RESEARCH REPORT

HEALTH EFFECTS INSTITUTE

Number 162
December 2011

Assessing the Impact of a Wood Stove Replacement Program on Air Quality and Children's Health

Curtis W. Noonan, Tony J. Ward, William Navidi, Lianne Sheppard, Megan Bergauff, Chris Palmer

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with a Critique by the HEI Health Review Committee

Research Report 162
Health Effects Institute
Boston, Massachusetts

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Publishing history: This document was posted at *www.healtheffects.org* in December 2011.

Citation for document:

Noonan CW, Ward TJ, Navidi W, Sheppard L, Bergauff M, Palmer C. 2011. Assessing the Impact of a Wood Stove Replacement Program on Air Quality and Children's Health. Research Report 162. Health Effects Institute, Boston, MA.

 Asterisk Typographics, Barre, Vt., Compositor. Printed by Recycled Paper Printing, Boston, Mass. Library of Congress Catalog Number for the HEI Report Series: WA 754 R432.

Cover paper: made with at least 55% recycled content, of which at least 30% is post-consumer waste; free of acid and elemental chlorine. Text paper: made with 100% post-consumer waste recycled content; acid free; no chlorine used in processing. The book is printed with soy-based inks and is of permanent archival quality.

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ABOUT HEI

The Health Effects Institute is a nonprofit corporation chartered in 1980 as an independent research organization to provide high-quality, impartial, and relevant science on the effects of air pollution on health. To accomplish its mission, the institute

- Identifies the highest-priority areas for health effects research;
- Competitively funds and oversees research projects;
- Provides intensive independent review of HEI-supported studies and related research;
- Integrates HEI's research results with those of other institutions into broader evaluations; and
- Communicates the results of HEI's research and analyses to public and private decision makers.

HEI typically receives half of its core funds from the U.S. Environmental Protection Agency and half from the worldwide motor vehicle industry. Frequently, other public and private organizations in the United States and around the world also support major projects or research programs. HEI has funded more than 280 research projects in North America, Europe, Asia, and Latin America, the results of which have informed decisions regarding carbon monoxide, air toxics, nitrogen oxides, diesel exhaust, ozone, particulate matter, and other pollutants. These results have appeared in the peer-reviewed literature and in more than 200 comprehensive reports published by HEI.

HEI's independent Board of Directors consists of leaders in science and policy who are committed to fostering the public–private partnership that is central to the organization. The Health Research Committee solicits input from HEI sponsors and other stakeholders and works with scientific staff to develop a Five-Year Strategic Plan, select research projects for funding, and oversee their conduct. The Health Review Committee, which has no role in selecting or overseeing studies, works with staff to evaluate and interpret the results of funded studies and related research.

All project results and accompanying comments by the Health Review Committee are widely disseminated through HEI's Web site (*www.healtheffects.org*), printed reports, newsletters and other publications, annual conferences, and presentations to legislative bodies and public agencies.

ABOUT THIS REPORT

Research Report 162, *Assessing the Impact of a Wood Stove Replacement Program on Air Quality and Children's Health*, presents a research project funded by the Health Effects Institute and conducted by Dr. Curtis W. Noonan of Center for Environmental Health Sciences, Department of Biomedical Sciences, 32 Campus Drive, The University of Montana, Missoula, Montana, and his colleagues. This report contains three main sections.

The HEI Statement, prepared by staff at HEI, is a brief, nontechnical summary of the study and its findings; it also briefly describes the Health Review Committee's comments on the study.

The Investigators' Report, prepared by Noonan and his colleagues, describes the scientific background, aims, methods, results, and conclusions of the study.

The Critique is prepared by members of the Health Review Committee with the assistance of HEI staff; it places the study in a broader scientific context, points out its strengths and limitations, and discusses remaining uncertainties and implications of the study's findings for public health and future research.

This report has gone through HEI's rigorous review process. When an HEI-funded study is completed, the investigators submit a draft final report presenting the background and results of the study. This draft report is first examined by outside technical reviewers and a biostatistician. The report and the reviewers' comments are then evaluated by members of the Health Review Committee, an independent panel of distinguished scientists who have no involvement in selecting or overseeing HEI studies. During the review process, the investigators have an opportunity to exchange comments with the Review Committee and, as necessary, to revise their report. The Critique reflects the information provided in the final version of the report.

PREFACE

HEI's Outcomes Research Program

The goal of most air quality regulations is to protect the public's health by implementing regulatory actions or providing economic incentives that help reduce the public's exposure to air pollutants. If this goal is met, air pollution should be reduced, and indicators of public health should improve or at least not deteriorate. Evaluating the extent to which air quality regulations succeed in protecting public health is part of a broader effort — variously termed *outcomes research, accountability research*, or *research on regulatory effectiveness* — designed to assess the performance of environmental regulatory policies in general. In recent decades, air quality in the United States and Western Europe has improved substantially, and this improvement is attributable to a number of factors, including increasingly stringent air quality regulations. However, the cost of the pollution-control technologies and mechanisms needed to implement and enforce these regulations is often high. It is therefore prudent to ask whether the regulations have in fact yielded demonstrable improvements in public health, which will provide useful information to inform future efforts.

Several U.S. government agencies have concluded that direct evidence about the extent to which air quality regulations have improved health (measured as a decrease in premature mortality and excess morbidity) is lacking. This finding is well documented by the National Research Council (NRC) in its report *Estimating the Public Health Benefits of Proposed Air Pollution Regulations* (NRC 2002), as well as by the California Air Resources Board, the U.S. Environmental Protection Agency (EPA), the U.S. Centers for Disease Control and Prevention (CDC), and other agencies.

In 2003, the Health Effects Institute published a monograph on outcomes research, Communication 11, *Assessing Health Impact of Air Quality Regulations: Concepts and Methods for Accountability Research* (HEI 2003). This monograph was written by the members of HEI's multidisciplinary Accountability Working Group after a 2001 workshop on the topic. Communication 11 set out a conceptual framework for outcomes research and identified the types of evidence required and the methods by which the evidence should be obtained. It has also guided the development of the HEI Health Outcomes Research program, which is discussed below.

Between 2002 and 2004, HEI issued four requests for applications (RFAs) for studies to evaluate the effects of actions taken to improve air quality. The study by Dr. Curtis Noonan and colleagues described in this Research Report (Noonan et al. 2011) was funded under RFA 04-4, "Measuring the Health Impact of Actions Taken to Improve Air Quality." HEI funded eight additional outcomes studies resulting from this and other RFAs (see Preface Table).

This preface describes both the framework of outcomes research as it relates to air quality regulations and HEI's Outcomes Research program.

BACKGROUND

The first step in assessing the effectiveness of air quality regulations is to measure emissions of the targeted pollutants to see whether they have in fact decreased as intended. A series of intermediate assessments, described in detail below, are needed in order to accurately measure the adverse health effects associated with air pollution to see whether they also decreased in incidence or severity relative to emissions. Some outcomes studies to date have used hypothetical scenarios (comparing estimated outcomes under existing and more stringent regulations) and risk estimates obtained from epidemiologic studies in an attempt to quantify past effects on health and to predict future effects (U.S. EPA 1999). However, more extensive validation of these estimates with data on actual outcomes would be helpful.

The long-term improvements in U.S. air quality have been associated with improved health in retrospective

HEI's Outcomes Research Program[a]

RFA / Investigator (Institution)	Study or Report Title	Intervention
RFA 02-1		
Douglas Dockery (Harvard School of Public Health, Boston, MA)	"Effects of Air Pollution Control on Mortality and Hospital Admissions in Ireland" (in review)	Coal ban in Irish cities
Annette Peters (GSF–National Research Center for Environment and Health, Neuherberg, Germany[b])	The Influence of Improved Air Quality on Mortality Risks in Erfurt, Germany (published as HEI Research Report 137, 2009)	Switch from brown coal to natural gas for home heating and power plants, changes in motor vehicle fleet after reunification of Germany
RFA 04-1		
Frank Kelly (King's College London, London, U.K.)	The Impact of the Congestion Charging Scheme on Air Quality in London: Part 1. Emissions Modeling and Analysis of Air Pollution Measurements. Part 2. Analysis of the Oxidative Potential of Particulate Matter (published as HEI Research Report 155, 2011)	Measures to reduce traffic congestion in the inner city of London
RFA 04-4		
Frank Kelly (King's College London, London, U.K.)	The London Low Emission Zone Baseline Study (published as HEI Research Report 163, 2011)	Measures to exclude most polluting vehicles from entering greater London
Richard Morgenstern (Resources for the Future, Washington, DC)	"Accountability Assessment of Title IV of the Clean Air Act Amendments of 1990" (in press)	Measures to reduce sulfur emissions from power plants east of the Mississippi River
Curtis Noonan (University of Montana, Missoula, MT)	Assessing the Impact of a Wood Stove Replacement Program on Air Quality and Children's Health (published as HEI Research Report 162, 2011)	Woodstove changeout program
Jennifer Peel (Colorado State University, Fort Collins, CO)	Impact of Improved Air Quality During the 1996 Summer Olympic Games in Atlanta on Multiple Cardiovascular and Respiratory Outcomes (published as HEI Research Report 148, 2010)	Measures to reduce traffic congestion during the Atlanta Olympics
Chit-Ming Wong (University of Hong Kong, Hong Kong)	"Impact of the 1990 Hong Kong Legislation for Restriction on Sulfur Content in Fuel" (in press)	Measures to reduce sulfur content in fuel for motor vehicles and power plants
RFPA 05-3		
Junfeng (Jim) Zhang (University of Medicine and Dentistry of New Jersey, Piscataway, NJ)	"Molecular and Physiological Responses to Drastic Changes in PM Concentration and Composition" (in review)	Measures to improve air quality during the Beijing Olympics

[a] Abbreviations: RFA, Request for Applications; RFPA, Request for Preliminary Applications.

[b] As of 2008, this institution has been called the Helmholtz Zentrum München–German Research Center for Environmental Health.

epidemiologic studies (Chay and Greenstone 2003; Laden et al. 2006; Pope et al. 2009). Considerable challenges, however, are inherent in the assessment of the health effects of air quality regulations. Different regulations go into effect at different times, for example, and may be implemented at different levels of government (e.g., national, regional, or local). Their effectiveness therefore needs to be assessed in ways that take into account the varying times of implementation and levels of regulation. In addition, other changes at the same time and place might confound an apparent association between pollution reduction and improved health, such as economic trends (e.g., changes in employment), improvements in health care, and behavioral changes (e.g., staying indoors when government warnings indicate pollution concentrations are high). Moreover, adverse health effects that might have been caused by exposure to air pollution can also be caused by other environmental risk factors (some of which may have changed over the same time periods as the air pollution concentrations). These challenges become more pronounced when regulations are implemented over long periods and when changes in air quality and health outcomes are not seen immediately, thus increasing the chance for confounding by other factors. For these reasons, scenarios in which regulations are expected to have resulted in rapid changes in air quality tend to be among the first, and most likely, targets for investigation, rather than evaluations of complex regulatory programs implemented over multiple years. Studies in Ireland by Clancy and colleagues (2002) and in Hong Kong by Hedley and colleagues (2002) are examples of such scenarios.

These inherent challenges are well documented in Communication 11 (HEI 2003), which was intended to advance the concept of outcomes research and to foster the development of methods and studies throughout the relevant scientific and policy communities. In addition, recent advances in data collection and analytic techniques provide an unprecedented opportunity to improve our assessments of the effects of air quality interventions.

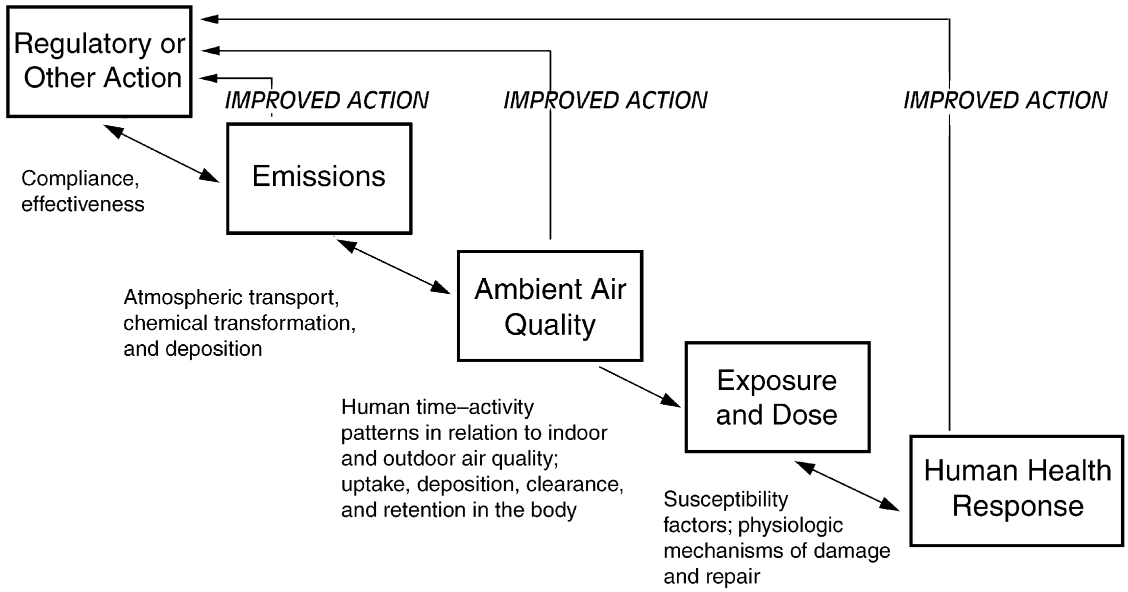
THE OUTCOMES EVALUATION CYCLE

The NRC's Committee on Research Priorities for Airborne Particulate Matter set out a conceptual framework for linking air pollution sources to adverse health effects (NRC 1998). This framework can be used to identify factors along an "outcomes evaluation cycle" (see Preface Figure), each stage of which affords its own opportunities for making quantitative measurements of the intended improvements.

At the first stage (regulatory action), one can assess whether controls on source emissions have in fact been put into place. At the second stage (emissions), one can determine whether controls on sources have indeed reduced emissions, whether emitters have changed their practices, and whether there have been unintended consequences. At the third stage (ambient air quality), one can assess whether controls on sources and reductions in emissions have resulted in improved air quality. At the fourth stage (personal or population exposure), one can assess whether the improvement in air quality has reduced people's actual exposure and whether susceptible subpopulations (those most likely to experience adverse health effects) have benefited. At this stage, it is important to take into account changes in time–activity patterns that could either increase or reduce exposure. The actual dose that an individual's organs may be exposed to should also be considered (i.e., whether reductions in exposure have led to reductions in concentrations in body tissues such as the lung). Finally, at the fifth stage (human health response), one can assess whether risks to health have declined, given the evidence about changes in health outcomes such as morbidity and mortality that have resulted from changes in exposure. The challenge at this stage is to investigate the health outcomes that are most directly related to exposure to air pollution.

At each stage in the outcomes evaluation cycle, the opportunity exists to collect evidence that either validates the assumptions that motivated the intervention or points to ways in which the assumptions were incorrect. The collection of such evidence can thus ensure that future interventions are maximally effective.

Ultimately, the framework for outcomes research will need to encompass investigations of the broader consequences of regulations, not just the intended consequences. Unintended consequences should also be investigated, along with the possibility that risks to public health in fact increased, as discussed by Wiener (1998) and others who have advanced the concept of a portfolio of effects of a regulation.



Outcomes Evaluation Cycle. Each box represents a stage in the process between regulatory action and human health responses to air pollution. Arrows connecting the stages indicate possible directions of influence. The text below the arrows identifies factors affecting the effectiveness of regulatory actions at each stage. At several of the stages, knowledge gained from studies on outcomes can provide valuable feedback for improving regulatory or other actions.

HEI'S OUTCOMES RESEARCH PROGRAM

HEI's Outcomes Research program currently includes nine studies. The study by Dr. Curtis Noonan and colleagues presented in this report is the fifth to be published; two additional studies are in press and are expected to be published in 2012. The remaining two studies are in review and are also expected to be published in 2012.

These studies involve the measurement of indicators along the entire outcomes evaluation cycle, from regulatory or other interventions to human health outcomes. Some of the studies focused on interventions that are implemented over relatively short periods of time, such as a ban on the sale of coal, the replacement of old wood stoves with more efficient, cleaner ones, reductions in the sulfur content of fuels, and measures to reduce traffic. Other groups focused on longer-term, wider-ranging interventions or events; for instance, one study assessed complex changes associated with the reunification of the former East and West Germany, including a switch from brown coal to natural gas for fueling power plants and home-heating systems and an increase in the number of modern diesel-powered vehicles in eastern Germany. HEI is also supporting research, including the development of methods, in an especially challenging area, namely, assessment of the effects of regulations implemented incrementally over extended periods of time, such as those resulting from Title IV of the 1990 Clean Air Act Amendments (U.S. EPA 1990), which aimed at reducing sulfur dioxide emissions from power plants by requiring compliance with prescribed emission limitations. Studies on health outcomes funded by HEI to date are summarized in the Preface Table and described in more detail in an interim evaluation of the HEI Outcomes Research program (van Erp and Cohen 2009).

FUTURE DIRECTIONS

As a part of its Strategic Plan for 2010 through 2015 (HEI 2010a), HEI has looked closely at opportunities for unique new contributions to health outcomes research. Key recommendations for future research were made at a December 2009 planning workshop (HEI 2010b), which led to HEI issuing a new Request for Applications in January 2011 for a second wave of outcomes research. RFA 11-1, "Health Outcomes Research — Assessing the Health Outcomes of Air Quality Actions," solicits applications for studies designed to assess the health effects of actions to improve air quality and to develop methods required for, and specifically suited to, conducting such research. Preference will be given to (1) studies that evaluate regulatory and other actions at the national or regional level implemented over multiple years; (2) studies that evaluate complex sets of actions targeted at improving air quality in large urban areas and major ports with well-documented air quality problems and programs to address them; and (3) studies that develop methods to support such health outcomes research (see *www.healtheffects.org/funding.htm*). HEI hopes to fund three or four studies, expected to start in 2012, to evaluate the effectiveness of longer-term regulatory actions.

In addition, HEI has funded the development of two Web sites intended to enhance transparency and provide other researchers with access to extensive data and software from HEI-funded studies:

1. Data and software from the National Morbidity, Mortality, and Air Pollution Study (NMMAPS), as described by Zeger and colleagues (2006) (data available at the Johns Hopkins Bloomberg School of Public Health Web site *www.ihapss.jhsph.edu*); and
2. Data from the National Particle Component Toxicity Initiative (NPACT) on concentrations of components of particulate matter with an aerodynamic diameter ≤ 2.5 µm ($PM_{2.5}$) collected at or near the 54 sites in the EPA's $PM_{2.5}$ Chemical Speciation Trends Network (STN) (data available at the Atmospheric and Environmental Research Web site *https://hei.aer.com*).

The data on pollution and health from a large number of U.S. cities, as documented by the NMMAPS team and made available on the Internet-Based Health and Air Pollution Surveillance System (iHAPSS) Web site, constitute a valuable resource that allows other researchers to undertake additional analyses, possibly including further outcomes studies. The STN Web site provides scientists an opportunity to investigate specific questions about concentrations of $PM_{2.5}$ components and their association with adverse health effects in regions covered by the STN network and to address questions related to outcomes research when interventions in these regions are being planned.

In January 2008, HEI co-organized and cosponsored, with the CDC's National Environmental Public Health Tracking Program and the EPA, a workshop titled "Methodologic Issues in Environmental Public Health Tracking of Air Pollution Effects." The workshop was part of an effort to implement the initiative outlined in HEI's Strategic Plan for 2005 through 2010 (HEI 2005) to "build networks with the U.S. Centers for Disease Control and Prevention and state public health tracking programs to facilitate accountability research."

The workshop built on the work of the CDC's National Environmental Public Health Tracking Program (see the CDC Web site *www.cdc.gov/nceh/tracking/*) in the development of standardized measures of air pollution–related effects on health at the state and local levels in the United States. It brought together representatives of state and federal agencies and academic researchers to discuss methodologic issues in developing standardized measures and made recommendations for their further development and application in assessing the health impacts of air pollution, including the impacts of actions taken to improve air quality. The recommendations were provided in a September 2008 report to the CDC, and the proceedings were published in the journal *Air Quality, Atmosphere & Health* in December 2009 (Matte et al. 2009). The CDC has subsequently funded a pilot project under the National Environmental Public Health Tracking Program to implement the recommendations of the workshop in selected states and metropolitan areas.

HEI will continue to seek opportunities to work with the CDC and the EPA to apply methods newly developed for tracking public health and assessing the effectiveness of environmental regulations.

Investigators who have identified a distinctive opportunity to evaluate the effects of environmental regulations on air pollution and human health are encouraged to contact HEI.

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HEI STATEMENT

Synopsis of Research Report 162

Wood Stove Replacement, Air Quality, and Health

BACKGROUND

Recent decades have seen substantial gains in air quality in the United States and Western Europe, with downward trends in concentrations of several major pollutants, including particulate matter (PM). These gains have been achieved largely through increasingly stringent air quality regulations and control measures. However, it is important to verify that regulations to improve air quality have indeed resulted in improved air quality and health.

The study led by Dr. Curtis W. Noonan evaluated a relatively large-scale program to replace about 1200 older, more polluting wood stoves with new, less polluting stoves in a rural mountain community (Libby, Montana), where residential wood combustion had been identified as a major source of fine PM ($PM_{2.5}$) during the heating season. Exposure to wood smoke is associated with increased respiratory symptoms in children and adults, decreased lung function in children, and increased emergency department visits and hospitalizations. In addition, wood smoke has been classified as "probably carcinogenic in humans" by the International Agency for Research on Cancer. Noonan and colleagues hypothesized that the intervention would substantially reduce community exposure to $PM_{2.5}$ derived from wood smoke, and thereby reduce children's respiratory symptoms and illness-related school absences.

APPROACH

Noonan and colleagues collected air quality and health data during four consecutive winters starting in 2005, the first year of the changeout program. The majority of changeouts took place during the second and third winters; the fourth winter constituted the post-changeout phase. Additional data were collected retrospectively, to cover two baseline winters before the start of the program.

The investigators measured $PM_{2.5}$ and some of its components outdoors, inside schools during different seasons, and in about 20 homes (before and after stove changeout) during the winter. They focused on compounds suggested to be specific markers for wood smoke, such as levoglucosan, abietic acid, and dehydroabietic acid, and evaluated whether these compounds could be used to track source-specific changes in air quality inside homes as well as in ambient air.

In parallel, they tracked illness-related school absences in children and parent-reported respiratory symptoms. Changes in wintertime reporting of symptoms and variations in school absences were also evaluated in relation to changes in ambient $PM_{2.5}$ concentrations in successive years.

RESULTS

Ambient winter concentrations of $PM_{2.5}$ gradually declined over the study period and were 30% lower in the final winter after the changeout program (year 4) than in the baseline years. By the end of the study period, Libby was no longer out of compliance with the National Ambient Air Quality Standard for $PM_{2.5}$.

Concentrations of levoglucosan, a fairly well-validated marker for wood smoke, were lower during the first three winters of the program than during the baseline winters, but increased again during the final winter. Concentrations of other potential markers, such as abietic acid and dehydroabietic acid, did not decrease in association with the changeout program. After stove changeout, indoor $PM_{2.5}$ concentrations were lower in a majority of the homes sampled, although there was substantial variability within and between homes. At the elementary and middle schools, indoor concentrations of markers for wood smoke and $PM_{2.5}$ were variable and not consistent with the timing of the changeout program.

This Statement, prepared by the Health Effects Institute, summarizes a research project funded by HEI and conducted by Dr. Curtis W. Noonan of the University of Montana–Missoula and colleagues. Research Report 162 contains both the detailed Investigators' Report and a Critique of the study prepared by the Institute's Health Review Committee.

Based on about 1700 surveys filled out by parents during the four years, there was a significant reduction in childhood wheezing associated with lower winter ambient $PM_{2.5}$ concentrations. The most robust associations were for itchy or watery eyes, sore throat, bronchitis, influenza, and throat infection. There were no differences in health outcomes (notably, wheezing) between children from homes with wood stoves and children from homes with other types of heating. School absence data showed that lower average ambient winter $PM_{2.5}$ concentrations were associated with fewer illness-related absences among older students, but with higher absence rates among students in grades 1 through 4.

INTERPRETATION

The wood stove changeout program should be considered a success because 95% of older, high-polluting wood stoves in Libby were replaced with more efficient certified wood stoves or with heating systems that did not burn wood. In its independent evaluation of the study, the HEI Review Committee thought the study had demonstrated that ambient $PM_{2.5}$ concentrations in the community were reduced during the course of the changeout program, and that this reduction was sustained over subsequent winters.

The 30% reduction in ambient $PM_{2.5}$ concentrations at the end of the intervention may be considered encouraging. However, although the newer stoves that were introduced in Libby were certified, they did not necessarily represent the cleanest technology available at the time. Moreover, even certified, lower-emitting stoves make substantial contributions to ambient PM concentrations — particularly when not operated optimally — as compared with heating systems using cleaner fuels. Certified wood stoves have been found to emit $PM_{2.5}$ at rates (about 2–7 grams per hour) that are one to two orders of magnitude higher than those associated with oil (0.07 g/hr) or gas (0.04 g/hr) furnaces.

Sampling from about 20 homes showed that indoor $PM_{2.5}$ concentrations generally decreased after the stove was replaced, but results were not consistent; a few homes actually had increased concentrations, which may be due to incorrect stove usage or other indoor sources of pollution.

Although the study demonstrates an impact of the changeout program on ambient $PM_{2.5}$ concentrations (even if relatively modest), the Committee concluded that there was weak evidence that such air quality changes were associated with improved respiratory health outcomes (wheezing) and fewer symptoms associated with wood smoke exposure, such as itchy or watery eyes. The lack of high-quality health outcomes data was considered the most limiting aspect of the study design.

The Committee thought Noonan and colleagues had chosen appropriate statistical methods to evaluate the intervention, although the study was limited by inherent challenges, such as the seasonal nature of the intervention, the small size of the Libby community, and the availability of only one year of pre-intervention survey data.

The investigators found no differences in health outcomes between children from homes with wood stoves and those from homes with other types of heating. This finding suggests that exposures may be more closely determined by the overall contribution of wood stoves to ambient air quality than by their contributions to air quality in individual homes, a result that is consistent with other studies in the literature.

CONCLUSIONS

In summary, the study showed that wood stove changeout programs can contribute to community-level improvements in ambient air quality. Generally, air quality inside homes also improved, but stoves remain relatively high emitters compared with oil or gas furnaces, and proper stove operation is an important determinant of emissions. This study provided some evidence of improved children's health in the community, with reduced rates of parent-reported wheezing, itchy or watery eyes, sore throat, bronchitis, influenza, and throat infection. Further research using hospital admission data or more direct health outcomes, such as medication use, or biomarkers of exposure and effect would be useful. In addition, more research is needed to identify reliable markers for wood smoke exposure.

INVESTIGATORS' REPORT

Assessing the Impact of a Wood Stove Replacement Program on Air Quality and Children's Health

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ABSTRACT

Many rural mountain valley communities experience elevated ambient levels of fine particulate matter (PM*) in the winter, because of contributions from residential wood-burning appliances and sustained temperature inversion periods during the cold season. A wood stove changeout program was implemented in a community heavily affected by wood-smoke–derived $PM_{2.5}$ (PM $\leq$ 2.5 µm in aerodynamic diameter). The objectives of this study were to evaluate the impact of this intervention program on ambient and indoor $PM_{2.5}$ concentrations and to identify possible corresponding changes in the frequency of childhood respiratory symptoms and infections and illness-related school absences.

Over 1100 old wood stoves were replaced with new EPA-certified wood stoves or other heating sources. Ambient $PM_{2.5}$ concentrations were 30% lower in the winter after the changeout program, compared with baseline winters, which brought the community's ambient air within the $PM_{2.5}$ standards of the U.S. Environmental Protection Agency (U.S. EPA). The installation of a new wood stove resulted in an overall reduction in indoor $PM_{2.5}$ concentrations in a small sample of wood-burning homes, but the effects were highly variable across homes. Community-level reductions in wood-smoke–derived $PM_{2.5}$ concentration were associated with decreased reports of childhood wheeze and of other childhood respiratory health conditions. The association was not limited to children living in homes with wood stoves nor does it appear to be limited to susceptible children (e.g., children with asthma). Community-level reductions in wood-smoke–derived $PM_{2.5}$ concentration were also associated with lower illness-related school absences among older children, but this finding was not consistent across all age groups.

This community-level intervention provided a unique opportunity to prospectively observe exposure and outcome changes resulting from a targeted air pollution reduction strategy.

INTRODUCTION

Mortality and morbidity studies have supported an association between ambient PM exposures and adverse effects on respiratory health (U.S. EPA 2004; Pope and Dockery 2006). Most large epidemiologic studies of PM and respiratory effects have focused on large urban environments where the primary PM sources are industrial processes and mobile emission sources (Katsouyanni et al. 1997; Samet et al. 2000). Residential-generated biomass smoke can be a substantial PM source in communities that have a large percentage of homes using wood heating (Fairley 1999; McGowan et al. 2002; Kim et al. 2003; Manchester-Neesvig et al. 2003; Larson et al. 2004; Ward et al. 2006; Ward and Lange 2010), as well as in many developing

This Investigators' Report is one part of Health Effects Institute Research Report 162, which also includes a Critique by the Health Review Committee and an HEI Statement about the research project. Correspondence concerning the Investigators' Report may be addressed to Dr. Curtis W. Noonan, Center for Environmental Health Sciences, Department of Biomedical Sciences, 32 Campus Drive, The University of Montana, Missoula, MT 59812, email: curtis.noonan@umontana.edu.

Although this document was produced with partial funding by the United States Environmental Protection Agency under Assistance Award CR–83234701 to the Health Effects Institute, it has not been subjected to the Agency's peer and administrative review and therefore may not necessarily reflect the views of the Agency, and no official endorsement by it should be inferred. The contents of this document also have not been reviewed by private party institutions, including those that support the Health Effects Institute; therefore, it may not reflect the views or policies of these parties, and no endorsement by them should be inferred.

* A list of abbreviations and other terms appears at the end of the Investigators' Report.

countries where biomass fuel for cooking is common (Naeher et al. 2007). A limited number of epidemiologic studies in communities with high levels of residential biomass smoke exposure have found morbidity and mortality associations that are similar to those observed in communities where PM exposure is primarily from industrial and mobile sources (Schwartz et al. 1993; Lipsett et al. 1997; Fairley 1999; Hales et al. 2000; McGowan et al. 2002; Boman et al. 2003; Sanhueza et al. 2009). Wood-burning stoves emit significant quantities of known toxic compounds, including polyaromatic compounds, benzene, aldehydes, carbon monoxide, nitrogen oxides, and respirable PM (Naeher et al. 2007). The particles emitted from wood combustion are predominantly smaller than 1 µm in aerodynamic diameter and have a peak mass size distribution ranging from 0.1 to 0.4 µm (Kleeman et al. 1999; Hays et al. 2002). These small particles can penetrate deep into the lung tissue with a great potential for causing adverse health outcomes such as airway inflammation or susceptibility to respiratory infection.

The Consumer Product Safety Commission estimates that there are 8.9 million wood stoves in use in the United States, and these stoves are the most intensively used type of space heater with an average annual usage per heater of 2100 hours (Zamula 1989; Air Quality Management Work Group 2005). Eighty to ninety percent of the wood stoves currently in use are old and inefficient (Air Quality Management Work Group 2005). Estimates vary greatly, but the rate of $PM_{2.5}$ emissions from old wood stoves is approximately 15 to 30 grams of $PM_{2.5}$ per hour compared with 2 to 7 grams per hour for newer EPA-certified wood stoves (U.S. EPA 2010).

Some communities experience sustained periods of stagnant air inversions because of regional topographic and meteorologic conditions. Biomass smoke PM can become concentrated in the air during these inversions, potentially resulting in noncompliance with the U.S. EPA National Ambient Air Quality Standard (NAAQS) for $PM_{2.5}$. One such community was Libby, Montana, which received the EPA designation of NAAQS *nonattainment* for annual $PM_{2.5}$ in 2004. The results of a chemical mass balance (CMB) source apportionment model revealed that residential wood combustion was the single major source of $PM_{2.5}$ throughout the winter months in Libby, contributing an average of 82% of the measured $PM_{2.5}$ concentration throughout the sampling program (Ward et al. 2006). Wood stoves in this community were predominantly of old stock, and the local agencies did not have a comprehensive wood-burning appliance registry program. With support from local, state, and federal agencies and from private interests, the community initiated a large-scale wood stove change-out program to replace over 1100 old wood-burning devices with new U.S. EPA-certified wood stoves or similar heating appliances. Long term compliance was monitored through a comprehensive registry program.

Aggressive campaigns to replace or remove uncertified wood stoves can have a substantial impact on local air quality. During the summer of 1989, 48% of the 406 uncertified stoves in Crested Butte, Colorado were replaced by U.S. EPA-certified units and another 33% were removed or disabled. The community experienced a 60% reduction in air pollution during the subsequent winter (Houck et al. 2005). In Missoula, Montana, public education efforts and stringent restrictions on wood burning coincided with a 45% reduction in winter PM_{10} ($PM \leq 10$ µm in aerodynamic diameter) concentrations, and the source contribution of residential wood combustion was reduced from 47% to 10.5% (Missoula City–County Health Department 1999). The experiences in these and other communities that, like Libby, Montana, are topographically isolated, suggested that targeting wood stoves can have a dramatic effect on winter air quality. However, there has been no evaluation of how such improvements in air quality affect the health of people living in these communities.

SPECIFIC AIMS

This study was undertaken in response to recent calls for studies to investigate the health effects of air quality interventions driven by regulatory standards (National Research Council 2002; HEI Accountability Working Group 2003). Utilizing both compliance monitoring data and additional air sampling over a four year period, changes in the air quality of Libby, Montana, were assessed for both ambient air and indoor air. The evaluation of community-wide changes in children's health was assessed by administering questionnaires to parents of children in grades 1–12 who lived in Libby and by tracking illness-related absences from Libby schools. The present study had the following specific aims:

1. Prospectively measure ambient (1a) and indoor (1b) $PM_{2.5}$ concentrations during and after the implementation of the community-wide wood stove replacement program to test the hypothesis that a wood stove intervention program will substantially reduce wood-smoke–derived PM exposures in the community.
2. Prospectively track respiratory symptoms and infections among children to test the hypothesis that intervention-driven reductions in $PM_{2.5}$ concentrations will result in reductions in children's respiratory symptoms.
3. Prospectively track school absences to test the hypothesis that intervention-driven reductions in $PM_{2.5}$ concentrations will reduce illness-related school absence.

METHODS

HUMAN STUDIES APPROVAL

This project was approved by the Institutional Review Board of The University of Montana.

STUDY PERIODS AND DEFINITION OF TERMS

In this context *changeout* refers to the removal of older, high-emitting wood stoves and replacement with U.S. EPA-certified wood stoves that met the $PM_{2.5}$ emission standard of < 7.5 g/hr. The changeout also included modification or refitting of existing certified stoves that were not operating efficiently and replacing older wood stoves with nonbiomass heating sources.

The air quality intervention in this community was targeted at a cold-temperature PM source. Thus, the air monitoring and corresponding health-related outcomes for this project was limited to winters. For the purposes of this study, the winter period includes November through February. Ambient air monitoring data were tracked for six winter periods. Data for the first two periods (baseline 1 and 2) were collected retrospectively:

1. baseline 1 (2003–2004) refers to the winter two years before the project started;
2. baseline 2 (2004–2005) refers to the winter one year before the project started;
3. year 1 (2005–2006) refers to the beginning of the wood stove changeout program and is the first winter of this project;
4. year 2 (2006–2007) refers to a highly active period in the wood stove changeout program and is the second winter of this project;
5. year 3 (2007–2008) corresponds to a second highly active period in the wood stove changeout program and is the third winter of this project;
6. year 4 (2008–2009) refers to the post-changeout period.

Additional detail on the timing of the wood stove changeout program and these project years are provided in the Results section.

STUDY LOCATION

Libby is a small community occupying approximately 3.4 square kilometers of Lincoln County, Montana. The city proper had an estimated population of approximately 2600 people in the year 2000. The median household income for that year was $24,276 compared with $41,994 for the United States. Approximately 10% of households were below the poverty level. Libby sits in a deep valley floor on the Kootenai River at an elevation of 628 meters. The surrounding mountains rise to an elevation of 1828 meters.

In past years the predominant employers in Libby had been the mining and wood products industries. The local lumber mill closed in 2003, but the area wood products industries had already scaled back considerably by the early 1990s. The largest mining facility in the area was a vermiculite operation that began in the early 1890s and continued until its closure in 1990. The mining site, located just outside of Libby, produced 80% of the world's vermiculite, a product used for insulation, soil amendments, and various other products. The vermiculite from this site was naturally contaminated with amphibole asbestos, and several studies and health surveillance activities have investigated asbestos-related disease and occupational as well as environmental asbestos exposures among adults in the town of Libby and surrounding communities (Peipins et al. 2003; Noonan 2006; Noonan et al. 2006; Larson et al. 2010). In 2002, Libby was added to the National Priorities List, or Superfund, and in 2009 the U.S. EPA declared the first-ever Public Health Emergency at a Superfund site, enabling the provision of federal health care assistance for the Libby community.

The $PM_{2.5}$ nonattainment area is a 370 square kilometer quadrangle region extending beyond Libby proper with an estimated population of 12,000. There is no natural gas line serving the city or the surrounding communities. Therefore, home heating is accomplished using electric, propane, oil, or wood-burning devices. The Libby Census County Division is slightly larger than the designated nonattainment area. Census-based estimates indicated that 1325 of 4178 households (≈32%) used wood as a main heating fuel. The best estimate of the number of conventional wood stoves (i.e., noncatalytic, pre-1990) in the Libby air shed, and the number used by the Lincoln County Health Department for planning purposes, was 1175 (Eagle and Houck 2007a).

Libby has been identified as a single-source air shed. During the heating season of 2003–2004 (baseline 1), a source apportionment program was conducted in Libby to identify the sources of $PM_{2.5}$. During this program, $PM_{2.5}$ mass concentrations averaged 28.2 $\mu g/m^3$. Results of the CMB modeling identified residential wood combustion as the source for more than 80% of the $PM_{2.5}$ measured during the heating season (Ward et al. 2006). There are no significant industrial sources in the Libby air shed. Various nonbiomass source contributions to $PM_{2.5}$ concentrations in the Libby air shed have been quantified, but no individual source was a large contributor. Another study of ambient polycyclic aromatic hydrocarbons and phenolic compounds found that the organic species with the highest concentrations measured were also signature chemical markers for wood combustion (Ward et al. 2009).

AMBIENT AIR

Compliance Monitoring

The Montana Department of Environmental Quality's (DEQ) $PM_{2.5}$ compliance site for the town of Libby has been located on the roof of the Lincoln County Health Department in downtown Libby since 2003. This site, known as the Courthouse Annex, is the representative sampling site for the entire Libby valley, and is located nearly one kilometer from the elementary school and approximately three kilometers from the middle school. Each of the homes involved in the residential indoor air sampling is located between 0.8 and 13.7 kilometers from the sampling site.

The Montana DEQ conducted two studies to determine the representativeness of the central site monitor for the community of Libby. To determine the geographical extent of the $PM_{2.5}$ concentrations from Libby, a winter monitoring study was conducted from November 2003 to March 2004 (Montana DEQ 2004) (baseline 1). The goal of this study was to establish the boundary that adequately encompassed the area violating the annual $PM_{2.5}$ NAAQS of 2004, as well as any area that may be contributing to the violation. A comparison of the $PM_{2.5}$ concentrations at the Courthouse Annex with $PM_{2.5}$ concentrations measured at three other sites on the edge of the community revealed that the Courthouse Annex concentrations exceeded the other sites in every case. The Courthouse Annex site is near the center of the Libby community, and is not affected by local sources. Final results from this study showed that the boundary surrounded a significant portion of the population as well as source areas. In addition, the boundary adequately addressed the unique meteorology and topography of the area.

A second monitoring study was conducted during the winter of 2008–2009 (year 4, post-changeout) to determine whether the Libby-Courthouse Annex site was still representative of its neighborhood five years after the original study (Montana DEQ 2009). Comparison of $PM_{2.5}$ data collected by remote β-gauge PM mass monitors (e-BAM, Met One Instruments, Grants Pass, OR) to the values monitored at the Courthouse Annex site revealed very consistent $PM_{2.5}$ concentrations over a distance of at least four city blocks. In addition, values measured at the Courthouse Annex site in the center of the commercial district compared closely with values measured at residential locations outside the commercial district, where there were more trees and also more wood stove use.

The compliance monitoring conducted by Montana DEQ and Lincoln County included running 24-hour $PM_{2.5}$ mass samplers (Federal Reference Method [FRM] BGI PQ200s) and continuous PM_{10} and $PM_{2.5}$ samplers (Rupprecht & Patashnick Co, East Greenbush, NY tapered element oscillating microbalances). FRM sampling was conducted on a one in three-day schedule. The net mass on the Teflon filters collected by the FRM samplers was determined gravimetrically by weighing each Teflon filter before and after sampling with a microbalance in a temperature- and relative humidity-controlled laboratory environment. $PM_{2.5}$ reference methods require that filters be equilibrated for 24 hours at a constant relative humidity (± 5%) between 30% and 40% and at a constant temperature (± 2°C) between 20°C and 23°C to minimize particle volatilization and aerosol liquid water bias (U.S. EPA 1998).

Additional Ambient Air Sample Collection

At the Libby compliance site, the Montana DEQ also installed a $PM_{2.5}$ speciation sampler collocated with the FRM $PM_{2.5}$ samplers. The Met One spiral ambient speciation sampler (Met One Instruments, Grants Pass, OR) collected 24-hour integrated samples on Teflon, nylon, and quartz filter media on a one in six-day schedule. $PM_{2.5}$ mass, elements, organic carbon (OC), elemental carbon (EC), and ions were all measured from the collected filters. In an effort to measure other specific organic compounds found associated with the ambient $PM_{2.5}$ concentrations, we collected an extra $PM_{2.5}$ quartz filter every six days following the U.S. EPA's fixed sampling schedule by utilizing one of the existing FRM $PM_{2.5}$ samplers at the site. Prefired 47-mm quartz filters (fired at 500°C for 2.5 hours) were purchased from Chester LabNet (Tigard, OR), and delivered to Lincoln County personnel in a cooler. The clean quartz filters to be used for sample collection were stored in a refrigerator at approximately 2°C. After the sample collection, the quartz filter samples were stored in a freezer at −20°C until analysis. Approximately 24 m^3 of air was sampled during each 24-hour episode. Quartz filter field blanks were also collected periodically throughout the program to address artifact contamination. These additional quartz filters were analyzed at The University of Montana for polar organic compounds, which included chemical markers for wood smoke such as levoglucosan and resin acids (abietic acid and dehydroabietic acid). Use of the spiral ambient speciation sampler was discontinued by the Montana DEQ before the final winter of the study. To obtain values for OC, EC, and polar organic compound concentrations for year 4, the extra quartz filters were sent to Desert Research Institute (Reno, NV) for analysis.

INDOOR AIR

Residential Indoor Air Sampling

Air sampling was conducted in 26 selected homes before wood stove replacement in the respective homes. Selection was limited to those homes with a completed inspection, approval for changeout per the county's program guidelines, and plans to replace the stove during the winter. The latter requirement allowed us to conduct pre- and post-changeout sampling in the same winter period for some homes. Homes were not eligible for selection if any resident was a tobacco smoker. The pre-changeout sampling for all homes was conducted in year 2. Post-changeout sampling for 24 of the homes was conducted during three winters (years 2–4).

The difference in ambient temperature between the pre-changeout and post-changeout measurements was greater than 10°F for more than half of the homes in this initial sample (10/19). There was concern that ambient temperature discrepancies on sampling days would affect burning behavior (fuel usage) in homes with wood stoves, thereby obscuring the impact of new wood stoves on indoor air quality. Additional post-changeout sampling during the subsequent two winter periods was conducted with an attempt to match the sampling day as closely as possible to ambient temperature of both the pre-changeout and the initial post-changeout sampling days for the corresponding homes. Altogether, the residential sampling program included five phases:

1. phase 1 = pre-changeout, year 2 (26 homes);
2. phase 2 = post-changeout, year 2 (24 homes);
3. phase 3 = post-changeout, year 3, matched on phase 1 ambient temperature (14 homes);
4. phase 4 = post-changeout, year 3, matched on phase 2 ambient temperature (14 homes);
5. phase 5 = post-changeout, year 4, no temperature match (16 homes).

During a sampling day, two sampling instruments operated in the home for 24 hours. A DustTrak (TSI, Shoreview, MN) was used to continuously monitor $PM_{2.5}$ concentrations at 60 second intervals. A single-filter sampler (Personal Environmental Monitor [PEM]) using a Leland Legacy Sample pump (SKC, Eighty Four, PA) was used to collect $PM_{2.5}$ on a prefired 37-mm quartz filter for analysis of OC, EC, and organic markers of wood smoke. Residents logged information on the frequency of wood stove loading for each residential indoor air sampling event. Residents also recorded information on home characteristics or activities that would affect indoor air quality. Home characteristics and activity variables included various types of cooking modes, usage of secondary heating sources, various types of indoor fans, burning of candles, incense, or oil lamps, various cleaning activities, outdoor or indoor construction, open windows or doors, and presence of household pets. For data collection instruments see Appendix D (available on the HEI Web site).

School Indoor Air Sampling

Periodic indoor air sampling was conducted both at the one grade school and the one middle school in the community during the fall, winter, and spring in years 1–4. The number of school indoor air sampling days per year ranged from 16 to 22. The two schools were located approximately 2.4 km from one another. No sampling was conducted at the Libby High School. Before choosing the sampling locations within the schools, the investigators evaluated the sites based on the following criteria: (1) areas designated by school administrators as acceptable; (2) adequate power requirements; (3) security of the site; and (4) safety of the site for children in proximity to this area. The gymnasium met these criteria at both schools. At the elementary school, PM samplers were installed on a stage approximately 4 feet above the gymnasium floor and 20 feet from an exterior door. At the middle school, the school gymnasium was in a building detached from the main building. The PM samplers were installed on an elevated platform approximately 20 feet above the gymnasium floor and 40 feet from an exterior door.

Two PEMs were used in each school; one sampler at each school was fitted with 37-mm Teflon filters for the gravimetric analyses, while another sampler at each school was fitted with 37-mm prefired quartz filters for the OC and EC analyses. Finally, a $PM_{2.5}$ cyclone was fitted with prefired quartz filters for the collection of indoor $PM_{2.5}$ to analyze polar organic compounds.

FILTER ANALYSES

$PM_{2.5}$ Polar Organic Compound Analysis

Quartz filter samples (of both indoor and outdoor air), field blanks, and trip blanks were delivered to The University of Montana Center for Environmental Health Sciences in Missoula, Montana, for analysis of the seven chosen chemical tracers for wood smoke (Bergauff et al. 2008). Analysis was performed on an Agilent 6890N gas chromatograph with an Agilent 5973 mass spectrometer (GC-MS) (Agilent, Santa Clara, CA). One half of each filter (from the school, residential, and ambient sampling programs) was spiked with the deuterated standards, then compounds were extracted with sonication into ethyl acetate containing 3.6 mM triethylamine. The volume of the solution was reduced, and the samples were split into two equivalent portions. One portion was derivatized to triethylamine (to be analyzed for the methoxyphenols) with a freshly prepared 2:3 mixture of acetic anhydride. The other portion was evaporated to dryness and then derivatized with a mixture of N-O-bis(trimethylsilyl)trifluoroacetamide, trimethylchlorosilane, and trimethylsilylimidazole to be analyzed for levoglucosan and the resin acids. The portion for levoglucosan and the resin acids was diluted with ethyl acetate containing 3.6 mM triethylamine. Both portions were then analyzed by GC-MS. For all compounds, highly selective quantitation was performed using the signal for representative ions for each compound extracted from the total ion chromatogram (Bergauff et al. 2008).

As indicated earlier, the speciation sampler had been discontinued before the final winter, so the extra quartz filters from year 4 went to an outside laboratory, Desert Research Institute, for the OC and EC analyses. In addition, Desert Research Institute analyzed these filters for polar organic compounds using the IMPROVE_A method developed by their laboratory (Chow et al. 1993; Desert Research Institute 2008), which was similar to the methods described above.

Analyses of OC, EC, and PM Mass

Teflon filters (PM mass) from the indoor school sampling and the quartz filter halves (OC and EC) from all indoor sampling programs were sent to Chester LabNet for analysis. A gravimetric analysis of PM mass was performed on the Teflon filters per the National Institute of Occupational Safety and Health Manual of Analytical Methods (NMAM) Method 0500, while NMAM 5040 was used on the quartz filters for the OC and EC analysis. For the OC and EC method, a gravimetric analysis was performed on a representative 1×1.5 cm^2 area of the quartz filter. The micrograms of OC or EC on the entire filter were calculated after measuring the diameter of the sample area, assuming an even distribution of OC and EC on the quartz filter (Ward et al. 2007; Ward et al. 2008).

SURVEY

Questionnaires were delivered to parents of school children in the community during four winter periods (years 1–4) to assess the frequency of the children's winter period respiratory symptoms and infections. The survey was adapted from core questionnaires on asthma, rhinitis, and eczema from the International Study of Asthma and Allergies in Childhood, which was designed to screen for the prevalence and severity of asthma and allergic disease in defined populations (Asher et al. 1995). The survey for the present study included additional questions on respiratory infections based on the National Health and Nutrition Examination Survey. Parents were asked to report on their child's symptoms and health conditions during the past two months, the type of heating used in their home, and the occurrence of tobacco smoking in the home. For the survey questionnaire see Appendix E (available on the HEI Web site).

Each winter, school administrators notified parents of the upcoming survey through their respective school newsletters. At the beginning of March, questionnaires were sent home with children through their home room teacher for the grade schools and middle schools and through their Health Enhancement teachers for Libby high school. Questionnaires were distributed and returned in a sealed manila envelope. Year 1 included the grade school (grades 1–4) and the middle school (grades 5–8). In year 2 the kindergarten students were moved to the grade school and 4th graders were moved to the middle school. The year 2 survey included kindergarten through 8th graders as well as the 9th and 10th grade students from the high school. 11th grade students were added to the survey in year 3, and 12th grade students were added in year 4. There were no kindergartners in the year 1 group, so this grade was dropped from the analysis of the subsequent years.

During two of the survey years an effort was made to increase response rates. After reviewing questionnaires that had been returned to the schools by mid-March, questionnaires were mailed to the homes of parents who had not yet returned a survey but who had responded in previous years. Of 351 questionnaires mailed to parents in year 2, 40 were returned. Of 488 questionnaires mailed to parents in year 3, 60 were returned.

In March 2005 (of baseline year 2), we pilot-tested a similar survey with parents of Libby school children using the methods already described (Noonan and Ward 2007). The questions in this earlier survey were similar but not identical to the questions used in the present study; the changes were based on our evaluation of the results and on feedback from the HEI Research Committee. For example, parents were asked to report on the occurrence of childhood wheeze in the previous four weeks and during the previous 12 months, whereas in the present study parents reported on childhood wheeze in the previous two months. The discrepancies between these survey instruments preclude direct comparisons, but the 2005 survey does offer some useful data for comparison with the results of this study.

SCHOOL ABSENCES

At the start of the study the schools used different absence tracking systems. The middle school had electronic archiving of absences dating to the 2003–2004 school year (baseline year 1). These archived records were obtained from the school district office and prospective years of absence data were requested directly from the school secretary. The grade school did not have electronically archived absence records until year 4. Absence call sheets were collected from the grade school for years 1 through 3 and manually entered into a dataset. Absence data from both schools included daily reporting entries for each absent student. Each entry included student identification number, grade, sex, times in and out (when relevant), and a comment field. The comment field was a text entry informed by the school communication with the student's parent or guardian. Based on the comment entry, absences were coded as one of the following: asthma absence, other respiratory or infectious absence (e.g., bronchitis, tonsillitis, sore throat, pleurisy, cold, and influenza), illness-related absence not otherwise specified, and non-illness absence.

An incident absence was an absence that followed attendance on the preceding school day. A prevalent absence was an absence that followed an absence on the preceding school day. Only incident absences were used in the analyses. Monthly school enrollment figures for each grade were provided by the school district office. To calculate daily school incident absence rates we included the number of children at risk for an absence in the model. The number of children at risk for absence was based on the number of children enrolled for a given school day, assuming static enrollment through the month, minus the total number of absences on the previous day.

OTHER DATA

Meteorologic Data

Daily temperature, wind speed, relative humidity, and precipitation data were obtained from the database collected by the Western Regional Climate Center, a division of the National Oceanic and Atmospheric Administration (Western Regional Climate Center 2010). The measurement station was located in Libby, at latitude 48° 23′ 00″ and longitude 115° 34′ 00″.

Surrogate for Seasonal Influenza Activity

An indicator variable for the influenza activity in the community during each winter period of years 1–4 was based on data obtained from the local hospital, which was the only hospital in Lincoln County. Influenza rates for winter periods were based on hospital discharge records that had influenza, code 487, listed as any of the five diagnostic codes on a patient's discharge papers for January through March of a given year. The denominators for these influenza rates were based on annual U.S. Census population estimates for Lincoln County (U.S. Bureau of the Census 2010). The total population for Lincoln County was approximately 18,800 compared with an approximate 12,000 within the Libby $PM_{2.5}$ nonattainment area.

DATA ANALYSIS

DESCRIPTIVE ANALYSIS

For descriptive analysis of ambient $PM_{2.5}$, OC, EC, and polar organic compounds we compared average winter concentrations in years 1 through 4 with combined prechangeout winter concentrations in baseline years 1 and 2. Air sampling and filter analyses data for indoor school measurements are summarized by mean and standard deviation (SD) according to school site and season (Appendix B).

CHANGE IN AMBIENT AIR (AIM 1A)

To estimate the effect of the wood stove changeout program on ambient $PM_{2.5}$ concentrations we fit a linear model by ordinary least squares. The outcome variable was daily ambient $PM_{2.5}$ concentrations during the winter periods. The primary independent variable was cumulative changeout, which was defined as the percentage of completed changeouts in a given day to the total number of completed changeouts (i.e., stoves reconditioned, replaced, or decommissioned). Other variables included in the model were daily ambient temperature, month, an indicator of workday versus weekend or holiday, and ambient $PM_{2.5}$ concentration lagged one day. The month indicators for November through February were not specific to the year, thus the estimated effect for each month was constrained to be the same for all years. The daily ambient $PM_{2.5}$ concentrations before November 10, 2005 were limited to an every third day measurement, so lagged $PM_{2.5}$ values were not available before that date.

We evaluated various permutations of the model described above. We replaced month indicators with cubic splines using one degree of freedom per month. We also assessed two-way interactions (e.g., cumulative changeout by ambient temperature). Finally, we fit the model with $PM_{2.5}$ concentrations lagged two or three days rather than one day. None of these sensitivity analyses changed the conclusions.

To separate the effects of changeout from the effect of the previous day's $PM_{2.5}$ concentration, we also fit the model described above without adjusting for lagged $PM_{2.5}$ concentrations. Consecutive daily $PM_{2.5}$ values from November 10, 2005 (which corresponds to year 1) through the end of the study are autocorrelated, whereas observations before that date are nearly independent due to the every third day measurement. To maintain a constant autocorrelation structure, we restricted this sensitivity analysis to the dates after November 2005. Due to the autocorrelation we fit a two-stage model, estimating the autocorrelation in the first stage and using it to "prewhiten" the data for the second stage fit. This is essentially the best linear unbiased estimator for errors with lag 1 autocorrelation.

Secondary analyses of central site ambient pollutants used the same approach but with different 24-hour average outcome measures, including OC, EC, and polar organic compounds. Secondary analyses of indoor school $PM_{2.5}$ concentrations used the same approach to evaluate 24-hour $PM_{2.5}$ mass, OC, EC, and polar organic compounds.

CHANGE IN RESIDENTIAL INDOOR AIR (AIM 1B)

To compare indoor $PM_{2.5}$ concentrations across the multiple home sampling visits we fit a generalized estimating equation assuming a Gaussian model and exchangeable covariance matrix. The dependent variable was indoor 24-hour average $PM_{2.5}$ concentration. The primary independent variable was an indicator of changeout, indicating either a pre-changeout sampling or a post-changeout sampling. Other explanatory variables included in the model were ambient $PM_{2.5}$ concentration corresponding to the same 24-hour indoor sampling period, ambient temperature, and the resident's baseline (i.e., pre-changeout) reporting of the number of cords of wood burned during a typical winter. Various activities were recorded by the resident during the sampling period and included in the model: cleaning (i.e., vacuuming, sweeping, or dusting); cooking (i.e., frying, deep-fat frying, baking, indoor grilling, or outdoor grilling); and burning of candles or incense. The presence of dogs or cats in the home was also included in the model. Any sampling events that occurred during a time when the resident reported indoor tobacco smoking were not included in the analysis.

To determine if any reductions in indoor $PM_{2.5}$ concentrations persisted in subsequent winters after the changeout we compared post-changeout indoor $PM_{2.5}$ concentrations during follow-up winters (years 3 and 4) with post-changeout indoor $PM_{2.5}$ concentrations during the initial winter (year 2). We fit a generalized estimating equation, adjusting for the variables described above.

SURVEY DATA (AIM 2)

After reviewing all survey response data from the grade and middle schools, we chose "presence of wheeze" from individual symptom reports as our primary outcome. Secondary outcomes were respiratory infectious conditions and other symptoms commonly considered to be affected by air pollution (e.g., attack of shortness of breath, shortness of breath after exercising, awakening from coughing, morning tightness of chest). To estimate the association of $PM_{2.5}$ concentration with wheeze prevalence, adjusting for age group of child, influenza, and wood stove use, we used a generalized estimating equation with a logit link, as follows:

$$\log \frac{p_{ij}}{1 - p_{ij}} = \beta_0 + \beta_1 \mathrm{PM}_j + \beta_2 \mathrm{Stove}_{ij} + \beta_3 \mathrm{Flu}_j + \gamma \mathrm{Age}_{ij}$$

where p_{ij} was the probability that subject i had a reported health outcome in year j, PM_j was the November–February average $PM_{2.5}$ concentration in year j, Stove_{ij} was a dummy variable indicating wood stove use in home of subject i in year j, and Flu_j was the community influenza rate in year j. We adjusted for age by dividing the subjects by grade into two-year groups (i.e., grades 1–2, 3–4, . . . , 11–12). We further assumed that repeated measures on a given subject were exchangeable. Results are presented as odds ratio (OR) and 95% confidence interval (CI).

Secondary analyses used alternate ambient PM exposure metrics such as OC, EC, and polar organic compounds from the central site.

SCHOOL ABSENCE DATA (AIM 3)

The primary analysis was restricted to the wood burning months of November through February. We fit overdispersed Poisson generalized linear models with a log link. The dependent variables were illness absences, respiratory absences (a subset of illness absences), and non-illness absences. The independent variables were average daily $PM_{2.5}$ concentration, daily $PM_{2.5}$ concentration lagged one day, annual $PM_{2.5}$ concentration (i.e., average winter $PM_{2.5}$ concentration), and annual community influenza rate. Because of equipment malfunction, daily ambient $PM_{2.5}$ concentration data were missing for a small number of days in the year 1 winter, 12/28/2005 through 1/24/2006. The missing values for these days were imputed via linear interpolation, using the county's federal reference method data that were available every third day. Also included in the model were indicators for day-of-week, grade, and month. The month indicators, November through February, were not specific to the year. Meteorologic variables were then considered for inclusion in the models. A secondary analysis was conducted to assess the within-year effect of daily variation in $PM_{2.5}$ concentrations. Dummy variables for each year were included in these latter models. Results are presented as OR and 95% CI.

The average winter change in ambient $PM_{2.5}$ concentration was the primary exposure of interest, so additional analyses were conducted after removing daily $PM_{2.5}$ concentrations and one-day lag $PM_{2.5}$ concentrations. We then stratified by grade groupings to determine if there were any differential effects of winter ambient $PM_{2.5}$ concentrations on school absences by age group. Stratification was based on the two schools, the elementary school for grades 1–4 and the middle school for grades 5–8. As described earlier, the grade groupings changed slightly at the two schools over the course of the study, but these grade groupings are consistent with the school grade groupings during the first winter of the study. For the stratified analyses we assessed both average winter $PM_{2.5}$ concentration and cumulative wood stove changeout. Winter $PM_{2.5}$ concentration and cumulative percentage of wood stove changeout were highly negatively correlated ($r = -0.98$). We therefore fit separate models, one with average winter $PM_{2.5}$ concentration as the primary exposure variable and one with cumulative stove changeout in place of average winter $PM_{2.5}$ concentration.

A secondary analysis was conducted to incorporate the archived absence data obtained from the middle school. Six winters, baseline 1 through year 4, were evaluated for grades 5–8. For each grade we compared the illness absences for pre-changeout winters, baseline 1, 2, and year 1, with post-changeout winters, years 2 through year 4. Absences for each grade and time period were modeled using Poisson regression with mean absences proportional to the number at risk and compared using the generalized likelihood ratio test. To evaluate the effect of average ambient $PM_{2.5}$ concentration and cumulative wood stove changeout over these six winter periods we fit an overdispersed Poisson generalized linear model with a log link as described earlier. As with the primary analyses, average ambient $PM_{2.5}$ concentration and cumulative wood stove changeout were fit as separate models.

QUALITY ASSURANCE / QUALITY CONTROL PROGRAM

A comprehensive QA/QC program was employed throughout for the ambient, school, and residential sampling programs and for the chemical analyses. The Montana DEQ was responsible for the QA/QC activities involving the ambient $PM_{2.5}$ FRM and tapered element oscillating microbalance samplers, while Intermountain Laboratories (Sheridan, WY) and the Research Triangle Institute (Research Triangle Park, NC) were responsible for the ambient program gravimetric and chemical speciation QA/QC analytical activities. The University of Montana was responsible for all sampling and analytical activities involving the school and residential programs. The University of Montana was also responsible for the analytical QA/QC activities involving the ambient chemical markers of wood smoke.

INDOOR SAMPLING

Using a certified DryCal flow meter (Bios International Corporation, Butler NJ), the flow rate for the PEM was measured both before and after each sampling event. Quartz filter field blanks were collected for approximately every 10 samples (10%). Field personnel followed the recommended maintenance and cleaning schedules for the DustTrak and PEM as described in their respective manuals throughout the program. The DustTrak was zeroed before each sampling event, with results documented on datasheets. Within The University of Montana laboratory, the QA/QC program for chemical markers of wood smoke included the analysis of blank filters (one blank filter was analyzed for every 10 samples), spikes, instrument calibration checks, and routine instrument maintenance. Chester LabNet (contracted to analyze for OC and EC) followed their own comprehensive QA/QC program in generating mass, OC, and EC results.

DETECTION LIMITS

Minimum detection limits (MDLs) for each of the parameters measured are presented in Appendix Table B.1. For $PM_{2.5}$ mass, MDLs are reported in the DustTrak manual. MDLs for OC and EC were reported by the contracted laboratory in micrograms. To calculate the MDLs for OC and EC, their respective values were divided by the average air volume collected with the PEM during each sample run (results reported in $\mu g/m^3$). MDLs for levoglucosan, the resin acids, and the methoxyphenols were also calculated. A calibration curve was created using the ratio of the peak height of each concentration of standard to the peak height of the deuterated internal standard. Peak-to-peak noise was estimated before and after the standard peak in each file to give an average value. The SD multiplied by 3, taken as 1/5 of the peak-to-peak noise, was used as the MDL. The ratio of 3/5 of the peak-to-peak noise to the deuterated internal standard was used to calculate the MDL from the calibration curves.

DATA ENTRY

All parent questionnaires were double-entered and compared using a spreadsheet. Inconsistencies between the two databases were then reconciled by referring back to the hard copy survey data, and a new accurate dataset was created for each year. The Principal Investigator (Noonan) checked the absence data for accuracy and inconsistencies. All air sampling data requiring data entry were reviewed by the coinvestigator (Ward). Data were uploaded to a server after being checked by a data validation program for expected data types, ranges, or both. The data were further reviewed for errors and inconsistencies by the biostatistician (Navidi).

Data collection and results related to the QA/QC Program can be found in Appendix F (available on the Web).

RESULTS

CHANGEOUT PROGRAM

The community wood stove changeout program occurred in two phases. The initial phase began in June 2005 and continued to June 2007. During this phase, 260 wood stoves donated by wood stove manufacturers were distributed to eligible low-income families in the community. Residents were eligible for this phase if they were already participating in the Low Income Energy Assistance Program or if their low income status was confirmed by the Northwest Montana Human Resources, a nonprofit third-party agency. All homes within the Air Quality Control District that met the low income eligibility requirements were able to participate in the initial phase of the program. A variety of wood stoves were offered through the manufacturers, but all devices adhered to the current industry low emissions standard according to U.S. EPA criteria. These criteria limited emissions to less than 7.5 grams of $PM_{2.5}$ per hour for noncatalytic stoves and less than 4.1 grams of $PM_{2.5}$ per hour for catalytic stoves. By comparison, older noncertified stoves typically had emissions of 15 to 30 grams of $PM_{2.5}$ per hour (U.S. EPA 2010). Recipients of these stoves also received free installation and any necessary modifications to hearth pads, chimneys, and duct work. Costs for this phase of the program were estimated at $2900 per home (Eagle and Houck 2007a).

The overlapping second phase began in February 2006 (the end of the year 1 winter of the present study), and documented installations continued through March 2008 (after the year 3 winter of the present study). During this phase the changeout program used a voucher system to directly fund eligible homeowners for the purchase and installation of new wood-burning devices. Homeowners with uncertified wood stoves could receive up to $1050 in vouchers and homeowners with uncertified wood furnaces could receive up to $1750 in vouchers. A key feature of this phase of the program was that homeowners were able to select a replacement heating device of their choosing as compared with the homeowners from phase one who had appropriate wood stoves selected for them from among the devices donated by cooperating manufacturers. Throughout both phases of the changeout program the Lincoln County Health Department and its partners engaged in extensive community outreach efforts to familiarize the community with the program. Community outreach efforts included biweekly newspaper advertisements, radio and television public service announcements, presentations to service clubs, and distribution of flyers and posters. The Health Department also organized several "stove fairs" and media events (Eagle and Houck 2007b).

Figure 1 illustrates the timeline for the wood stove program. An estimated 1175 conventional (noncatalytic, pre-1990) wood-burning appliances were in use before the changeout program. By the end of the program, 1147 older wood stoves were replaced with new U.S. EPA-certified wood stoves (n = 736), wood inserts (n = 57), wood furnaces (n = 11), pellet devices (n = 170), or other nonbiomass combustion heating devices such as propane, oil, or electric (n = 86). This number also includes existing stoves that were reconditioned to meet U.S. EPA emission guidelines (n = 81) or that were surrendered (n = 6). Despite the uncertainty surrounding the original estimate of 1175 existing conventional wood stoves before the changeout program, it was clear that the program reached a high percentage (> 95%) of wood-burning homes. To track the timing of the wood stove changeout program we have used the cumulative percentage of completed changeouts with 1147 as the denominator.

AMBIENT AIR

As illustrated in Figure 1, winter ambient $PM_{2.5}$ concentrations in Libby were considerably lower in the three winters during and after the changeout program (years 2, 3, and 4) compared with the baseline winters before the changeout program (baseline 1 and 2). Winter average $PM_{2.5}$ concentrations were 22.4% lower in year 2, 25.3% lower in year 3, and 30.1% lower in year 4 compared with the average of baseline 1 and 2 values (see Table 1). The wood stove changeout program had been initiated before the year 1 winter, but the average $PM_{2.5}$ concentration during this winter was similar to those of the baseline winters; the mean difference was −0.19 μg/m^3 (95% CI = −2.9 to 3.3) (Table 1). Only 112 (9.8%) of wood stoves had been changed before the year 1 winter, and an additional 38 (3.3%) stoves were changed during that winter.

The U.S. EPA implemented the revised 2006 24-hour ambient standard for $PM_{2.5}$ (35 μg/m^3) during the program.

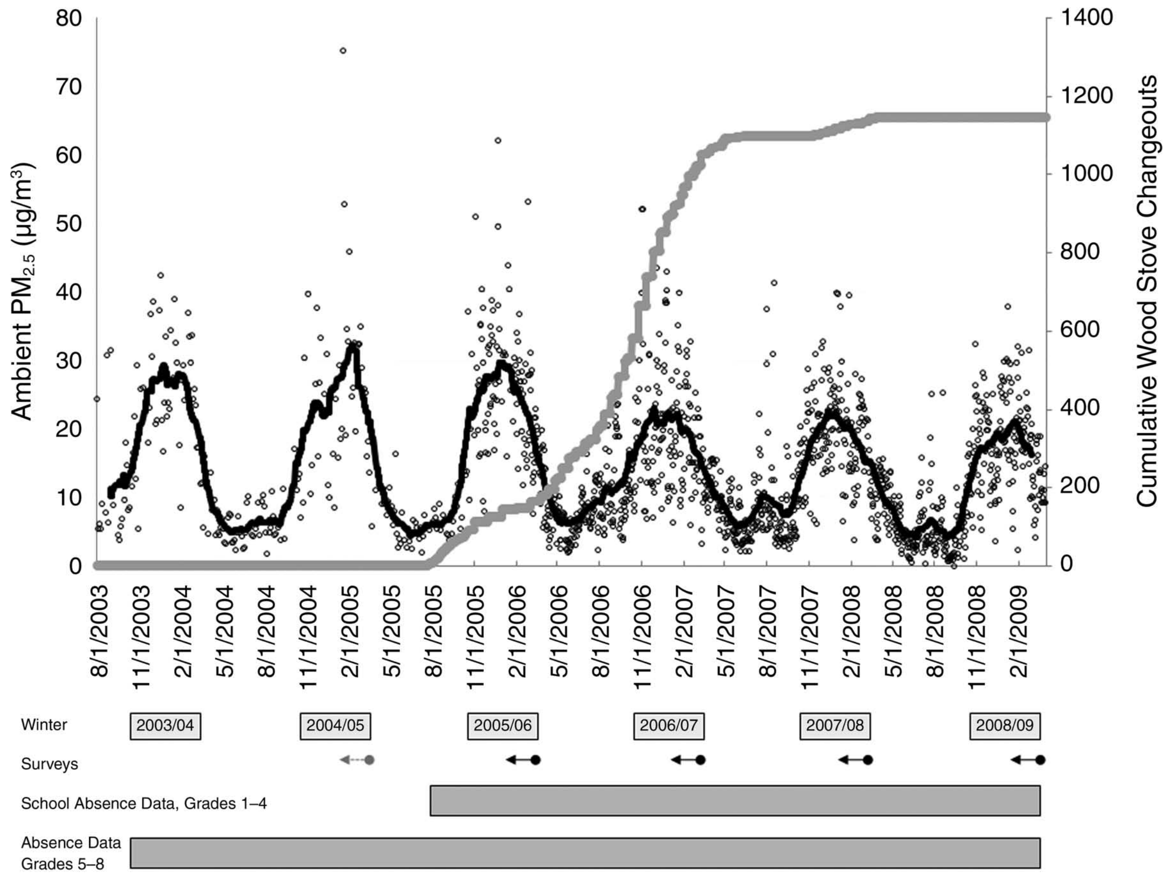
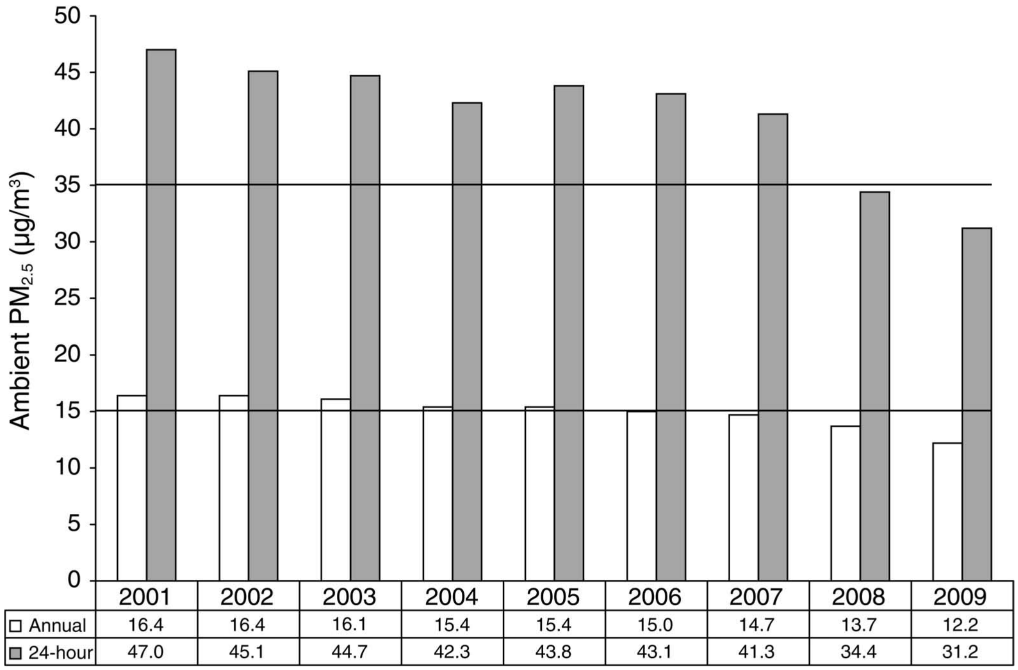


Figure 1. Study overview, ambient $PM_{2.5}$ concentrations, and cumulative changeout of wood stoves from August 2003 through February 2009. Open circles are ambient $PM_{2.5}$ concentrations every third day prior to November 2005 and daily thereafter. Black line = 60 day moving average for $PM_{2.5}$ concentration. Grey line = cumulative wood stove changeouts. Four parent-reported child health surveys (black arrows) were conducted during the study period. One survey (grey arrow) was conducted prior to the study, but the questions were slightly different and not analyzed. Availability of absence data is indicated by grey bars. Analyses of absence data were limited to winter periods, November through February. (2003/04 = baseline 1, 2004/05 = baseline 2, 2005/06 = year 1, 2006/07 = year 2, 2007/08 = year 3, 2008/09 = year 4.)

Table 1. Winter Ambient Mean Concentrations of $PM_{2.5}$, OC, EC, and Polar Organic Compounds[a]

	Baseline 1	Baseline 2	Year 1	Year 2	Year 3	Year 4
Compliance monitoring (*n*)	36	39	95	120	121	120
$PM_{2.5}$	27.31 ± 7.47	27.04 ± 12.49	26.98 ± 9.68	21.1 ± 9.69	20.3 ± 6.68	19.0 ± 6.61
Carbon fractions (*n*)	17	20	19	20	20	21
OC	18.19 ± 3.90	17.91 ± 6.38	16.19 ± 5.75	12.05 ± 4.82	12.63 ± 3.34	11.60 ± 4.81
EC	1.67 ± 0.51	1.36 ± 0.48	1.35 ± 0.45	1.14 ± 0.54	1.51 ± 0.67	3.08 ± 1.79
Polar organic compounds (*n*)	19	18	20	20	19	21
Levoglucosan	2844 ± 865	3036 ± 1462	1897 ± 779	1513 ± 746	1537 ± 511	2059 ± 669
Dehydroabietic acid	353.2 ± 110.9	281.7 ± 97.8	426.2 ± 136.0	234.2 ± 105.0	323.3 ± 65.4	2336.8 ± 1247.2
Abietic acid	14.9 ± 6.9	18.1 ± 12.6	28.3 ± 12.5	16.6 ± 11.1	42.7 ± 16.8	58.4 ± 37.6

[a] Values are mean concentrations for sample size *n*, ± SD. $PM_{2.5}$, OC, and EC concentrations are in μg/m^3; polar organic compound concentrations are in ng/m^3.



	2001	2002	2003	2004	2005	2006	2007	2008	2009
□ Annual	16.4	16.4	16.1	15.4	15.4	15.0	14.7	13.7	12.2
■ 24-hour	47.0	45.1	44.7	42.3	43.8	43.1	41.3	34.4	31.2

Figure 2. Three-year running annual and 24-hour (98th percentile) average concentrations of $PM_{2.5}$ for years 2001–2009. Horizontal black bars at 15 and 35 μg/m^3 represent annual and 24-hour standards, respectively.

In the years before the changeout program, 2001–2004, the 98th percentile three-year running 24-hour average in Libby ranged from 42.3 to 47.0 μg/m^3. This value gradually decreased to 34.4 μg/m^3 for 2008 and 31.2 μg/m^3 for 2009. In October 2009, the U.S. EPA designated Libby, Montana, as an attainment area for the 24-hour $PM_{2.5}$ air quality standard. The three-year running annual mean average has similarly declined in recent years to a value of 12.2 μg/m^3 for 2009 (Figure 2).

We evaluated the quantitative effect of cumulative wood stove changeout on ambient $PM_{2.5}$ concentrations by estimating the daily percentage of changeout program completed, with 1147 representing 100% completion. We fit a linear model for the winter periods adjusted for ambient temperature, ambient $PM_{2.5}$ concentrations lagged one day, month, and an indicator for workday versus weekend or holiday. A 10% increase in cumulative wood stove changeout corresponded to an estimated 0.43-μg/m^3 reduction in $PM_{2.5}$ concentration (95% CI = 0.24 to 0.62) (Figure 3, solid grey line). This regression was heavily dependent on the year 2 winter, during which 31% of stoves were changed out compared with approximately 3% of stoves during the winters of years 1 and 3. The adjusted slope estimate when the regression was limited to the year 2 winter was similar to the estimate for all three winters (i.e., a 0.69-μg/m^3 reduction in $PM_{2.5}$ concentration per 10% increase in cumulative wood stove changeout) (Figure 3, dashed line).

To separate the effect of changeout on daily ambient $PM_{2.5}$ concentration from the effect of the previous day's $PM_{2.5}$ concentration we fit a two-stage model without the one-day lagged $PM_{2.5}$ values (see Figure 3). The fact that the second stage residuals were not strongly autocorrelated (Durbin-Watson statistic 1.89, P = 0.152) was evidence that the two-stage model was successful in accounting for the autocorrelation in the $PM_{2.5}$ series. With this model, the estimated effect of a 10% increase in cumulative changeout — adjusting for ambient temperature, month and workday indicator — was a 0.87-μg/m^3 reduction in $PM_{2.5}$ concentration (95% CI = 0.54–1.20). The corresponding slope estimate when this model was limited to the year 2 winter was −1.13.

The winter ambient $PM_{2.5}$ filter-based concentrations of OC and EC are presented in Table 1. The mean winter period OC concentration during the two baseline years was 18.04 μg/m^3 (SD = 5.32). The winter immediately after the beginning of the wood stove intervention program (year 1) did not have substantially lower concentrations of OC; the mean difference was 1.85 μg/m^3 (95% CI = −0.91 to 4.61). Each of the final three winters of the study period had 30.0% to 35.7% lower OC concentrations compared with the average of the baseline years. The

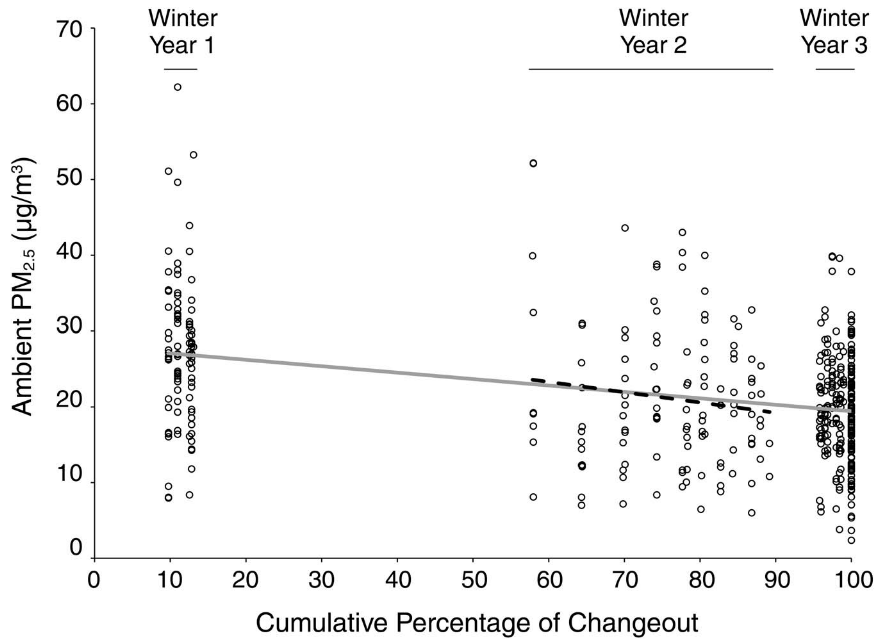


Figure 3. Plot of daily ambient $PM_{2.5}$ concentration against cumulative percentage of changeout during winter periods only. Solid grey line = fitted linear regression with slope of −0.84 per 10% increase in cumulative changeout (years 1–3). Dashed line = fitted linear regression with slope of −1.36 per 10% increase in cumulative changeout (year 2 only). Fitted lines and slopes are not adjusted for ambient temperature or other factors.

OC fraction of $PM_{2.5}$ mass was 0.65 (SD = 0.05) during the baseline winters. Only the final winter was different, with a reduction in the OC fraction of 0.08 (95% CI = 0.05 to 0.11).

The mean winter period EC concentration during the two baseline years was 1.50 μg/m^3 (SD = 0.51). An increase in EC concentration was observed in the final winter (year 4) with a mean increase of 1.57 (95% CI = 1.09 to 2.06). This EC concentration was notably higher than previous winter EC concentrations, but the average was not driven by outliers. The EC fraction of $PM_{2.5}$ mass was 0.05 (SD = 0.01) during the baseline winters. Both of the final two winters experienced increases in EC as a fraction of $PM_{2.5}$ mass with mean increases of 0.02 μg/m^3 (95% CI = 0.01 to 0.03) for year 3 and 0.09 μg/m^3 (95% CI = 0.08 to 0.11) for year 4.

The ambient $PM_{2.5}$ filter-based concentrations of polar organic compounds during the winter periods are presented in Table 1. During the two baseline years the mean levoglucosan concentration was 2937 (SD = 1180) ng/m^3. All four of the subsequent winters had lower concentrations of levoglucosan, with reductions ranging from 29.9% to 48.5% compared with the average of the baseline years. Other chemical markers of wood smoke were also evaluated, including resin acids and methoxyphenols. Ambient concentrations of abietic acid and dehydroabietic acid increased from the baseline years but not consistently across the years. Average concentrations of dehydroabietic acid and abietic acid were notably higher in the final winter (year 4), but there were no detectable outliers driving the averages. Concentrations of methoxyphenols were low with no notable increases or decreases during the study period.

RESIDENTIAL INDOOR AIR

Post-changeout sampling was completed from one to four times in 24 of the 26 homes for which we had pre-changeout sampling. One of the homes was abandoned and could no longer be accessed after the wood stove changeout. The second home had switched to a pellet stove rather than a wood stove and was excluded from subsequent sampling. Of those residences with both pre- and post-changeout sampling there was a fairly wide spatial distribution, ranging from 0.8 to 13.7 kilometers from the central air monitoring site. All but one of the homes was owner occupied, and most of the homes (19/24) were single family fixed residences with the remainder being trailers. The winter period wood use estimated by residents averaged 5.8 cords, ranging from 1.5 to 10 cords. Resident-reported size of homes was about 138.5 m^2 on average, ranging from 55.7 m^2 to 278.7 m^2.

Attempts to match sampling on temperature were reasonably successful, but this strategy was imperfect. When matching post-changeout sampling days to ambient

Table 2. Indoor Residential $PM_{2.5}$ Sampling and Corresponding Ambient $PM_{2.5}$ and Temperature on Sampling Days[a]

Wood Stove Status	Winter Sampled Relative to Pre-Change Sample	Number of Homes	Average Indoor $PM_{2.5}$ ± SD (μg/m³)	Average Ambient $PM_{2.5}$ ± SD (μg/m³)	Average Ambient Temperature ± SD (°C)
Pre-change	—	21	45.0 ± 33.0	25.3 ± 12.4	0.48 ± 5.62
Post-change	Same winter	21	20.4 ± 26.5	18.3 ± 8.12	−3.74 ± 4.23
Post-change[b]	Second winter	12	23.8 ± 15.3	18.5 ± 6.76	−0.22 ± 6.36
Post-change[c]	Second winter	12	21.2 ± 13.3	19.5 ± 3.53	−2.13 ± 5.29
Post-change	Third winter	15	18.0 ± 14.5	17.7 ± 6.85	−2.55 ± 2.71

[a] For homes of non-smokers with at least one post-changeout sample.
[b] Post-changeout sample day chosen to match closely to ambient temperature of pre-changeout sample day.
[c] Post-changeout sample day chosen to match closely to ambient temperature of first winter post-changeout sample day.

temperature of the previous winter's pre-changeout sample day, the temperature difference was 0.91°C (95% CI = −1.12 to 2.94). When matching post-changeout sampling days to ambient temperature of the previous winter's post-changeout sample day the temperature difference was 1.33°C (95% CI = −2.40 to 5.06). When including the final winter of sampling, the average ambient temperature on all post-changeout sampling days was slightly lower than the ambient temperature on the pre-changeout sampling days (−2.89°C [95% CI = −5.40 to −0.39]).

We had attempted to exclude homes with residents who smoked tobacco, but smoking was reported during one or more of the sampling events for three of the 24 homes. Homes with reported tobacco smoking during any of the sampling events were excluded from the analyses. Pre-changeout indoor $PM_{2.5}$ concentrations and the combined average of all post-changeout concentrations for the remaining 21 homes are presented in Table 2 and Figure 4. The average indoor $PM_{2.5}$ concentration for the pre-changeout sampling was 45.0 μg/m³ (SD = 33.0). The average indoor

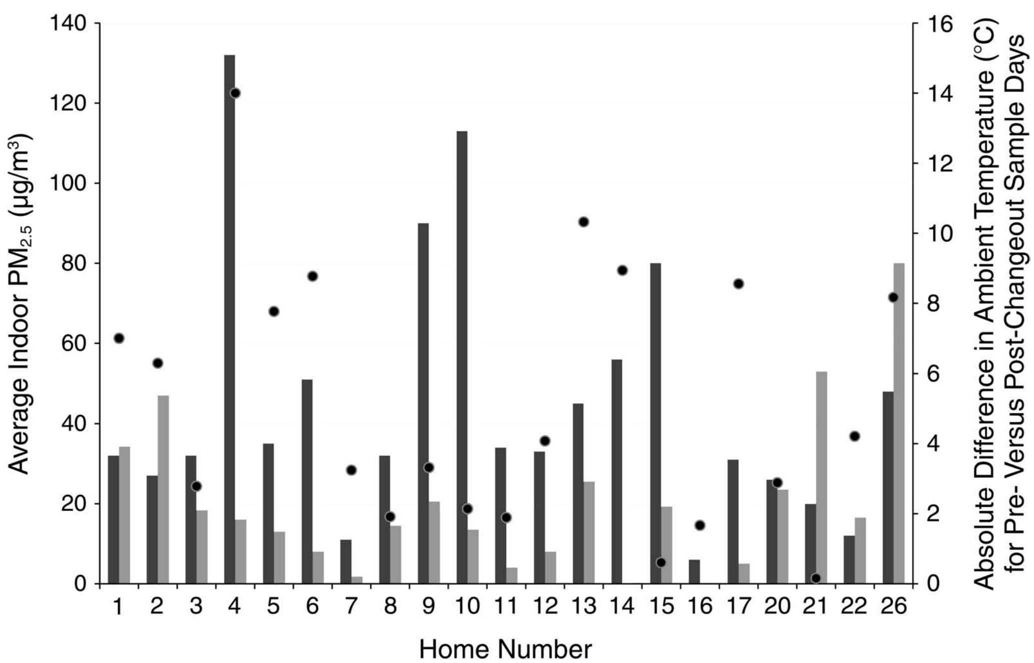


Figure 4. Indoor $PM_{2.5}$ concentrations pre- (dark grey bars) and post- (light grey bars) wood stove changeout for homes of nonsmokers. Numbers along the *x*-axis identify specific homes. Data were available from 21 of the original 26 homes. Light grey bars represent averages across one to four post-changeout sampling days. Dark circles represent the absolute difference in ambient temperature for the pre-changeout sample day compared to the average ambient temperature for all post-changeout sample days in the corresponding home. Temperatures on pre-changeout sampling days were generally higher than on post-changeout sampling days.

$PM_{2.5}$ concentration for the post-changeout sampling with one to four sampling events per homes was 21.3 μg/m^3 (SD = 19.1). Overall reductions after wood stove changeout were observed in 16 of 21 homes (Figure 4). High variability was observed for post-changeout sampling within some homes, but 14 of the 21 homes still had lower $PM_{2.5}$ concentrations for all post-changeout sampling compared with pre-changeout $PM_{2.5}$ concentrations (see Appendix C, Figure C.1). Among the homes that failed to demonstrate an overall reduction in indoor $PM_{2.5}$ concentration, the resident of home #2 reported cooking activities that corresponded to spikes of 280–300 μg/m^3 $PM_{2.5}$ on the continuous sampler. Home #21 also had a reported cooking event corresponding to 191 μg/m^3 $PM_{2.5}$ during one of the post-changeout sampling days. The other post-changeout sampling day for this home yielded an average $PM_{2.5}$ concentration of 10 μg/m^3, a 50% reduction from the pre-changeout measure. During the one post-changeout measure for home #26 there were no indications of particular activities associated with elevated concentrations, but we observed outlier spike values > 5000 μg/m^3. We were unable to identify anything distinctive about homes #1 and #22 that would explain the higher readings post-changeout, but both homes had a mixture of post-changeout sampling days that were higher and lower than the pre-changeout sampling days. For all homes of nonsmokers, the mean reduction from pre-changeout to post-changeout was −20.8 μg/m^3 (95% CI = −34.7 to −6.95), adjusting for ambient $PM_{2.5}$ concentration, ambient temperature, baseline number of cords burned in a winter, presence of indoor pets, and reported cooking, cleaning, or burning candles or incense during the sampling period.

We evaluated the post-changeout measures across three winter seasons to determine if the impact of newly introduced wood stoves on reducing indoor $PM_{2.5}$ concentration was attenuated over multiple years of use. Thus, this analysis was limited to homes that had more than one post-changeout sampling. During the initial winter (year 2) the post-changeout average indoor $PM_{2.5}$ concentration in 18 homes was 12.8 μg/m^3 (SD = 14.8). The average indoor $PM_{2.5}$ concentration during the second winter (year 3) increased to 27.4 μg/m^3 (SD = 22.0), an increase of 17.2 μg/m^3 (95% CI = 8.9 to 25.5), adjusting for ambient $PM_{2.5}$ concentration, ambient temperature, average wood usage (number of cords) in pre-changeout years, presence of indoor pets, and various reported activities during the sampling event. In the final winter (year 4), however, the post-changeout $PM_{2.5}$ concentrations were similar to those of the initial post-changeout sampling, 18.0 μg/m^3 (SD = 14.5) or an adjusted increase of only 4.34 μg/m^3 (95% CI = −6.31 to 15.0).

SCHOOL INDOOR AIR

Winter sampling in the two schools was conducted during all four study years. Sampling in the late spring was conducted during the academic years that corresponded to study years 1 through 3. Sampling in the early fall in the two schools was conducted during the academic years that corresponded to study years 2 through 4. Descriptive results for all winter sampling periods are presented in Appendix Tables B.2 and B.3. Across years 1–3, the concentrations of $PM_{2.5}$ and OC fraction declined at the grade school site, however this pattern did not continue in the year 4 winter sampling period. Overall, none of the polar organic compounds or $PM_{2.5}$ measures at either school showed a pattern that would be consistent with the timing of the wood stove intervention program.

Indoor concentrations of $PM_{2.5}$ by season are presented in Appendix Figures C.2 and C.3 for the grade school and the middle school, respectively. In addition to there being no observable pattern by year, there was no observable pattern by season; some of the highest concentrations occurred in the nonwinter periods. Levoglucosan concentrations inside the schools similarly failed to parallel the reduction in ambient levoglucosan over the study period, although levoglucosan concentrations were highest in the winter periods for both schools (Appendix Figures C.4 and C.5).

SURVEY

A total of 1713 questionnaires were returned during the study period. For each of the four winters, 381 to 459 questionnaires were returned. For grades one through eight, returned questionnaires corresponded to a 44% to 51% response rate based on enrollment in March of each year (Table 3). Efforts to obtain parent questionnaires as students progressed through the high school were less successful with response rates at 43% in March 2007 (year 2) but falling to 21% of all attending high school students in the final year (year 4). Table 4 describes the characteristics of children and their homes among survey respondents. Wood stove use was reported in the homes of 41.6% to 43.3% of students over the study period. Smoking was reported in 30.8% to 36.7% of homes.

Table 3. Survey Responses for First Through Eighth Grades

	Year 1	Year 2	Year 3	Year 4
Number of respondents	381	341	322	334
Enrollment	748	720	732	693
Response rate (%)	51.0	47.4	44.0	48.2

Table 4. Percentage of Respondents According to Selected Characteristics for each Survey Year

	Year 1 (n = 381)	Year 2 (n = 459)	Year 3 (n = 430)	Year 4 (n = 430)
Sex				
Boy	53.8	49.5	47.8	52.0
Girl	46.2	51.5	52.2	48.0
Grade				
1–2	20.2	18.5	17.9	21.6
3–4	22.6	19.6	19.1	23.0
5–6	24.4	16.6	22.6	19.8
7–8	32.8	19.2	15.4	11.9
9–10	—	26.1	17.9	13.0
11–12	—	—	7.2	10.7
Type of heating				
Wood	43.3	42.2	43.3	41.6
Propane	29.9	24.2	27.0	24.9
Gas	3.7	3.9	4.4	3.5
Oil	20.5	21.4	21.2	16.1
Electricity	38.9	43.4	43.0	39.5
Tobacco use in home	36.7	32.9	30.8	33.6
Also responded in Year 1	—	43.6	38.1	28.4

Table 5 presents the percentage of questionnaires with reported respiratory symptoms and respiratory infections for each year. Within years there was no difference for reporting of wheeze among those children living in homes with wood stoves versus those living in homes without wood stoves (Table 6). We were able to record the change-out date for some of the homes reporting wood stove use. For the first three years of the study there was no difference in the reporting of wheeze between those living in homes in which the stove had been changed by the time of the respective survey compared with those living in homes in which the stove had not yet been changed (Table 7). Note that a changeout date was not reported for all homes that had indicated having a wood stove. By the final year of the study, it was assumed that all respondents living in homes with wood stoves had participated in the change-out program.

Summaries of the community-level variables during the study period are presented in Table 8. The largest percentage of wood stove changeouts occurred between the year 1 and year 2 winters. Ambient temperature did not vary dramatically over the four winters. The mean temperature varied by only 0.88°C. As anticipated, wind speed during the winter periods was extremely low, with average wind gusts never exceeding 7.4 km/hr. The surrogate measure of influenza activity in the community suggested dramatic

Table 5. Percentage of Questionnaires with Reported Respiratory Symptoms and Infections[a]

Outcome	Year 1 (n = 365)	Year 2 (n = 451)	Year 3 (n = 417)	Year 4 (n = 416)
Respiratory symptoms				
Wheeze	11.78	6.87	9.11	6.00
Morning tightness of chest	7.07	4.92	6.90	5.18
Attack of shortness of breath	5.33	3.51	4.49	2.81
Attack of shortness of breath after exercise	10.66	6.90	9.86	9.50
Wake up at night by an attack of shortness of breath	2.95	1.54	4.01	1.41
Wake up at night by an attack of coughing	30.77	23.68	29.31	20.61
Other symptoms and infections				
Itchy and/or watery eyes	33.69	25.46	24.59	22.30
Sore throat	67.29	60.89	57.14	52.24
Cold	77.98	71.77	72.77	71.19
Bronchitis	5.48	1.79	2.40	1.91
Influenza	45.41	42.09	26.92	17.18
Throat infection	18.11	13.72	10.65	7.45
Middle-ear infection	8.99	4.91	5.76	3.82

[a] Responses were collected during March of the given year and covered the previous two months. The sample size was based on those responding to the wheeze question. The sample size for other symptoms and infections may vary slightly because of missing responses.

Table 6. Total Respondents (and Percentage of Questionnaires with Reported Wheeze) by Wood Stove Use and Year[a]

	Year 1	Year 2	Year 3	Year 4
Wood stove use in home, *n* (%)	156 (12.82)	189 (7.94)	182 (8.79)	173 (4.05)
No wood stove use in home, *n* (%)	209 (11.00)	262 (6.11)	235 (9.36)	243 (7.41)
P value for difference[b]	0.60	0.45	0.84	0.16

[a] Responses were collected during March of the given year and covered the previous two months.
[b] Chi-square test.

Table 7. Respondents (and Percentage of Questionnaires with Reported Wheeze) by Changeout Status and Year Among Those Living in Homes with Wood Stoves[a]

	Year 1	Year 2	Year 3
Changed wood stove prior to survey, *n* (%)	19 (10.53)	100 (9.00)	156 (9.62)
Not changed wood stove prior to survey, *n* (%)	92 (10.87)	42 (9.52)	8 (12.50)
P value for difference[b]	0.97	0.92	0.79

[a] Responses were collected during March of the given year and covered the previous two months. By the final survey year (year 4) all respondents from homes with wood stoves had completed a changeout.
[b] Chi-square test.

Table 8. Summary of Factors Captured over Winter Study Periods

	Year 1	Year 2	Year 3	Year 4
Wood stoves changed by start of winter (%)[a]	9.76	57.89	95.82	100
Wood stoves changed by end of winter (%)[a]	13.08	89.19	98.95	100
Average $PM_{2.5}$ (μg/m^3)	27.0	21.1	20.3	19.0
Average temperature (°C)	−2.10	−2.56	−2.53	−2.98
Average wind speed (km/hr)	0.27	0.35	0.50	0.40
Average wind gust (km/hr)	5.8	7.1	7.4	6.6
Average relative humidity (%)	81.8	87.0	82.1	89.0
Total precipitation (cm)	21.3	24.6	22.9	19.8
Influenza rate[b]	3.17	1.54	3.69	1.92

[a] Percentage of total wood stoves replaced, reconditioned, or removed by the start of winter (November 1) and end of winter (last day of February) of the corresponding year.
[b] Rate per 1000 people based on hospital discharge records with indication of influenza. Secondary analysis of absence data for the middle school included two additional pre-changeout years during which the estimated influenza rates were 1.13 (baseline 1) and 5.78 (baseline 2) per 1000 people.

Table 9. Effects Estimates Across Four Winter Periods for Children's Health Outcomes Associated with a 5-µg/m^3 Decrease in Ambient $PM_{2.5}$ Concentration and with Wood Stove Usage in the Home

Outcome	Ambient $PM_{2.5}$ Reduction OR (95% CI)[a]	Wood Stove Use in Home OR (95% CI)[a]
Respiratory symptoms		
Wheeze	0.7334 (0.5545–0.9701)	1.0420 (0.7193–1.5095)
Morning tightness of chest	0.9347 (0.6601–1.3237)	0.6270 (0.4074–0.9649)
Attack of shortness of breath	0.7961 (0.5425–1.1681)	0.8400 (0.5238–1.3471)
Shortness of breath after exercise	1.0309 (0.7725–1.3758)	0.9767 (0.6835–1.3957)
Wake up at night by an attack of shortness of breath	1.0268 (0.6149–1.7147)	0.9612 (0.4971–1.8587)
Wake up at night by an attack of coughing	0.9037 (0.7545–1.0824)	1.0476 (0.8298–1.3225)
Other symptoms and infections		
Itchy/watery eyes	0.6681 (0.5536–0.8064)	0.9639 (0.7641–1.2159)
Sore throat	0.6844 (0.5702–0.8215)	0.9580 (0.7805–1.1759)
Cold	0.7462 (0.6028–0.9236)	0.9335 (0.7383–1.1803)
Bronchitis	0.4542 (0.2723–0.7577)	1.6019 (0.8473–3.0283)
Influenza	0.4766 (0.3954–0.5746)	0.9766 (0.7814–1.2206)
Throat infection	0.5490 (0.4244–0.7103)	0.8301 (0.6089–1.1317)
Middle-ear infection	0.7092 (0.4896–1.0273)	1.2984 (0.8131–2.0735)

[a] The ORs were adjusted for student age group and community influenza rate.

differences across the four years, with rates of influenza per 1000 people ranging from 1.54 in the year 2 winter to 3.69 in the year 3 winter.

Table 9 presents effect estimates for respiratory symptoms and other health outcomes associated with a 5-µg/m^3 reduction in $PM_{2.5}$ concentration, adjusting for age group of child, wood stove use in the home, and the surrogate measure of community influenza activity.

Some health outcomes were significantly reduced with reduced ambient $PM_{2.5}$ concentrations. There was a 26.7% (95% CI = 3.0 to 44.6) reduced odds of reported wheeze for a 5-µg/m^3 decrease in average winter $PM_{2.5}$ concentration. In secondary analyses, other respiratory symptoms were not strongly associated with $PM_{2.5}$ concentration while symptoms most likely to be caused by irritant effects of wood smoke were reduced, notably risk of itchy/watery eyes (33.2% [95% CI = 19.4 to 44.6]) and sore throat (31.6% [95% CI = 17.9 to 43.0]). Lower ambient $PM_{2.5}$ concentrations were also associated with reduced odds for reported cold infection (25.4% [95% CI = 7.6 to 39.7]), bronchitis (54.6% [95% CI = 24.2 to 72.8]), influenza (52.3% [95% CI = 42.5 to 60.5]), and throat infection (45.1% [95% CI = 29.0 to 57.6]) among children.

Parents were asked to report the frequency of some additional nonrespiratory symptoms that were assumed a priori to not be associated with ambient $PM_{2.5}$ concentrations (see Appendix Table B.4). Some of these symptoms (e.g., nausea and vomiting) were associated with changes in ambient $PM_{2.5}$ concentrations in the main analysis. Given that these conditions could be associated with other illnesses such as influenza, which were also found to be associated with ambient $PM_{2.5}$ concentration, we reevaluated the reports of these symptoms in a sensitivity analysis using individually-reported influenza rather than the community-level indicator of influenza incidence in the model. After this adjustment, reporting of nausea and vomiting was no longer associated with changes in ambient $PM_{2.5}$ concentration (Appendix Table B.4). Associations between ambient $PM_{2.5}$ concentrations and reporting of wheeze and infectious health conditions remained unchanged when including individually-reported influenza rather than the community-level indicator of influenza incidence in the model. The risk reduction estimate for reporting of wheeze per 5-µg/m^3 decrease in average winter $PM_{2.5}$ concentration with the individual-level influenza indicator in the model was 0.74 (95% CI = 0.56 to 0.98) versus the estimate with the community-level influenza indicator in the model, 0.73 (95% CI = 0.55 to 0.97) (See Appendix Table B.4).

Wood stove use in the home was also included in the analysis model. The use of a wood stove, adjusting for ambient $PM_{2.5}$ concentration, age group of child, and community influenza, was associated with increased reporting of only one of the symptoms or health conditions. Morning tightness in chest had lower odds with respect to wood stove use (Table 9). We also evaluated the potential for wood stove use in the home to modify the relationship between ambient $PM_{2.5}$ concentrations and reporting of symptoms and conditions. When stratifying the primary analysis by wood stove use in the home, there were no discernible differences in the effect estimates for ambient $PM_{2.5}$ concentrations and reporting of symptoms and conditions (data not shown).

The data from year 1 were highly influential because that was the only year with elevated ambient winter $PM_{2.5}$ concentrations, whereas $PM_{2.5}$ concentrations were consistently low in the subsequent three winters. In consideration of this influential first winter period we conducted sensitivity analyses on these findings, restricting the evaluation to those children for whom survey data had been received for the first winter of the study and for one or more subsequent winters. The point estimates for most outcomes were similar to those in the primary analysis, but the CIs were much wider. In the restricted analysis, frequency of reported wheeze was 30% lower (95% CI = −7 to 54) for a 5-μg/m^3 decrease in average winter $PM_{2.5}$ concentration, while in all subjects it was 26.7% (95% CI = 3.0 to 44.6); reductions in $PM_{2.5}$ concentrations also significantly reduced reporting of itchy/watery eyes, sore throat, bronchitis, influenza, and throat infection (Appendix Table B.5).

Parent reporting of childhood symptoms and health conditions was also evaluated with respect to ambient measures of OC, EC, and markers of wood smoke. Associations between these markers and reported symptoms and conditions among children are presented in Appendix Table B.6. Risk estimates for changes in OC, but not EC concentrations, were consistent with what was observed for ambient $PM_{2.5}$ concentrations. Changes in polar organic compound concentrations did not yield risk estimates that were consistent with what was observed for ambient $PM_{2.5}$ or OC concentrations.

SCHOOL ABSENCES

Absence data during the winter periods of the four study years, corresponding to academic years 2005–2006 through 2008–2009, were obtained for grades 1 through 8. During these four winters, there were 5034 incident illness-related absences (i.e., not preceded by an absence day). The percentages of illness-related absences and person-days at risk for each school grade are presented in Table 10. When considering only the four study years, there is no clear pattern in the crude illness-related absence frequency across years. There were limited recordings of illnesses with sufficient information to code the illness as respiratory-related, and these may not have been recorded consistently between the two schools. During the winters of years 1 through 4 there was an average of only 155 absences per year that could be coded as respiratory-related. Over 80% of these were recorded at the grade school.

Illness-related absence was the outcome of interest for the primary analysis. The effect estimate associated with a 5-μg/m^3 reduction in average ambient winter $PM_{2.5}$ concentration, adjusted for age group of child, day-of-week, month, community influenza, and wind gust, was an 8.9% reduction in illness-related absence (95% CI = 4.0 to 13.6) (Table 11). Daily changes in $PM_{2.5}$ concentration were not associated with illness-related absences after adjusting for age group of child, year, day of week, month, wind gust, and one-day lagged $PM_{2.5}$ concentration. Community influenza rate could not be included in this model because of the presence of the year indicator. Surprisingly, reduced ambient $PM_{2.5}$ concentrations were associated with increased nonillness-related absences (Tables 11 and 12).

Table 10. Percentage of Incident Illness-Related School Absences and Person-Days at Risk by Year and Grade During Winter Months (November–February)[a]

Grade	Baseline 1	Baseline 2	Year 1	Year 2	Year 3	Year 4
First	—	—	1.72 (5683)	2.81 (5756)	1.97 (6143)	2.69 (4937)
Second	—	—	1.84 (3810)	2.41 (5553)	1.97 (5320)	2.24 (6153)
Third	—	—	2.06 (5774)	2.81 (4409)	1.55 (5947)	2.38 (4825)
Fourth	—	—	1.54 (4144)	3.29 (6324)	3.41 (5226)	2.29 (6195)
Fifth	1.94 (5766)	3.26 (7863)	3.25 (5717)	2.89 (4459)	2.46 (6940)	2.69 (4831)
Sixth	2.91 (9079)	2.20 (6011)	3.53 (7563)	3.32 (6512)	2.69 (4866)	2.57 (5758)
Seventh	2.50 (6870)	3.72 (9364)	3.23 (6246)	3.51 (7438)	3.03 (6365)	2.32 (4661)
Eighth	3.08 (8678)	2.89 (7780)	3.70 (8704)	3.14 (7070)	2.62 (8052)	2.56 (5542)

[a] Values are percent absent (person-days at risk).

Table 11. Effects Estimates Across Four Winter Periods for School Absence in Grades 1–8 Associated with a 5-µg/m^3 Decrease in Ambient $PM_{2.5}$ Concentration

Outcome	Average Winter $PM_{2.5}$ OR (95% CI)[a]	Daily $PM_{2.5}$ OR (95% CI)[b]
All absences	1.0397 (1.0031–1.0777)	1.0115 (0.9962–1.0269)
Illness absences	0.9110 (0.8642–0.9604)	0.9930 (0.9717–1.0148)
Non-illness absences	1.1027 (1.0523–1.1555)	1.0214 (1.0012–1.0420)

[a] The ORs were adjusted for age, day-of-week, month, community influenza rate, and wind gust.

[b] The ORs were adjusted for age, year indicator, day-of-week, month, wind gust, and one-day lag ambient $PM_{2.5}$ concentration.

Table 12. Effects Estimates Across Four or Six Winter Periods for Illness-Related School Absence Associated with a 5-µg/m^3 Decrease in Ambient $PM_{2.5}$ Concentration or with a 10 Percent Increase in Cumulative Wood Stove Changeout

Grades	Average Winter $PM_{2.5}$ OR (95% CI)[a]	Cumulative Stove Changeout OR (95% CI)[a]
Four winters: year 1–year 4		
Grades 1–8	0.9110 (0.8642–0.9604)	0.9871 (0.9792–0.9960)
Grades 1–4	1.1115 (1.0138–1.2187)	1.0202 (1.0050–1.0356)
Grades 5–8	0.8198 (0.7692–0.8737)	0.9704 (0.9607–0.9812)
Six winters: baseline 1–year 4		
Grades 5–8	0.9425 (0.8985–0.9886)	0.9920 (0.9851–1.0000)

[a] The ORs were adjusted for student age group, day-of-week, month, community influenza rate, and wind gust. Average winter $PM_{2.5}$ and cumulative stove changeout were separate models.

The effect estimate for cumulative wood stove changeout, modeled separately, was consistent with these findings. Stratification by grades suggested that the protective effect of reduced $PM_{2.5}$ concentration on illness-related absences was limited to older students, and students in grades one through four actually had increased illness absences with reduced average ambient winter $PM_{2.5}$ concentration or increasing cumulative wood stove changeout (Table 12).

The secondary analysis included six winters of absence data that were available for the middle school. Illness absence percentages in the three pre-changeout winters were higher than illness absence percentages in the three post-changeout winters for all four grades. When considering the first three winters compared with the last three winters, the frequency of illness-related absences was 3.07% versus 2.84%. The modeled results for middle school illness absences using six winters of data was similar to the results for middle school illness absences using only four winters of data. A 5-µg/m^3 reduction in average ambient winter $PM_{2.5}$ concentration corresponded to a 5.6% reduction in illness-related absence (95% CI = 1.1 to 10.2) (Table 12).

DISCUSSION

Some communities have been required to engage in pollution-reduction programs and policy changes to attain the NAAQS for $PM_{2.5}$ and other criteria air pollutants. It is often difficult to track the effect of any one program or policy on a given air pollutant because of complex air sheds, multiple sources, and other factors that change over time. The current study indicates that a community-wide wood stove changeout program can be a successful approach for achieving substantial and sustained reductions of ambient $PM_{2.5}$ concentrations. Tracking the pace of the changeout program in this small community suggested that a 10% increase in cumulative wood stove changeout corresponded to a 0.43-µg/m^3 reduction in ambient $PM_{2.5}$ concentration. This slope estimate was limited to the winter periods, and the majority of changeouts (62%) occurred in the nonwinter periods. Over 31% of changeouts did occur during one winter period, however, yielding a similar slope estimate of a 0.69-µg/m^3 reduction in $PM_{2.5}$ concentration per 10% increase in cumulative wood stove changeout.

From a regulatory perspective the overall wood stove changeout program, with over 95% completion, achieved a reduction in winter ambient $PM_{2.5}$ concentration of approximately 30%, which brought the community's ambient air within the $PM_{2.5}$ NAAQS standards of the U.S. EPA. It is difficult to discern from the available data if this regulatory achievement would have been realized with something less than the full-scale community-wide changeout program. The most dramatic drop in ambient $PM_{2.5}$ concentration occurred from the year 1 winter to the year 2 winter. The first phase of the program was targeted at lower income households, and the homes were geographically dispersed. It is possible that the changeout homes in the initial phase of the program had stoves that were older and higher emitting than changeout homes in the second phase of the program. Nevertheless, the majority of changeouts (≈58%) occurred before the year 2 winter. Of these, over 60% happened during phase two of the changeout program, the phase that did not have a targeted approach based on household income. Over 89% of the changeout program had been completed by the end of the year 2 winter, so it is not surprising that further dramatic reductions in ambient $PM_{2.5}$ concentrations were not observed in subsequent winters.

For several reasons this community was ideally suited for a study of the effect of a source-targeted intervention on ambient $PM_{2.5}$ concentrations. First, the community experienced dramatic and predictable seasonal fluctuations in $PM_{2.5}$ concentrations. Second, winter-time elevations in PM were derived almost exclusively from one source — wood stoves. Third, the community managed a well-funded intervention that achieved near-complete compliance. All of these factors contributed to the community's success in achieving substantial reductions in ambient winter $PM_{2.5}$ concentrations. There are few similar wood stove programs with documented community-wide reductions in PM, and none that addressed the more recent $PM_{2.5}$ NAAQS or associated health outcomes.

Our survey results suggested that there were beneficial effects on children's health resulting from the wood stove changeout program, and that such benefits were not limited to children with asthma. Our primary outcome, reduction in parent-reporting of childhood wheeze, was associated with reductions in ambient $PM_{2.5}$ concentration, with an OR of 0.73 (95% CI = 0.55 to 0.97) per 5-μg/m^3 reduction in winter ambient $PM_{2.5}$ concentration. Other respiratory symptoms that are commonly associated with childhood asthma were not associated with reductions in $PM_{2.5}$ concentrations, and there was no consistent pattern with respect to directionality. Some previous cross-sectional studies of school children did not observe associations between survey-based reporting of asthma-related symptoms and ambient PM (Dockery et al. 1989; Braun-Fahrlander et al. 1997). In communities with substantial contributions to ambient air from residential wood combustion, increased ambient PM concentrations have been associated with increased asthma symptoms in cohort studies of asthma (Yu et al. 2000; Slaughter et al. 2003) and with hospitalizations or emergency department visits for asthma (Lipsett et al. 1997; Norris et al. 1999; Sheppard et al. 1999). Among other factors, these latter studies are not directly comparable with our study because they were conducted as daily time-series studies rather than by season.

The frequency of most asthma-related symptoms was low in this study (6%–12%), but reporting of the child being woken at night due to cough was observed among 21%–31% of children across the years. Although this question has been used to assess symptoms among children with asthma, coughing at night can occur with children who have respiratory infections but not asthma. The reports for two other symptoms that could occur among children without asthma, itchy/watery eyes and sore throat, were similarly high and strongly associated with $PM_{2.5}$ concentration. These symptoms may indicate a response to the irritant effect of smoke exposure. Eye discomfort has been observed among subjects in several studies of adults living in homes with high levels of biomass smoke exposure from cook stoves (Diaz et al. 2007; Romieu et al. 2009). It is reasonable to speculate that the influence of wood smoke exposure on eye symptoms is relevant to the more general child population and not limited to susceptible individuals. A recent study of symptoms reported among Children's Health Study participants exposed to wildfire smoke found positive associations between smoke exposure and reporting of symptoms. Although the association with respiratory symptoms was modified by an indicator of compromised breathing (i.e., small airway size), the association with eye symptoms was not limited to susceptible children (Mirabelli et al. 2009).

We also found that parental reporting of infectious health conditions was associated with ambient $PM_{2.5}$ concentration. Reductions in ambient wood smoke PM concentrations were associated with reduced reporting of bronchitis and influenza among children. Our risk estimate was based on a small number of bronchitis reports (n = 20 during the first winter), but these findings were consistent with a recent study in British Columbia that found residential wood smoke exposure to be associated with an increased risk of outpatient and inpatient visits for infant bronchiolitis (Karr et al. 2009). Our observed associations between $PM_{2.5}$ concentration and reports of cold, sore throat, and throat infection suggested that changes in reported health outcomes were not limited to the lower respiratory tract. Upper respiratory conditions and associated symptoms have also been observed among children exposed to smoke from nearby wildfires (Künzli et al. 2006). Previous studies not specific to biomass exposure have also reported associations between PM and otitis media (Brauer et al. 2006; Brauer et al. 2007). We did observe a suggestive association between $PM_{2.5}$ concentration and ear infection, but our study population, children 5 years and older, may have been less susceptible than younger children to ear infection, as indicated by the low frequency of reporting for this condition in our study.

We observed that community-level reductions in woodsmoke–derived $PM_{2.5}$ concentrations were associated with fewer illness-related school absences, but these observations were not consistent across all ages. School absences can be an indicator of disease severity and outcomes that have an indirect economic and social impact. Absenteeism has been used in previous studies that assessed the effect of environmental pollution, but not all studies have found a strong association with PM concentrations (Ransom and Pope 1992; Chen et al. 2000; Gilliland et al. 2001; Park et al. 2002). When stratifying by grades, we observed an association between ambient $PM_{2.5}$ and illness-related absences for older children. Among younger children, however, the association between reductions in $PM_{2.5}$ concentrations and illness absences was in the opposite direction. This contradiction, coupled with the overall increased odds for nonillness absences associated with decreasing $PM_{2.5}$ concentrations, suggests using caution when interpreting the school absence data. The reason for the distinction by age group was not clear, but it is possible that the middle school personnel more accurately recorded cause-of-absence. Evidence supporting this possibility is found in the reporting of illness-related absences; consistently more illness-related absences were reported among 5th through 8th graders (middle school) compared with those of the grade school students (Table 10). Perhaps more illuminating is the two-fold higher incidence of illness-related absences among 4th graders in year 1, when that grade was housed at the grade school, compared with the subsequent three years after the 4th grade had been relocated to the middle school. These raw data suggest that the middle school absence data may be more credible, but this does little to explain the increased risk estimates for illness-related absences among younger children or for increased nonillness-related absences among all children as ambient $PM_{2.5}$ concentration decreased.

Analyses for survey-based health outcomes and school absences were evaluated with respect to ambient $PM_{2.5}$ concentrations. People spend the majority of their time indoors, as much as 95% in some areas (Fishbein and Henry 1991; Jenkins et al. 1992). Although children tend to spend more time outdoors than adults, this study was limited to the cold winter periods during which both children and adults would be spending most of their time in indoor environments. We assessed indoor air in some homes during the study period. Sampling of indoor residential environments suggested that the reduction in indoor $PM_{2.5}$ concentration after wood stove changeouts was greater than what was observed in the ambient environment (Ward et al. 2008). Indoor $PM_{2.5}$ concentrations remained reduced in follow-up years compared with prechangeout concentrations, suggesting that improved indoor air quality is sustained over time after a wood stove replacement program. These observations are based on a small number of sampled homes and a limited follow-up time. These findings of reduced indoor $PM_{2.5}$ concentrations after wood stove replacement are consistent with our previous work in other communities affected by wood smoke (Ward et al. 2010a; Noonan et al. 2011), but similar reductions were not observed in another community (Allen et al. 2009). It is not clear from these data how ambient $PM_{2.5}$ concentration and residential air infiltration may have influenced our indoor measurements. Air infiltration can affect indoor environments either locally due to re-entry of emitted $PM_{2.5}$ from the home, or from other ambient sources of $PM_{2.5}$ (Allen et al. 2003; Barn et al. 2008). Previous analyses that evaluated indoor changes in $PM_{2.5}$ concentrations, stratified on whether homes experienced increases or decreases in ambient $PM_{2.5}$ concentrations on pre- versus post-changeout sampling days, did not suggest a strong influence from ambient PM concentrations (Ward et al. 2008). Our present study adjusted estimates of reductions in indoor $PM_{2.5}$ concentrations for central site ambient $PM_{2.5}$ concentrations. Thus, the large decreases in indoor $PM_{2.5}$ concentrations that occurred after stove changeouts were likely the result of the changeouts. Either less wood smoke escaped inside the homes from the new stoves than from the old stoves or greater burn efficiencies and the associated lower chimney emissions decreased the amount of wood smoke that could enter the houses from outside.

The results of our indoor air sampling were not directly linked with the health outcomes measured in this study, but our survey data do offer some insight into the effect of wood stoves in the homes of participating families. The prevalence of wheeze did not differ among children living in homes with wood stoves versus those living in homes without wood stoves. More importantly, there were no differences when evaluating percentages of children with wheeze among those whose wood stove had already been changed versus those whose wood stove had not yet been changed. The lack of effect associated with wood stove use in the home or wood stove improvement could be due to a number of things. First, individuals and families who were susceptible to wood smoke effects may have self-selected out of wood stove use before the study. Second, central ambient levels of wood smoke can affect indoor exposures via infiltration, allowing for indoor wood smoke exposures among homes without wood stoves. We were unable to directly evaluate indoor- versus outdoor-generated wood smoke, but as noted earlier, other studies have described substantial residential infiltration of ambient smoke (Allen et al. 2003; Barn et al. 2008). Third, it is possible that the presence of a wood stove in the homes of responding parents led to a reporting bias. To evaluate the potential for biased reporting we collected information on symptoms that were not anticipated to be associated with PM exposure. We found that nonrespiratory symptoms such as nausea and vomiting were not associated with the presence of a wood stove in the home, suggesting that presence of a wood stove did not result in over-reporting of general health symptoms. Also, after adjusting for individual-level rather than community-level reporting of influenza, these nonrespiratory symptoms were not associated with ambient $PM_{2.5}$ concentration.

Several limitations affect the generalizability of our health outcomes findings. As indicated earlier, our exposure assessment with respect to health outcomes was limited to ambient measures. A great deal of variability in personal-level exposure would be anticipated, owing to different home environments and activity patterns, which would result in uncertain but likely nondifferential effects on our risk estimates. As with any survey-based study, our findings were highly dependent upon the population that chose to participate in the study, as well as on the accuracy with which the participants understood and responded to the symptom and disease questions. Based on the similar risk estimates observed when our analysis was restricted to those who had responded in the most influential year and in at least one of the subsequent years, we are less concerned about a differential effect in nonparticipation over time. Nevertheless, the overall response rates were low across the years (less than 50%), so it is likely that our findings may not be representative of the entire childhood population in the community. Although our school absence data suggested reduced illness absences among older children as ambient $PM_{2.5}$ concentration decreased, the school absence results were not consistent. The findings for younger children were in the opposite direction, and our ability to evaluate cause-specific illness absences was limited, with very few illnesses recorded specifically as asthma or other respiratory illnesses. School nurses did provide the student identifiers of children known to have asthma, but this variable was not influential in the models owing to the limited numbers of such children.

Another limitation of our study was that we were unable to capture outcome data for earlier pre-changeout years. In most analyses we had only one year of data to estimate the pre-changeout effect. A notable exception was our ability to analyze archived school absence data for a subset of students, yielding results that were consistent with and supportive of our primary analysis. Another potential concern with the time frame of the study was that we did not capture symptom reporting before the introduction, and community awareness, of the wood stove changeout program. This may suggest a potential for self-selection bias in the first survey winter among participating parents who had a more heightened awareness of the wood stove changeout program and the associated health concerns. Two observations support the argument against this concern. First, we conducted a restricted analysis with only the survey respondents who had participated in the first winter and in one or more subsequent winters. We did not observe differences in reported wheeze or risk estimates for ambient $PM_{2.5}$ concentrations and reported health conditions between this restricted analysis and the full analysis. Second, prior to this study we had conducted a similar parent survey in March 2005. The questions from this survey were slightly different from the questions of the current study, so direct comparisons were not possible. In the March 2005 survey the frequency of parent-reported “attacks of wheezing” in the previous 12 months and in the previous four weeks was 14.1% and 7.3%, respectively (Noonan and Ward 2007). By comparison, the wheeze prevalence for a two month recall period reported in March 2006 during the first winter of the present study (year 1) was 11.8%. Among respondents in the March 2005 survey, 32.5% used wood stoves as their primary heating source; among respondents in the March 2006 survey, 32.3% used wood stoves as their primary heating source. This suggests that the implementation of the wood stove changeout program in the summer of 2006 did not influence overall participation in the survey study among wood stove users. Similar to the present study, the previous survey suggested no association between the use of a wood stove in the home and reporting of respiratory symptoms among children (Noonan and Ward 2007).

Despite the success of the changeout program in meeting the $PM_{2.5}$ NAAQS, it could be argued that the overall reduction in ambient $PM_{2.5}$ concentration was lower than anticipated given the magnitude and comprehensiveness of the intervention program. Residential wood combustion remains the predominant source of $PM_{2.5}$ in the community. A CMB source apportionment study was conducted in Libby, Montana, during the baseline 1 winter, before the community-wide wood stove changeout program, to identify the sources of $PM_{2.5}$ within the valley. Results from this study showed that residential wood stoves were the major source, contributing approximately 80% of the ambient $PM_{2.5}$ concentration throughout the winter months. During the year 3 winter, a follow-up CMB source apportionment study was conducted to evaluate the effectiveness of the changeout. Results from this study showed that average winter $PM_{2.5}$ mass was reduced by 20%, and wood-smoke–related $PM_{2.5}$ concentration (as identified by the CMB model) was reduced by 28% when compared with the pre-changeout baseline 1 winter (Ward et al. 2010b). These findings demonstrate that even though over 1100 older model wood stoves were replaced as part of the changeout program, wood smoke $PM_{2.5}$ from the new (lower emission) stoves was still the major source of ambient $PM_{2.5}$ in the winter.

Although these reductions were substantial and achieved the community's goal of meeting compliance standards, the overall reductions in ambient $PM_{2.5}$ concentration were not as large as might have been predicted. Previous large-scale wood stove replacement programs or restrictions resulted in reductions of PM concentration greater than 40%, but these evaluations were based on PM_{10} size fractions (Missoula City–County Health Department 1999; Houck et al. 2005). Particle emissions from new U.S. EPA-certified wood stoves are estimated to be 70% lower than older wood stoves (U.S. EPA 2010). Gross application of such emission reductions over all wood stoves in the community suggested a reduction of ambient $PM_{2.5}$ concentration greater than the observed 25%. To fully understand the anticipated versus observed improvements in ambient air quality, the half-life of $PM_{2.5}$ should be considered. Once emitted, $PM_{2.5}$ can remain airborne for days or weeks under the inversion conditions commonly experienced in the Libby valley. In addition, low sun angles and often overcast conditions reduce the potential for photodegradation of the $PM_{2.5}$ for extended periods of time during the winter months. Despite the reduction in $PM_{2.5}$ mass resulting from the replacement of old polluting wood stoves with new lower emission stoves, the trapping of these $PM_{2.5}$ emissions in the air shed could have resulted in the lower than expected effect of the changeout program on ambient concentrations of $PM_{2.5}$. Changes in heating practices also could have influenced the overall effect of the program. Residences with dual heating options may have preferentially increased their wood combustion compared with previous years because of the newer, more efficient stove. We have no evidence that there was a dramatic increase in biomass burning for homes after the changeout. Indeed, almost 11% of homes participating in the program surrendered their wood stove and switched to alternative heating sources such as electricity or propane (Eagle and Houck 2007b). The potential for increased wood burning among some wood-burning homes, however, cannot be entirely ruled out. Finally, as documented in another community affected by wood smoke, the full realization of anticipated emissions reductions after a wood stove replacement is dependent upon proper burning practices (Ward et al. 2010a). The Libby changeout program included a training component targeted toward recipients of new wood stoves, but there was no follow-up assessment to evaluate the efficacy of this training.

In addition to overall changes in $PM_{2.5}$ concentrations during the wood stove changeout program, we were able to track some of the changes in particle chemistry. Although the EC fraction was low in this community, there was a notable increase in the final winter period (year 4). Unfortunately the state speciation program at this site had been discontinued before this final year. OC and EC analysis of filters from this final winter were analyzed by Desert Research Institute using the IMPROVE_A method (Chow et al. 1993; Desert Research Institute 2008), whereas in previous years OC and EC were analyzed through the state compliance program by Research Triangle Institute using the NMAM 5040 method (National Institute of Occupational Safety and Health 1999). Changing from one method to another likely accounted for the aberrant observation of increased EC concentration during the final winter.

A similar concern exists with the polar organic compounds, as the same quartz filters in the final winter (year 4) were analyzed by Desert Research Institute for levoglucosan and resin acids rather than by our own laboratory. Even before this final winter, however, we noted increases in abietic and dehydroabietic acid concentrations in year 3 compared with year 2. These observations suggested that the chemistry of the $PM_{2.5}$ had changed as a result of the wood stove changeout. Previous studies have indicated that changes in the relative concentrations of the resin acids can be explained either by changes in the type of fuel burned or by photolysis of the compounds when PM is exposed to sunlight over a period of time (Simoneit et al. 1993; Corin et al. 2000; Leithead et al. 2006). These observations are unlikely to explain our results, however, because the tree species used for fuel in the region remained the same before and after the changeout. There is also little reason to suspect differences in the photolysis conditions of indoor PM pre- and post-changeout. Others have observed differences in the chemical composition of PM depending on air flow during combustion or on stove design (Purvis et al. 2000; Jordan and Seen 2005). Purvis and colleagues (2000) not only observed significant decreases in PM_{10} and $PM_{2.5}$ emissions from a modern wood stove relative to an older design, but also reported unchanged or increased emissions of various semivolatile organics, including polycyclic aromatic hydrocarbons. Our results suggest that while more complete combustion in the modern U.S. EPA-certified wood stove does result in significant decreases in emissions of $PM_{2.5}$ and combustible pyrolysis products such as levoglucosan, chemically stable compounds such as resin acids that are released through vaporization rather than combustion or pyrolysis (Simoneit et al. 1993) may continue to be released at similar or even higher concentrations.

The small size of the study community served as both an advantage and disadvantage in addressing the HEI call for health outcome studies. The greatest advantage of this study location was the ability to observe the effect of a targeted community-wide intervention program that had a measurable effect on ambient $PM_{2.5}$ concentration in a relatively short timeframe. One disadvantage of this study location was the historical legacy of asbestos contamination and the corresponding asbestos-related disease in the community. Throughout the study period, this community was an active Superfund site with ongoing asbestos remediation activities. Given the recent asbestos-related health surveillance activities, it is possible that the community had a heightened awareness of environmentally-related health outcomes. It is also possible that the community had experienced research fatigue, which may have led to lower participation rates among certain parents. We suspect that this factor is of limited concern for this study that focused on children's respiratory health. Previous asbestos-related health research in the community focused on asbestos-related health outcomes among adults with past occupational and environmental exposures (Peipins et al. 2003; Noonan et al. 2006; Larson et al. 2010). Another potential disadvantage of this study location was the small population size, resulting in limited ability to observe small-magnitude effects. Despite this concern, and the resulting imprecision in risk estimates, we still observed reductions in the frequency of reported wheeze and other symptoms and infections with CIs that supported the a priori study hypotheses. A study focused more precisely on clinical objective measures in a susceptible population such as people with asthma may have been warranted. Our survey data suggested, however, that the benefits of reduced ambient winter $PM_{2.5}$ concentrations included more general infectious health conditions and were not limited to a specific population, such as children with asthma.

This research activity did not use a purposive sampling design; it was a natural experiment that tracked a community-led wood stove changeout program. This quasi-experimental design limited our ability to account for changes over time other than wood stoves and PM concentrations. To partially address this concern we were able to evaluate our results with respect to changes in meteorologic conditions and community-level influenza frequency. The inclusion of a control community in future studies would help to better characterize the effect of time-dependent variation in ambient conditions and other factors. The prospective approach of this study was an important strength. Because this was a natural rather than purposive design, however, we had limited ability to control the timing of the study observations. Ideally we would have collected an additional year of survey data before the start of the wood stove changeout program. This experience confirms the need for rapid planning and responsiveness to secure the greatest scientific yield from community-level programs that are attempting to decrease the ambient concentrations of criteria air pollutants.

ACKNOWLEDGMENTS

The authors would like to thank the parents of the Libby children for their participation in this study. We also thank the Libby school administrators and staff, notably Kirby Maki, Helena Cassady, Ken Foss, Ron Goodman, Keith Ivers, Margaret O'Brien-Johnson, Gigi Scofield, Kim Edlund, and Ellen Mills. State compliance monitoring data and advice were provided by Bob Habeck and Elton Erp of the Montana DEQ. Administrative and data entry support was provided by Mike Leary, Corey Regini, Emily Weiler, and Stephanie Domitrovitch. Assistance with data analysis was provided by Colorado School of Mines students Steven Geinitz, Tonya Lauriski-Karriker, and Claire Le Lait. We also thank Amy Smart of the Lincoln County Health Department and Jeanie Gentry of Saint John's Lutheran Hospital for acquiring hospital discharge data. During this study we received important advice from the HEI Research Committee. Most importantly, we would like to thank Kathy Hooper, Director of Lincoln County Environmental Health Department, for her tireless support of this study.

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APPENDIX A. HEI QUALITY ASSURANCE STATEMENT

The conduct of this study was subjected to independent audit by Mr. David Bush of T&B Systems, Inc. Mr. Bush is an expert in quality assurance for air quality monitoring studies and data management. The audit included reviews of study activities for conformance to the study protocol and reviews of the overall data quality. The dates of the audit activities are listed below, along with the phase of the study and a brief summary of findings.

August 8–10, 2007

The auditor conducted an on-site audit at The University of Montana in Missoula and at the monitoring locations in Libby, Montana. The audit consisted of a review of the existing study documentation, interviews with key study personnel, and observation of monitoring, data collection, and laboratory activities. Questions arose regarding pre-winter 2005/2006 supplemental air quality data, and these data were ultimately not used during analysis. No other significant issues were noted.

July 2011

The auditor reviewed the draft final report, as well as the finalized data sets, verifying validation of the final data and consistency between the submitted data and the final report. Only minor inconsistencies were noted. All issues were appropriately addressed by the study investigators.

Written reports of the audits were provided to the HEI project manager, who transmitted the findings to the Principle Investigators. The quality assurance audit demonstrated that the study was conducted by an experienced team with a high concern for data quality.

David H. Bush
Quality Assurance Officer

APPENDIX B. Supplementary Tables

Table B.1. Minimum Detection Limits for $PM_{2.5}$, OC, and EC, and the Seven Selected Tracers for Wood Smoke in the Air

Parameter	Minimum Detection Limit
$PM_{2.5}$	1.0 µg/m^3
OC	0.098 µg/m^3
EC	0.007 µg/m^3
Levoglucosan	7.7 ng/m^3
Dehydroabietic Acid	0.6 ng/m^3
Abietic Acid	0.5 ng/m^3
Vanillin	0.9 ng/m^3
Acetovanillin	0.5 ng/m^3
Guaiacol	0.03 ng/m^3
4-Ethylguaiacol	0.1 ng/m^3

Table B.2. Air Sampling in Grade School During Four Winter Periods[a]

	Year 1 (n = 15)	Year 2 (n = 7)	Year 3 (n = 11)	Year 4 (n = 10)
$PM_{2.5}$ (µg/m^3)	27.08 ± 5.76	26.64 ± 16.37	21.50 ± 8.45	30.52 ± 11.26
OC (µg/m^3)	13.5 ± 1.30	12.11 ± 1.30	11.97 ± 1.29	17.52 ± 4.23
EC (µg/m^3)	0.39 ± 0.12	0.50 ± 0.24	0.36 ± 0.23	1.01 ± 0.65
Levoglucosan (ng/m^3)	264.1 ± 116.1	163.0 ± 128.5	706.4 ± 284.9	410.0 ± 144.6
Dehydroabietic acid (ng/m^3)	133.6 ± 35.9	74.5 ± 21.4	162.4 ± 39.6	50.4 ± 22.9
Abietic acid (ng/m^3)	8.36 ± 3.62	4.53 ± 4.77	19.54 ± 13.18	73.16 ± 40.90
Vanillin (ng/m^3)	2.33 ± 1.54	0.45 ± 0.00	8.83 ± 6.56	0.45 ± 0.00
Acetovanillin (ng/m^3)	0.25 ± 0.00	0.25 ± 0.00	16.20 ± 11.42	0.25 ± 0.00
Guaiacol (ng/m^3)	0.10 ± 0.07	0.25 ± 0.15	0.34 ± 0.57	0.25 ± 0.15
4-Ethylguaiacol (ng/m^3)	0.17 ± 0.07	0.24 ± 0.12	0.26 ± 0.35	0.07 ± 0.06

[a] Values are mean ± SD. Analyses conducted by Montana DEQ for years 1–3 and by the Desert Research Institute for year 4.

Table B.3. Air Sampling in Middle School During Four Winter Periods[a]

	Year 1 (n = 12)	Year 2 (n = 8)	Year 3 (n = 10)	Year 4 (n = 9)
$PM_{2.5}$ (µg/m^3)	7.67 ± 1.53	15.17 ± 6.87	6.54 ± 1.97	20.40 ± 24.15
OC (µg/m^3)	8.30 ± 1.53	7.27 ± 0.71	6.84 ± 0.89	9.81 ± 1.43
EC (µg/m^3)	0.23 ± 0.12	0.19 ± 0.08	0.11 ± 0.11	0.32 ± 0.16
Levoglucosan (ng/m^3)	572.9 ± 223.2	341.0 ± 185.0	597.9 ± 120.3	405.1 ± 119.7
Dehydroabietic acid (ng/m^3)	163.3 ± 46.2	113.1 ± 71.9	132.4 ± 13.6	27.4 ± 9.31
Abietic acid (ng/m^3)	7.08 ± 2.25	6.92 ± 7.92	11.26 ± 4.18	37.59 ± 26.87
Vanillin (ng/m^3)	3.61 ± 3.01	0.45 ± 0.00	3.69 ± 3.41	0.45 ± 0.00
Acetovanillin (ng/m^3)	0.32 ± 0.27	0.25 ± 0.00	10.40 ± 11.43	0.25 ± 0.00
Guaiacol (ng/m^3)	0.14 ± 0.09	0.40 ± 0.43	0.14 ± 0.05	0.16 ± 0.12
4-Ethylguaiacol (ng/m^3)	0.17 ± 0.07	0.14 ± 0.13	0.09 ± 0.08	0.12 ± 0.11

[a] Values are mean ± SD. Analyses conducted by Montana DEQ for years 1–3 and by the Desert Research Institute for year 4.

Table B.4. Effects Estimates Across Four Winter Periods for Children's Health Outcomes Associated with a 5-µg/m^3 Decrease in Ambient $PM_{2.5}$ Concentration, Adjusting for Community-Level Influenza Rate or for Individual Influenza Reporting

Outcome	Adjusted for Community-Level Influenza Rate OR (95% CI)[a]	Adjusted for Individual-Level Reporting of Influenza OR (95% CI)[a]
Symptoms		
Wheeze	0.733 (0.555–0.970)	0.739 (0.559–0.978)
Morning tightness of chest	0.935 (0.660–1.324)	1.015 (0.707–1.459)
Attack of shortness of breath	0.796 (0.543–1.168)	0.807 (0.546–1.191)
Shortness of breath after exercise	1.031 (0.773–1.376)	1.046 (0.780–1.402)
Wake up at night by an attack of shortness of breath	1.027 (0.615–1.715)	0.869 (0.507–1.489)
Wake up at night by an attack of coughing	0.904 (0.755–1.082)	0.947 (0.785–1.143)
Other symptoms and infections		
Itchy/watery eyes	0.668 (0.554–0.806)	0.720 (0.599–0.866)
Sore throat	0.684 (0.570–0.822)	0.796 (0.665–0.953)
Cold	0.746 (0.603–0.924)	0.824 (0.673–1.009)
Bronchitis	0.454 (0.272–0.758)	0.445 (0.271–0.730)
Influenza	0.477 (0.395–0.575)	—
Throat infection	0.549 (0.424–0.710)	0.671 (0.517–0.861)
Middle-ear infection	0.709 (0.490–1.023)	0.704 (0.489–1.011)
Symptoms with *a priori* null effect		
Dizziness	0.937 (0.747–1.176)	1.130 (0.888–1.438)
Nausea	0.774 (0.640–0.936)	1.007 (0.815–1.244)
Vomiting	0.627 (0.501–0.783)	1.000 (0.787–1.271)
Stomach pain	0.896 (0.748–1.072)	1.156 (0.955–1.401)
Diarrhea	1.083 (0.878–1.335)	1.646 (1.296–2.091)

[a] The ORs were adjusted for student age group and two different indicators of influenza.

Table B.5. Effects Estimates Across Four Winter Periods for Children's Health Outcomes Associated with a 5-µg/m^3 Decrease in Ambient $PM_{2.5}$ Concentration for All Subjects or for a Restricted Set of Subjects

Outcome	All Subjects OR (95% CI)[b]	Restricted Analysis[a] OR (95% CI)[b]
Symptoms		
Wheeze	0.733 (0.555–0.970)	0.703 (0.463–1.067)
Morning tightness of chest	0.935 (0.660–1.324)	1.399 (0.824–2.376)
Attack of shortness of breath	0.796 (0.543–1.168)	0.779 (0.438–1.383)
Shortness of breath after exercise	1.031 (0.773–1.376)	1.256 (0.825–1.913)
Wake up at night by an attack of shortness of breath	1.027 (0.615–1.715)	2.101 (0.738–5.984)
Wake up at night by an attack of coughing	0.904 (0.755–1.082)	0.860 (0.649–1.141)
Other symptoms and infections		
Itchy/watery eyes	0.668 (0.554–0.806)	0.703 (0.533–0.928)
Sore throat	0.684 (0.570–0.822)	0.738 (0.570–0.956)
Cold	0.746 (0.603–0.924)	0.773 (0.571–1.046)
Bronchitis	0.454 (0.272–0.758)	0.335 (0.147–0.762)
Influenza	0.477 (0.395–0.575)	0.474 (0.354–0.634)
Throat infection	0.549 (0.424–0.710)	0.358 (0.221–0.578)
Middle-ear infection	0.709 (0.490–1.023)	0.688 (0.408–1.158)
Symptoms with a priori null effect		
Dizziness	0.937 (0.747–1.176)	1.006 (0.711–1.423)
Nausea	0.774 (0.640–0.936)	0.877 (0.662–1.163)
Vomiting	0.627 (0.501–0.783)	0.675 (0.475–0.960)
Stomach pain	0.896 (0.748–1.072)	0.910 (0.704–1.176)
Diarrhea	1.083 (0.878–1.335)	1.068 (0.775–1.472)

[a] The analyses were restricted to those with data for year 1 and for one or more subsequent years.
[b] The ORs were adjusted for student age group and community influenza rate.

Table B.6. Effects Estimates Across Four Winter Periods for Children's Health Outcomes Associated with Decreases in Ambient Concentrations of PM Components[a]

Outcome	OC OR (95% CI)[b]	EC OR (95% CI)[b]	Levoglucosan OR (95% CI)[c]	Dehydroabietic Acid OR (95% CI)[c]	Abietic Acid OR (95% CI)[c]
Wheeze	0.896 (0.809–0.991)	1.159 (0.924–1.455)	0.9999 (0.9992–1.0005)	1.011 (0.990–1.033)	1.009 (0.998–1.021)
Itchy/watery eyes	0.867 (0.810–0.929)	1.208 (1.047–1.394)	1.0000 (0.9995–1.0004)	1.015 (1.002–1.028)	1.011 (1.004–1.018)
Sore throat	0.876 (0.819–0.937)	1.210 (1.059–1.384)	1.0001 (0.9997–1.0005)	1.016 (1.004–1.028)	1.011 (1.004–1.017)
Cold	0.897 (0.829–0.970)	1.083 (0.938–1.252)	0.9998 (0.9994–1.0003)	1.006 (0.993–1.019)	1.005 (0.998–1.012)
Bronchitis	0.750 (0.626–0.900)	1.260 (0.795–1.996)	0.9990 (0.9977–1.0004)	1.014 (0.973–1.056)	1.018 (0.995–1.042)
Influenza	0.785 (0.733–0.842)	1.965 (1.686–2.290)	1.0010 (1.0005–1.0014)	1.060 (1.045–1.074)	1.035 (1.027–1.043)
Throat infection	0.813 (0.740–0.894)	1.495 (1.201–1.860)	1.0002 (0.9996–1.0008)	1.033 (1.013–1.054)	1.022 (1.011–1.036)

[a] The ORs were adjusted for student age group and community influenza rate.
[b] Per 1-µg/m^3 decrease in average winter concentration.
[c] Per 1-ng/m^3 decrease in average winter concentration.

APPENDIX C. Supplementary Figures

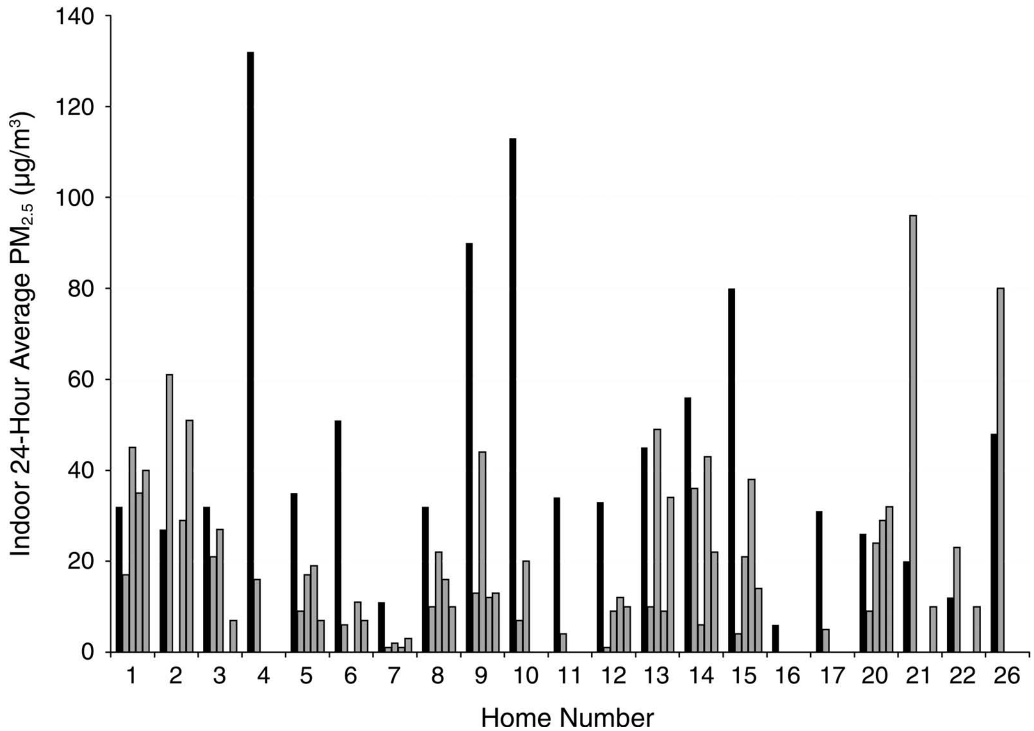

Figure C.1. Indoor $PM_{2.5}$ concentrations pre- (black bars) and post- (grey bars) wood stove changeout for homes of nonsmokers. Numbers along the *x*-axis identify specific homes. Data were available for 21 of the original 26 homes.

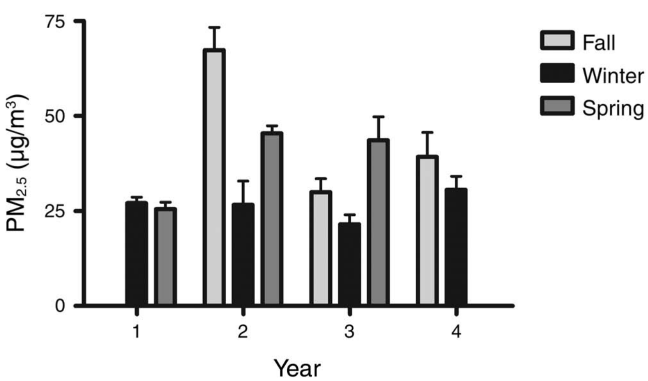

Figure C.2. Indoor $PM_{2.5}$ concentrations by sampling season at the grade school.

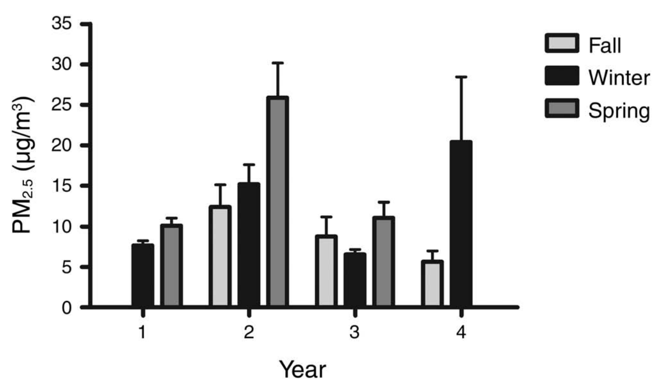

Figure C.3. Indoor $PM_{2.5}$ concentrations by sampling season at the middle school.

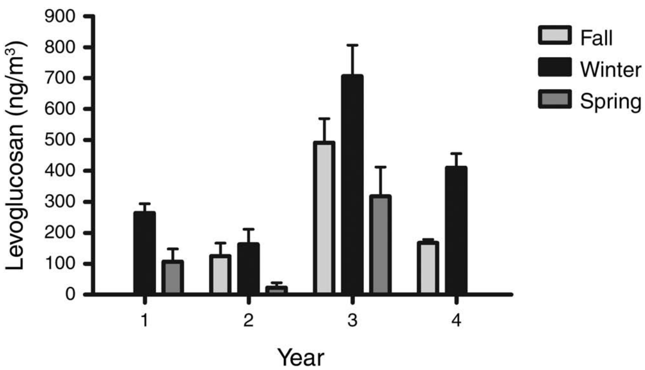

Figure C.4. Indoor levoglucosan concentrations by sampling season at the grade school.

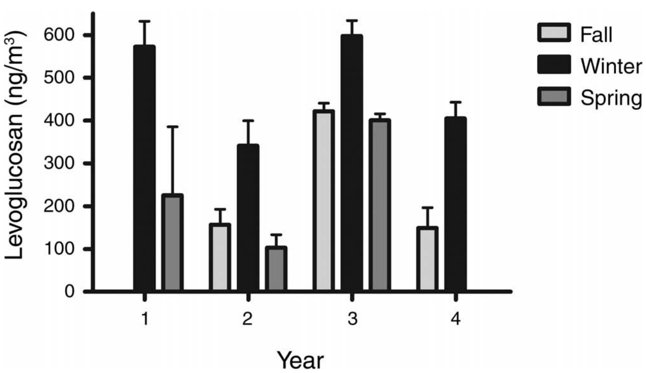

Figure C.5. Indoor levoglucosan concentrations by sampling season at the middle school.

APPENDICES AVAILABLE ON THE WEB

Appendices D, E, and F contain supplemental material not included in the printed report. They are available on the HEI Web site *http://pubs.healtheffects.org.*

Appendix D. Data Collection for Residential Sampling

Appendix E. Survey

Appendix F. Quality Assurance/Quality Control Results

ABOUT THE AUTHORS

Curtis W. Noonan is an associate professor of epidemiology at the Center for Environmental Health Sciences, Department of Biomedical Sciences at The University of Montana. He received his Ph.D. in environmental health/epidemiology from Colorado State University. His research focuses on health effects from air pollution due to biomass combustion and other sources.

Tony J. Ward is an assistant professor at the Center for Environmental Health Sciences, Department of Biomedical Sciences at The University of Montana. He received his Ph.D. in chemistry from The University of Montana in 2001. His research focuses on exposure assessment for indoor and outdoor pollutants.

William C. Navidi is a professor in the Department of Mathematical and Computer Sciences at the Colorado School of Mines. He received his Ph.D. from the University of California, Berkeley. His research focuses on the effects of air quality on children's health and methods for case–control sampling.

Lianne Sheppard is a professor of biostatistics and occupational and environmental health at the University of Washington. She received her Ph.D. from the University of Washington. Her research focuses on observational study methods, grouping, environmental and occupational exposures, and air pollution health effects.

Megan Bergauff is a postdoctoral fellow at the Center for Environmental Health Sciences, Department of Biomedical Sciences at The University of Montana. She received her Ph.D. in chemistry from The University of Montana in 2009. Her research focuses on separation chemistry and biomarker research.

Chris Palmer is a professor in the Department of Chemistry at The University of Montana. He received his Ph.D. in analytical chemistry from the University of Arizona in 1991. His research focuses on development of novel polymeric materials for application in microscale liquid phase separations and in the application of separations methods to environmental analysis.

OTHER PUBLICATIONS RESULTING FROM THIS WORK

MacNamara ML, Noonan CW, Ward TJ. 2011. Correction factor for continuous monitoring of wood smoke fine particulate matter. Aerosol Air Qual Res 11:315–322.

Ward TJ, Noonan CW, Palmer CP. 2010. Fine particulate matter source apportionment following a large woodstove changeout program in Libby, Montana. J Air Waste Manag Assoc 60:688–693.

Bergauff M, Ward TJ, Noonan CW, Palmer CP. 2009. The effect of a woodstove changeout on ambient levels of $PM_{2.5}$ and chemical tracers for wood smoke in Libby, Montana. Atmos Environ 43:2938–2943.

Ward TJ, Palmer CP, Houck JE, Navidi WC, Geinitz S, Noonan CW. 2009. A community woodstove changeout and impact on ambient concentrations of polycyclic aromatic hydrocarbons and phenolics. Environ Sci Technol 43:5345–5350. [PMCID: PMC2735050]

Bergauff M, Ward T, Noonan C, Palmer C. 2008. Determination and evaluation of selected organic chemical tracers for wood smoke in airborne particulate matter. Int J Environ Anal Chem 88:473–486.

Ward TJ, Palmer C, Bergauff M, Hooper K, Noonan CW. 2008. Results of a residential indoor $PM_{2.5}$ sampling program before and after a woodstove changeout. Indoor Air 18:408–415.

Noonan CW, Ward TJ. 2007. Environmental tobacco smoke, woodstove heating and risk of asthma symptoms. J Asthma 44:735–738.

ABBREVIATIONS

CI	confidence interval
CMB	chemical mass balance
DEQ	Department of Environmental Quality
EC	elemental carbon
FRM	federal reference method
GC-MS	gas chromatograph mass spectrometer
ISAAC	International Study of Asthma and Allergies in Childhood
NMAM	National Institute of Occupational Safety and Health Manual of Analytical Methods
MDL	minimum detection limit
NAAQS	National Ambient Air Quality Standard
OC	organic carbon
OR	odds ratio
PEM	personal environmental monitor
PM	particulate matter
$PM_{2.5}$	PM ≤ 2.5 µm in aerodynamic diameter
PM_{10}	PM ≤ 10 µm in aerodynamic diameter
RFA	Request for Applications
QA/QC	quality assurance / quality control
SD	standard deviation
U.S. EPA	U.S. Environmental Protection Agency

CRITIQUE
Health Review Committee

Research Report 162, *Assessing the Impact of a Wood Stove Replacement Program on Air Quality and Children's Health*, C.W. Noonan et al.

INTRODUCTION

Recent decades have seen substantial gains in air quality in the United States and Western Europe, with downward trends in concentrations of several major pollutants, including particulate matter (PM*). In large part, these gains have been achieved through increasingly stringent air quality regulations and control measures. Because risk assessments have estimated that air pollution is associated with a substantial burden of premature mortality and excess morbidity, even at current ambient pollution levels, it would be important to verify that air quality regulations have resulted in improved air quality and improved health. However, evidence for such verification — particularly in terms of health outcomes — has not been systematically collected. Thus, there is a need for research to assess the performance of environmental regulatory policy, an effort that has been termed accountability research or, more recently, health outcomes research.

HEI launched an initiative to improve the evidentiary basis for assessing the health impact of regulations and other actions to improve air quality with its 2000–2005 Strategic Plan (Health Effects Institute 2000). In 2003, HEI published a monograph setting out a conceptual framework to address research needs; it identified the types of evidence required as well as the methods by which that evidence could be obtained (HEI Accountability Working Group 2003). Nine studies were funded through four Requests for Applications (RFAs), issued between 2002 and 2004, that solicited studies to measure the health impacts of actions taken to improve air quality (see Preface). The study led by Dr. Curtis Noonan was funded under RFA 04-4, which sought proposals for studies of the health effects of real-world experiments, that is, planned actions taken with the intent of improving air quality or other actions that might have resulted in changes in air quality. Proposed studies would measure both changes in air quality resulting from these actions and changes in health status or health effects in the affected populations.

Noonan and colleagues proposed to evaluate a relatively large-scale wood stove changeout program taking place in a rural mountain community (Libby, Montana). In this community, which was at the time out of compliance with the National Ambient Air Quality Standard for $PM_{2.5}$ (PM ≤ 2.5 µm in aerodynamic diameter), residential wood combustion had been identified as a major source of $PM_{2.5}$ during the heating season. The changeout program was designed to replace older, higher-polluting wood-burning appliances with newer, lower-polluting heating appliances. The new heating appliances available for changeout were certified by the U.S. Environmental Protection Agency (U.S. EPA). Noonan hypothesized that the intervention would substantially reduce community exposure to $PM_{2.5}$ derived from wood smoke, leading to reductions in children's respiratory symptoms and illness-related school absences. The HEI Health Research Committee thought the changeout program presented a unique opportunity to study the air quality and health effects of a clearly defined intervention that was designed to address the major source of $PM_{2.5}$ within a single air shed. Noonan had initially proposed to measure lung function and biologic markers of inflammation (in exhaled breath). Because the Committee thought it was not clear whether lung function was a sensitive enough measure and whether potentially small differences in exposure would lead to detectable differences in the proposed biomarkers, those components were not included in the final study design.

This Critique is intended to aid the sponsors of HEI and the public by highlighting both the strengths and limitations of the study and by placing the Investigators' Report into scientific and regulatory perspective.

Dr. Noonan's 3-year study, "Assessing the impact on air quality and children's health of actions taken to reduce $PM_{2.5}$ levels from woodstoves," began in March 2006. Total expenditures were $666,430. The draft Investigators' Report from Noonan and colleagues was received for review in August 2010. A revised report, received in January 2011, was accepted for publication in February 2011. During the review process, the HEI Health Review Committee and the investigators had the opportunity to exchange comments and to clarify issues in both the Investigators' Report and in the Review Committee's Critique. As a coinvestigator of the Noonan report, Dr. Lianne Sheppard was not involved in its evaluation by the Review Committee.

* A list of abbreviations and other terms appears at the end of the Investigators' Report.

SCIENTIFIC AND REGULATORY BACKGROUND

The use of solid fuels, including wood, for cooking and heating is known to be the major source of household air pollution in developing countries (Smith and Mehta 2003). Residential wood burning also remains a common form of household heating in the developed world, with direct implications for both household exposures and ambient concentrations. Wood burning in fireplaces (for ambience) is also common in much of the developed world and a growing source of concern for policy makers. Source apportionment studies conducted in the Pacific Northwest, Canada, Northern Europe, Australia, and New Zealand have consistently shown that wood smoke is a major source of ambient PM during winter months in these regions (Naeher et al. 2007).

Similar to other forms of smoke from biomass combustion, wood smoke is a complex mixture of compounds, including inorganic gases (e.g., carbon monoxide, ozone, nitrogen dioxide), hydrocarbons (e.g., 1,3-butadiene, benzo[*a*]pyrene), oxygenated organic compounds, free radicals, and PM (Naeher et al. 2007). Exposure to air pollution from household wood burning in developing countries has been consistently associated with acute lower respiratory infections in young children and chronic obstructive pulmonary disease in adults, albeit at much higher concentrations than in developed countries (Smith and Mehta 2003). The epidemiologic evidence from studies conducted in developed countries suggests that exposure to wood smoke is associated with increased respiratory symptoms (e.g., otitis media, acute lower respiratory infections) in children and adults, decreased lung function in children, and increased emergency department visits and hospitalizations (Naeher et al. 2007). Recent controlled human exposure studies have shown increases in markers of inflammation and oxidative stress in blood, urine, breath condensate, and bronchoalveolar lavage fluid of volunteers exposed to wood smoke at 200–400 μg/m^3 of PM (Barregard et al. 2006, 2008; Sehlstedt et al. 2010; Ghio et al. 2011; Riddervold et al. 2011). In addition, wood smoke has been classified as "probably carcinogenic in humans" (Straif et al. 2006; International Agency for Research on Cancer 2010).

Wood-burning appliances have been a largely unregulated source of air pollution until fairly recently. Since 1988, a New Source Performance Standard issued by the U.S. EPA has required that all new wood stoves must comply with U.S. EPA particulate emissions guidelines issued under the Clean Air Act. However, this does not affect existing stoves. Particulate emissions from such EPA-certified new stoves are estimated to be about 50% lower than those from uncertified models. Certified stoves must meet a $PM_{2.5}$ emissions limit of 7.5 g/hr for noncatalytic wood stoves and 4.1 g/hr for catalytic wood stoves, based on independent testing by a U.S. EPA-accredited laboratory (U.S. EPA 2011a). (In catalytic stoves, exhaust is passed through a coated ceramic honeycomb inside the stove where the smoke gases and particles are oxidized.) $PM_{2.5}$ emissions from noncertified wood stoves are approximately 15 to 30 g/hr compared with 2 to 7 g/hr for all certified wood stoves and 1 to 4 g/hr for the newest certified stoves (U.S. EPA 2011b), which makes uncertified stoves a prime target for replacement with the goal of improving air quality. However, even emissions from certified stoves remain substantially higher than emissions from appliances that burn other types of fuel, such as oil (0.07 g/hr) or gas (0.04 g/hr) furnaces (Schreiber and Chinery 2008; U.S. EPA 2011b).

Air emission standards in the United States typically regulate the specific type of device, the fuels, and heat outputs. A recent report indicated that this may allow gaps and variation in coverage, and some residential and small-to-medium–sized biomass units may not be subject to environmental regulations (Handley et al. 2009). In contrast, regulations in Europe are issued according to heat output and type of feeding device (manual or automatic), which provides 100% coverage. More importantly, European $PM_{2.5}$ standards for wood-burning appliances are significantly lower at about 0.02 to 0.05 lb/million BTU heat output compared with state regulations in the United States at, for example, 0.1 lb/million BTU in Massachusetts and 0.6 lb/million BTU in New York State (Handley et al. 2009). This indicates that in the United States there is room for improvement in terms of reducing emissions from wood-burning appliances.

Given the fact that the U.S. regulations affect only new but not existing stoves, local and regional agencies in several states of the United States and in Canada have offered incentive programs to residents that subsidize the replacement of older, more polluting wood-burning stoves and boilers with newer, certified models. Depending on available funding, programs may use a variety of incentives, such as providing a full, free upgrade to a new stove, vouchers or rebates toward the purchase of a new wood stove, tax deductions, or vouchers to replace a wood stove with a heating device that does not burn wood. Funds offered through these programs range from several hundred to several thousand dollars per stove; many programs provide larger incentives for low-income households. Some programs were offered during a limited time (several months or one year), whereas others are longer term in nature.

The program in Libby, Montana, was one of the first extensive changeout programs in the United States and was a collaboration between the Montana Department of Environmental Quality (DEQ), the U.S. EPA, the Lincoln County Department of Public Health, the Hearth, Patio & Barbecue Association, the State of Montana, and the town of Libby. The program aimed to replace approximately 1200 uncertified wood stoves and 100 wood-burning boilers and furnaces to reduce winter concentrations of PM in this community. The wood stove changeout was planned to take place over the course of 2005–2007. During the initial phase of the changeout program, low-income households were offered a full upgrade to a certified wood stove donated by the Hearth, Patio & Barbecue Association. During the second phase of the program, additional residents were offered a voucher toward a certified wood stove of their choice. It should be noted that the new stoves were certified but were not necessarily in the cleanest category that were available at the time (i.e., catalytic stoves that emit less than 4.1 g/hr of $PM_{2.5}$). Several households opted to give up their wood stove altogether or decided to switch to a different heating source.

APPROACH

Noonan and colleagues evaluated whether the wood stove replacement program in Libby, Montana, led to reduced ambient and indoor air pollutant concentrations and improved respiratory health in children. They measured concentrations of $PM_{2.5}$ and some of its components (e.g., markers for wood smoke) outdoors, inside schools during different seasons, and in about 20 homes during the winter months (with additional sampling in subsequent winters). In parallel, they tracked parent-reported respiratory symptoms as well as illness-related school absences in children. Changes in wintertime reporting of symptoms and variations in school absences were also evaluated in relation to changes in ambient $PM_{2.5}$ concentrations in successive years.

The specific aims of the study were to

1. Prospectively measure ambient and indoor $PM_{2.5}$ concentrations during and after the implementation of the stove replacement program in Libby, Montana. This included sampling inside homes before and after stove replacement.
2. Prospectively track respiratory symptoms and infections among Libby children using a survey disseminated to parents through the schools.
3. Prospectively track school absences using records at the elementary, middle, and high schools.

A methods development aspect of the study was designed to evaluate whether specific markers for wood smoke could be used to track source-specific changes in air quality inside homes as well as in ambient air.

METHODS

STUDY DESIGN

The study started in 2005 (year 1), when the changeout program had just been initiated. Because the program start was delayed, few stoves were replaced before the first winter; the majority of change-outs took place during years 2 and 3 (2006 and 2007). The study was therefore extended to include a fourth winter (2008), which constituted the post-changeout phase.

The investigators prospectively collected air quality and health data during four consecutive winters (years 1–4). They collected additional data retrospectively, to cover two winters before the start of the changeout program (baseline years 1 and 2; i.e., 2003 and 2004). The investigators focused on the winter season only, which they defined as November through February.

For practical reasons, most stoves were replaced outside of the heating season. However, the investigators wanted to measure air quality in the homes during the winter season and to compare indoor air quality before and after stove changeout. The homes that could be studied were therefore limited to a relatively small number of homes (about 20) in which the stove was replaced during the heating season.

AMBIENT AIR QUALITY

Daily average ambient $PM_{2.5}$ concentrations were obtained from the Montana DEQ for the period 2001 through 2009. The chosen collection site was intended to capture pollutant concentrations that would be representative of pollutant exposures experienced by Libby residents. The site is located within three kilometers of the middle and elementary schools, where indoor sampling was conducted (see “School Indoor Air Quality” below) and between 0.8 and 13.7 kilometers from the homes where indoor monitoring was conducted. In addition, the DEQ collected 24-hour integrated samples once every six days at the ambient monitor to enable $PM_{2.5}$ speciation. The investigators collected additional samples for analysis of organic species. Retrospective air quality data were obtained for baseline years 1 and 2.

HOUSEHOLD INDOOR AIR QUALITY

The investigators collected 24-hour $PM_{2.5}$ samples in winter, both before and after wood stove replacement. Pre-changeout sampling was conducted in 26 homes, but valid post-changeout sampling was available for only 21 homes with nonsmoking residents. A DustTrack continuous monitor collected samples every minute. A single-filter gravimetric personal environmental monitor provided daily average concentrations for analysis of organic carbon (OC) and elemental carbon (EC) as well as organic compounds that are potential markers for wood smoke, including the most common marker — levoglucosan — and the additional markers dehydroabietic acid, abietic acid, vanillin, acetovanillin, guaiacol, and 4-ethylguaiacol. During the monitoring periods, residents were asked to record information on the frequency of stove loading, as well as any characteristics that could affect air quality (such as the use of secondary heating sources; burning of incense or candles; vacuuming, construction, or cooking; and the presence of pets).

SCHOOL INDOOR AIR QUALITY

The investigators collected 24-hour gravimetric indoor air samples of $PM_{2.5}$ in the gymnasiums of the elementary school and of the middle school in fall, winter, and spring of years 1 through 4. No sampling was conducted at the high school. Two single-filter personal environmental monitors measured daily average $PM_{2.5}$ concentrations for analysis of OC, EC, and markers for wood smoke.

ANALYSES OF $PM_{2.5}$ AND MARKERS FOR WOOD SMOKE

Indoor and ambient samples were analyzed for a number of potential markers for wood smoke (levoglucosan, the resin acids abietic acid and dehydroabietic acid, vanillins, and guaiacols) by the Center for Environmental Health Sciences, University of Montana. Analyses of $PM_{2.5}$ mass, OC, and EC, were conducted by Chester LabNet. Because the ambient sampler operated by DEQ was discontinued during the final year of the study, the additional samples that were collected by the investigators were analyzed by Desert Research Institute using similar methods.

HEALTH OUTCOMES

Children's respiratory health was assessed by distributing a survey through the schools to the parents after each of the four winters (in March). Initially all children in the elementary and middle schools were targeted. Grades that moved to the high school were part of the follow up in subsequent years. The survey included questions about respiratory symptoms adapted from a questionnaire used by the International Study of Asthma and Allergies in Childhood and questions about respiratory infections adapted from the National Health and Nutrition Examination Survey. Parents were also asked to provide information on home heating and tobacco smoking. The investigators conducted a pilot survey in baseline year 2. Based on the results and input from the HEI Research Committee, the questions of the final survey were slightly different. Because the results could not be directly compared, the pilot survey was not included in the main analyses.

Information on school absences was obtained from archived records at the schools. Only so-called incident absences (i.e., absences not preceded by an absence day) were used in the analyses. Absence data were collected for years 1–4. Additional absence data were available from the middle school for baseline years 1 and 2. Recorded data included information on the reason for the absence, allowing the investigators to analyze illness-related absences.

SUMMARY OF KEY RESULTS

Ambient winter concentrations of $PM_{2.5}$, as measured at the central monitoring site in Libby, MT, declined over the study period and were 30% lower in the final winter after the changeout program compared with the baseline years (i.e., 19 μg/m^3 in year 4 compared with 27 μg/m^3 in baseline years 1 and 2, see Table 1 of the Investigators' Report). Exceedences of the National Ambient Air Quality Standard for $PM_{2.5}$ were reduced to the extent that Libby was no longer out of compliance with the standard by the end of the study period. The investigators calculated that exchanging 10% of the noncertified wood stoves (~115 stoves) with certified models was associated with a reduction of 0.87 μg/m^3 (95% confidence interval [CI] = 0.54 to 1.20) in daily average ambient $PM_{2.5}$ concentrations.

Ambient concentrations of levoglucosan, a fairly well-validated marker for wood smoke, were lower during the first three winters of the program (2005 through 2007) than during the baseline winters (2003 and 2004), but increased again during the final winter (2008). Ambient concentrations of other potential markers for wood smoke, such as abietic acid and dehydroabietic acid, did not decrease in association with the changeout program. Decreased indoor $PM_{2.5}$ concentrations were observed in 16 of the 21 homes of nonsmoking residents after changeout, although there was substantial variability in the measurements within and between homes, and the average concentrations across homes varied over subsequent winters. At the elementary and middle schools, indoor concentrations of $PM_{2.5}$ and markers for wood smoke were variable and not consistent with the timing of the wood stove intervention program.

Based on about 1700 surveys filled out by parents of schoolchildren during the four years (with an annual return rate of 20%–50%), the investigators reported a 26% reduction in the effect estimate for childhood wheeze associated with a 5-μg/m^3 reduction in winter ambient $PM_{2.5}$ concentration (see Table 9 of the Investigators' Report). The most robust associations were observed for itchy or watery eyes, sore throat, bronchitis, influenza, and throat infection. There were no differences in health outcomes (notably, wheezing) between children (all ages combined) from homes with wood stoves and children from homes with other types of heating.

Analysis of school absence data showed that reductions in average ambient winter $PM_{2.5}$ concentrations were associated with fewer illness-related absences, but the association was not consistent across age groups: older students showed reduced absence rates, but there were higher absence rates among students in grades 1 through 4.

HEALTH REVIEW COMMITTEE CRITIQUE

STOVE EXCHANGE PROGRAM

Through extensive community outreach, the partners involved in the Lincoln County Woodstove Changeout program facilitated the replacement of more than 95% of approximately 1200 conventional wood-burning appliances with new EPA-certified stoves or other heating devices. The intervention was carried out successfully, although completion was slower than anticipated. It should be pointed out that the new stoves that were introduced in Libby were certified, but were not necessarily in the cleanest category available at the time (i.e., catalytic stoves that emit less than 4.1 g/hr of $PM_{2.5}$).

Given that the changeout program was quite successful, the study was informative. However, the prolonged time frame of the intervention did require an additional winter of sampling. This delay changed the nature of the evaluation conducted by Noonan and colleagues to some extent, but is probably more representative of interventions.

WOOD SMOKE AND AMBIENT AIR QUALITY

In its independent evaluation of the study, the HEI Review Committee thought the study had demonstrated that ambient air $PM_{2.5}$ concentrations in the community were reduced during the course of the changeout program, and that this reduction was sustained over subsequent winters.

The investigators estimated that changing 10% of the wood stoves was associated with a 0.87 μg/m^3 (95% CI = 0.54 to 1.20) reduction in ambient $PM_{2.5}$ concentration. Although this cumulative exchange metric appears to be easy to interpret, the Review Committee thought it may be misleading to conclude that the rate of reduction of $PM_{2.5}$ concentrations was constant throughout the study period. It is possible that the most polluting stoves were replaced first, with a larger effect of the intervention on ambient $PM_{2.5}$ concentrations during the early stages of the changeout program.

The 30% observed reduction in ambient $PM_{2.5}$ concentrations at the end of the intervention may be considered encouraging, even though earlier reports had estimated that reductions of 60% to 80% were possible (Air Quality Management Work Group 2005). One can calculate after the fact that the reduction would have been 38% if one assumes that all wood combustion was done in uncertified wood stoves, and if one takes into account that more than 95% of uncertified wood stoves in this community were replaced, that wood combustion was estimated to account for 80% of $PM_{2.5}$ concentrations in the pre-exchange period, and that emissions from each stove were expected to decrease by at least 50%. The difference between the observed and expected reductions in $PM_{2.5}$ concentrations may be partially explained by the fact that emissions from certified wood stoves may (and did) vary with the conditions of stove operation, and by the presence of other sources of wood smoke (e.g., fireplaces) that were not affected by the changeout. It is likely that even certified stoves will make substantial contributions to ambient PM concentrations — particularly when not operated optimally — as compared with heating appliances that use cleaner fuels. Certified wood stoves have been found to emit $PM_{2.5}$ at rates (about 2–7 g/hr) that are one to two orders of magnitude higher than those of oil (0.07 g/hr) or gas (0.04 g/hr) furnaces (Schreiber and Chinery 2008; U.S. EPA 2011b).

It remains unclear whether the use of wood-burning devices increased during the post-exchange period (in a manner unrelated to temperature differences among winters). It is well known that the choice of home heating devices is influenced by economic factors, such as changes in the cost of various fuels. Although the investigators provided information on the percentage changeout of existing stoves, it is not clear whether the total number of wood stoves in the community increased in the meantime, because more people may have chosen wood as a cheaper fuel, because of an increasing population size, or both. Even modest changes in population size or in fuel prices during the study period could affect $PM_{2.5}$ emissions in such a small community.

In addition, there may be other sources of $PM_{2.5}$ in the area, such as wood- or coal-burning fireplaces and boilers that were not covered by the changeout program. New York State has noted a tripling in the sales of outdoor wood boilers since the early 1990s that is related to increased costs of natural gas and oil for home heating (Schreiber and Chinery 2008). Although outdoor wood boilers help to partially reduce air pollution inside the home by moving the source outside, they produce considerably more emissions than certified wood stoves and have the potential to contribute significantly to outdoor air pollution that can penetrate indoors. It remains unclear to what extent there may be similar trends in other states, including Montana. Finally, it cannot be ruled out that some of the wood smoke in the Libby mountain valley may not have been locally produced but could have originated in adjacent valleys. It would be helpful to evaluate air pollution patterns in adjacent communities and atmospheric transport patterns to parse out the contributions of the changeout program.

SAMPLING OUTDOOR AND INDOOR AIR

Over the study period, the investigators observed varying concentrations of markers for wood smoke, including levoglucosan, abietic acid, and dehydroabietic acid. They also noted changes (both increases and decreases) in EC and OC concentrations over time. Although ambient levoglucosan concentrations were reduced initially, concentrations increased again during the final winter; concentrations of other markers for wood smoke — such as the resin acids abietic acid and dehydroabietic acid — did not consistently track with indoor $PM_{2.5}$ concentrations. In fact, changes in concentrations of levoglucosan were less consistent than those for $PM_{2.5}$ mass. This is puzzling, because levoglucosan would be expected to track better with wood burning than would $PM_{2.5}$ mass. The investigators proposed that some of the observed increases may have occurred because filter samples were analyzed by a different laboratory. The Committee thought that the lack of consistent reductions in these markers could also be explained by unexpected increases in wood burning or other factors.

Although Libby is a relatively small community, the extent to which a single central site monitor accurately represents exposures of the population must be questioned. The investigators characterized Libby as a single-source air shed; however, differences in concentrations of levoglucosan measured at the two school sites suggest that there was substantial spatial variability in the study area. Given highly dispersed, localized sources such as wood stoves, a central monitor may not capture changes at the neighborhood level, where homes with wood stoves may affect air quality at neighboring houses. For example, if a subject lived next door to a home that did not have a wood stove, the effect of the intervention might be less than for a subject who lived in a home surrounded by neighboring homes with wood stoves.

The Review Committee thought that Noonan and colleagues sampled a reasonable number of homes, although it was a small fraction of the homes in the changeout program. The number of available homes was somewhat limited because most changeouts occurred outside of the heating seasons, whereas the investigators targeted homes where the changeout took place during the winter. This would allow both pre- and post-changeout indoor $PM_{2.5}$ concentrations to be measured in the same winter. Some homes had to be excluded from the analyses because the occupants used an electric or propane heater instead of the wood stove or because it became apparent that the occupants smoked. During the study, it appeared that sampling in the homes was complicated by variation in stove usage with different ambient temperatures. To address this problem, the investigators added additional sampling on days that matched the ambient temperature on the previous sampling days, both before and after the changeout. This was a valid approach, even though it would be difficult to completely account for a possible temperature effect given the small number of homes. Some of the variation in indoor $PM_{2.5}$ concentrations may have been due to variation in stove operation, because improperly operated stoves emit larger amounts of particles.

RESPIRATORY HEALTH OUTCOMES AND ABSENTEE DATA IN SCHOOLCHILDREN

Although the study demonstrates an impact of the changeout program on ambient $PM_{2.5}$ concentrations (even if relatively modest), the Review Committee concluded that there was weak evidence that such air quality changes were associated with improved symptoms and illness in children. The lack of high-quality health outcomes data to reflect symptoms and illness was considered the most limiting aspect of the study design. In addition, periodic health surveys may not adequately capture acute illnesses. The investigators had originally proposed to evaluate more direct outcomes, such as biomarkers of exposure or health. However, this type of evaluation was not included in the study because the Research Committee thought that it was not clear whether lung function — the measurement proposed by the investigators — was a sensitive enough measure and whether potentially small differences in exposure could be detected using the proposed biomarkers. Future studies should continue to explore approaches that include biomarkers, however.

The investigators found no differences in health outcomes between children from homes with wood stoves and those from homes with other types of heating. This finding may suggest that health outcomes are more closely determined by the overall contribution of wood stoves to ambient air quality than by their contribution to air quality in individual homes. While this would be consistent with other studies in the literature (e.g., Allen et al. 2009) that do not show an effect of stove exchanges on indoor air quality, Noonan and colleagues did find that on average, indoor $PM_{2.5}$ concentrations were reduced in homes where stoves were exchanged. We note that the health outcomes analyses did not include indoor concentrations because such data were not available, so it cannot be directly determined from this analysis whether or not indoor wood smoke exposure affects health outcomes.

The Committee thought that Noonan and colleagues had chosen appropriate statistical methods to evaluate the intervention, although the study was limited by inherent challenges of the study setting, such as the seasonal nature of the intervention, the small size of the Libby community, the availability of only one year of pre-intervention survey data, and the small number of homes sampled. Nevertheless, the Committee agreed with the investigators that the finding of decreases in some symptoms associated with decreases in particulate pollution is at least suggestive of a beneficial effect of the intervention on children's respiratory symptoms. However, as is true for all observational studies, the uncertainty in the effect estimates given should be assumed to be larger than that reflected in the confidence intervals, which do not reflect uncertainty owing to residual confounding or intra-annual variation due to unknown risk factors.

The investigators made an a priori decision to define November through February as the winter season. The data suggest, however, that elevated $PM_{2.5}$ concentrations were present from October through March. Temperatures in October and March were also consistent with a winter (or heating) season classification. The Committee thought that limiting the winter season to November through February may have excluded two months (October and March) when wood smoke might have been present in sufficiently high concentrations to contribute additional information to the analyses, providing more statistical power to the analysis of absence data. In retrospect, defining the heating season using prespecified criteria for temperature, evidence of wood stove use, or both might have been better.

More pre-intervention data were available for the investigation of changes in school absences, which was helpful. However, the Committee agreed with the investigators that several features of the results suggested that interpreting the association of changes in outcomes with changes in air pollution as evidence for causality was unwarranted. These included the lack of consistency across age groups, the increase in nonsickness absences associated with decreased pollution, and other unexplained inter-annual differences among absence rates.

SUMMARY AND CONCLUSIONS

This study by Noonan and colleagues was funded under HEI's outcomes research program. Such studies are aimed at asking the following questions (as described in van Erp and Cohen 2009). Was the regulation implemented as specified? Did it achieve the intended reductions of pollutant concentrations and a subsequent reduction of the exposure of human populations? Did it have the anticipated public health benefits? What remains to be done?

The wood stove changeout program should be considered a success because 95% of older, high-polluting wood stoves in Libby, Montana, were replaced with more efficient certified wood stoves or with heating systems that did not burn wood. The current study shows that the intervention contributed to a sustained improvement in air quality that resulted in Libby achieving attainment status with respect to the $PM_{2.5}$ standard. However, the air quality improvement was not as large as might have been expected based on the dominant contribution of wood burning to ambient $PM_{2.5}$ concentrations in the area and the approximately 50% expected reduction in emissions anticipated from each certified stove compared with uncertified models; this could in part be due to other sources of wood smoke and variations in stove operation. Sampling from a small number of about 20 homes showed that indoor $PM_{2.5}$ concentrations generally decreased after the stove was replaced, but results were not completely consistent across homes. Several homes actually showed increased concentrations, which may be due to incorrect stove usage, other indoor sources of pollution, differences in the amount of burning during the sampling periods, or other factors.

This study attempted to evaluate whether additional markers for wood smoke — besides the relatively well-validated marker levoglucosan — could be used to track source-specific changes in air quality inside homes as well as in ambient air. However, changes in concentrations of levoglucosan were inexplicably less consistent than those for $PM_{2.5}$ mass. Other markers for wood smoke were even more variable and did not decrease consistently in concert with the changeout program. Further assessment of these data and additional research is needed before these organic compounds can be considered to be reliable markers for wood smoke exposure.

After the changeout, there was some evidence from the survey data that reduced ambient $PM_{2.5}$ concentrations were associated with improved respiratory outcomes (wheezing) in children and other symptoms associated with wood smoke exposure, such as itchy or watery eyes. Response rates were low and there were only four years of data upon which to base this analysis, only one of which represented the pre-intervention time period. The investigators found no differences in health outcomes between children from homes with wood stoves and those from homes with other types of heating. Also, in secondary analyses, changes in health outcomes were not associated with wood stove usage in the home. This finding may suggest that exposures are more closely determined by the overall contribution of wood stoves to ambient air quality rather than by their contribution to air quality in individual homes, a result that is consistent with other studies in the literature (e.g., Allen et al. 2009). Changes in health outcomes were also not associated with concentrations of markers for wood smoke (including the measurement of the common marker levoglucosan). This finding may be related to features of the wood smoke marker data in this study. There was an association of the intervention with illness-related school absenteeism, but it was not consistent across age groups.

Prospective outcomes studies provide an opportunity to collect detailed baseline data before the planned regulation or action takes effect and could include the evaluation of biomarkers and personal exposure, for which data are usually unavailable in retrospective studies. The challenge with such studies, however, is having sufficient advance notice of the intervention and thus sufficient lead time to develop a study protocol and have funding in place (van Erp and Cohen 2009). The indoor air quality subcomponent of this study was purely prospective, but the main ambient air quality study and assessment of health impacts was only partially prospective because the wood stove changeout program had already started. Sometimes, as was the case here, it may be useful to evaluate both shorter and longer time windows. Other unique features of this study were its focus on: (1) wood smoke as a contributor to poor ambient air quality; potentially shedding light on differential toxicity of air pollution mixtures with different compositions; (2) a rural, small community; (3) air quality and children's health, especially the relationship with infectious disease outcomes; and (4) the use of school absenteeism as a measure of air quality impacts. The latter is sometimes included in cost–benefit analyses, but there are not many studies supporting the use of absenteeism data.

In summary, this study showed that wood stove changeout programs can contribute to community-level improvements in air quality. The study also showed that air quality inside homes improved (although there were a few exceptions). It should be pointed out, however, that stoves remain relatively high emitters compared with oil or gas furnaces, and proper stove operation remains an important determinant of emissions.

This study provided some evidence of improved children's health in the community, based on parental surveys, with evidence of reduced rates of symptom reporting for wheeze, itchy or watery eyes, sore throat, bronchitis, influenza, and throat infection. Further research using hospital admission data or more direct health outcomes, such as medication use or biomarkers of exposure and health effects, would be useful. In terms of study design, investigators designing future studies may wish to consider panel studies, which use subjects as their own controls, or study designs that include a control community that is similar to the population under study, except for the air quality intervention, so that effects can be more clearly delineated.

ACKNOWLEDGMENTS

The Health Review Committee thanks the ad hoc reviewers for their help in evaluating the scientific merit of the Investigators' Report. The Committee is also grateful to Maria Costantini and Kate Adams for their oversight of the study, to Sumi Mehta and Annemoon van Erp for their assistance in preparing its Critique, to Carol Moyer for the science editing of this report and its Critique, and to Suzanne Gabriel, Fred Howe, Bernard Jacobson, and Flannery Carey McDermott for their roles in preparing this Research Report for publication.

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RESEARCH
REPORT

Number 162
December 2011