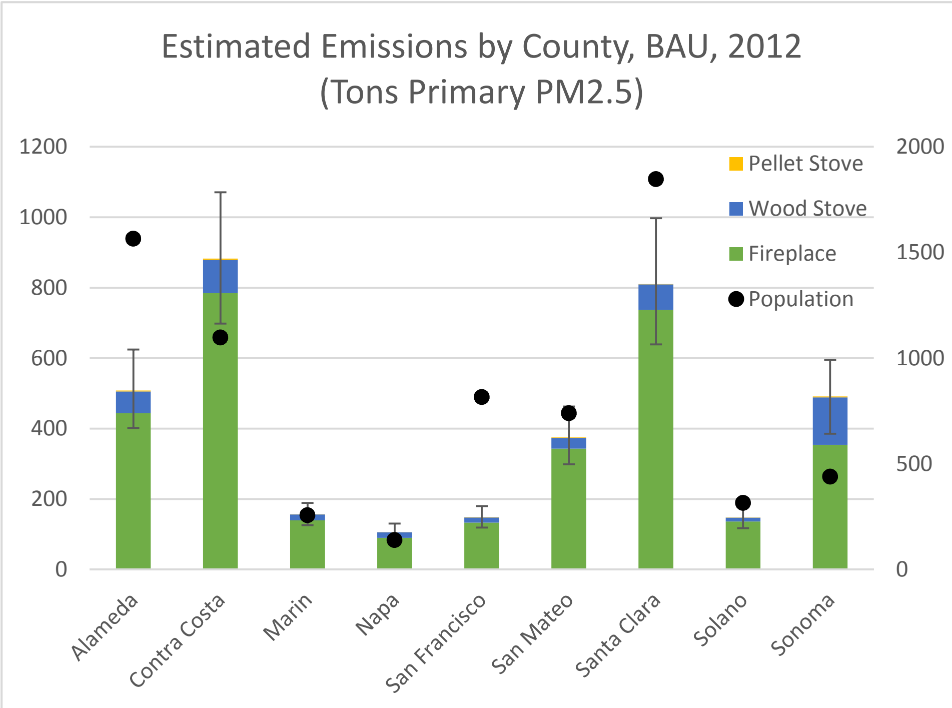

Figure 4.2. Estimated emissions of primary $PM_{2.5}$ (tons) from residential household heating with solid fuels, by county, for BAAQMD air shed in 2012. Population of each county (in units of 1,000 residents, axis on right side) is shown by black dots. (Population data from 2011.) Error bars indicate 90% uncertainty bounds for county sector emission totals.

Table 4.4. Estimated tons of primary $PM_{2.5}$ emissions from residential heating in BAAQMD (with 90% uncertainty bounds) by county and device type, for 2012. Totals may not add due to rounding.

County	Fireplace	Pellet stove	Wood stove	Total (all devices)
Alameda	440 (350, 550)	<10	62 (32, 94)	509
Contra Costa	780 (600, 970)	<10	94 (49, 143)	885
Marin	140 (110, 170)	<10	17 (9, 26)	157
Napa	90 (70, 110)	<10	16 (8, 24)	107
San Francisco	130 (100, 170)	<10	14 (7, 22)	149
San Mateo	340 (270, 430)	<10	31 (16, 46)	375
Santa Clara	740 (570, 910)	<10	72 (38, 109)	816
Solano	140 (110, 170)	<10	11 (5, 16)	149
Sonoma	350 (280, 440)	<10	135 (70, 204)	493
BAAQMD	*3200 (2700,3800)*	*10*	*450 (240,680)*	*3600 (3000,4300)*

4.4.1.1 Per capita analysis

Across all counties in the airshed, per capita emissions in this sector are dominated by emissions originating from fireplaces, even in counties where wood stoves and pellet stoves are more prevalent, such as Sonoma and Napa. Figure 4.3 shows the per capita distribution of $PM_{2.5}$ emissions, by device type and county. The figure shows the primary $PM_{2.5}$ emissions per 1000 occupied housing units (which, for this analysis, is treated as being equivalent to a household unit.) Note that the emissions from pellet stoves are much less than for fireplaces or wood stoves, so they are difficult to see in the Figure.

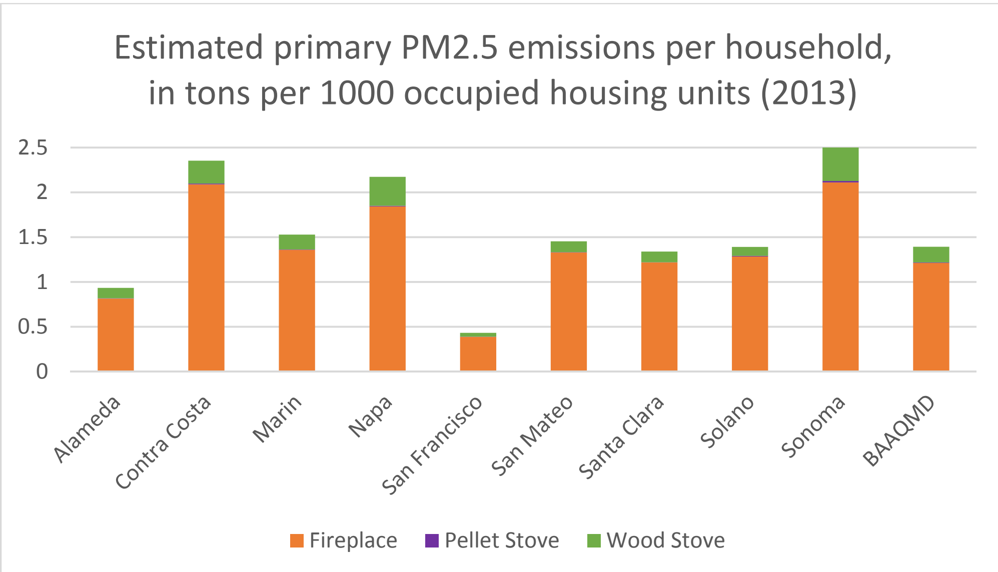

Figure 4.3. Estimated primary $PM_{2.5}$ emissions per household, in tons per year, for BAU in 2013.

4.4.1.2 Comparison with other emission inventories

My mean estimate of 3600 tons of primary $PM_{2.5}$ from residential wood combustion in 2013 is conservative, as compared to two other emission inventories that include estimates of $PM_{2.5}$ from residential wood combustion in the Bay Area (see Figure 4.4). This is likely due to the lower emission rates (and corresponding emission factors) applied in this model. (See Section 4.2 for a comparison of the emission rates and factors used in other models.) For comparison, here are the emission estimates from two other emission inventories that evaluate San Francisco Bay Area $PM_{2.5}$ emissions from residential wood burning:

- CARB estimates that an annual total of 5300 tons of $PM_{2.5}$ were generated from residential combustion of solid fuels in fireplaces and wood stoves in 2013. This is a significant portion (83%) of the 6400 tons of $PM_{2.5}$ estimated to result from all residential fuel use (heating, water heating, and cooking) in the same year. The CARB estimate is outside of the 90% uncertainty bounds for my mean estimate (3000, 4300).
- BAAQMD estimates that an annual total of 4200 tons of $PM_{2.5}$ were generated from "domestic combustion" in 2011; this same value was projected for 2014. No specific device types or fuel types were specified in the data available. Note that the BAAQMD estimate of 4200 tons is within the 90% uncertainty bounds for my estimate (3000, 4300).

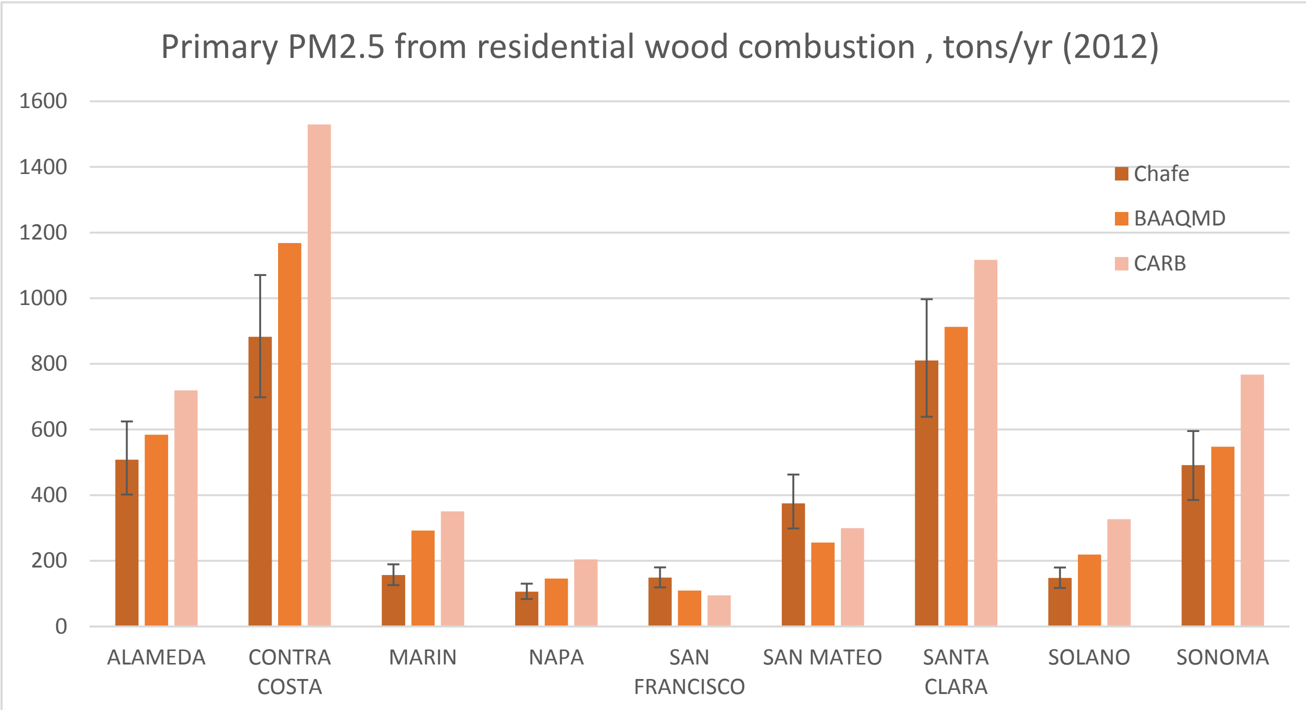

Figure 4.4. Comparison of $PM_{2.5}$ from residential wood combustion (tons/yr in 2012) between three different emission estimations (Chafe [this analysis], BAAQMD emission inventory, and CARB emission inventory). Error bars on Chafe estimates indicate 90% uncertainty bounds.

Across all three inventories, Contra Costa has the highest estimated mass emissions of $PM_{2.5}$ from residential wood combustion in 2012, accounting for approximately 25% of total sector emissions across BAAQMD, despite having only 15% of the area's population. (See Figure 4.5.) Conversely, San Francisco has 11% of the region's population but produces less than 5% of the district's $PM_{2.5}$ from residential wood combustion.

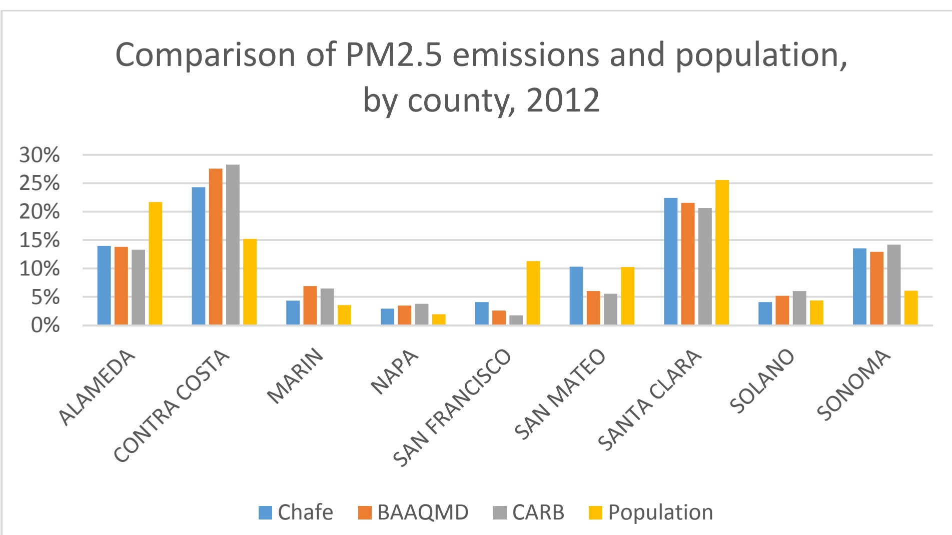

Figure 4.5. Comparison of $PM_{2.5}$ emissions and population by county, 2012.

On both an absolute and per capita basis, by county, CARB has the highest estimates for all counties except for San Francisco and San Mateo. For San Francisco and San Mateo, on a per capita basis, my estimates are highest. For San Mateo, BAAQMD estimates are the lowest; for all other counties, BAAQMD estimates are in between my estimates and the CARB estimates, on a per capita basis. (See Figure 4.6.)

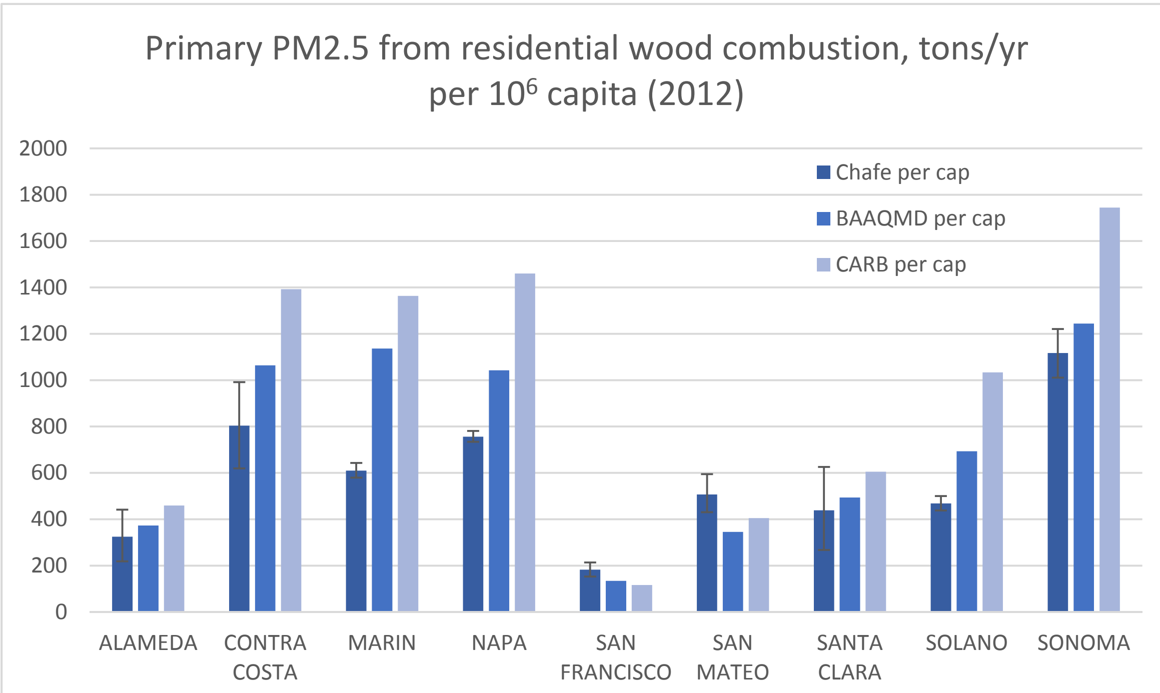

Figure 4.6. Comparison of $PM_{2.5}$ from residential wood combustion by county on a per capita basis (tons/yr per million inhabitants in 2012) between three different emission estimations (Chafe [this analysis], BAAQMD emission inventory, and CARB emission inventory). Error bars on Chafe values indicate 90% uncertainty bounds.

4.4.2 Frequency of burning (f,s) at household level

The proportion of households surveyed who expected not to burn wood at any time during the year ranged from 26% in Sonoma to 66% in San Mateo. Among households who expected to burn wood at some time during the survey season, the proportion who burn frequently (≥1 time per week, *f*) ranged from 41-58%, depending on the county. Conversely, the proportion of households who burn sporadically (<1 time per week, *s*) ranged from 42-59% depending on the county.

Frequent burners (*f*): This group accounted for 48.4% of those who said they would burn at some point during the year (True North, 2014). Data were available for BAAQMD as a whole on the number of days per week each of the respondents in this group estimated that they would burn (1-7 days). I found that frequent burners reported that they combust wood on average 3.1 days per week. Using the assumption that the burning season is 6 months long, I estimate that households in the frequent burning group use a wood burning device about 82 days per year.

Sporadic burners (*s*): This group accounted for 51.6% of the BAAQMD households who said they expected to burn at least once a year (True North, 2014). Survey respondents who selected this response option then provided further detail on their planned burning frequency (2-3 times per month, once per month, or less than once per month). Using the detailed burn frequency responses and assuming, as above, that the heating season is six months long, I found that, on average, sporadic burners combust wood 7 days per year.

4.4.3 Scenario Results

The emission impacts of introducing five hypothetical scenarios were explored with this model. The primary $PM_{2.5}$ emissions associated with implementation of each scenario, by device type, are shown in Figure 4.7 and further detailed in Table 4.5 below. As shown in Figure 4.7, scenarios 1 and 2 reduced BAU wood stove emissions by assuming that wood stove emission rates improved with introduction of EPA-certified stoves across the counties in BAAQMD. In Scenario 3, some fireplace emissions are avoided by assuming that 15% of frequent fireplace users convert to gas fireplace inserts, but existing wood stove emissions remain prominent. In Scenario 4, fireplace emissions are completely removed, but wood stove and pellet stove emissions remain. In Scenario 5, most fireplace emissions and all wood stove emissions are avoided but pellet stove emissions increase, as it is assumed that frequent fireplace users and all wood stove users convert to using pellet stoves.

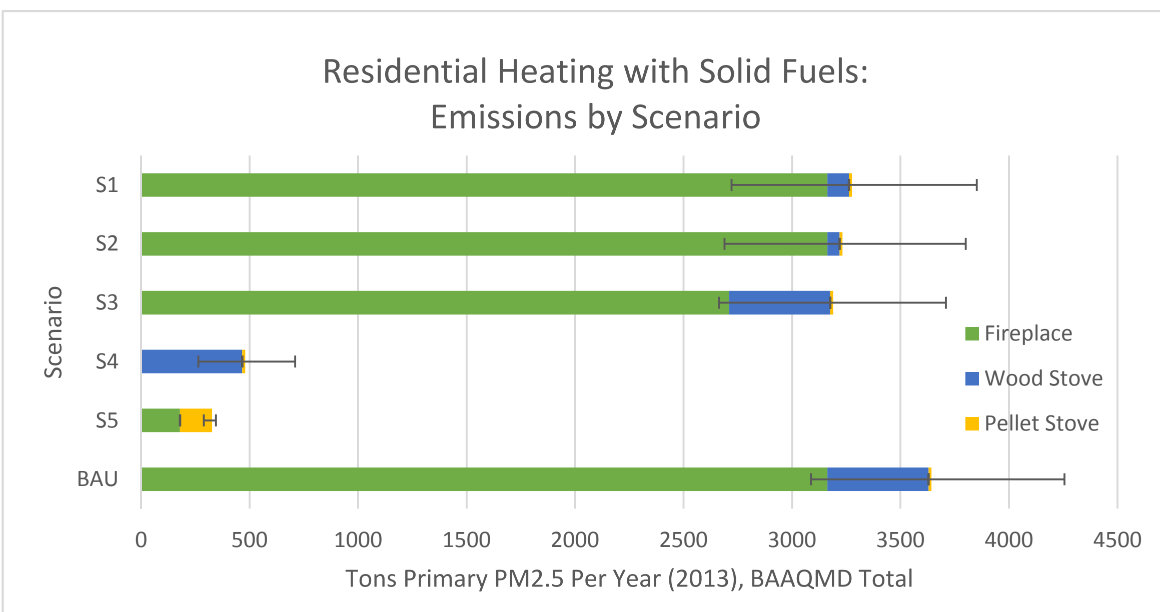

Figure 4.7. Tons of primary $PM_{2.5}$ expected from residential solid fuel combustion for heating under each of five hypothetical scenarios. Error bars indicate 90% uncertainty bounds for the total $PM_{2.5}$ in BAAQMD for each scenario.

Two groups of scenarios emerged: Scenarios 1, 2, and 3 each resulted in mean estimates of $PM_{2.5}$ emissions reductions equivalent to approximately 10% of the BAU emissions from the residential solid fuel space heating sector. Scenarios 4 and 5 resulted in mean estimate of emissions reductions around 90% of BAU emissions from the sector. There was substantial uncertainty associated with emission reductions (differences from BAU) expected from Scenario 1, 2, and 3; and 90% uncertainty bounds for these scenarios crossed zero. For Scenarios 4 and 5, however, the lower (conservative) end of the uncertainty bounds was around 2500 tons/yr in avoided primary $PM_{2.5}$, or approximately 70% of sector emissions.

Expected mass reductions of $PM_{2.5}$, from BAU, estimated to result from implementation of each scenario described above, are presented in Table 4.5 and explained in more detail below. The health and economic benefits associated with these reductions are then described in the following subsection.

Table 4.5. Summary of scenario results: expected reductions of primary $PM_{2.5}$, approximate percent reduction in residential wood combustion $PM_{2.5}$, and number of device units to be converted.

Scenario	Expected $PM_{2.5}$ reductions (tons/yr)	Approximate reduction in residential wood combustion $PM_{2.5}$	Number of device units to be converted
1: Convert wood stoves to EPA NSPS Step 1	350 (0, 1200)	10%	81,000-170,000
2: Convert wood stoves to EPA NSPS Step 2	380 (0, 1200)	10%	81,000-170,000
3: 15% of frequently used fireplaces convert to gas inserts	450 (0, 1300)	10%	34,000
4: Fireplaces converted to gas inserts or removed	3,200 (2500, 3800)	90%	480,000-990,000
5: Convert frequently used fireplaces and all wood stoves to pellet stoves	3,300 (2700, 4000)	90%	310,000-400,000

Note that, where 90% uncertainty bounds included zero, the lower bound is reported as zero rather than a negative number. (See Table 4.5.) This is because implementation of the scenarios is not expected to worsen air quality, although it is possible that, given uncertainty around wood burning emission rates and use patterns, the scenarios would not provide any tangible health benefits. This point is discussed in more detail in Section 4.

4.4.3.1 Scenario 1: EPA NSPS Step 1

Changing all existing wood stoves to stoves with an "EPA NSPS Step 1" emission rate (4.5 g/hr) yields a BAAQMD-wide reduction average estimate of 350 tons/year (0, 1200), with the central estimate representing approximately a 10% reduction from BAU sector emissions as a whole. This hypothetical intervention would require that all 170,000 wood stoves currently installed in the BAAQMD jurisdiction—or at least the 81,000 that are used in a given year—are brought up to EPA NSPS Step 1 emission standards.

4.4.3.2 Scenario 2: EPA NSPS Step 2

Changing all existing wood stoves to stoves with an "EPA NSPS Step 2" emission rate (2.5 g/hr) yields a BAAQMD-wide reduction average estimate of 380 tons/year (0, 1200), with the central estimate representing approximately a 10% reduction from sector as a whole. This hypothetical intervention would require that all 170,000 wood stoves currently installed in the BAAQMD jurisdiction—or at least the 81,000 that are used in a given year—are brought up to EPA NSPS Step 2 emission standards.

4.4.3.3 Scenario 3: 15% of frequent fireplace users convert to gas inserts

If 15% of frequent (≥1 per week) fireplace users converted to using gas inserts instead, BAAQMD-wide primary $PM_{2.5}$ emissions from residential wood combustion would fall by a BAAQMD-wide reduction estimated at 450 tons/year (0, 1300), with the central estimate representing approximately a 10% reduction in primary $PM_{2.5}$ emissions from the sector. This hypothetical intervention would require that the 34,000 fireplaces that are used frequently be converted to gas inserts. This would bring the number of wood burning fireplaces installed across BAAQMD down from approximately 990,000 to 960,000.

4.4.3.4 Scenario 4: Changing all fireplaces to gas inserts (or removing fireplaces)

If all fireplaces that are used in the Bay Area were either replaced with gas-burning inserts or removed (0 g/hr $PM_{2.5}$), there would be a BAAQMD-wide reduction of 3200 tons/year primary $PM_{2.5}$ (2500, 3800), or approximately 90% reduction in sector emissions. This hypothetical intervention would require that all 990,000 fireplaces currently installed in the BAAQMD jurisdiction—or at least the 480,000 that are estimated to be used in a given year—were either replaced with gas-burning fireplace inserts or removed.

4.4.3.5 Scenario 5: Converting all frequent fireplace users, and all wood stove users, to pellet stoves

If all frequent (≥1 per week) fireplace users, and all current wood stove users, converted to using pellet stoves instead, BAAQMD-wide primary $PM_{2.5}$ emissions from residential wood combustion would fall by 3300 tons/year (2700, 4000), approximately a 90% reduction in primary $PM_{2.5}$ emissions from the sector. This hypothetical intervention would require that the 230,000 fireplaces that are used frequently, as well as all of the 170,000 wood stoves currently installed in the BAAQMD jurisdiction—or at least the 81,000 that are used in a given year—be converted to pellet stoves. This would bring the number of installed pellet stoves across BAAQMD up from 90,000 (54,000 of which are estimated to be used in the survey year) to approximately 400,000-490,000, depending on whether all wood stoves were replaced or only those that are expected to be used at least once annually.

4.4.3.6 Sensitivity analysis of scenario reductions

To determine uncertainty bounds associated with the reductions projected to be achieved in each scenario, I ran a Monte Carlo simulation involving both the BAU mean estimate of tons/year $PM_{2.5}$ and the specific scenario tons/year $PM_{2.5}$. I assumed a normal distribution and used the 90% uncertainty bounds generated during the first part of this analysis. This uncertainty analysis incorporated both uncertainty around the BAU emission estimate as well as the scenario estimates.

As described above, the 90% uncertainty bounds on the scenario reduction estimates include zero in Scenarios 1, 2, and 3. This is due to both the large spread in uncertainty bounds in the BAU estimate of 3600 (3000, 4300) tons/year $PM_{2.5}$ for BAAQMD (58-62% contribution to variance), and to a lesser extent the uncertainty associated with the specific scenario emission estimates (38-42% contribution to variance). The 90% uncertainty bounds for Scenario 4 and Scenario 5 do not include zero, indicating that, despite uncertainty about emission rates and device use, emissions reductions should be expected. This is discussed in more detail in Section 4.5.

4.4.4 Health effects of population exposure to AAP from residential wood combustion in the Bay Area

I use an EPA methodology (US Environmental Protection Agency, 2013b) to calculate the estimated health benefits and financial savings associated with the avoided $PM_{2.5}$ calculated above. I find that reducing emissions of $PM_{2.5}$ from residential wood combustion in the Bay Area would have substantial health and financial benefits, as detailed in Table 4.6.

The scenarios described above could reduce premature mortality in the Bay Area by 20-300 deaths per year, depending on the intervention and calculation method employed. Lost work days could be reduced by 2,000-19,000 per year. Financial benefits are on the order of $130 million - $2.7 billion, expressed in 2010 US Dollars, depending on the calculation method and discount rate (3% or 7%) applied.

Notably, Scenario 5 which would require that approximately 300,000-400,000 frequently used (≥1 per week in the winter) fireplaces be converted to gas inserts, yields estimates of 140-310 (120, 370) avoided premature deaths (depending on whether Krewski 2009 or Lepeule 2012 risk estimates are applied). Scenario 5 would also avert 19,000 (16,000, 23,000) lost days of work, and yield upwards of $1 billion ($870 million, $3.2 billion) in financial benefits.

Table 4.6. Estimated health and economic benefits associated with avoided $PM_{2.5}$ under five modeled scenarios.

Indicator	Scenario 1: EPA Step 1 Scenario	Scenario 2: EPA Step 2 Scenario	Scenario 3: 15% Gas insert scenario
Premature mortality-Krewski (2009)	15 (0, 49)	16 (0, 50)	19 (0, 55)
Premature mortality-Lepeule (2012)	33 (0, 110)	36 (0, 110)	43 (0, 120)
Respiratory emergency room visits	7 (0, 24)	8 (0, 25)	10 (0, 27)
Acute bronchitis	23 (0, 74)	25 (0, 76)	29 (0, 83)
Lower respiratory symptoms	290 (0, 947)	320 (0, 980)	370 (0, 1,100)
Upper respiratory symptoms	420 (0, 1386)	460 (0, 1,400)	550 (0, 1,600)
Minor Restricted Activity Days	12,000 (0, 39,000)	13,000 (0, 41,000)	15,000 (0, 44,000)
Work loss days	2,000 (0, 6,600)	2,200 (0, 6,800)	2,600 (0, 7,400)
Asthma exacerbation	1000 (0, 3,300)	1100 (0, 3,500)	1,300 (3,800)
Cardiovascular hospital admissions	5 (0, 15)	5 (0, 15)	6 (0, 17)
Respiratory hospital admissions	4 (0, 12)	4 (0, 12)	5 (0, 13)
Non-fatal heart attacks (Peters)	15 (0, 50)	17 (0, 51)	20 (0, 56)
Non-fatal heart attacks (All others)	2 (0, 5)	2 (0, 6)	2 (0, 6)
Million $2010 (Krewski 2009, 3% discount rate)	$130 ($0, $420)	$140 ($0, 430)	$160 ($0, $470)
Million $2010 (Lepeule 2012, 3% discount rate)	$290 ($0, $940)	$310 ($0, $965)	$370 ($0, $1,100)
Million $2010 (Krewski 2009, 7% discount rate)	$110 ($0, $370)	$120 ($0, $380)	$150 ($0, $420)
Million $2010 (Lepeule 2012, 7% discount rate)	$260 ($0, $840)	$280 ($0, $870)	$330 (0, $950)

Indicator	Scenario 4: Gas insert scenario	Scenario 5: Pellet scenario
Premature mortality-Krewski (2009)	130 (110, 160)	140 (120, 170)
Premature mortality-Lepeule (2012)	300 (240, 360)	310 (260, 370)
Respiratory emergency room visits	66 (53, 80)	70 (57, 83)
Acute bronchitis	200 (160, 240)	210 (180, 250)
Lower respiratory symptoms	2,600 (2,100, 3,100)	2,700 (2,200, 3,200)
Upper respiratory symptoms	3,800 (3,000, 4,600)	4,000 (3,300, 4,800)
Minor Restricted Activity Days	110,000 (86,000, 130,000)	110,000 (93,000, 130,000)
Work loss days	18,000 (14,000, 22,000)	19,000 (16,000, 23,000)
Asthma exacerbation	9,100 (7,300, 11,000)	9,600 (7,900, 11,000)
Cardiovascular hospital admissions	41 (33, 50)	43 (35, 52)
Respiratory hospital admissions	32 (25, 38)	33 (27, 40)
Non-fatal heart attacks (Peters)	140 (110, 160)	140 (120, 170)
Non-fatal heart attacks (All others)	15 (12, 18)	16 (13, 19)
Million $2010 million (Krewski 2009, 3% discount rate)	$1,100 ($910, $1,400)	$1,200 ($980, $1,400)
Million $2010 (Lepeule 2012, 3% discount rate)	$2,600 ($2,000, $3,100)	$2,700 ($2,200, $3,200)
Million $2010 (Krewski 2009, 7% discount rate)	$1,000 ($810, $1,200)	$1,000 ($870, $1,300)
Million $2010 (Lepeule 2012, 7% discount rate)	$2,300 ($1,800, $2,800)	$2,400 ($2,000, $2,900)

4.5. Discussion

4.5.1 Structure of emission model

The purpose of the model presented here was to evaluate the relative health and economic effects expected to accrue as a result of decreases in emissions from the residential household combustion sector in the San Francisco Bay Area. To understand the magnitude of emission reductions from various hypothetical scenarios, I structured the emission model to include multiple device types (fireplaces, wood stoves, and pellet stoves), frequency of use (more or less than once per week during the burn season), duration of time that devices are used (hours per day per household, which was combined with frequency to produce an estimate of hours per year), and emission rates (g/hr $PM_{2.5}$).

I structure my model around use of an emission rate rather than emission factor for three main reasons: 1) Detailed information about differences in quantity and type of fuel use were not available in BAAQMD survey results for the various classes of wood burning devices in use in the Bay Area. To see benefits from using an emission factor rather than an emission rate, it is necessary to have detailed fuel information. 2) I surmise that respondents are able to more accurately recall or forecast the amount of time that they will use a wood burning device than to accurately estimate and communicate the quantity of fuel they expect to use over the course of a burn event or season. 3) Specific emission rate information, with uncertainty estimates, was available specifically for the Bay Area (Gullett et al., 2003).

Both BAAQMD and CARB emission inventories are structured around estimation of the mass of fuel used (tons of biomass per year), to which an emission factor (lbs $PM_{2.5}$ per ton of fuel burned) is applied. Unfortunately, this made it difficult to directly compare the assumptions and input data used in each model. Emission factors published in CARB and BAAQMD models, as well as those published by Gullett et al. alongside the emission rates used here, are discussed in the next section.

4.5.2 Emission rates and factors

In this analysis, I used emission rates published in Gullett et al. (2003), as detailed above. Emission factors published in the same paper, resulting from the same experimental tests, are listed in Table 4.7, below.

BAAQMD uses the following to inform its emission factors, when calculating emissions from residential wood combustion, with the resulting emission factors shown in Table 4.7:

> "Wood stove and fire place emission factors were based on data obtained from ARB and EIIP volume IV. Composite emission factors were calculated for wood stove criteria pollutants based on an average of conventional and EPA phase II wood stove emission factors. The number of conventional versus EPA phase II wood stoves were derived based on a statistical analysis from the 05-06 winter survey." (BAAQMD, 2014)

CARB published the emission factors used in its calculations of $PM_{2.5}$ from residential wood combustion, as represented in its emission inventory. CARB distinguishes a wide range of technologies, fuels, and associated emission factors (fireplaces, fireplace inserts, certified and non-certified wood stoves, and pellet stoves), as noted in Table 4.7. (California Air Resources Board, 2015). Though CARB's list of emission factors is more extensive and more specific in terms of fuel and device pairings, the factors do not include uncertainty terms and do not seem to be specific to the Bay Area.

Note that, as shown in Table 4.7, the emission factors published in Gullett et al. (6-33 lbs/ton) are much lower for the fireplace tests than those used in the CARB emission inventory (23-46 lbs/ton) or in BAAQMD emission inventory calculations (31.1 lbs/ton). This discrepancy likely explains why the mass PM2.5 emissions I calculate in my analysis are lower than those reported for the residential wood combustion sector in the BAAQMD and CARB emission inventories.

Table 4.7. Emission factors (lbs $PM_{2.5}$ per ton fuel burned) used in CARB emission inventory (California Air Resources Board, 2015), used in BAAQMD emission inventory calculations (Dinh, 2016) and reported in Gullett (2003). Gullett et al. values, published in the paper's Table 7, were converted from g/kg to lbs/ton.

Device type	Fuel type	CARB	BAAQMD	Gullett et al. (2003)
Fireplace	Cord wood, bundles	22.7	31.1	5.6 - 11.1
Fireplace	Manufactured log	46.4		33.2
Wood stove: conventional (non-EPA certified)	Cord wood	29.5	27.5	
Wood stove: Phase II EPA certified, non-catalytic	Cord wood	14.1	14.6	19.2
Wood stove: Phase II EPA certified, catalytic	Cord wood	19.6	14.6	
Fireplace inserts: conventional (non-EPA certified)	Cord wood, bundles	29.5		
Fireplace inserts: Phase II EPA certified, non-catalytic	Cord wood, bundles	14.1		
Fireplace inserts: Phase II EPA certified, catalytic	Cord wood, bundles	19.6		
Fireplace inserts: all	Compressed wood log	25.0		
Pellet stove	Pellets	2.9		

Some argue, however, that the emission factors used in this analysis, while lower than others, effectively *overestimate* effects of residential wood burning on air pollution. The emission factors published in Gullett et al. (2003) were disputed in published correspondence from members of the Hearth, Patio, and Barbeque Association (a solid fuel device trade and lobbying group) who contended that the experimental conditions described in Gullett et al. did not mimic actual burn conditions in the Bay Area and therefore likely overstated effects of emissions on air quality (Crouch and Houck 2004).

Most reported emission rates are calculated based on laboratory tests, rather than in-home or "field" experiments. This means that stoves that meet standards based on laboratory test results may not perform as cleanly (i.e., with as low rate of $PM_{2.5}$ emissions) when routinely used in households (Houck and Tiegs, 1998). This problem is compounded by the fact that stove performance often deteriorates over time (Houck and Tiegs, 1998).

Fireplace emission rates are variable, given the range of behavior in using fireplaces. Houck and Tiegs (1998) identify several factors that cause high emission rates among wood burning devices used in areas of the Western United States that have mild climates, such as the Bay Area: 1) the highest emission rates occur during the initial (kindling) phase of a fire, sometimes leading to about 50% of total emissions from the first 17% of a single burn, and milder climates tend to have shorter burns that are allowed to smolder and go out, rather than being re-stoked, which results in more "cold starts"; and 2) stoves in the Western United States tend to have larger fireboxes than stoves purchased elsewhere, and larger fireboxes often yield lower combustion temperatures, which lead to higher emissions for a given burn rate (mass of fuel burned per unit time) (Houck and Tiegs, 1998). Given these factors, another paper suggests that fireplace emission rates may be as high as 60 g/hr $PM_{2.5}$, which is more than twice the emission rate used here (Houck and Tiegs, 1998).

4.5.3 Assumptions

4.5.3.1 Use of wood for heating

I assume, for the purposes of this analysis, that all residential wood combustion in the Bay Area, as represented in this analysis, is for the purposes of space heating. In reality, some portion of the wood combustion could be for cooking. Some portion could also be classified as for aesthetic purposes only, rather than for heating, although there is not a clear distinction between these two categories, either in the literature or among many in the general public.

4.5.3.2 Exfiltration and fugitive emissions of $PM_{2.5}$

As with the results presented in Chapters 2 and 3, I assume that the escape fraction, or exfiltration rate, of $PM_{2.5}$ from household wood combustion is 100%; that is, I assume that all $PM_{2.5}$ that is produced from the combustion contributes to AAP and that there is negligible deposition on chimneys, inside the house due to fugitive emissions, or on other infrastructure. There are few field-tested results available on indoor or internal deposition of residential combustion particulate matter, but the studies that have been published generally indicate a minimal amount of deposition, in terms of the percent of all particulate matter generated (Lam et al., 2012a).

4.5.3.3 Primary vs. total $PM_{2.5}$

The health and economic results presented in Section 4.3 are based on estimation of primary $PM_{2.5}$ released as a result of biomass combustion for residential space heating. $PM_{2.5}$ is also formed through atmospheric chemical interactions with precursors (NO_x, SO_2, semi-volatile organic compounds) emitted through the same combustion. Since primary $PM_{2.5}$ represents only a fraction of the total $PM_{2.5}$ attributable to this sector, the results presented here likely represent an underestimation of the true health and economic benefits of avoided emissions from this sector.

4.5.4 Interpretation of scenario reduction results

Though Scenarios 1, 2, and 3 have lower uncertainty bounds in the negative range, indicating a hypothetical increase in emissions, it is not expected that emissions would in practice increase. All scenarios involve large-scale transitions to technologies that have lower emission rates. Even taking into account any uncertainty around emission rates for the new technologies that would be implemented in each of the scenarios (gas fireplace inserts, more efficient wood stoves, pellet stoves), one would expect to see a reduction in emissions, assuming that use patterns remain similar. Even if use were to increase to some degree, as a result of the rebound or take-back effect, emissions reductions would most likely still occur. The major challenge, made clear by this sensitivity analysis, is in detecting and understanding the exact magnitudes of emission reductions possible in this sector, given the relative dearth of comparable data on emission factors and emission rates in the household solid fuel combustion sector.

4.5.5 Linearity in health effects calculations

The health effects calculations presented above include factors, published by the US EPA, used in estimating the avoided mortality and morbidity associated with avoided emissions (measured in tons) of $PM_{2.5}$ from the residential wood burning sector. Avoided mortality estimates presented in Table 4.6 include only infant deaths (below one year of age) and adult deaths over age 25 (Krewski et al.) or age 30 (Lepeule et al). The age ranges covered by the morbidity end points are described in more detail in Section 5.3. Use of the factors specified in Table 4.3 implies a linear approach to the relationship between air pollutant emissions and health outcomes. This differs from the method used in Chapter 2 and Chapter 3. The difference between the approaches is described in more detail in Section 5.4.

Recent research, motivated in large part by the 2010 Global Burden of Disease project, has shown that there is substantial evidence of non-linearity in the relationship between air pollution concentrations and health outcomes, over the wide range of concentrations from unpolluted ambient air to that inhaled through active smoking (Burnett et al., 2014).

The 2010 Global Burden of Disease evaluation of the household air pollution and AAP risk factors used a distribution around 7 $\mu g/m^3$ annual mean $PM_{2.5}$ as the counterfactual, or theoretical minimum risk exposure distribution (TMRED). The TMRED represents the counterfactual level when estimating the impact of a non-optimal level of exposure to a risk factor (Burnett et al., 2014). In the Bay Area, background $PM_{2.5}$ levels are estimated, based on long-term data collected in rural Pt. Reyes, north of San Francisco, to be 5.5 $\mu g/m^3$ (Fairley, 2011).

4.5.6 Devices: The fireplace dilemma

Using the model presented here, I found that the vast majority (87%) of Bay Area residential heating combustion emissions originate from solid fuel combustion in fireplaces. This analysis also shows that if just 15% of the *frequent* fireplace users across the Bay Area (34,000 households who report using fireplaces more than once per week during winter) switched to using gas fireplaces instead of burning wood (Scenario 3), the Bay Area would see more emission reductions than if all existing wood stoves (81,000 that are in use, or 170,000 total) in the Bay Area were converted to cleaner-burning Step 2 EPA-certified stoves (Scenario 2).

The health benefits that could be achieved through device change-outs are remarkable. If all of the estimated 230,000 fireplaces that are used more than once a week were converted to cleaner-burning pellet stoves (along with 81,000 wood stoves), at least 120 and as many as 370 premature deaths per year could be averted. For context, that is about 27-85% of the deaths caused by motor vehicle accidents in the Bay Area each year, and two to four times the number of deaths caused by the flu (influenza) each year (California Department of Public Health Safe and Active Communities Branch, 2016; State of California Department of Public Health, 2016).

While in colder climates, where space heating is more routinely required, a higher percentage of households might invest in wood stoves or pellet stoves, these two categories of devices are in the minority in terms of the wood burning devices that households surveyed report having installed in their homes in BAAQMD. There are an estimated 990,000 fireplaces, but only 90,000 pellet stoves and 170,000 wood stoves, in the 1.13 million occupied housing units that have any type of wood burning device in the nine BAAQMD counties (Fairley, 2014).

Fireplaces have, in a sense, become an "elephant in the room," in the discussion of Bay Area residential wood smoke regulation. Although fireplaces will soon be prohibited in new construction, per BAAQMD Regulation 6 Rule 3, the agency stopped short of requiring that fireplaces be retrofitted or removed when an existing housing unit is sold. Moreover, there are no limits on fireplace emission rates or emission factors, given the relative difficulty in retrofitting fireplaces to create more efficient combustion conditions. This is in opposition to wood stoves and pellet stoves, for which increasingly efficient designs are available, and which are subject to increasingly strict limits on the emissions (in mass per hour terms) under new wood burning rules issued by BAAQMD (Bay Area Air Quality Management District, 2015b).

One reason for the lack of regulation of fireplace emissions is the fact that fireplaces are not considered true space-heating devices. Since they often result in a net loss of heat from houses, they are considered recreational devices rather than heaters. While this is scientifically accurate, most fireplace users are likely unaware of this fact and regard fireplaces as a source of heat to employ on cold days, as they do heat the area directly in front partly through radiation.

Prohibiting construction of new fireplaces, as the new wood burning regulations issued by BAAQMD does, is a laudable step toward decreasing emissions from fireplaces. However, this measure does not immediately reduce the number of existing fireplaces in the Bay Area, nor does it affect the number of fireplaces being used each winter. It is dependent on fireplaces being replaced with gas inserts (a fairly expensive undertaking), closed off during remodeling, or removed when a housing unit is eventually demolished.

Given that nearly one million fireplaces appear to be here to stay, at least for the time being, there is a need for further education regarding environmental and population health effects of using them, especially when meteorological conditions trap pollutants within the airshed. BAAQMD has, over the past few years, undertaken a public service campaign to inform residents about the health effects of residential wood smoke. The air district will continue to pursue its efforts in the public education realm by now requiring that pamphlets be distributed when homes with fireplaces are rented or sold, as detailed in the 2015 version of Regulation 6 Rule 3 (Bay Area Air Quality Management District, 2015b).

4.5.7 Applicability of other emission reduction measures

When, in 2006, the US EPA lowered the national 24-hour $PM_{2.5}$ standard from 65 to 35 μg/m^3, BAAQMD adopted Regulation 6 Rule 3 as a way to mitigate wood smoke emissions and meet the federal standard for $PM_{2.5}$. As detailed in Section 1.4, Regulation 6 Rule 3 established the air district's authority to call mandatory burn bans or "Spare the Air Days" on days when $PM_{2.5}$ is forecast to reach unhealthy levels. It also provided guidance on the types of solid fuels that could be burned and set standards aimed at restricting the highest emissions from residential fireplaces.

In 2015, the District released updates to the rules that govern residential wood combustion for heating within the San Francisco Bay Area. The previous version of the rules had been finalized in 2008. Below I summarize what policy levers are currently being used and might be used in the near future to reduce emissions of $PM_{2.5}$ from residential wood burning in BAAQMD. These measures are summarized in Table 4.8.

4.5.7.1 Device switching and emission standards

The air district is employing device restrictions and switch-outs, including financial support, to reduce emissions from residential wood burning. Most of these rules are mandatory (in new construction, for example), but some are voluntary (encouraging switch-out of less efficient wood stoves). Some pertain to an entire device class (such as fireplaces), while others use emissions standards to distinguish among devices within a category (such as wood stoves).

Beginning in November 2016, builders will be prohibited from installing wood burning devices in new construction (Bay Area Air Quality Management District, 2015b). Also, when existing chimneys are remodeled and the cost is over a certain threshold, owners will be required to convert fireplaces to EPA-certified inserts or stoves, or to install gas-fired or electric heaters (Bay Area Air Quality Management District, 2015b).

BAAQMD is employing established emission standards to limit $PM_{2.5}$ emissions from household wood burning devices. Beginning in 2016, any wood heater sold or resold within the air district boundaries must meet emissions rating of 4.5 g/hr, which is equivalent to EPA NSPS Step 1 (modeled here in Scenario 1). By 2020, the stoves must be below 2.5 g/hr (and 2.0 g/hr if tested with cordwood), which is equivalent to EPA NSPS Step 2 (modeled here in Scenario 2) (Bay Area Air Quality Management District, 2015b).

Financial support for owner-occupied unit switch-outs was announcement by BAAQMD in late 2015, in conjunction with amendments to Regulation 6 Rule 3. The air district announced that a $3 million grant program would assist households with wood burning device change-outs, providing >50% of the funds necessary per device. Priority is expected to be given to low-income households and areas with high wood smoke (Bay Area Air Quality Management District, 2015a).

Table 4.8. BAAQMD options for reducing population exposure to residential wood heating emissions.

Policy lever	**Applicability to San Francisco Bay Area**	**Notes**
US EPA New Source Performance Standard	In use	New wood stove use rules released by BAAQMD in 2015 reference NSPS.
State policies	Unlikely	CARB has largely delegated regulation to regional air districts
Local ordinances	In use	49 local ordinances in existence in Bay Area. BAAQMD released model ordinance in mid-1990s.
Wood stove exchanges	In use	$3 million grant to provide >50% changeout cost announced in 2015.
Subsidies for switching fuels	Unlikely	
Subsidies for switching devices	In use	Air district expected to open a grant program for fireplaces and wood stoves in 2016. Priority to low-income residents and high wood smoke areas.
Switch to district heating	Unlikely	Not enough heating demand in SF Bay Area.
HEPA filtration	Possible	Would reduce indoor air pollution but not outdoor air pollution.
Educational campaigns	In use	Ads on buses, brochures for rental units, apps available.
Temporal restrictions on burning	In use	Winter Spare the Air Days announced through media, social media, apps, website.

4.5.7.2 Fuel limitations and switching

The air district also restricts which fuels can legally be burned in residential solid fuel devices. The following fuels are prohibited, according to Regulation 6 Rule 3: garbage, treated wood, non-seasoned wood, used or contaminated wood pallets, plastic products, rubber products, waste petroleum products, paints and paint solvents, coal, animal carcasses, glossy or colored paper, salt water driftwood, particle board, and any material not intended by a manufacturer for use as a fuel in a wood burning device. The purpose of this restriction is to limit both the type of substances emitted, as well as the quantity, since many of these fuels are wetter than seasoned firewood and would smolder rather than burn cleanly.

Landlords and other property owners are also responsible for making sure that residential heating options fit within guidelines, when it comes to fuels: By 2018, landlords must ensure that, if their rental property is in an area with piped natural gas service, they provide a permanently installed source of heat that is not wood burning (Bay Area Air Quality Management District, 2015b).

4.5.7.3 Sub-regional (local) ordinances

Since the mid-1990s, when the air district released a model ordinance for local jurisdictions to use in combatting wood smoke problems, at least 49 local governments have adopted wood smoke ordinances around the Bay Area (Bay Area Air Quality Management District, 2012b).

4.5.7.4 Educational components

BAAQMD recognizes that education and behavioral change are necessary to achieving further reductions in residential wood burning. In 2012, the agency surmised that "emissions from residential wood burning will decrease further in future years in response to continued public education, the gradual phasing out of housing with uncertified wood stoves, conversion of fireplaces to natural gas inserts, and other factors," (Bay Area Air Quality Management District, 2012b).

Education has been a significant part of the air district's push to reduce emissions from wood burning for the past decade. In the air district's 2010 Clean Air Plan, regulators wrote that "[t]he primary focus during the first year of rule implementation [Regulation 6, Rule 3] was to educate the public about the new rule, how to comply and the rule's relevance to public health. The Winter Spare the Air Alert advertising and outreach campaign utilized TV, print, billboard, radio, direct mail, public events, door-to-door canvassing and the Air District website. The District's No Burn phone line received over 500,000 calls. Enforcement focused on providing information to residents on how to comply with the rule, issuing warning letters to first-time violators who did not comply, and developing enforcement action for repeat violators. While household wood burning was reduced by approximately 50% throughout the entire 2008-2009 Winter Spare the Air season, there were still exceedances of the 35 $\mu g/m^3$ [$PM_{2.5}$] standard (Bay Area Air Quality Management District, 2010).

Now that the rule has been updated, starting in June 2016, when any properties within district boundaries are being rented or sold, the person selling or leasing the property must provide information on the health hazards of burning wood or other solid fuels for heat. In addition, anyone offering a new or used wood burning device for sale must include information about the health effects of burning wood, including this statement: "Wood smoke contains harmful particulate matter (PM) which is associated with numerous negative health effects." (Bay Area Air Quality Management District, 2015b)

Firewood offered for sale must include information about wood burning regulation and moisture content. All firewood must include a tag with the following information: "Use of this and other solid fuels may be restricted at times by law. Please check 1-877-4-NO-BURN or http://www.8774noburn.org/before burning." If the wood is seasoned, it will also state: "This wood meets air quality regulations for moisture content to be less than 20 % (percent) by weight for cleaner burning." If it is not seasoned, it must state: "This wood does NOT meet air quality regulations for moisture content and must be properly dried before burning." (Bay Area Air

Quality Management District, 2015b) Unfortunately, this rule is likely to be difficult to enforce, given that many firewood sales are done informally, rather than through stores.

4.6. Conclusions

Household combustion of solid fuels, mostly wood, in fireplace for aesthetic purposes or in other wood burning devices for space heating, is the top contributor to $PM_{2.5}$ air pollution in the San Francisco Bay Area during winter months. The Bay Area is out of attainment for the 24-hour $PM_{2.5}$ standard, and non-attainment days fall in the winter months, when meteorology favors buildup of $PM_{2.5}$ and emissions from residential fuel combustion for heating are especially high (Fairley, 2012).

This chapter presents a model created to 1) estimate the emissions of $PM_{2.5}$ attributable to residential wood combustion in fireplaces, wood stoves, and pellet stoves, and 2) evaluate the emissions, health, and economic implications of five hypothetical scenarios designed to reduce $PM_{2.5}$ from this sector. The scenarios include replacing existing wood stoves with lower-emission EPA-certified wood stoves, retrofitting fireplaces with gas inserts, and converting conventional wood burning devices to pellet stoves. Model inputs include telephone survey results about wood burning practices in Bay Area households, published emission rates designed to mimic Bay Area residential wood burning conditions, and factors estimating the health and economic benefits associated with reduced $PM_{2.5}$ emissions.

Modeled results based on survey data indicate that fireplaces are responsible for the vast majority of residential wood combustion emissions of $PM_{2.5}$ in the San Francisco Bay Area. There are opportunities to avoid premature deaths, serious morbidity outcomes, and financial losses by encouraging replacement of fireplaces with gas inserts, removing existing fireplaces, and upgrading existing wood stoves.

The scenarios presented here show that the most important factor is in lessening the use of conventional fireplaces. From a public health perspective, especially with a focus on $PM_{2.5}$ air pollution, it matters less whether households that currently use fireplaces for heating (or aesthetics) convert to pellet stoves, gas inserts, or electric heating. What matters is that they stop using their fireplaces, because the $PM_{2.5}$ emission rates for fireplaces are so much higher than for any other device in consideration here. (From a climate change perspective, some researchers argue that converting to electricity rather than natural gas might be wiser, given the high radiative forcing associated with the methane production and delivery system.)

This analysis shows that there is a need to generate more device- and fuel-specific emission factors and emission rates for burn patterns in the San Francisco Bay Area. Many of the emission rates and emission factors used in the calculation of mass emissions from the sector, in regional and state-wide emission inventories, rely on relatively generalized emissions information. Residential solid fuel combustion emissions, whether measured per time or per unit fuel burned, are highly variable and difficult to characterize. As illustrated by the model presented above, and specifically the Monte Carlo sensitivity testing, current emission rates (specific to the San Francisco Bay Area) contribute the most to uncertainty in mass emission estimates for the sector. Our understanding of the public health, economic, and air pollution benefits associated with device change-outs and programs to encourage shifts away from inefficient household solid fuel combustion is hindered by a lack of comparable emission factors and emission rates for the sector, especially emission information specific to the San Francisco Bay Area.

Chapter 5 : Household Cooking and Heating with Solid Fuels: Overarching Methodological Issues

In this chapter, I summarize the similarities and differences between methodological approaches, models and data, and descriptive metrics used in Chapters 2, 3, and 4.

5.1. Overview of Data Sources and Models Used

5.1.1 GAINS

GAINS is an energy-driven emissions inventory that provides estimates of primary anthropogenic $PM_{2.5}$ attributable to various activity sector and fuel combinations, by year, at the national or subnational level, for 179 countries. Energy use data are drawn mostly from the International Energy Agency's World Energy Outlook 2011, with some additional information provided by local partners in countries such as India and China (Amann et al., 2011; International Energy Agency, 2011b).

Within GAINS, I used a scenario that draws on data from the International Energy Agency's (IEA) World Energy Outlook 2011 publication (International Energy Agency, 2011a). The IEA energy database distinguishes several categories of solid fuels used in the residential sector, e.g., several coal types, briquettes, biomass, and charcoal. These are principally compatible with the GAINS structure; however, distinctions between fuel wood, crop residue, and dung must be derived from other sources. Further difficulty arises in distinguishing between the use of fuel in households for heating and cooking, specifically in regions where both uses represent a sizable fraction of total use, e.g., northern China. IIASA has ongoing collaboration with several partners in China, India, and Pakistan and uses published sources of information (local reports and peer reviewed research) to fill these gaps. In a number of countries in Asia, GAINS allocates activities also at the subnational level, e.g., provinces in China or India, where information from regional GAINS studies (Amann et al., 2008b; Purohit et al., 2010) is used to scale IEA energy data. The split between cooking and heating in Europe was developed using data from European Commission consultations under the Convention for Long Range Transboundary Air Pollution.

In general, GAINS estimates current and future emissions based on activity data, fuel-specific uncontrolled emission factors, the removal efficiency of emission control measures and the extent to which such measures are applied (Amann et al., 2011). For household cooking with solid fuels from 1990-2010, no technical control measures are applied in the model. Klimont et al. (2002) and Kupiainen and Klimont (2004, 2007) describe the methodology applied to calculate emissions of $PM_{2.5}$, including extension of the model to include primary particulate BC and OC. Any formation of secondary organic aerosol is excluded due to the lack of estimates about the contribution of household sources to precursors in many regions.

5.1.2 MESSAGE

MESSAGE provides global, regional, and spatially explicit emissions of (CH_4), sulfur dioxide (SO_2), nitrogen oxides (NO_X), carbon monoxide (CO), VOCs, BC, OC, and $PM_{2.5}$ (at a 1°x1° resolution). The downscaling methods for the socio-economic and demographic drivers as well as emissions are described in Grubler et al. (2007) and Riahi et al. (2011).

MESSAGE covers all GHG-emitting sectors, including power plants, industry (combustion and process), road transport, households, international shipping and aviation, agricultural waste burning, and biomass burning (deforestation, savannah burning, and vegetation fires) for a full basket of GHGs and other radiatively active gases. In the residential sector, MESSAGE includes an explicit representation of the energy use of rural and urban households with different income levels. Fuel choices at the household level consider the full portfolio of commercial fuels as well as traditional biomass for cooking, heating and specific use of electricity of household appliances (Ekholm et al., 2011).

To estimate the impacts of these spatially explicit emissions, atmospheric concentrations of average ambient population-exposure weighted anthropogenic $PM_{2.5}$ and also specifically the household-related fraction are further derived using the TM5 –FASST source-receptor model. Modeled $PM_{2.5}$ includes contributions from (i) primary $PM_{2.5}$ released from anthropogenic sources and forest fires, and (ii) secondary inorganic aerosols formed from anthropogenic emissions of SO_2, NO_x and NH_3 (including water vapor). The data are reported on a spatial level and are then aggregated by country and GBD region.

5.1.3 TM5-FASST

TM5-FASST provides population-exposure weighted $PM_{2.5}$ concentrations, by country and year. These estimates were developed for the forthcoming outdoor air pollution CRA chapter, based on energy use estimates from the MESSAGE model, also hosted by IIASA. Concentrations represent primary anthropogenic $PM_{2.5}$ and associated secondary particulate formation (calculated by the TM5 transport model) but do not include salt or dust. Population-exposure weighting was done at the sub-grid cell level (0.1° x 0.1°). Data were provided by Rita van Dingenen (EU JRC).

5.1.4 Global Burden of Disease

IHME provides the regional definitions used to express the Global Burden of Disease findings from Chapters 2 and 3 of this study, as well as standard covariates, such as population figures by country and year. Data were provided in spreadsheet format by Heather Adair (WHO). Regional definitions used by IHME and in Chapters 2 and 3 are presented in Appendix C.

5.1.5 True North/BAAQMD Survey

BAAQMD commissioned a telephone survey of Bay Area inhabitants to better understand winter season residential wood burning behavior, awareness, and attitudes. The survey company contacted 1,300 randomly selected residents in the Bay Area airshed to participate in the survey in 2013-2014, during winter months. Sampled respondents were offered the option of participating by telephone or online. The survey firm used probability-based sampling techniques to construct a sample representative of the adult population within BAAQMD boundaries (True North, 2014). From this sample, I use information about number of wood burning devices, frequency of wood burning, duration of burning events, and variation by county.

5.2. Calculation of burden of disease

In Chapters 2 and 3, I estimate the burden of disease associated with exposure to outdoor $PM_{2.5}$ air pollution that can be attributed to household cooking or heating by applying the derived proportions of ambient $PM_{2.5}$ to the GBD 2010 burden of disease estimates for AAP (Lim et al., 2012). GBD 2010 burden of disease estimates were calculated using an integrated exposure response function, as described in Burnett et al. (2014). Results were scaled by applying the proportion of $APM_{2.5}$ due to household cooking with solid fuels (the risk factor) to the burden estimates while preserving the exposure-response relationships used to determine the overall burden of disease attributable to AAP.

5.2.1 Population-weighted values

Results reported in Chapters 2 and 3 are often expressed in terms of population-weighted annual $PM_{2.5}$ concentrations. Population-weighting was performed to determine AAP concentrations published in Brauer (2012) and used in calculations for these studies. The population-weighting was performed by assigning population and air pollution values to 0.1° x 0.1° grid cells, multiplying the values within each cell, summing the resulting cell-specific values by region, and then dividing that sum by the total population (of all cells) for a region (Brauer et al., 2012).

This technique was replicated when calculating regional results for other outcomes in Chapters 2 and 3, such as the proportion of mass $PM_{2.5}$ emissions attributable to household sectors. In that case, population weighting was done when results were calculated at the country level, before they were then reported at the (population-weighted) regional level. The sensitivity analysis section of Chapter 2 summarizes some of the potential uncertainty introduced by this technique and why it was not possible to do a more spatially-explicit analysis for household cooking emissions at the global scale.

The major effect of population-weighting AAP concentrations is to account for the representative exposure experienced by the population living within a given region. If the population is predominately located in and around cities, and the cities have higher $PM_{2.5}$ concentrations than rural areas, then population-weighting will raise the average concentration reported for the region, as compared to the (non-weighted) average concentration. Conversely, if most of the population lives in rural areas with lower $PM_{2.5}$ concentrations, then population-weighting will lower the reported average concentration, reflecting the fact that inhabitants will have lower exposures, on average.

5.2.2 From population-weighted concentrations to exposure

In using the population-weighted concentrations described above to estimate the burden of disease associated with exposure to AAP from household solid fuel combustion, I assume that the derived proportion of the concentrations are representative of population-level exposure to air pollution from household combustion. In many developing country contexts, houses are characterized by high air exchange rates, indicating that background indoor and outdoor air pollution concentrations are similar (Smith et al., 2000) and that exposures are influenced by the "neighborhood" (ambient) pollution that is explored here (Smith et al., 1994; Zhang and Smith, 2007).

In some cases, population-weighted concentrations for given geographic regions may underestimate exposure, if point sources affect densely-populated areas. To correct for this, TM5 uses a subgrid parametrization to redistribute the computed concentrations that might affect urban areas, for example (Brauer et al., 2012). Brauer et al. separated sub-regional sections into urban and rural areas, to produce analyses such as that shown in Figure 5.1. This coding by density allowed for urban/rural sensitivity analysis, as detailed in Chapter 2.

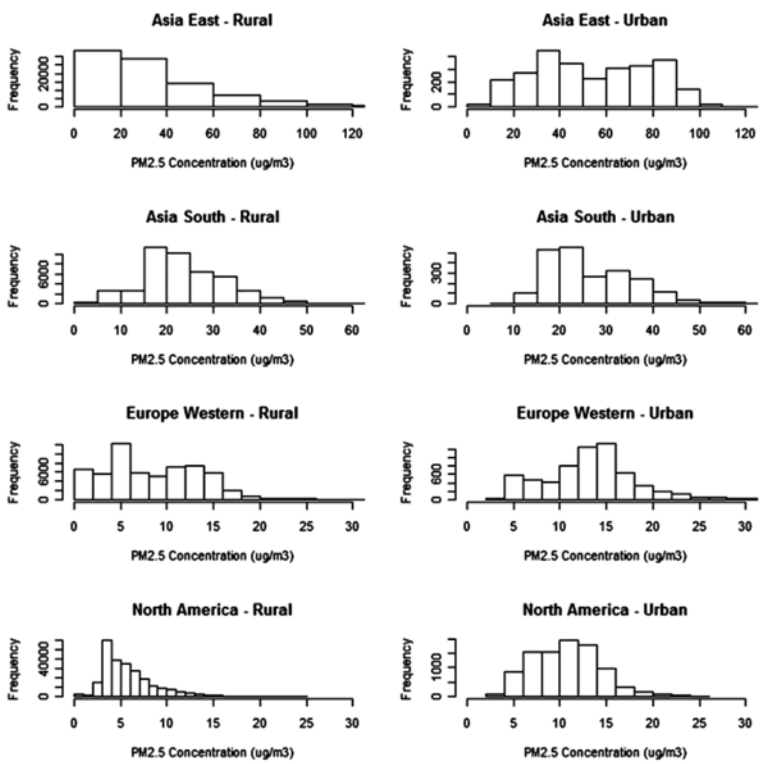

Figure 5.1. Histograms of selected regional (2005) annual average $PM_{2.5}$ concentrations for urban and rural grid cells. Frequency denotes the number of grid cells with concentrations in a given range. Note the difference in scales between regions. Reproduced from Brauer et al. (2012).

Even with attention to urban/rural classification, use of population-weighted AAP concentrations, and recognition of neighborhood-level effects of household combustion pollution, it is possible that results still under-represent and underestimate true exposures to the air pollution generated by household combustion. This is because household combustion sources are, by definition of being in living areas, so much closer to human breathing zones than other sources, such as industrial sources. This proximity likely yields higher intake fractions than most other sources, although at least one study has found intake fraction of $PM_{2.5}$ emissions from residential buildings to be lower than that from traffic (Taimisto et al., 2011). In addition, modeled $PM_{2.5}$ may underrepresent total $PM_{2.5}$ concentrations from this sector because secondary aerosols that are formed downwind from the semivolatile organic compounds released in woodsmoke are not yet incorporated into most atmospheric chemistry models and analyses.

5.3 Calculation of avoided ill-health and mortality

In Chapter 4, I estimate the health benefits associated with reduction of $PM_{2.5}$ attributable to household heating with wood in the San Francisco Bay Area. These health benefits are expressed in terms of avoided premature deaths and avoided ill-health. In this section, I describe the general methods used to determine the factors that were applied in this analysis to estimate the health benefits. These factors were constructed by the US EPA and are published in two relevant reports by that agency (US Environmental Protection Agency, 2013a, b). The descriptions below are summaries of more detailed explanations from the two US EPA reports. I focus here on the demographic groups included and excluded from each class of health benefits reported in Chapter 4.

Reduced incidence of premature mortality from exposure to $PM_{2.5}$ was estimated for some adults and some children, based on (prospective) cohort study estimates and expert elicitation estimates. Two relative risk values are used: one for adults age >30, analyzing the American Cancer Society cohort (Krewski et al., 2009) and one for adults age >25, analyzing the Harvard Six City Study cohort (Lepeule et al., 2012). Krewski et al. represents application of the relative risks derived from the "American Cancer Society" cohort outcomes and employs all-cause mortality risk estimate based on the random-effects Cox proportional hazard model that incorporates 44 individual and 7 ecological covariates (RR=1.06, 95% confidence intervals 1.04–1.08 per 10μg/m^3 increase in $PM_{2.5}$). Lepeule et al. represents application of the relative risks derived from the "Harvard Six Cities Study" cohort outcomes, and uses all-cause mortality risk estimate based on a Cox proportional hazard model that incorporates 3 individual covariates. (RR=1.14, 95% confidence intervals 1.07–1.22 per 10 μg/m^3 increase in $PM_{2.5}$). In this analysis, avoided infant deaths are restricted to 0-1 years (whereas children's deaths are sometimes studied from 0-5yrs in other studies) (Woodruff et al., 1997). To avoid double counting, EPA focused on applying the risk coefficients from the long-term cohort studies rather than short-term studies, since cohort studies likely capture any long-term and short-term impacts.

The avoided morbidity outcomes pertain to either adults, children, or both age groups, depending on the specific factor. Non-fatal heart attacks avoided are estimated for adults >18 years, with different death rates applied for ages 18-44, 45-64, and over 65. Hospital admissions avoided for respiratory symptoms are estimated for all ages, though impacts are estimated differently for ages 0-17, 18-64, and over 65. Cardiovascular hospital admissions avoided are estimates for adults 20-64 and over 65. Emergency room visits for asthma are estimated for children <18 years. Acute bronchitis is estimated for children 5-17 years. Lower respiratory symptoms avoided are estimated for children 7-14 years, and upper respiratory symptoms avoided are estimated for children 9-11 years. Asthma exacerbation avoided was estimated for children 6-18 years; in adults, it was classified instead as avoided lost days of work and avoided minor restricted activity days. Lost work days avoided were estimated for adults 18-65, on the basis of personal symptoms or having to care for a sick family member. Minor restricted activity days avoided were estimated for adults 18-65.

As noted in the underlying EPA documentation, application of these factors to a specific location must be done with caution (US Environmental Protection Agency, 2015b). When developing the factors used here, however, the EPA noted that most if not all of the benefits expected from the policy analyzed (to lower allowable levels of particulate air pollution in the United States) were expected to accrue in California. The analysis in Chapter 4 pertains solely to the San Francisco Bay Area in California. As stated in Chapter 4, the health benefits results presented are rough estimations and would benefit from further refinement.

5.4 Differences in health calculations used in Chapters 2, 3, and 4

Chapters 2 and 3 use the Global Burden of Disease as the basis for calculation of health effects associated with AAP from household fuel combustion. Chapter 4 employs a different method to estimate health effects from similar household fuel combustion, using linearly derived concentration-response functions published by the US EPA.

5.4.1 Global Burden of Disease health effect calculations

The Global Burden of Disease health effects, expressed in premature deaths and DALYs, are calculated at the regional scale for all regions of the world. They incorporate non-linear relationships between concentrations of ambient $PM_{2.5}$ (population-weighted annual averages) and a variety of health outcomes. Determination of the estimated effect of household fuel combustion on AAP, expressed in premature deaths or DALYs per year, is made by deriving the proportion of total $PM_{2.5}$ emissions that are attributable to household fuel use for specific energy services. In this way, the Global Burden of Disease serves as a link between energy use data, emissions information, satellite-derived air pollution calculations, and health impact estimates.

5.4.2 US EPA health effect calculations

The US EPA methodology employed in Chapter 4 assigns linear health benefit factors to each avoided ton of $PM_{2.5}$ released in a given energy use sector. The chapter focuses on residential heating with wood, so health benefit factors for that sector are employed. Underlying this method is the assumption that each avoided ton of $PM_{2.5}$ produces the same health benefits as any other avoided ton of $PM_{2.5}$ (US Environmental Protection Agency, 2013a, b). In reality, we know that health impacts of particulate air pollution are tied to the concentrations of the pollutant; in other words, a given step change in $PM_{2.5}$ concentrations, say from 25 $\mu g/m^3$ to 26 $\mu g/m^3$, will have different health effects than a change in concentration from 250 $\mu g/m^3$ to 251 $\mu g/m^3$.

Chapter 6 : Regulatory Instruments and Climate Change Implications of Household Biomass Combustion

“*There is an urgent need to design and implement an effective approach to limiting black carbon emissions from home heating sources as their use continues to rise*,” (Pearson et al., 2013).

In Chapter 2 and Chapter 3, I outlined the relationship between outdoor air pollution and household combustion of solid fuels for cooking and heating. The extent to which residential cooking and heating contribute to AAP is determined, in part, by mandatory and voluntary policies and regulations (as described in Chapters 4). Often these regulations are created at the local level and are motivated by a desire to protect human health from the negative effects of short- and long-term exposure to fine particulate matter pollution. However, household fuel choices are also, usually indirectly, affected by state, national, and/or regional (e.g., European Community) policies on climate change mitigation.

Household solid fuel combustion regulatory measures and policies will be especially useful if they are designed so that they can shift as our nuanced understanding of the air pollution, health, and climate change implications of household solid fuel use emissions evolve. Biomass and coal burned in cooking or heating stoves often releases pollutants that are harmful to human health and can contribute to climate change. This is especially true when combustion is incomplete (Canada Standards 2012). Climate change mitigation is a secondary (though more complicated) reason for regulation of residential wood burning, since the sector accounts for 12% of the Bay Area’s BC emissions, for example (Bay Area Air Quality Management District 2016).

In this chapter, I review regulatory measures available to decrease emissions from household heating emissions, and household cooking emissions where applicable. I then summarize implications of household solid fuel emission regulation for climate change mitigation, with an emphasis on household biomass combustion.

6.1. Regulatory measures related to heating and cooking emissions

As detailed in Chapter 3, residential solid fuel combustion for heating is a major source of fine particulate air pollution ($PM_{2.5}$) in Europe and North America, generating an estimated 142 kilotonnes of $PM_{2.5}$ per year in Europe, 174 kilotonnes in the United States, and 160 kilotonnes in Canada, according to the most recent estimates available (Canadian Council of Ministers of the Environment, 2012; European Commission, 2015; US Environmental Protection Agency, 2013b).

In this section, I consider the similarities and differences between existing regulatory and voluntary guidelines designed to reduce emissions of fine particulate matter ($PM_{2.5}$) from household solid fuel heating devices.

6.1.1 Emission-based regulations and guidelines for household heating devices

Here, I review three sets of emission-based regulations and guidelines that pertain to household space heating combustion devices. The Ecodesign standards were recently entered into force by the European Commission and will affect space heating combustion device emissions across the European Union. The US EPA New Source Performance Standards set similar limits for combustion products sold in the US. The WHO Indoor Air Quality Guidelines are international in scope and set influential standards for household solid fuel emissions, though they were developed primarily for cooking devices rather than heating devices. The regulations and guidelines are summarized in Tables 6.1-6.3 below.

Table 6.1. Regulatory limits and guidelines based on emission rates (mass per time).

Source	**Device**	**Fuel**	**g/hr $PM_{2.5}$**	**Notes**
Canada Standards Association	Non-catalytic wood burning devices	[Not specified]	4.5	CSA B415.10 (2010)
US EPA NSPS Step 1 (2015c)	All wood stoves and pellet stoves sold in US	Crib wood testing	4.5	Enforceable after Dec 31 2015
Canada Standards Association	Catalytic wood burning devices	[Not specified]	2.5	CSA B415.10 (2010)
US EPA NSPS Step 2	All wood stoves and pellet stoves sold in US	Cord wood testing	2.5	Enforceable after May 16 2020
US EPA NSPS Step 2	All wood stoves and pellet stoves sold in US	Crib wood testing	2.0	Enforceable after May 16 2020
WHO Indoor Air Quality Guidelines	Vented appliances- intermediate emission rate target	[Not specified]	0.429	Level needed to meet WHO Air Quality Guidelines. Intended for cooking appliances. Provided for information.
WHO Indoor Air Quality Guidelines (2014b)	Vented appliances- emission rate target	[Not specified]	0.048	Level needed to meet WHO Air Quality Guidelines. Intended for cooking appliances. Provided for information.
WHO Indoor Air Quality Guidelines	All appliances	Coal (unprocessed)	0	Recommendation 3: Unprocessed coal should not be used as a household fuel

Table 6.2. Regulatory limits based on emission factors (mass of pollutant per mass of fuel).

Source	**Device**	**Fuel**	**g/kg $PM_{2.5}$**	**Notes**
US EPA NSPS (Phase 1)	Fireplaces: low-mass, engineered, pre-fabricated, masonry and site-built masonry	[not specified]	7.3	Voluntary program
European Commission Ecodesign (2015)	Open fronted local space heaters	All solid fuels	6	EU 2015/1185. Enforceable after 1 Jan 2022
US EPA NSPS (Phase 2)	Fireplaces: low-mass, engineered, pre-fabricated, masonry and site-built masonry	[not specified]	5.1	Voluntary program
European Commission Ecodesign	Closed fronted local space heaters	Solid fossil fuel	5	EU 2015/1185. Enforceable after 1 Jan 2022
European Commission Ecodesign	Closed fronted local space heaters	Solid biomass fuels, except compressed wood pellets	2.4-5.0 (depending on measurement method)	EU 2015/1185. Enforceable after 1 Jan 2022
European Commission Ecodesign	Closed fronted local space heaters	Wood pellets	1.2-2.5 (depending on measurement method)	EU 2015/1185. Enforceable after 1 Jan 2022

Table 6.3. Regulatory limits for hydronic heaters and small boilers.

Source	Device	Fuel	g/kg $PM_{2.5}$	Notes
Canada Standards Association	Indoor boilers and furnaces	[not specified]	0.4 g/MJ (0.93 lb/mmBtu)	CSA B415.10 (2010)
Canada Standards Association	Outdoor wood hydronic heaters	[not specified]	0.13 g/MJ (0.30 lb/mmBtu)	CSA B415.10 (2010)
European Commission Ecodesign	Solid fuel boilers <500 kW (manual)	Woody biomass or fossil fuel	40 mg/m^3 seasonal emissions average	EU 2015/1189. Enforceable after 1 Jan 2022
European Commission Ecodesign	Solid fuel boilers <500 kW (manual)	Woody biomass or fossil fuel	60 mg/m^3 seasonal emissions average	EU 2015/1189. Enforceable after 1 Jan 2022
US EPA NSPS Step 1	Residential hydronic heater	[not specified]	0.32 lb/mmBtu (18 g/hr)	Enforceable after May 15 2015
US EPA NSPS Step 2	Residential hydronic heater	Cord wood testing	0.15 lb/mmBtu	Enforceable after May 16 2020
US EPA NSPS Step 2	Residential hydronic heater	[not specified]	0.10 lb/mmBtu	Enforceable after May 16 2020

6.1.2 Ecodesign standards (Europe)

Over the past decade, the European Commission has worked towards regulation of solid fuel local space heaters, particularly those that use various forms of woody biomass fuel (wood logs, pellets and biomass bricks). Broader policy initiatives set the stage for the EU's work in this area and specific regulations to address energy efficiency and emissions have been developed for solid fuel space heaters (ENER Lot 20) and solid fuel boilers (ENER Lot 15) under the Ecodesign directive (European Commission, 2009, 2015).

The Ecodesign regulations for solid fuel space heaters were adopted in 2015, following a 2009 directive mandating the European Commission to develop standards for solid fuel heaters, and then several years of subsequent revision of draft regulations. The regulations apply to solid fuel local space heaters with nominal heat output of <50 kW and are classified under Ecodesign regulation 2015/1185. Though the regulations entered into force in 2015, they will not come into effect until 2022 (European Commission, 2015).

The European Commission estimates that the annual energy consumption related to solid fuel local space heaters was 627 PJ (15.0 Mtoe) in the European Union in 2010, producing 9.5 Mt of carbon dioxide (CO_2) emissions. Without efficiency regulation, the corresponding energy consumption in 2030 was estimated to grow to 812 PJ (19.4 Mtoe) from solid fuel local space heaters and 530 PJ (12.7 Mtoe) from solid fuel boilers, producing slightly less (8.8 Mt) CO_2 from local space heaters (European Commission, 2015).

The European Commission estimates that in 2010, annual emissions of particulate matter (PM) were 142 kton/year from solid fuel local space heaters in member states (European Commission, 2015). With specific measures adopted by Member States and technological development, these emissions are expected to fall to 94 kton/year in 2030.

According to the Commission proposals, implementation of Ecodesign standards would lead to significant energy savings and reductions of CO_2 emissions from solid fuel local space heaters and boilers compared to baseline projections. In 2030 the requirements for those products, combined with energy labeling, were expected to save around 41 petajoules (0.9 Mtoe) per year for local space heaters and 18 PJ (0.4 Mtoe) from solid fuel boilers, corresponding to 0.4 million tonnes and 0.2 million tonnes of CO_2 respectively (European Commission, 2015). They are also expected to reduce $PM_{2.5}$ emissions by 27 kilotonnes per year for solid fuel local space heaters and 10 kt for solid fuel boilers by 2030 (European Commission, 2015).

Some countries in Europe (including Austria, Denmark, Germany, Norway and Sweden) have issued national emission standards for small residential heating installations, which are already in effect. The most comprehensive at this time is a German law of 2010 (quoted in Bond et al., 2013).

6.1.3 New Source Performance Standard (USA)

In the US, the EPA updated its new source performance standard (NSPS) for residential wood stoves in 2015. The new standard contains lower emission rate limits for wood burning stoves (see Table 6.1 for specific emission rate limits). The original standard, created under the Clean Air Act to limit particular matter emissions, was published in 1988, and set limits of 7.5 g/hr for noncatalytic wood heating appliances and 4.1 g/h for catalytic wood heating appliances.

The new NSPS is composed of two progressively lowering emission rate limits. The first level, Step 1, came into force in 2015, and requires that adjustable burn rate wood stoves, single burn rate wood stoves, and pellet stoves, meet a PM emissions limit of 4.5 g/hr. This is equivalent to the 1995 Washington State limit for non-catalytic wood stoves. The second level, Step 2, will come into force in 2020, and requires that wood stoves have an emission rate lower than 2.0 g/hr. This is equivalent to the 1995 Washington State emission rate limit for catalytic wood stoves.

Notably, the updated NSPS applies only to wood heaters that will be sold in the future. It does not apply to devices that burn coal, gas, or oil. It does not cover devices installed prior to implementation of the standards, nor does it apply to fireplaces (since they are not considered effective sources of heat). Many of the increasingly popular residential wood burning devices, such as hydronic heaters and forced air furnaces, are addressed by a separate standard mentioned in the NSPS documents. The NSPS also does not apply to devices that are used primarily for cooking, such as camp stoves, cook stoves with an oven, or traditional Native American bake ovens.

An EPA voluntary certification standard for low-mass fireplaces (5.1 g/h) was included under the NSPS revision, as well as a standard for masonry heaters (2.0 g/h daily average; 0.32 lb/million British Thermal Unit (mmBTU) [around 0.14 g/megajoule]) and single burn-rate stoves (3.0 g/h).

Hydronic heaters, whether located indoors or outdoors, must meet a Step 1 limit of 0.32 lb/mmBtu in 2015, and the Step 2 limit of 0.10 lb/mmBtu in 2020. A hydronic heater is a wood fired boiler, often located outside the building for which it is generating heat – in a shed, for example – that heats a liquid (water or water/antifreeze mix) and then uses this to circulate heat. To promote the production and sale of cleaner and more efficient outdoor hydronic heaters, EPA currently runs a voluntary certification program for manufacturers. Certified outdoor hydronic heaters at the most stringent certification level ("phase 2") are about 90% cleaner than uncertified models. Even outdoor hydronic heaters qualifying for phase 2 certification, however, still emit one to two orders of magnitude more $PM_{2.5}$ on an annual average emission rate basis than residential oil or gas furnaces. A number of state and provincial jurisdictions have also adopted setback distances (distances from buildings or other structures deemed to need protection) of 30–150 m, depending on emissions certification, for outdoor hydronic heaters.

Additionally, small forced air furnaces (<65,000 Btu/hr) will need to meet a Step 1 emission requirement of 0.93 lb/mmBtu in 2016, which is identical to Canada's CSA standard for indoor furnaces and boilers. By 2017, larger forced air furnaces (≥65,000 Btu/hr), which make up approximately 75% of forced air furnace sales, will also need to meet this requirement. All forced air furnaces will need to have emission rates below 0.15 lb/mmBtu, which is stricter than the current CSA standard for either indoor or outdoor boilers or furnaces, by 2020.

Though the standards detailed here focus on reducing emissions of particulate matter, the NSPS (like the Ecodesign standards) includes minimum efficiency requirements for a number of appliances, with the aim of reducing CO emissions.

6.1.4 Canada Standards Association (Canada)

Model standards for wood burning emissions exist in Canada as well, and residential wood burning has been prioritized as a sector in which contaminant emissions can be reduced. (Canada generates an estimated 104,087 tonnes/year of $PM_{2.5}$ from residential wood combustion.) CCME participated in an initiative to update the Canadian Standards Association (CSA) standards for new wood burning appliances (CSA Group, 2010). These standards were adopted in 2010, lowering the PM emission rate to 4.5 g/h for noncatalytic wood heating appliances and to 2.5 g/h for catalytic wood heating appliances. They also established emissions limits of 0.4 and 0.13 g/megajoule for indoor boilers/furnaces and outdoor hydronic heaters, respectively.

In Canada, the relevant standard (CSA B415.10) is "a consensus-based standard intended to provide appliance manufacturers, regulatory agencies and testing laboratories in Canada with methods for determining thermal efficiencies, particulate emissions and flue gas flow rates of solid fuel burning appliances." It is not a regulatory standard as the NSPS is in the US.

6.1.5 WHO Indoor Air Quality Guidelines

WHO recently released indoor air quality guidelines for household fuel combustion (WHO, 2014a). The guidelines describe the household combustion technologies and fuels (and associated performance levels) needed to prevent the negative health effects currently attributable to this source of air pollution. The implied focus of the report and guidelines is household cooking with solid fuels, in part due to the model inputs and assumptions made in developing the guidelines and emission rate targets (see Table 6.1). However, the guidelines are written with language that makes them applicable to household space heating as well. Recommendations pertinent to household space heating include:

- setting emission rate targets (see the guidelines for specific target values) for both vented and unvented household stoves (for $PM_{2.5}$ and CO);
- encouraging governments to accelerate efforts to meet air quality guidelines, in part by increasing access to and encouraging sustained use of clean fuels and improved stoves, including maintenance and replacement of the stoves over time;
- preventing use of unprocessed coal as a household fuel, given that indoor emissions from household combustion of coal are carcinogenic to humans, according to the International Agency for Research on Cancer (IARC, 2010) – note that unprocessed coal is distinguished here from so-called “clean” or “smokeless” coal, for which less research on health effects has been done;
- discouraging household combustion of kerosene since there is strong evidence that heating with kerosene leads to indoor concentrations of $PM_{2.5}$, nitrogen dioxide (NO_2) and sulfur dioxide (SO_2) that exceed WHO guidelines, and household use of kerosene also poses burn and poisoning hazards;
- encouraging governments to maximize health gains while designing climate-relevant household energy actions.

6.1.6 Implications of national and international regulatory measures for heating emissions

In general, the development of the standards summarized here, as well as other voluntary and mandatory household solid fuel combustion emission standards, appear to be helpful in incrementally reducing emissions from certified appliances, encouraging awareness of the health and environmental effects associated with emissions of products of incomplete combustion from household heating (and cooking) devices, and perhaps most importantly, setting a standard that can be referenced by international, national, state, regional, and local municipalities seeking to limit pollution from these devices.

In Chapter 4, I mention the existence of nearly fifty city and county ordinances that regulate wood burning devices in the San Francisco Bay Area alone. Many, though not all, of those ordinances refer to EPA certified wood stoves and pellet stoves when describing which devices may or may not be installed in new construction, installed as part of retrofits, or used during particular days when air quality is forecast or measured to be unhealthy (Bay Area Air Quality Management District, 2012a). Most, if not all, of these ordinances were passed before the EPA updated its NSPS emission limits, but the ordinances were often written to account for future updates to the limits, referring to “EPA-certified” appliances rather than to a specific emission rate or emission factor within the ordinance itself (see City of Oakland, California Ordinance

Number 12671, for example), though some refer to specific emission rates and would need to be amended to reflect current EPA emission limits as they change over time (see City of Berkeley Municipal Code, Chapter 15.16, which refers to a 7.5 g/hr emission rate to set compliance, for example).

6.2. Climate Change Mitigation Implications of Household Solid Fuel Combustion Regulation

6.2.1 Climate Change Mitigation Potential of Household Biomass Fuel

One of the reasons that residential wood burning persists in developed countries, including the United States, is the popular understanding that it is a "climate-neutral" heating option. This opinion was voiced by attendees at a series of public workshops held in the San Francisco Bay Area to gather comment on proposed amendments to BAAQMD's wood burning rule in 2015. More broadly, and at higher levels of government, the perceived climate benefit of residential wood combustion guides planning as governments consider ways to reduce fossil fuel use and concomitant carbon emissions, especially in European countries where climate change mitigation targets are well established.

Biomass is often touted as a renewable fuel that can assist with climate change mitigation and reduce reliance on fossil fuels (Alliance for Green Heat 2015). Whether a specific type and source of biomass is indeed considered a renewable fuel, meaning that it can be regenerated over time, is dependent on factors such as the type of biomass, the long-term ecological health of the biomass growing location, and whether the biomass is planted or naturally regenerates after harvest. However, these factors are not always considered in analyses of renewability (Cherubini and Strømman, 2011).

To complicate matters even further, the climate implications of a specific fuel depend on combustion conditions as well. As shown in Figure 6.1, the device in which a fuel is burned has a role in determining impacts on climate.

6.2.1.1 European Union

In the European Union, many countries have developed climate change mitigation strategies that favor expansion of biomass combustion at the household level. For example, the United Kingdom's Renewable Heat Incentive, introduced in 2014, explicitly includes payment to households using biomass boilers as part of the strategy to reduce the country's GHG emissions by 80% (from 1990 levels) by 2050 (Ofgem, 2014).

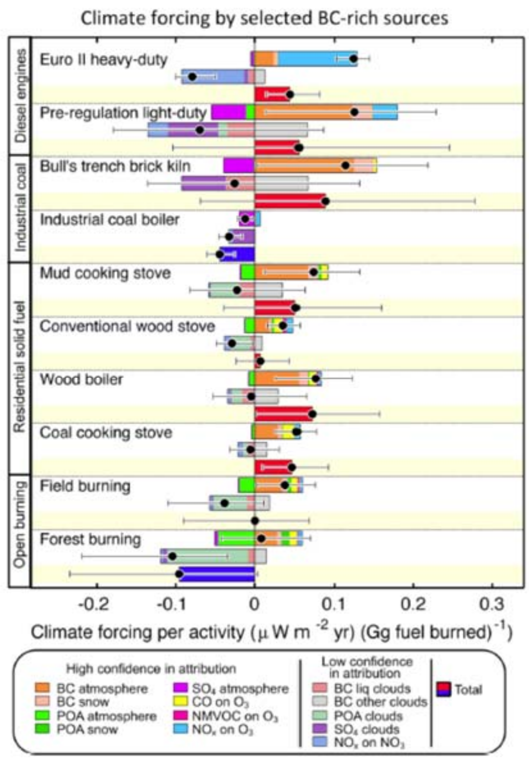

Figure 6.1. The BC:OC ratio changes depending on fuel type, device type, and energy service provided. See for example mud cooking stove and conventional wood stove. Figure from Bond et al. (2013).

Biomass fuels were also included in the European Commission's strategy for reaching the "202020" targets (20% reduction in GHG emissions, 20% of final energy consumption from renewable energy and 20% increase in energy efficiency by 2020) (European Climate Foundation, 2010). Heating is currently the largest end use of biomass in the EU, and is projected to continue to be into the near future (90.4 Mtoe in 2020) (European Commission, 2014). (See Figure 6.2.) However, much of the new biomass use in the EU has been for electricity production rather than household heating (European Climate Foundation, 2010). The World Bank and others have recognized that these trends will likely have bad climate consequences, writing that "There is an urgent need to design and implement an effective approach to limiting black carbon emissions from home heating sources as their use continues to rise," (Pearson et al., 2013).

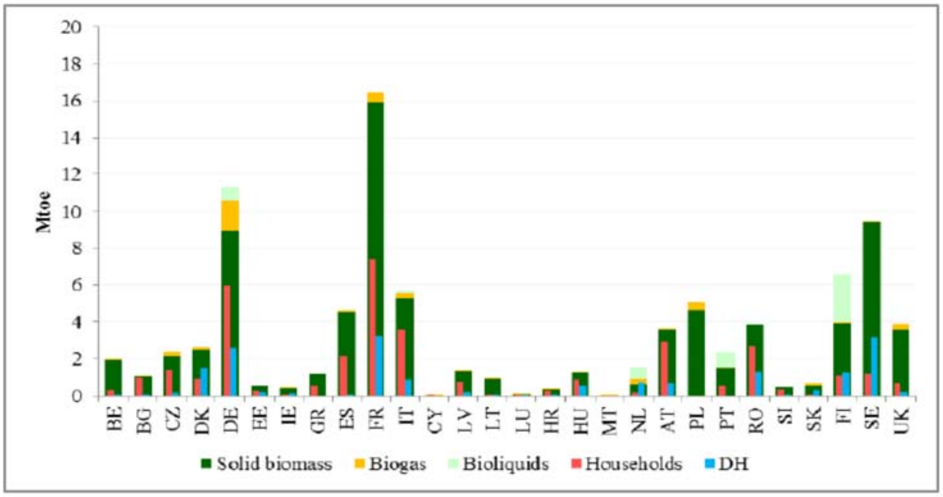

Figure 6.2. Heating and cooling demand from solid biomass and biogas in households and district heating (2020, Mtoe). Source: National Renewable Energy Action Plans.

6.2.1.2 San Francisco Bay Area

BAAQMD, which is aware of and interested in exploring the climate change implications of heating fuels, maintains a GHG emissions inventory. The most recent version available, which was published in 2008, states that household solid fuel combustion across the Bay Area accounts for 6,242 metric tons/year of methane (CH4) and 67 metric tons/yr of N2O, or a total of 151,742 metric tons/yr in CO_2 equivalent (CO_2e) terms. It also causes the release of 628,550 metric tons/yr of biogenic CO_2, which is four times the CO_2e (by mass) of the N2O and CH4 released by the same fuel burning. Household combustion of solid fuels accounts for 33% of all biogenic CO_2 released by any included source in the Bay Area.

Included emissions from household solid fuel combustion (CH4 and N2O) equate to 2% of all CO_2e emissions from residential fuel use (natural gas, liquid fuels, and solid fuels). Note that while all household solid fuel use can be assumed to be for space heating, the same is not true for the other included fuel uses, which might include water heating and perhaps cooling as well.

Notably, this inventory, as its name (Greenhouse Gas Emissions Inventory) implies, does not include climate active particles, such as BC. Inclusion of all climate active pollutants emitted by residential solid fuel combustion would likely significantly change the balance of warming and cooling substances emitted.

6.2.2 Co-benefits for local air quality and health

Of the many ways that climate and health intersect, household solid fuel use is an area that has important short-term climate implications, one of the highest associated burdens of disease of any analyzed risk factors, and interventions that might lessen both the climate and health effects (Smith et al., 2014; Smith et al., 2009; Unger et al., 2010; Wilkinson et al., 2009).

On the positive side, climate change policies that are generally pro-household biomass combustion can sometimes protect local air quality as well. For example, policies that lead to more complete combustion of solid fuels, such as stove change-out programs, can reduce emissions of climate-active pollutants (such as BC) that harm local air quality. When climate change mitigation strategies have benefits for health as well, they are often called "co-benefits" or "no-regret" approaches.

6.2.2.1 Co-benefits for health from climate change mitigation

Co-benefits are health benefits that come along with actions that are primarily motivated by an interest in mitigating climate change; or climate mitigation benefits produced by actions that are primarily motivated by an interest in improving public health. Reducing emissions of health-relevant air pollutants, especially those that are also climate-active pollutants (especially CH4 and BC), can have short- and medium-term co-benefits for health; they can also immediately reduce exposure to associated particulate air pollution. Accounting for these health co-benefits can produce a more complete economic picture of the costs and benefits associated with efforts to reduce heating-related emissions, such as wood stove change-out programs.

Originally called "win-win" or "no regrets," the term "co-benefit" denotes, in the climate change context, a mitigation strategy that carries related but outside benefits, usually to human welfare or health (Nemet et al., 2010). The Organization for Economic Cooperation and Development (OECD) distinguishes this from benefits to local air quality, which it terms a potential "double dividend" of GHG mitigation efforts (Bollen et al., 2009).

Co-benefits are usually expressed in units of health (avoided deaths, disability-adjusted life years, health-related dollars) per units of climate active pollutant mitigated (tons of CO_2 or CO_2e). The concept of disability-adjusted life years (DALYs) was first described around 1990 and has been widely adopted since then, with the goal of integrating morbidity and mortality into a single metric. It is used, perhaps most prominently, in the Global Burden of Disease and Comparative Risk Assessment studies, which calculate health burdens by disease and by underlying risk factor in such a way that they can compared (Ezzati et al., 2004).

Several studies conclude that the most significant opportunities for health co-benefits exist in countries with higher existing levels of climate active pollutant emissions; these are often developing countries in which emissions reduction laws have not been enacted or are not widely enforced (Nemet et al., 2010). In many cases, as in China, failure to consider co-benefits has resulted in incomplete and inflated estimates of climate active pollutant mitigation costs (Aunan et al., 2006). Smith and others conclude that most co-benefit estimates represent a conservative valuation approach because they often only include select climate pollutants and only the health endpoints for which the most conclusive evidence exists (Smith and Haigler, 2008).

6.2.2.2 Examples of co-benefits from reducing household solid fuel emissions

The World Bank found that replacing current wood stoves and residential boilers used for heating, with pellet stoves and boilers, and replacing chunk coal fuel with coal briquettes (mostly in Eastern Europe and China), could provide significant climate benefits and would save about 230,000 lives annually, with the majority of these health benefits occurring in OECD nations (Pearson et al., 2013). Note that the continued use of coal for residential heating is not recommended; however, the use of coal briquettes was factored into the scenario detailed here.

Another study coordinated by the UN Environment Programme (UNEP) and the World Meteorological Organization (WMO) found that widespread dissemination of pellet stoves (in industrialized countries) and coal briquettes (in China), for BC mitigation, could improve health, since these interventions lead to reductions in $PM_{2.5}$. Major reductions in annual premature deaths include expected as a result of these interventions include about 22,000 less deaths in North America and Europe, 86,000 less deaths in East, Southeast Asia and the Pacific, and 22,000 less deaths in South, West and Central Asia.

6.2.2.3 Tensions between biomass use and health

However, policies that advocate broadly for increased biomass combustion at the household level may actually decrease local air quality while potentially mitigating climate change. This tension has been documented in several EU government documents, one of which shows that diverse stakeholders are concerned about the issue.

The European Commission itself noted, in a 2013 report, that, when it comes to household solid fuel combustion, "[t]he problem is not only continuing coal use, but also increase in biomass use, driven partly by renewables policy and (more recently) by the economic crisis," (European Commission, 2013).

In a public consultation on European Union air policy, 19 of 40 stakeholder respondents (from the EU Stakeholder Expert Group on the Air Review), including member states, business associations, and environmental organizations, favored better alignment between air quality policy and climate change policy, agreeing that "[t]rade-offs with climate change policy must be taken into account," including, specifically, the "negative impacts on [air quality] by more biomass combustion," (TNO, 2012).

6.2.3 Further work on climate change aspects of household biomass combustion

The issues laid out in this chapter form the basis for future work on the topic of household combustion of biomass, especially for heating, as it relates to climate change mitigation policies and strategies. Table 6.4 suggests just three of the synergies or tensions in this sector that could come to bear through current or proposed near-future policy developments in California.

As climate mitigation strategies are proposed and debated, there will be a need to better understand the exact relationship between household biomass use for heating, local air quality, and human health, especially in relation to other heating fuel options. Many environmental organizations are transitioning their messaging to advocate for electrification of the household heating sector in California, based primarily on climate-related concerns about long-term use of natural gas, and will be evaluating how to provide guidance on biomass fuels. Without better information about the sector, including reliable emission factors, good underlying data on the prevalence of solid fuel use in the sector, and a strong understanding of whether it will be feasible to encourage device change-outs in the near future, it will be difficult to evaluate biomass against other fuel options.

Table 6.4. Synergies and tensions in California climate change and air pollution mitigation strategies and policies.

Synergies	Reasoning/Rationale	California policy example
Air districts issue temporary bans on residential wood burning	Reduces $PM_{2.5}$ emissions, including BC, and improves local air quality	BAAQMD restrictions on residential burning
Use of cap and trade revenue to fund wood stove change-out programs	Reduces $PM_{2.5}$, including BC, by decreasing emissions from wood stoves	Proposal in draft concept paper for three-year CARB Greenhouse Gas Revenue Fund (GGRF) Investment Plan
Tensions		
Encouraging increased/continued use of biomass combustion, to reduce reliance on fossil fuels	Could increase $PM_{2.5}$ emissions, including BC, if combustion is not complete	Money from cap/trade revenue (Greenhouse Gas Revenue Fund) diverted to fund biomass energy (ex: AB590—proposed bill)

Chapter 7 : Conclusion

7.1. Summary of previous chapters

In the preceding chapters, I asked how household combustion of solid fuels for cooking and space heating affects outdoor air pollution and human health, and how changes in policies or regulations relevant to the residential solid fuel use sector might affect emissions in the future. My focus has been on fine particulate matter ($PM_{2.5}$), just one of many air pollutants created through the incomplete combustion of solid fuels.

In Chapter 2, I found that household cooking with solid fuels caused an estimated 12% of population-weighted ambient $PM_{2.5}$ worldwide in 2010. Exposure to this air pollution caused the loss of 370,000 lives and 9.9 million disability-adjusted life years (DALYs) globally in the same year.

In Chapter 3, I found that household heating with solid fuels caused an estimated 21% of population-weighted ambient $PM_{2.5}$ in 2010 in Central Europe, 13% in Eastern Europe, 12% in Western Europe, and 8% in North America. Exposure to this air pollution results caused approximately 60,000 premature deaths in Europe, and nearly 10,000 deaths in North America, as well as an estimated 1.0 million disability-adjusted life years (DALYs) in Europe and 160,000 DALYs in North America.

In Chapter 4, I found that fireplaces are the source of the vast majority (84%) of $PM_{2.5}$ from residential wood combustion in the San Francisco Bay Area, despite their use primarily as an aesthetic or recreational combustion activity. I also found that replacing fireplaces with gas would yield significant health and economic benefits. Retrofitting frequently used fireplaces (300,000 units) to gas inserts in the Bay Area's nine counties would reduce sector emissions by about 90%, avoiding 140-310 premature deaths and 19,000 lost days of work each year, and creating upwards of $1 billion in annual financial benefits from improved public health.

In Chapter 5, I describe the methodological similarities and differences between the approaches taken to estimate air pollution and health effects associated with household use of solid fuels, at two different scales (global and local) in the previous chapters. I summarize the data inputs and metrics used to evaluate health impacts, emphasizing that while premature deaths are a common metric shared among the chapters, the ways of calculating impacts differ fairly significantly.

In Chapter 6, I outline current and proposed regulatory and policy mechanisms related to the household solid fuel use sector, most of which have the goal of reducing $PM_{2.5}$ emissions associated with solid fuel combustion. I then relate those mechanisms to the current challenge of designing policies that simultaneously work toward climate change mitigation goals and local air quality goals. One of the specific challenges embedded in this process is the relatively dearth of data on household heating with biomass and its nuanced climate implications. I suggest areas for future work on the topic of climate change and household biomass use, outlining areas where this debate is currently visible in California.

7.2. Research needs and ways forward

Across the analyses explained in the previous chapters, several common points emerge as cross-cutting findings.

7.2.1 Household Heating Data

Particulate emissions created by household combustion have long been an under-appreciated and often disregarded source of outdoor air pollution and ill-health. However, projections from within the United States and in most countries around the world indicate that emissions from the household solid fuel use sector will persist for decades to come, especially for household space heating, as findings presented in earlier chapters indicate.

There is a need to collect household solid fuel use data even more carefully, systematically, and widely than is occurring now. This is especially true for household heating, which has been somewhat overlooked as international attention focuses on household heating. Collecting, comparing, and refining measurements of emission factors and emission rates is especially important, something that was highlighted in the findings of Chapter 4. Specifically, there is a need for more information on solid heating fuels, fuel use patterns, device installation and use, and associated emissions. Fewer than 40 surveys provide reliable data on primary heating fuels; only 14 of these are conducted in low- or middle-income countries (World Health Organization, 2016).

7.2.2 Cooking-heating Continuum

Throughout this dissertation, household solid fuel use is cleanly separated as being either for the purpose of providing energy for cooking or for the purpose of providing space heating. In reality, this is a somewhat misleadingly characterization, as there are many situations in which these two fundamental energy services overlap. This is a concept easily observed in many rural Chinese villages, for example, where coal stoves often provide space heating and are also used for cooking tasks. Some EPA-certified stoves sold in the United States are marketed as having upper surfaces with two different temperatures, to accomplish different water-boiling or cooking tasks while the stove provides space heating.

Going forward, it will be important that household fuel use surveys ask questions about all energy services provided by all appliances in a household, to better understand what energy services might need to be replaced when shifting away from a relatively inefficient stove or open fire. For a family that is used to cooking and heating with one stove, introducing an efficient cookstove will only reduce emissions if a low-emission source of heat is available as well. Gathering more complete data on the overlap between cooking and heating may enable the design of more successful interventions, in terms of both reducing emissions and protecting health.

7.2.3 Urban-Rural Continuum

In Chapters 2 and 3, I make a distinction between urban and rural areas, pointing in some sections to the relative lack of an internationally-accepted definition for these terms, at least in terms of data classifications. In reality, there is much more of a continuum between rural and urban areas than might be indicated by binary analyses. The work done by Brauer et al. (2012) to better characterize household emissions outside of cities is an important step forward in understanding the magnitude of air pollution and associated health effects for the half of the world living beyond city boundaries. One of the next steps must be an improvement in on-the-ground monitoring equipment in sensitive peri-urban and rural areas, to complement the satellite measurement on which we rely now (Balakrishnan et al., 2014).

7.2.4 Terminology

Using exact terminology when describing this sector will assist researchers both within and outside the field in better understanding research questions and findings, especially in interdisciplinary projects.

Using the terms "household" or "residential" is preferable to "domestic," for example, which can be understood as referring to emissions or fuel use within a given country. There is also a tendency within some disciplines and research communities to refer to solid biomass fuels as "biofuels," which is confusing given that second-generation (synthesized liquid) fuels derived from biomass are more commonly thought of when that word is used.

Even the term "biomass" itself can be confusing. As a type of fuel, is a term often used too ambiguously within climate change policy (and local air quality policy) documents. It can refer to direct combustion of wood at the household level, pellet production with lower-quality fuels (such as agricultural waste), centralized industrial-scale combustion of low quality fuels for electricity generation, and can even encompass biofuels (liquid fuels made from biomass), as mentioned above.

Within climate change documents, the terms "sustainable," "carbon neutral," and "zero emissions" are often misused with regard to biomass combustion, especially in climate change mitigation policy documents. There is a need for careful use of these terms in conjunction with household cooking or heating research, since they may be misconstrued or misinterpreted, especially in policy settings. To be climate neutral, biomass fuels must be both renewably harvested and used with near perfect combustion efficiency to avoid production of short-lived climate pollutants such as BC.

7.3 Future research needs

The international community faces major, and in many cases unprecedented, environmental and health challenges as a result of increasing globally averaged surface temperatures. The Intergovernmental Panel on Climate Change Fifth Assessment report (AR5) determined that "Human influence on the climate system is clear, and recent anthropogenic emissions of greenhouse gases are the highest in history. Recent climate changes have had widespread impacts on human and natural systems," (IPCC, 2014). Anthropogenic greenhouse gas concentrations are influenced by human-related emissions of the so-called "Kyoto gases" (CO_2, CH_4, N_20, HFCs, PFCs, and SF_6) and other climate active pollutants (such as BC and OC) from industrial processes, electricity generation, agricultural production, transportation, waste, and household cooking, heating and lighting.

As researchers and policy makers consider how to effectively manage air pollution problems (which continue to grow in many developing countries), ensure health care for acute and chronic illnesses, and mitigate climate active pollutants, household solid fuel use will need to be considered, perhaps even more than in past years. Open questions, such as how carbon credits based on household fuel use are calculated, what impact various household fuels have on the climate, what role co-benefit calculations will have in policy making, and who will fund climate-motivated interventions, make this a complicated and interesting area of investigation.

For decades, research on climate change and environmental health has happened largely in parallel. Though this is a natural outcome of disciplinary academic systems, it represents a sometimes limited way of understanding the many complex global problems of which we are now aware. My work seeks to bridge the gap between these two intimately related areas of study, by studying their intersections (especially as it pertains to household solid fuel use in developing countries) and the ways these intersections might be better understood to create more effective climate and health policy in the future.

7.4. Future work

There is an urgent need to better understand trends in this energy sector worldwide, so that the role of residential wood combustion for heating in meeting health and sustainability goals can be rigorously assessed at both local and global scales. Below I describe two projects that will allow me to assist with the task of better understanding the emission factors, fuel use, and device use related to household heating in other regions; and to better understand the role of household biomass use in ongoing climate change mitigation policymaking.

7.4.1 Data collection on solid fuel heating in Asia and Latin America

As explained in Chapter 3, I was not able to do a global analysis of health effects and AAP from solid fuel heating because of the lack of data for many world regions. There is an urgent need to better understand household solid fuel use for heating in Latin America and Asia, especially China.

Two recent studies of household energy use in China have helped begin this process. One found that, in 2010, 88% of heat supplied in Northern China was provided by district heating, with a very small share of 12% coming from individual small coal boilers (Khanna et al., 2014). Additionally, approximately 34.1% of households in China had no means of household heating, in some cases because they live in a climate zone that does not require heating. The other, Duan et al. found that, nationwide, 10.3% of households used a central heating system, and 8.9% of households used individual heating, in which natural or liquid petroleum gas was burned to heat water. The proportions of households using coal, electricity and biomass for heating were 16.7%, 15.6%, and 12.8%, respectively (Duan et al. 2014). In rural areas, most households lacked central heating, so residents largely burned traditional solid fuels (21.5% used coal and 19.0% used biomass) for heating. In urban areas, approximately 22.0% of households had central heating. Most urban residents used electricity/solar power (23.6%), followed by coal (10.5%) for the household heating. This is apparently the first national survey to report household heating data in China.

In partnership with researchers who run the GAINS and MESSAGE models at the IIASA, and in-country colleagues, I will analyze recently published data (Duan et al., 2014) and unpublished survey results to update and expand estimates of household heating fuel use and associated emissions in China, India, and other countries, with the goal of integrating this data into existing emission inventories. This work will extend current collaboration with colleagues who work on household heating and air pollution issues at Peking University and Tsinghua University (China), Sri Ramachandra University (India), University of British Columbia (Canada), and Aarhus University (Denmark), among others.

7.4.2 Household biomass fuels in climate policy documents

Beginning with CARB's draft strategy to reduce short-lived climate pollutants, and relevant negotiated texts released after the December 2015 UN Framework Convention on Climate Change Conference of the Parties (known more colloquially as the "Paris Climate Summit"), I will assess policy documents for treatment of terms such as "biomass," "sustainability" and "renewability." My hypothesis is that the term "biomass" is often used in such a general way, in climate policy documents, that it loses most meaning, because biomass can be divided into specific fuel types (ethanol, pellets, firewood, agricultural waste) that have very different combustion characteristics, especially when seen from a lifecycle analysis perspective; and the range of combustion conditions possible (biomass-to-electricity plant, efficient pellet stove, fireplace) is very wide and has substantial impact on the emissions produced. The goal of this objective will be to point out places where climate change policy guidance on biomass fuels is lacking necessary level of detail to the point that it is not helpful for guiding decisions about sustained use of biomass fuels for household heating; and to raise the profile of this important emissions sector (household heating with biomass) in the context of climate change policy formation.

References

Ahmad, S., Baiocchi, G., Creutzig, F., 2015. CO2 Emissions from Direct Energy Use of Urban Households in India. Environ. Sci. Technol. 49, 11312-11320.

Ahmad, S., Puppim de Oliveira, J.A., 2015. Fuel switching in slum and non-slum households in urban India. Journal of Cleaner Production 94, 130-136.

Alliance for Green Heat, 2011. 2010 census shows wood is fastest growing heating fuel in US, Heated Up!, Takoma Park, MD.

Amann, M., Bertok, I., Borken-Kleefeld, J., Cofala, J., Heyes, C., et al., 2011. Cost-effective control of air quality and greenhouse gases in Europe: Modeling and policy applications. Environmental Modelling & Software 26, 1489-1501.

Amann, M., Bertok, I., Borken, J., Chambers, A., Cofala, J., et al., 2008a. GAINS Asia: Methodology. IIASA, Laxenburg, Austria.

Amann, M., Kejun, J., Jiming, H., Wang, S., Xing, Z., et al., 2008b. GAINS Asia Scenarios for Cost-Effective Control of Air Pollution and Greenhouse Gases in China International Institute for Applied Systems Analysis, Laxenburg, Austria.

Ancelet, T., Davy, P., Trompetter, W., Markwitz, A., Weatherburn, D., 2013. Carbonaceous aerosols in a wood burning community in rural New Zealand. Atmospheric Pollution Research 4, 245-249.

Anenberg, S., Horowitz, L., Tong, D., West, J., 2010. An Estimate of the Global Burden of Anthropogenic Ozone and Fine Particulate Matter on Premature Human Mortality Using Atmospheric Modeling. Environ Health Perspectives 118, 1189-1195.

Anenberg, S.C., Balakrishnan, K., Jetter, J., Masera, O., Mehta, S., et al., 2013. Cleaner Cooking Solutions to Achieve Health, Climate, and Economic Cobenefits. Environ. Sci. Technol. 47, 3944-3952.

Apte, J.S., Kirchstetter, T.W., Reich, A.H., Deshpande, S.J., Kaushik, G., et al., 2011. Concentrations of fine, ultrafine, and black carbon particles in auto-rickshaws in New Delhi, India. Atmos. Environ. 45, 4470-4480.

Aunan, K., Fang, J., Hu, T., Seip, H.M., Vennemo, H., 2006. Climate change and air quality measures with co-benefits in China. Environmental Science and Technology 40, 4822-4829.

Balakrishnan, K., Cohen, A., Smith, K.R., 2014. Addressing the Burden of Disease Attributable to Air Pollution in India: The Need to Integrate across Household and Ambient Air Pollution Exposures. Environ. Health Perspect. 122, A6-A7.

Balakrishnan, K., Dhaliwal, R.S., Shah, B., 2011. Integrated Urban-Rural Frameworks for Air Pollution and Health-Related Research in India: The Way Forward. Environ. Health Perspect. 119, a12-a13.

Balmes, J., 2016. Effects of wood smoke exposure on cardiopulmonary responses in healthy & susceptible humans. California Air Resources Board,.

Balmes, J.R., 2010. When smoke gets in your lungs. Proceedings of the American Thoracic Society 7, 98-101.

Barnes, D.F., Floor, W.M., 1996. Rural Energy in Developing Countries: A Challenge for Economic Development. Annual Review of Energy and the Environment 21, 497-530.

Baumgartner, J., Schauer, J.J., Ezzati, M., Lu, L., Cheng, C., et al., 2011. Indoor Air Pollution and Blood Pressure in Adult Women Living in Rural China. Environ. Health Perspect. 119, 1390-1395.

Bay Area Air Quality Management District, 2012a. Historic Wood Smoke Model Ordinances Adopted by Bay Area Cities and Counties. BAAQMD.

Bay Area Air Quality Management District, 2012b. Understanding Particulate Matter: Protecting Public Health in the San Francisco Bay Area. BAAQMD, San Francisco.

Bay Area Air Quality Management District, 2014. Bay Area Emissions Inventory Summary Report: Criteria Air Pollutants. BAAQMD, San Francisco.

Bay Area Air Quality Management District, 2015a. Air District strengthens wood burning rule San Francisco.

Bay Area Air Quality Management District, 2015b. Regulation 6 Particulate Matter and Visible Emissions Rule 3 Wood-Burning Devices, San Francisco.

Bennett, D.H., McKone, T.E., Evans, J.S., Nazaroff, W.W., Margni, M.D., et al., 2002. Peer Reviewed: Defining Intake Fraction. Environ. Sci. Technol. 36, 206A-211A.

Bollen, J.C., Brink, C.J., Eerens, H.C., Manders, A.J.G., 2009. Co-benefits of climate policy. Netherlands Environmental Assessment Agency,.

Bond, T.C., Doherty, S.J., Fahey, D.W., Forster, P.M., Berntsen, T., et al., 2013. Bounding the role of black carbon in the climate system: A scientific assessment. Journal of Geophysical Research: Atmospheres 118, 5380-5552.

Bond, T.C., Streets, D.G., Yarber, K.F., Nelson, S.M., Woo, J.H., et al., 2004. A technology-based global inventory of black and organic carbon emissions from combustion. Journal of Geophysical Research-Atmospheres 109, D14203.

Bonjour, S., Adair-Rohani, H., Wolf, J., Bruce, N.G., Mehta, S., et al., 2013. Solid fuel use for household cooking: country and regional estimates for 1980-2010. Environ. Health Perspect. 121, 784-790.

Bowyer, J., 2012. Life Cycle Impacts of Heating with Wood in Scenarios Ranging from Home and Institutional Heating to Community Scale District Heating Systems, in: Dovetail Partners Inc (Ed.). Legislative-Citizen Commission on Minnesota Resources,.

Brauer, M., Amann, M., Burnett, R.T., Cohen, A., Dentener, F., et al., 2012. Exposure Assessment for Estimation of the Global Burden of Disease Attributable to Outdoor Air Pollution. Environ. Sci. Technol. 46, 652-660.

Brauer, M., Freedman, G., Frostad, J., van Donkelaar, A., Martin, R.V., et al., 2016. Ambient Air Pollution Exposure Estimation for the Global Burden of Disease 2013. Environ. Sci. Technol. 50, 79-88.

Broderick, D., Houck, J., 2003. Results of Wood Burning Survey – Sacramento, San Joaquin, and San Francisco Areas, University of California Berkeley/California Air Resources Board – GIS Study. OMNI Consulting Services.

Burnett, R., Pope III, C., Ezzati, M., Olives, C., Lim, S., et al., 2014. An integrated risk function for estimating the global burden of disease attributable to ambient fine particulate matter exposure. Environ. Health Perspect. 122, 397-403.

California Air Resources Board, 2013. CEPAM: 2013 Almanac - Standard Emissions Tool: Emission Projections by Summary Category, Sacramento, CA.

California Air Resources Board, 2015. Residential Wood Combustion Methodology (Emission Inventory Documentation). CARB, Sacramento.

California Department of Public Health Safe and Active Communities Branch, 2016. CDPH Vital Statistics Death Statistical Master and Multiple Cause of Death files

California Environmental Protection Agency, 2009. Wood Burning Handbook: Protecting the Environment and Saving Money.

Canadian Council of Ministers of the Environment, 2012. Code of Practice for Residential Wood Burning Appliances.

Cao, G., Zhang, X., Zheng, F., 2006. Inventory of black carbon and organic carbon emissions from China. Atmos. Environ. 40, 6516-6527.

Caseiro, A., Bauer, H., Schmidl, C., Pio, C.A., Puxbaum, H., 2009. Wood burning impact on PM10 in three Austrian regions. Atmos. Environ. 43, 2186-2195.

Centre, E.C.J.R., 2014. FASST - FAst Scenario Screening Tool

Chafe, Z., Brauer, M., Heroux, M.-E., Klimont, Z., Lanki, T., et al., 2015. Residential heating with wood and coal: health impacts and policy options in Europe and North America. World Health Organization Regional Office for Europe, Bonn, Germany.

Chafe, Z.A., Brauer, M., Klimont, Z., Dingenen, R.V., Mehta, S., et al., 2014. Household Cooking with Solid Fuels Contributes to Ambient PM2.5 Air Pollution and the Burden of Disease. Environ. Health Perspect. 122, 1314-1320.

Chen, Y., Ebenstein, A., Greenstone, M., Li, H., 2013. Evidence on the impact of sustained exposure to air pollution on life expectancy from China's Huai River policy. Proceedings of the National Academy of Sciences 110, 12936-12941.

Cherubini, F., Strømman, A.H., 2011. Life cycle assessment of bioenergy systems: State of the art and future challenges. Bioresource Technology 102, 437-451.

Chowdhury, Z., Le, L.T., Masud, A.A., Chang, K.C., Alauddin, M., et al., 2012. Quantification of Indoor Air Pollution from Using Cookstoves and Estimation of Its Health Effects on Adult Women in Northwest Bangladesh. Aerosol Air Qual. Res. 12, 463-475.

Chowdhury, Z., Zheng, M., Schauer, J.J., Sheesley, R.J., Salmon, L.G., et al., 2007. Speciation of ambient fine organic carbon particles and source apportionment of PM2.5 in Indian cities. J. Geophys. Res. 112, D15303.

Clancy, L., Goodman, P., Sinclair, H., Dockery, D.W., 2002. Effect of air-pollution control on death rates in Dublin, Ireland: an intervention study. The Lancet 360, 1210-1214.

Cofala, J., Bertok, I., Borken-Kleefeld, J., Heyes, C., Klimont, Z., et al., 2012. Emissions of Air Pollutants for the World Energy Outlook 2012 Energy Scenarios. Final Report. International Energy Agency, Paris, France.

Cohen, A.J., Anderson, H.R., Bart Ostro, Pandey, K.D., Krzyzanowski, M., et al., 2004. Urban Air Pollution, in: Ezzati, M., Lopez, A., Murray, C.J.L. (Eds.), Comparative Quantification of Health Risks: Global and Regional Burden of Disease Attributable to Selected Major Risk Factors Geneva. World Health Organization, pp. 1353-1433.

Dasch, J.M., 1982. Particulate and gaseous emissions from wood-burning fireplaces. Environ. Sci. Technol. 16, 639-645.

Dherani, M., Pope, D., Mascarenhas, M., Smith, K.R., Weber, M., et al., 2008. Indoor air pollution from unprocessed solid fuel use and pneumonia risk in children aged under five years: a systematic review and meta-analysis. Bull. World Health Organ. 86, 390-398C.

Dinh, T., 2016. Emission factors used by the District in the calculation of wood stove and fire place emissions.

Duan, X., Jiang, Y., Wang, B., Zhao, X., Shen, G., et al., 2014. Household fuel use for cooking and heating in China: Results from the first Chinese Environmental Exposure-Related Human Activity Patterns Survey (CEERHAPS). Applied Energy 136, 692-703.

Ekholm, T., Krey, V., Pachauri, S., Riahi, K., 2011. Determinants of household energy consumption in India. Energy Policy 38, 5696-5707.

Environment and Climate Change Canada, 2015. Air Pollution Emission Inventory Report 1990-2014.

Environmental Protection Agency, 2013. National Ambient Air Quality Standards for Particulate Matter (40 CFR Parts 50, 51, 52, 53 and 5). Fed. Regist. 78.

European Climate Foundation, 2010. Biomass for Heat and Power: Opportunity and Economics.

European Commission, 2009. Directive 2009/125/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for the setting of ecodesign requirements for energy-related products. Official Journal of the European Union L 285.

European Commission, 2013. Commission Staff Working Document Impact Assessment Accompanying the Documents: Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions a Clean Air Programme for Europe; Proposal for a Directive of the European Parliament and of the Council on the limitation of emissions of certain pollutants into the air from medium combustion plants; Proposal for a Directive of the European Parliament and of the Council on the reduction of national emissions of certain atmospheric pollutants and amending Directive 2003/35/EC; Proposal for a Council Decision on the acceptance of the Amendment to the 1999 Protocol to the 1979 Convention on Long-Range Transboundary Air Pollution to Abate Acidification, Eutrophication and Ground-level Ozone Brussels.

European Commission, 2014. State of play on the sustainability of solid and gaseous biomass used for electricity, heating and cooling in the EU. European Union, Brussels.

European Commission, 2015. Commission Regulation (EU) 2015/1185 Official Journal of the European Union L 193/1.

European Environment Agency, 2015. Emissions of primary PM2.5 and PM10 particulate matter, Indicator Assessment.

Executive Body for the Convention on Long-range Transboundary Air Pollution Joint Task Force on the Health Aspects of Air Pollution, 2014. Residential heating with wood and coal: health impacts and policy options in Europe and North America. United Nations Economic and Social Council Economic Commission for Europe (UNECE),, Geneva.

Ezzati, M., Hoorn, S.V., Lopez, A.D., Danaei, G., Rodgers, A., et al., 2004. Comparative Quantification of Mortality and Burden of Disease Attributable to Selected Risk Factors, Global Burden of Disease and Risk Factors, pp. 241-268.

Ezzati, M., Kammen, D.M., 2001. Quantifying the effects of exposure to indoor air pollution from biomass combustion on acute respiratory infections in developing countries. Environ. Health Perspect. 109, 481-488.

Ezzati, M., Kammen, D.M., 2002. The health impacts of exposure to indoor air pollution from solid fuels in developing countries: knowledge, gaps, and data needs. Environ. Health Perspect. 110, 1057.

Fairley, D., 2011. Trends in Bay Area Ambient Particulates. Bay Area Air Quality Management District,.

Fairley, D., 2012. Source of Bay Area Fine Particles: 2010 Update and Trends. BAAQMD.

Fairley, D., 2014. Wood-burning devices in the SF Bay Area – Update, Bay Area Air Quality Management District Office Memorandum. BAAQMD.

Fernandes, S.D., Trautmann, N.M., Streets, D.G., Roden, C.A., Bond, T.C., 2007. Global biofuel use, 1850 2000. Global Biogeochem. Cycles 21, GB2019.

Forouzanfar, M.H., Alexander, L., Anderson, H.R., Bachman, V.F., Biryukov, S., et al., 2015. Global, regional, and national comparative risk assessment of 79 behavioural, environmental and occupational, and metabolic risks or clusters of risks in 188 countries, 1990–2013: a systematic analysis for the Global Burden of Disease Study 2013. The Lancet.

Fuller, G.W., Sciare, J., Lutz, M., Moukhtar, S., Wagener, S., 2013. New Directions: Time to tackle urban wood burning? Atmos. Environ. 68, 295-296.

Gaidajis, G., Angelakoglou, K., Aktsoglou, D., 2014. Wintertime particulate mass concentrations in urban environment and the impact of economic crisis. Journal of Environmental Science and Health, Part A 49, 1653-1660.

GEA, 2012. Global Energy Assessment - Toward a Sustainable Future, Cambridge University Press, Cambridge, UK and New York, NY, USA and the International Institute for Applied Systems Analysis, Laxenburg, Austria.

Gianelle, V., Colombi, C., Caserini, S., Ozgen, S., Galante, S., et al., 2013. Benzo(a)pyrene air concentrations and emission inventory in Lombardy region, Italy. Atmospheric Pollution Research 4.

Grubler, A., O'Neill, B., Riahi, K., Chirkov, V., Goujon, A., et al., 2007. Regional, national, and spatially explicit scenarios of demographic and economic change based on SRES. Technological Forecasting and Social Change 74, 980-1029.

Gullett, B.K., Touati, A., Hays, M.D., 2003. PCDD/F, PCB, HxCBz, PAH, and PM Emission Factors for Fireplace and Woodstove Combustion in the San Francisco Bay Region. Environ. Sci. Technol. 37, 1758-1765.

Health Effects Institute, 2010. Outdoor Air Pollution and Health in the Developing Countries of Asia: A Comprehensive Review Health Effects Institute, Boston.

Houck, J.E., Tiegs, P., 1998. Residential Wood Combustion Technology Review: Volume 1 Technical Report, in: McCrillis, R.C. (Ed.). US Environmental Protection Agency, Washington, DC.

Huang, W., Cao, J., Tao, Y., Dai, L., Lu, S.-E., et al., 2012. Seasonal Variation of Chemical Species Associated With Short-Term Mortality Effects of PM2.5 in Xian, a Central City in China. Am. J. Epidemiol. 175, 556-566.

Huneeus, N., Schulz, M., Balkanski, Y., Griesfeller, J., Prospero, J., et al., 2011. Global dust model intercomparison in AeroCom phase I. Atmos. Chem. Phys. 11, 7781-7816.

IIASA, 2012. GAINS: Greenhouse Gas and Air Pollution Interactions and Synergies. International Institute for Applied Systems Analysis, http://gains.iiasa.ac.at/models/ [accessed 27 November 2012].

IIASA, 2014. Greenhouse gas – air pollution interactions and synergies (GAINS) model, in: International Institute for Applied Systems Analysis (Ed.), Laxenburg, Austria.

Institute for Health Metrics and Evaluation, 2010a. Global Burden of Diseases, Injuries and Risk Factors Study Operations Manual, Seattle.

Institute for Health Metrics and Evaluation, 2010b. Global Burden of Diseases, Injuries and Risk Factors Study Operations Manual, Seattle.

International Energy Agency, 2011a. World Energy Outlook 2011, in: Oecd/Iea (Ed.), World Energy Outlook 2009. International Energy Agency, Paris.

International Energy Agency, 2011b. World Energy Outlook 2011, in: OECD/IEA (Ed.). IEA, Paris.

IPCC, 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, in: Pachauri, R.K., Meyer, L.A. (Eds.). IPCC, Geneva.

Isaac, M., van Vuuren, D.P., 2009. Modeling global residential sector energy demand for heating and air conditioning in the context of climate change. Energy Policy 37, 507-521.

Jetter, J., Zhao, Y., Khan, B., Yelverton, T., Hays, M., et al., 2012. Pollutant emissions and energy efficiency under controlled conditions for household biomass cookstoves and implications for metrics useful in setting international test standards. Environ. Sci. Technol. 46, 10827-10834.

Johnston, F.H., Hanigan, I.C., Henderson, S.B., Morgan, G.G., 2013. Evaluation of interventions to reduce air pollution from biomass smoke on mortality in Launceston, Australia: retrospective analysis of daily mortality, 1994-2007. BMJ 346.

Khanna, N., Zhou, N., Ke, J., Fridley, D., 2014. Quantifying PM2.5 Emissions from China's Building Sector and Co-Benefits of Energy Efficiency, 2014 ACEEE Summer Study on Energy

Efficiency in Buildings. The American Council for an Energy-Efficient Economy, Pacific Grove, CA.

Kim, E., Hopke, P.K., 2008. Source characterization of ambient fine particles at multiple sites in the Seattle area. Atmos. Environ. 42, 6047-6056.

Klimont, Z., Cofala, J., Xing, J., Wei, W., Zhang, C., et al., 2009. Projections of SO2, NOx and carbonaceous aerosols emissions in Asia. Tellus Series B Chemical and Physiological Meteorology 61, 602-617.

Knight, K., Rosa, E., 2011. Household dynamics and fuelwood consumption in developing countries: a cross-national analysis. Population & Environment August, 1-14.

Krewski, D., Jerrett, M., Burnett, R.T., Ma, R., Hughes, E., et al., 2009. Extended Follow-Up and Spatial Analysis of the American Cancer Society Study Linking Particulate Air Pollution and Mortality Health Effects Institute, Boston.

Kupiainen, K., Klimont, Z., 2007. Primary emissions of fine carbonaceous particles in Europe. Atmos. Environ. 41, 2156-2170.

Kurmi, O.P., Semple, S., Simkhada, P., Smith, W.C.S., Ayres, J.G., 2010. COPD and chronic bronchitis risk of indoor air pollution from solid fuel: a systematic review and meta-analysis. Thorax 65, 221-228.

Lam, N.L., Chen, Y., Weyant, C., Venkataraman, C., Sadavarte, P., et al., 2012a. Household Light Makes Global Heat: High Black Carbon Emissions from Kerosene Wick Lamps. Environ. Sci. Technol. 46, 13531–13538.

Lam, N.L., Smith, K.R., Gauthier, A., Bates, M.N., 2012b. Kerosene: A Review of Household Uses and their Hazards in Low- and Middle-Income Countries. Journal of Toxicology and Environmental Health, Part B 15, 396-432.

Lamarque, J.F., Bond, T.C., Eyring, V., Granier, C., Heil, A., et al., 2010. Historical (1850-2000) gridded anthropogenic and biomass burning emissions of reactive gases and aerosols: methodology and application. Atmospheric Chemistry and Physics 10, 7017-7039.

Larson, T., Gould, T., Simpson, C., Liu, L.J.S., Claiborn, C., et al., 2004. Source apportionment of indoor, outdoor, and personal PM2.5 in Seattle, Washington, using positive matrix factorization. J. Air Waste Manag. Assoc. 54, 1175-1187.

Larson, T.V., Koenig, J.Q., 1994. Wood Smoke: Emissions and Noncancer Respiratory Effects. Annu. Rev. Public Health 15, 133-156.

Lei, Y., Zhang, Q., He, K.B., Streets, D.G., 2011. Primary anthropogenic aerosol emission trends for China, 1990-2005. Atmos. Chem. Phys 11, 931-954.

Lelieveld, J., Evans, J.S., Fnais, M., Giannadaki, D., Pozzer, A., 2015. The contribution of outdoor air pollution sources to premature mortality on a global scale. Nature 525, 367-371.

Lepeule, J., Laden, F., Dockery, D., Schwartz, J., 2012. Chronic Exposure to Fine Particles and Mortality: An Extended Follow-up of the Harvard Six Cities Study from 1974 to 2009. Environ. Health Perspect. 120, 965-970.

Leung, D.Y.C., 2015. Outdoor-indoor air pollution in urban environment: Challenges and opportunity. Frontiers in Environmental Science 2.

Lim, S.S., Vos, T., Flaxman, A.D., Danaei, G., Shibuya, K., et al., 2012. A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990-2010: a systematic analysis for the Global Burden of Disease Study 2010. The Lancet 380, 2224-2260.

Macdonald, R., 2009. Evaluation of Gas, Oil and Wood Pellet Fueled Residential Heating System Emissions Characteristics. Brookhaven National Laboratory, Upton, NY.

Marshall, J.D., Apte, J.S., Coggins, J.S., Goodkind, A.L., 2015. Blue Skies Bluer? Environ. Sci. Technol.

McGowan, J.A., Hider, P.N., Chacko, E., Town, G.I., 2002. Particulate air pollution and hospital admissions in Christchurch, New Zealand. Aust. N. Z. J. Public Health 26, 23-29.

Mondal, N., Roy, A., Mukherjee, B., Das, D., Ray, M., 2010. Indoor Air Pollution from Biomass Burning Activates Akt in Airway Cells and Peripheral Blood Lymphocytes: A Study among Premenopausal Women in Rural India. Toxicol. Pathol., 1-14.

Mortimer, K., Gordon, S.B., Jindal, S.K., Accinelli, R.A., Balmes, J., et al., 2012. Household air pollution is a major avoidable risk factor for cardiorespiratory disease. CHEST Journal 142, 1308-1315.

Mosley, S., 2001. The Chimney of the World: A History of Smoke Pollution in Victorian and Edwardian Manchester. White Horse Press, Cambridge.

Muye, R., Shu, T., Kirk, S., Guofeng, S., Huizhong, S., et al., 2015. Direct Energy Consumption Associated Emissions by Rural-to-Urban Migrants in Beijing. Environ. Sci. Technol.

Nemet, G., Holloway, T., Meier, P., 2010. Implications of incorporating air-quality co-benefits into climate change policymaking. Environmental Research Letters 5, 014007.

Ochir, C., Smith, K., Hill, L., 2014. Air Pollution and Health in Ulaanbaatar. Ministry of the Environment and Green Development, Ulaanbaatar, Mongolia.

Ohara, T., Akimoto, H., Kurokawa, J., Horii, N., Yamaji, K., et al., 2007. An Asian emission inventory of anthropogenic emission sources for the period 1980-2020. Atmos. Chem. Phys. 7, 4419-4444.

Pachauri, S., 2011. Reaching an international consensus on defining modern energy access. Energy Systems 3, 235-240.

Pachauri, S., Brew-Hammond, A., Barnes, D.F., Bouille, D.H., Gitonga, S., et al., 2012. Chapter 19 - Energy Access for Development, Global Energy Assessment - Toward a Sustainable Future, Cambridge University Press, Cambridge, UK and New York, NY, USA and the International Institute for Applied Systems Analysis, Laxenburg, Austria, pp. 1401-1458.

Pant, P., Harrison, R.M., 2012. Critical review of receptor modelling for particulate matter: A case study of India. Atmos. Environ. 49, 1-12.

Pearson, P., Bodin, S., Nordberg, L., Pettus, A., 2013. On thin ice : how cutting pollution can slow warming and save lives. World Bank, Washington, DC.

Peters, A., Rückerl, R., Cyrys, J., 2011. Lessons From Air Pollution Epidemiology for Studies of Engineered Nanomaterials. J. Occup. Environ. Med. 53, S8-S13.

Pope, D.P., Mishra, V., Thompson, L., Siddiqui, A.R., Rehfuess, E.A., et al., 2010. Risk of low birth weight and stillbirth associated with indoor air pollution from solid fuel use in developing countries. Epidemiol. Rev. 32, 70-81.

Purohit, P., 2010. Clarifying Household Cooking Activity Categories within GAINS.

Purohit, P., Amann, M., Mathur, R., Gupta, I., Marwah, S., et al., 2010. GAINS ASIA: Scenarios for cost-effective control of air pollution and greenhouse gases in India. IIASA, Laxenburg, Austria.

Rao, S., Chirkov, V., Dentener, F., Van Dingenen, R., Pachauri, S., et al., 2012. Environmental Modeling and Methods for Estimation of the Global Health Impacts of Air Pollution. Environ Model Assess 17, 613-622.

Rao, S., Riahi, K., 2006. The role of non-CO2 greenhouse gases in climate change mitigation: Long-term scenarios for the 21st century. Energy Journal 27, 177-200.

Riahi, K., Dentener, F., Gielen, D., Grubler, A., Jewell, J., et al., 2012. Chapter 17 - Energy Pathways for Sustainable Development, Global Energy Assessment - Toward a Sustainable Future, Cambridge University Press, Cambridge, UK and New York, NY, USA and the International Institute for Applied Systems Analysis, Laxenburg, Austria, pp. 1203-1306.

Riahi, K., Rao, S., Krey, V., Cho, C., Chirkov, V., et al., 2011. RCP 8.5-A scenario of comparatively high greenhouse gas emissions. Climatic Change 109, 33-57.

Rogalsky, D.K., Mendola, P., Metts, T.A., Martin, W.J., 2nd, 2014. Estimating the number of low-income americans exposed to household air pollution from burning solid fuels. Environ. Health Perspect. 122, 806-810.

Saarnio, K., Niemi, J.V., Saarikoski, S., Aurela, M., Timonen, H., et al., 2012. Using monosaccharide anhydrides to estimate the impact of wood combustion on fine particles in the Helsinki Metropolitan Area. Boreal Environment Research 17, 163-183.

Saffari, A., Daher, N., Samara, C., Voutsa, D., Kouras, A., et al., 2013. Increased Biomass Burning Due to the Economic Crisis in Greece and Its Adverse Impact on Wintertime Air Quality in Thessaloniki. Environ. Sci. Technol. 47, 13313-13320.

Sanhueza, P.A., Torreblanca, M.A., Diaz-Robles, L.A., Schiappacasse, L.N., Silva, M.P., et al., 2009. Particulate Air Pollution and Health Effects for Cardiovascular and Respiratory Causes in Temuco, Chile: A Wood-Smoke-Polluted Urban Area. J. Air Waste Manag. Assoc. 59, 1481-1488.

Shen, G., Yang, Y., Wang, W., Tao, S., Zhu, C., et al., 2012. Emission Factors of Particulate Matter and Elemental Carbon for Crop Residues and Coals Burned in Typical Household Stoves in China. Environ. Sci. Technol. 44, 7157-7162.

Smith, K., McCracken, J.P., Weber, M., Hubbard, A., Jenny, A., et al., 2011. Effect of reduction in household air pollution on childhood pneumonia in Guatemala (RESPIRE): a randomised controlled trial. Lancet 378, 1717-1726.

Smith, K., Pillarisetti, A., 2012. Breve historia del humo de leña y sus implicaciones para Chile. Estudios públicos 126, 163-179.

Smith, K.R., 1989. Dialectics of improved stoves. Economic and Political Weekly 24, 517-522.

Smith, K.R., 1993. Fuel combustion, air pollution exposure, and health: the situation in developing countries. Annual Review of Energy and the Environment 18, 529-566.

Smith, K.R., 2006. Rural Air Pollution: A Major But Often Ignored Development Concern, Commission on Sustainable Development Thematic Session on Integrated Approaches to Addressing Air Pollution and Atmospheric Problems, UN Commission on Sustainable Development Thematic Session on Integrated Approaches to Addressing Air Pollution and Atmospheric Problems, New York City.

Smith, K.R., Apte, M.G., Yuqing, M., Wongsekiarttirat, W., Kulkarni, A., 1994. Air pollution and the energy ladder in asian cities. Energy 19, 587-600.

Smith, K.R., Bruce, N., Balakrishnan, K., Adair-Rohani, H., Balmes, J., et al., 2014. Millions Dead: How Do We Know and What Does It Mean? Methods Used in the Comparative Risk Assessment of Household Air Pollution. Annu. Rev. Public Health 35, 185-206.

Smith, K.R., Haigler, E., 2008. Co-Benefits of Climate Mitigation and Health Protection in Energy Systems: Scoping Methods. Annu. Rev. Public Health 29, 11-25.

Smith, K.R., Jerrett, M., Anderson, H.R., Burnett, R.T., Stone, V., et al., 2009. Public health benefits of strategies to reduce greenhouse-gas emissions: health implications of short-lived greenhouse pollutants. The Lancet 374, 2091-2103.

Smith, K.R., Mehta, S., 2003. The burden of disease from indoor air pollution in developing countries: comparison of estimates. International Journal of Hygiene and Environmental Health 206, 279-289.

Smith, K.R., Samet, J.M., Romieu, I., Bruce, N., 2000. Indoor air pollution in developing countries and acute lower respiratory infections in children. Thorax 55, 518-532.

State of California Department of Public Health, 2016. CA-Vital Statistics Query (CA-VSQ)

Stone, E.A., Schauer, J.J., Pradhan, B.B., Dangol, P.M., Habib, G., et al., 2010. Characterization of emissions from South Asian biofuels and application to source apportionment of carbonaceous aerosol in the Himalayas. J. Geophys. Res. 115, D06301.

Streets, D.G., Bond, T.C., Carmichael, G.R., Fernandes, S.D., Fu, Q., et al., 2003. An inventory of gaseous and primary aerosol emissions in Asia in the year 2000. J. Geophys. Res 108, 8809.

Strubegger, M., Totschnig, G., Zhu, B., 2004. MESSAGE: A technical model description, in: Schrattenholzer, L., Miketa, A., Riahi, K., Roehrl, R.A. (Eds.), Achieving a Sustainable Global Energy System: Identifying Possibilities Using Long-Term Energy Scenarios. Edward Elgar, Cheltenham, UK, pp. 168-214.

Taimisto, P., Tainio, M., Karvosenoja, N., Kupiainen, K., Porvari, P., et al., 2011. Evaluation of intake fractions for different subpopulations due to primary fine particulate matter (PM2.5) emitted from domestic wood combustion and traffic in Finland. Air Quality, Atmosphere & Health 4, 199-209.

Textor, C., Schulz, M., Guibert, S., Kinne, S., Balkanski, Y., et al., 2006. Analysis and quantification of the diversities of aerosol life cycles within AeroCom. Atmos. Chem. Phys. 6, 1777-1813.

TNO, 2012. Survey of views of stakeholders, experts and citizens on the review of the EU Air Policy Part I: Main results.

True North, 2014. Winter Spare the Air Study 2013-2014 Winter Wood Smoke Season. BAAQMD.

U.S. Congress Office of Technology Assessment, 1992. Fueling development: Energy technologies for developing countries, Washington, DC.

UNEP, 2011. Integrated Assessment of Black Carbon and Tropospheric Ozone. United Nations Energy Program and World Meteorological Organization, Nairobi.

Unger, N., Bond, T., Wang, J., Kocha, D., Menone, S., et al., 2010. Attribution of climate forcing to economic sectors. Proc Natl Acad Sci PNAS Early Edition.

Unosson, J., Blomberg, A., Sandstrom, T., Muala, A., Boman, C., et al., 2013. Exposure to wood smoke increases arterial stiffness and decreases heart rate variability in humans. Part. Fibre Toxicol. 10, 20.

Ürge-Vorsatz, D., Cabeza, L.F., Serrano, S., Barreneche, C., Petrichenko, K., 2015. Heating and cooling energy trends and drivers in buildings. Renewable and Sustainable Energy Reviews 41, 85-98.

US Environmental Protection Agency, 2011. Air Quality Modeling Technical Support Document: Source Sector Assessments Research Triangle Park.

US Environmental Protection Agency, 2013a. Regulatory Impact Analysis for the Final Revisions to the National Ambient Air Quality Standards for Particulate Matter, Research Triangle Park, NC.

US Environmental Protection Agency, 2013b. Technical Support Document: Estimating the Benefit per Ton of Reducing PM2.5 Precursors from 17 Sectors, in: Office of Air and Radiation (Ed.). US EPA, Research Triangle Park.

US Environmental Protection Agency, 2015a. Air Pollutant Emissions Trends Data: Average Annual Emissions.

US Environmental Protection Agency, 2015b. Regulatory Impact Analysis for Residential Wood Heaters NSPS Revision: Final Report. US EPA Office of Air Quality Planning and Standards,, Washington DC.

US Environmental Protection Agency, 2015c. Standards of Performance for New Residential Wood Heaters, New Residential Hydronic Heaters and Forced-Air Furnaces. Federal Register, Washington DC.

van Ruijven, B.J., van Vuuren, D.P., de Vries, B.J.M., Isaac, M., van der Sluijs, J.P., et al., 2011. Model projections for household energy use in India. Energy Policy 39, 7747-7761.

Wang, Y., Zhuang, G., Tang, A., Yuan, H., Sun, Y., et al., 2005. The ion chemistry and the source of PM2.5 aerosol in Beijing. Atmos. Environ. 39, 3771-3784.

Ward, T., Lange, T., 2010. The impact of wood smoke on ambient PM2.5 in northern Rocky Mountain valley communities. Environmental Pollution 158, 723-729.

Wilkinson, P., Smith, K.R., Davies, M., Adair, H., Armstrong, B.G., et al., 2009. Public health benefits of strategies to reduce greenhouse-gas emissions: household energy. The Lancet 374, 1917-1929.

Woodruff, T.J., Grillo, J., Schoendorf, K.C., 1997. The relationship between selected causes of postneonatal infant mortality and particulate air pollution in the United States. Environ. Health Perspect. 105, 608-612.

World Bank, 2011. Household cookstoves, environment, health, and climate change : a new look at an old problem. World Bank,, Washington, DC.

World Bank Group and Institute for Health Metrics and Evaluation, 2014. Transport for Health: The Global Burden of Disease from Motorized Road Transport. World Bank / IHME, Washington, DC.

World Health Organization, 2006. WHO Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide: Global update 2005. World Health Organization, Geneva.

World Health Organization, 2014a. Frequently Asked Questions: Ambient and Household Air Pollution and Health Updated 2014. WHO, Geneva.

World Health Organization, 2014b. WHO Indoor air quality guidelines: household fuel combustion. World Health Organization, Geneva.

World Health Organization, 2016. Burning Opportunity: Clean Household Energy for Health, Sustainable Development, and Wellbeing of Women and Children WHO, Geneva.

Zhang, J., Mauzerall, D.L., Zhu, T., Liang, S., Ezzati, M., et al., 2010. Environmental health in China: progress towards clean air and safe water. The Lancet 375, 1110-1119.

Zhang, J., Smith, K.R., 2007. Household Air Pollution from Coal and Biomass Fuels in China: Measurements, Health Impacts, and Interventions. Environ. Health Perspect. 115, 848-855.

Zhang, Q., Streets, D.G., Carmichael, G.R., He, K.B., Huo, H., et al., 2009. Asian emissions in 2006 for the NASA INTEX-B mission. Atmos. Chem. Phys 9, 5131-5153.

Appendices

Appendix A

Abbreviations used in the manuscript:

AAP	Ambient air pollution
$APM_{2.5}$	Population-weighted ambient fine particulate air pollution
BAAQMD	Bay Area Air Quality Management Agency
BC	Black carbon
CARB	California Air Resources Board
CO_2e	Carbon dioxide equivalent
DALY	Disability-adjusted life year
EPA	Environmental Protection Agency
GAINS	Greenhouse Gas and Air Pollution Interactions and Synergies Model
GBD 2010	2010 Global Burden of Disease/ Comparative Risk Assessment Project
GHG	Greenhouse gas
IEA	International Energy Agency
IER	Integrate exposure-response
IIASA	International Institute for Applied Systems Analysis
IHME	Institute for Health Metrics and Evaluation
OC	Organic carbon
OECD	Organization for Economic Cooperation and Development
$PPM_{2.5\text{-cook}}$	Primary particulate matter with average diameter < 2.5 micrometers (µm) attributable to combustion of solid fuels for household cooking
$PPM_{2.5\text{-hh}}$	Primary particulate matter with average diameter < 2.5 micrometers (µm) attributable to all household fuel combustion (cooking and heating)
$PM_{2.5}$	Particulate matter with average diameter < 2.5 micrometers (µm)

$PM_{2.5\text{-cook}}$	Particulate matter with average diameter < 2.5 micrometers (µm), attributable to combustion of solid fuels for household cooking
$PM_{2.5\text{-heat}}$	Particulate matter with average diameter < 2.5 micrometers (µm), attributable to combustion of solid fuels for household heating
$PM_{2.5\text{-hh}}$	Particulate matter with average diameter < 2.5 micrometers (µm) attributable to all household fuel combustion (cooking and heating)
PM_{10}	Particulate matter with average diameter < 10 micrometers (µm)
Tg	Teragram
TMRED	Theoretical minimum risk exposure distribution
TM5-FASST	Fast Scenario Screening Tool for Global Air Quality and Instantaneous Radiative Forcing
VOC	Volatile Organic Compound
WHO	World Health Organization

Appendix B

This appendix describes the regional definitions used to calculate and report results in Chapters 2 and 3; these were also used in the 2010 Global Burden of Disease Project coordinated by the IHME. They include the two major energy use, emissions, and air pollution models that I rely on for my analysis: the GAINS model and TM5-FASST.

Table B1: Regional Groupings Used in Global Burden of Disease 2010 (With Regional Population in 2010 of Countries Included in this Analysis and Total Population in Region). Note: countries that were not included in this analysis, because of data gaps, are shown in italics.

Region (Population in Millions)	Countries in Region
Asia Central (81/81)	Armenia, Azerbaijan, Georgia, Kazakhstan, Kyrgyzstan, Mongolia, Tajikistan, Turkmenistan, Uzbekistan
Asia East (1,383/1,383)	China, Democratic People's Republic of Korea, *Hong Kong*
Asia Pacific High Income (181/181)	Brunei Darussalam, Japan, Republic of Korea, Singapore
Asia South (1,591/1,591)	Afghanistan, Bangladesh, Bhutan, India, Nepal, Pakistan
Asia Southeast (609/610)	Indonesia, Cambodia, Lao People's Democratic Republic, Sri Lanka, *Maldives*, Myanmar, Malaysia, Philippines, Thailand, *Timor-Leste*, Viet Nam
Australasia (26/26)	Australia, New Zealand
Caribbean (38/40)	Antigua and Barbuda, *Aruba*, Bahamas, Belize, Barbados, Cuba, Dominica, Dominican Republic, Grenada, *Guadeloupe,* Guyana, Haiti, Jamaica, *Martinique, Netherlands Antilles*, Saint Kitts and Nevis, Saint Lucia, Suriname, Trinidad and Tobago, Saint Vincent and the Grenadines
Europe Central (108/108)	Albania, Bulgaria, Bosnia and Herzegovina, Czech Republic, Croatia, Hungary, The former Yugoslav Republic of Macedonia, Montenegro, Poland, Romania, Serbia, Slovakia, Slovenia
Europe Eastern (206/206)	Belarus, Estonia, Lithuania, Latvia, Republic of Moldova, Russian Federation, Ukraine
Europe Western (414/414)	Andorra, Austria, Belgium, Switzerland, Cyprus, Germany, Denmark, Spain, Finland, France, United Kingdom, Greece, Ireland, Iceland, Israel, Italy, Luxembourg, *Monaco,* Malta, Netherlands, Norway, Portugal, San Marino, Sweden
Latin America Andean (53/53)	Bolivia, Ecuador, Peru

Latin America Central (230/230)	Colombia, Costa Rica, Guatemala, Honduras, Mexico, Nicaragua, Panama, El Salvador, Venezuela
Latin America Southern (61/61)	Argentina, Chile, Uruguay
Latin America Tropical (205/205)	Brazil, Paraguay
North Africa Middle East (473/478)	United Arab Emirates, Bahrain, Algeria, Egypt, Iran, Iraq, Jordan, Kuwait, Lebanon, Libyan Arab Jamahiriya, Morocco, Oman, Qatar, *Palestinian Territories,* Saudi Arabia, Syrian Arab Republic, Tunisia, Turkey, Yemen
North America High Income (348/348)	Canada, United States of America
Oceania (0/9)	*American Samoa, Cook Islands, Fiji, Micronesia, Guam, Kiribati, Marshall Islands, Micronesia, Niue, Nauru, New Caledonia, Palau, Papua New Guinea, Solomon Islands, Tonga, Tuvalu, Vanuatu, Samoa*
Sub-saharan Africa Central (98/98)	Angola, Central African Republic, Democratic Republic of the Congo, Congo, Gabon, Equatorial Guinea
Sub-saharan Africa East (357/357)	Burundi, Comoros, Djibouti, Eritrea, Ethiopia, Kenya, Madagascar, Mozambique, Mauritius, Malawi, Rwanda, Sudan, Somalia, Seychelles, United Republic of Tanzania, Uganda, Zambia
Sub-saharan Africa Southern (70/70)	Botswana, Lesotho, Namibia, Swaziland, South Africa, Zimbabwe
Sub-saharan Africa West (339/339)	Benin, Burkina Faso, Côte d'Ivoire, Cameroon, Cape Verde, Ghana, Guinea, Gambia, Guinea-Bissau, Liberia, Mali, Mauritania, Niger, Nigeria, Senegal, Sierra Leone, Sao Tome and Principe, Chad, Togo

Table B2: Regional Groupings Used in TM5-FASST.

Region	Countries in Region
ARG	Argentina, Falkland Islands, Uruguay
AUT	Austria, Liechtenstein, Slovenia
BLX	Belgium, Luxembourg, Netherlands
CAN	Canada, Greenland
CHN	China, Hong Kong, Macau
EAF	Burundi, Central African Republic, Chad, Comoros, DR Congo, Djibouti, Eritrea, Ethiopia, Kenya, Madagascar, Mauritius, Reunion, Rwanda, Seychelles, Somalia, Sudan, Tanzania, Uganda
ESP	Gibraltar, Portugal, Spain
FRA	France, Andorra
GBR	Ireland, United Kingdom
GOLF	Bahrain, Iran, Iraq, Kuwait, Oman, Qatar, Saudi Arabia, United Arab Emirates, Yemen
GRC	Cyprus, Greece
IDN	Indonesia, Timor-Leste
ITA	Italy, Malta, Monaco, San Marino, Vatican City State
MEME	Israel, Jordan, Lebanon, Palestinian Territory, Syrian Arab Republic
MON	North Korea, Mongolia
MYS	Brunei Darussalam, Malaysia, Singapore
NDE	India, Maldives, Sri Lanka
NOA	Algeria, Libya, Morocco, Tunisia, Western Sahara
NOR	Iceland, Norway, Svalbard and Jan Mayen Islands
PAC	Fiji, French Polynesia, Guam, Kiribati, Marshall Islands, Micronesia, Nauru, New Caledonia, Niue, Norfolk Island, Northern Mariana Islands, Palau, Papua New Guinea, Pitcairn, Samoa, Solomon Islands, Tokelau, Tonga, Tuvalu, Vanuatu, Wallis and Futuna Islands
POL	Estonia, Latvia, Lithuania, Poland
RCAM	Anguilla, Antigua and Barbuda, Aruba, Bahamas, Barbados, Belize, Cayman Islands, Costa Rica, Cuba, Dominica, Dominican Republic, El Salvador, Grenada, Guadeloupe, Guatemala, Haiti, Honduras, Jamaica, Martinique, Montserrat, Netherlands Antilles, Nicaragua, Panama, Puerto Rico, Saint Kitts and Nevis, Saint Lucia, Saint Vincent and the Grenadines, Trinidad and Tobago, Turks and Caicos Islands, Virgin Islands (British), Virgin Islands (U.S.)
RCEU	Albania, Bosnia and Herzegovina, Croatia, Macedonia, Serbia and Montenegro
RCZ	Czech Republic, Slovakia
RIS	Kyrgyzstan, Tajikistan, Turkmenistan, Uzbekistan
RSA	Lesotho, South Africa, Swaziland
RSAM	Bolivia, Colombia, Ecuador, French Guiana, Guyana, Paraguay, Peru, Suriname, Venezuela
RSAS	Afghanistan, Bangladesh, Bhutan, Nepal, Pakistan

RSEA	Cambodia, Laos, Myanmar
RUS	Armenia, Azerbaijan, Georgia, Russian Federation
SAF	Angola, Botswana, Malawi, Mayotte, Mozambique, Namibia, Zambia, Zimbabwe
SWE	Denmark, Faroe Islands, Sweden
UKR	Belarus, Moldova, Ukraine
USA	Bermuda, Saint Pierre and Miquelon, United States
WAF	Benin, Burkina Faso, Cameroon, Cape Verde, Congo, Cote D'Ivoire, Equatorial Guinea, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Liberia, Mali, Mauritania, Niger, Nigeria, Saint Helena, Sao Tome and Principe, Senegal, Sierra Leone, Togo

Table B3: Regional Groupings Used in GAINS.

Region	Countries in Region
Former USSR (Asia)	Tajikistan, Turkmenistan, Uzbekistan
Middle East	Bahrain, Iran, Iraq, Jordan, Kuwait, Lebanon, Oman, Qatar, Saudi Arabia, Syrian Arab Republic, United Arab Emirates, Yemen
North Africa	Algeria, Libyan Arab Jamahiriya, Morocco, Sudan, Tunisia
Other Africa	Angola, Benin, Botswana, Burkina Faso, Burundi, Cameroon, Cape Verde, Central African Republic, Chad, Comoros, Democratic Republic of Congo, Congo, Cote d'Ivoire, Djibouti, Equatorial Guinea, Eritrea, Ethiopia, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Kenya, Lesotho, Liberia, Madagascar, Malawi, Mali, Mauritania, Mauritius, Mozambique, Namibia, Niger, Nigeria, Rwanda, Sao Tome and Principe, Senegal, Seychelles, Sierra Leone, Somalia, Swaziland, Tanzania, Togo, Uganda, Western Sahara, Zambia, Zimbabwe
Other Latin America	Antigua and Barbuda, Bahamas, Barbados, Belize, Bolivia, Colombia, Costa Rica, Cuba, Dominica, Dominican Republic, Ecuador, El Salvador, French Guiana, Grenada, Guatemala, Guyana, Haiti, Honduras, Jamaica, Nicaragua, Panama, Paraguay, Peru, Puerto Rico, Saint Kitts and Nevis, Saint Lucia, Saint Vincent and the Grenadines, Suriname, Trinidad and Tobago, Uruguay, Venezuela

Appendix C

This appendix lists input data used in calculating population-weighted averages by region, such as population and emissions data, in Chapter 2 and Chapter 3.

Table C1. Population and Emissions Data Used in the Analysis.

Region	**Population 1990[a]**	**Population 2010[a]**	**$PPM_{2.5\text{-cook}}$ 1990[b]**	**$PPM_{2.5\text{-cook}}$ 2010[b]**	**$PM_{2.5\text{-hh}}$ 1990[c]**	**$PM_{2.5\text{-hh}}$ 2010[c]**
High-income Asia Pacific	170	181	14%	6%	12%	12%
Central Asia	69	81	3%	1%	23%	21%
East Asia	1175	1383	83%	85%	34%	18%
Southeast Asia	455	609	90%	89%	28%	18%
South Asia	1106	1591	82%	82%	21%	32%
Australasia	20	26	4%	2%	2%	4%
Caribbean	31	38	83%	83%	12%	14%
Central Europe	112	108	0%	0%	19%	23%
Eastern Europe	223	206	0%	0%	19%	19%
Western Europe	381	414	0%	0%	10%	14%
Andean Latin America	39	53	83%	83%	7%	11%
Central Latin America	167	230	83%	83%	8%	9%
Southern Latin America	49	61	82%	81%	13%	18%
Tropical Latin America	154	205	83%	83%	5%	9%
High-income North America	284	348	0%	0%	4%	10%
North Africa and Middle East	318	473	34%	31%	15%	14%
Central sub-Saharan Africa	55	98	99%	99%	6%	19%
Eastern sub-Saharan Africa	211	357	97%	94%	6%	18%
Southern sub-Saharan Africa	52	70	95%	93%	13%	41%
Western sub-Saharan Africa	199	339	99%	99%	10%	30%
World	*5269*	*6872*	*62%*	*65%*	*20%*	*21%*

[a]Population x 10^6

[b] Percent of primary $PM_{2.5}$ household emissions attributable to household cooking (GAINS)

[c] Percent of combustion-derived emissions attributable to household cooking and heating (TM5-FASST)

Appendix D

This graphic describes in detail the emissions and particles covered by various models used in the analysis in Chapter 2 and Chapter 3.

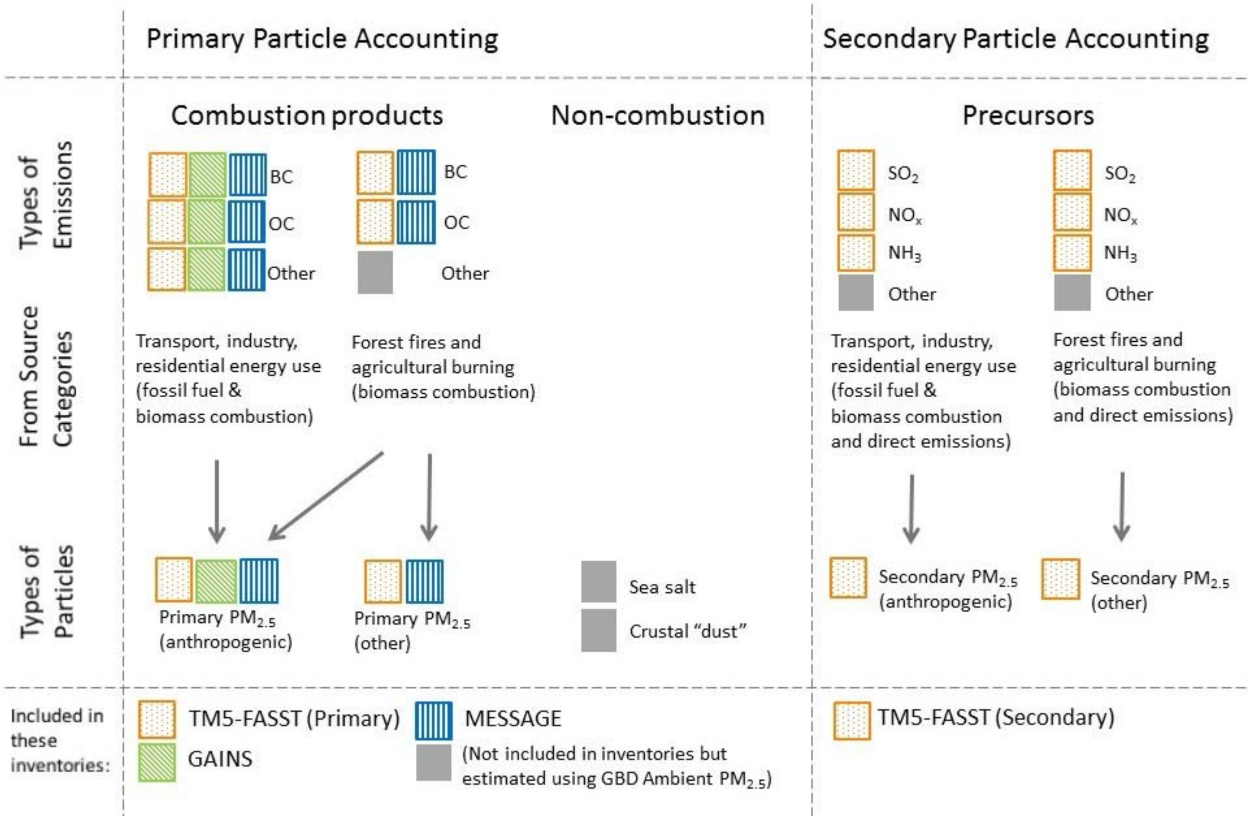

Figure D1. Emissions and particle coverage in the major databases and models used in this analysis. Note that sea salt, dust, and some secondary particle precursors are not included in the models used here; however, they are represented in the total ambient $PM_{2.5}$ concentrations calculate for GBD 2010, published in Brauer et al. (2012) and used in the final stages of the analysis presented in this analysis.