CRITICAL REVIEW OF THE

HEALTH EFFECTS OF WOODSMOKE

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1 INTRODUCTION

As the ability to control fire is often applied as the characteristic distinguishing pre-human and human evolution and wood is the oldest of human fuels, it is literally true that exposure to woodsmoke[1] is as old as humanity itself. Even today, biomass in the form of wood and agricultural wastes is a significant source of energy worldwide, with more than 10% of total fuel use. Of this, about 90% is used in its traditional forms as household heating and cooking fuels in developing countries, the rest being modern forms such as power plant fuel, principally in developed countries [1]. Because such household use dominates total fuel demand in many developing countries, particularly in rural areas where half of humanity still lives, it is probably true to say today as it has been since the control of fire that biomass is the main source of energy for most of humanity.

Surprisingly, however, although the percentage that wood makes of total fuel demand sharply declines with economic development, the absolute amount remains relatively constant. The average use of biomass fuel per capita in OECD (Organisation for Economic Co-operation and Development), which contains the world's richest countries, for example, is quite similar to that in the Asia-Pacific Region, which has the world's largest developing countries [1]. Of course per capita use varies substantially with local circumstances. For example, because of wood availabilities, Finland, Sweden, and Canada burn more biomass fuel per capita than other countries and South Korea and Singapore burn less [2].

Over the past few decades, rising fossil energy costs, the availability of new technologies, and the desire to use renewable sources has led to increases in the use of wood and other biomass fuels in North America. In Canada, such fuels increased at about 2.4% annually during the 1990s, more than half again as fast as overall energy demand [3].

During this same period, the knowledge of, and consequent concern about, the health effects of air pollution has increased dramatically around the world, leading to stricter air pollution regulation and controls. Open burning of trash was one of the first control measures introduced, but now wood burning in North America has come partly under regulation, based on emissions of criteria pollutants from commercial and industrial technologies and, to a lesser degree, new household wood heating stoves. There are still important non-regulated sources of woodsmoke, however, including existing household heating stoves and fireplaces, although these have been the target of local ordinances in a number of areas where woodsmoke dominates outdoor air pollution during some seasons. There is concern, however, that to reach standards for important pollutants, such as small particles, there may be need to forcefully address these household sources in more areas.

There are also important non-point sources of woodsmoke, particularly wildland and agricultural fires, whether purposeful or accidental. The apparent increases of accidental wildfires in some areas due perhaps to climate change; the increase in purposeful fires for management in others; the rise in human population density near fire-prone areas; and, the international attention given

[1] Here, we use the term "smoke" to refer to the entire mixture of gases, solid particles, and droplets emitted by combustion.

Table 1: Summary of the Toxic Chemical Agents Identified in Woodsmoke.

Chemical class	Number of compounds	Mode of toxicity	Representative compounds *
Toxic gases	4+	Irritant, acute toxicity	Carbon monoxide Ammonia Nitrogen dioxide Sulfur dioxide
VOCs (C2-C7)	30+	Irritant, possibly carcinogenic	Methyl chloride Methylene chloride
Saturated hydrocarbons	25+	Irritant, neurotoxicity	Hexane
Unsaturated hydrocarbons	40+	Irritant, carcinogenic, mutagenic	1,3-butadiene Acrolein
Mono-aromatics	28+	Irritant, carcinogenic, mutagenic	Benzene Styrene
Polycyclic aromatic hydrocarbons (PAHs)	20+	Carcinogenic, mutagenic, Immunotoxic	Benzo[163]pyrene, Dibenz[a,h]anthracene
Organic alcohols and acids	25+	Irritant, acute toxicity, Teratogenic	Methanol Acetic acid
Aldehydes	20+	Irritant, carcinogenic, mutagenic	Formaldehyde, Acetaldehyde
Phenols	33+	Irritant, carcinogenic, mutagenic, teratogenic	Catechol Cresol (methyl-phenols)
Quinones	3	Irritant, allergenic, Redox active, causes oxidative stress and inflammation response, possibly carcinogenic	Hydroquinone Fluorenone Anthraquinone
Free radicals		Redox active, cause oxidative stress and inflammation response, possibly carcinogenic	Semi-quinone type radicals
Inorganic compounds	14+	Carcinogenic, acute toxicity	Arsenic Lead Chromium
Fine particulate matter		Inflammation, may be allergenic	$PM_{2.5}$
Chlorinated dioxins		Irritant, may be carcinogenic or teratogenic	
Particulate acidity		Irritant	Sulfuric acid

* Compounds in italics are either criteria air pollutants, or are included on the EPA list of hazardous air pollutants. At least 26 hazardous air pollutants are known to be present in woodsmoke

to the problem because of spectacular fires in Southeast Asia and elsewhere has resulted in a growing concern about the potential health impacts of such events. A question arises whether ameliorative actions such as evacuations may be necessary for some populations.

The sentiment that woodsmoke, being a natural and ancient substance, must be benign to humans is still sometimes heard. It is now well established, however, that woodburning stoves and fireplaces as well as wildland and agricultural fires emit significant quantities of known health-damaging pollutants, including carcinogenic and other toxic organic compounds such as polyaromatic compounds, benzene, and aldehydes; respirable particulate matter with diameters allowing it to penetrate into the deep lung; carbon monoxide (CO); nitrogen oxides (NO_x), and free radicals, among many other pollutants [4-10]. See Table 1.

The main gaseous pollutants in woodsmoke, such as carbon monoxide and nitrogen oxides, add to the atmospheric levels of these regulated gases coming from other combustion sources and thus have always been regulated alike with them. As these gases are indistinguishable no matter where they come from, there is no urgent need to examine their particular health implications in woodsmoke. There are two other main reasons, however, why woodsmoke may be a special case requiring separate health evaluation.

Woodsmoke contains a vast array of solid, liquid, and gaseous constituents that change, sometime rapidly, with time, temperature, sunlight, and interaction with other pollutants, water vapor, and surfaces. Many of the constituents are known to be hazardous for human health, but are not specifically regulated or even fully evaluated. Current methods of health-effects assessment, however, do poorly in estimating impacts by summing the effects of separate constituents. The best approach is to examine the entire mixture. The most well-known case of this sort is another biomass smoke, that from tobacco burning. Although there have been more than 4000 compounds identified in tobacco smoke, many dozens of which possess toxic properties, there is still no well understood link between individual constituents and the vast array of health effects known to be caused by the intact mixture.

The first question for this Report, therefore, is *whether emissions of the mixture called woodsmoke need to be regulated separately from the separate constituents*, some of the most important of which are already under regulation.

Small particles (less than a few microns in diameter) are thought to be the best single indicator of the health impacts of most combustion sources. Although woodsmoke particles are often within the size thought to be most damaging to human health, their chemical composition is different from those derived from fossil fuel combustion on which most health-effects studies have focused. Except for size, most regulations do not distinguish particles by composition, although there is actually a large range of particle types. There is concern, however, that because of their different composition, woodsmoke particles may not produce the same amount of ill-health as other combustion particles per unit mass as has been indicated for general ambient air pollution in developed countries and, thus, should not be regulated in the same manner.

The second question for this Report, therefore, is *whether woodsmoke particles pose similar levels of risk as other ambient particles of similar sizes.*

Several topics are examined in this Report to address these two key issues:

- The chemical and physical nature of woodsmoke
- The toxicology of woodsmoke, based on animal exposures and laboratory tests;
- The extent of human exposure to woodsmoke in developed countries;
- The special exposures and epidemiology of smokes from wildland fires and agricultural burning and related controlled human laboratory exposures; and,
- The epidemiology of woodsmoke exposures in developed countries.

A short summary of the exposures and health effects of biomass smoke in developing countries is provided as an additional line of evidence.

The Conclusion to this Report returns to the two key issues above to summarize: (1) what is currently known about the health effects of inhaled woodsmoke at exposure levels experienced in North America; and, (2) whether there exists sufficient reason to believe that woodsmoke particles are adequately different to warrant separate treatment from other regulated particles. Recommendations concerning needed additional woodsmoke research to fill remaining gaps in knowledge are enumerated.

[2] Although "woodsmoke" is the substance of primary interest in this report, evidence related to smoke from other biomass (agricultural residues, grass, etc) is also examined where relevant.

2 CHEMICAL COMPOSITION OF BIOMASS SMOKE

Wood consists primarily of two polymers: 50-70 weight percent cellulose, and approximately 30% by weight lignin) [11]. Other biomass fuels (e.g. grasses, wheat stubble) also contain these polymers, although their relative proportions differ compared to wood. In addition small amounts of low molecular weight organic compounds (e.g. resins, waxes, sugars) and inorganic salts are also present in wood. During combustion, pyrolysis occurs and the polymers break apart producing a variety of smaller molecules. Biomass combustion is typically inefficient and a multitude of partially oxidized organic chemicals are generated in biomass smoke. Biomass

Table 2: Summary of Major Biomass Pollutants.

Compound	Examples	Source	Notes
Inorganic gases	Carbon monoxide (CO)	Incomplete combustion of organic material	Transported over distances
	Ozone (O_3)	Secondary product of nitrogen oxides and hydrocarbons	Only present downwind of fire, transported over distances
	Nitrogen Dioxide (NO_2)	High temperature oxidation of nitrogen in air	Reactive – concentrations decrease with distance from fire
Hydrocarbons	Benzene	Incomplete combustion of organic material	Some transport – also react to form organic aerosols
Aldehydes	Acrolein	Incomplete combustion of organic material	
	Formaldehyde (HCHO)	Incomplete combustion of organic material	
Particles	Inhalable particles (PM10)	Condensation of combustion gases; Incomplete combustion of organic material; entrainment of vegetation and ash fragments	Coarse + fine particles. Coarse particles are not transported over long distances and contain mostly soil and ash
	Respirable Particles	Condensation of combustion gases; Incomplete combustion of organic material	For biomass smoke, approximately equal to fine particles
	Fine particles (PM2.5)	Condensation of combustion gases; Incomplete combustion of organic material	Transported over long distances; Primary and secondary production
Polycyclic aromatic hydrocarbons (PAHs)	Benzo[163]pyrene (BaP)	Condensation of combustion gases; incomplete combustion of organic material	Specific species vary with composition of biomass and combustion conditions

smoke contains a large number of chemicals, many of which have been associated with adverse health impacts. The major chemicals present in biomass smoke and their sources are listed in Table 2. These chemicals include both particles and gaseous compounds.

Table 3 and Table 4 summarize the major chemical classes detected in woodsmoke, and detailed chemical speciation of the several hundred individual compounds that have been detected in smoke samples is reported in the original references [12, 13, 14-17]. The studies cited in Table 3and Table 4 by Rogge et al, Schauer et al, Fine et al and McDonald et al all attempted to re-create conditions of residential wood combustion. In contrast the studies by Oros et al aimed to be more representative of wildfire emissions. Lee et al have also described comprehensive chemical composition of smoke from prescribed burns recently [18]. Although less well characterized, a similar mixture of chemicals are reported in smoke emissions from other types of biomass including grasses, rice straw, sugarcane and ferns [11, 19, 20].[3]

Table 3: Average Fine Particle Emission Rate and Bulk Chemical Composition for Wood Combustion Emissions.

Compound Class	Concentration	References
Fine particle emissions rate (g/kg of wood burned)	1.6-9.5	[13-15]
Organic carbon (wt % of fine particle mass)	1.3-650	[13-15]
Elemental carbon (wt % of fine particle mass)	0.65-82	[13-15]
Ionic species (wt % of fine particle mass)	0.014-7.8	[13-15]
Elemental Species (wt % of fine particle mass)*	0.01-5.3	[13-15]
Methane (mg Kg-1 wood burned)	4100	[13]
Total non-methane hydrocarbons C2-C7 (mg/Kg wood burned)	390-4000	[13, 15]
Aldehydes and Ketones (mg/Kg wood burned)	0.94-4450	[12]**- [13-15]
CO (g/Kg wood burned)	130	[15]

*Chloride included as an element
**Only aldehydes reported

In general, it is difficult to make quantitative comparisons between emission factors for specific organic compounds reported by different authors, because many of the reports are semi-quantitative in that the analytical methods used were not comprehensively validated for each analyte, authentic standards were frequently not available to calibrate instrument response, variable combustion conditions (fuel type, moisture content, combustion device) were used, and emission factors were reported in a variety of units.

[3] It should be noted that to date, most studies have used gas chromatography/mass spectrometry (GC/MS) to characterize the chemical content of woodsmoke. GC is a very efficient tool for separating complex mixtures of organic chemicals. Combined with MS, the technique allows for highly sensitive, specific and accurate detection and quantification of a range of organic chemicals in environmental samples. GC/MS fails to detect compounds that are involatile or thermally labile, however. The application of novel methods such as liquid chromatography-mass spectrometry (LC/MS) that are appropriate for analysis of non-volatile or thermally labile compounds will further expand the list of chemicals known to be present in biomass smoke.

Table 4: Composition by Chemical Class for Wood Combustion Emissions.

Organic Chemical	Particle-phase Concentration (mg/Kg wood burned)	References	Vapor-phase Concentration (mg/Kg wood burned)	References
Alkanes (C2-C7)	0.47-570	[12, 14]	1.01-300	[13, 15]
Alkenes (C2-C7)	0.58-280	[12, 14]	92-1,300	[15]
Alkanols	0.24-5,400	[14, 16, 17]	120-9,200	[15]
Carboxylic acids	6,200-755,000	[12, 14, 16, 17]	2.4	[13]
Alkyl Esters	0.37-4,450	[14, 16, 17]		
Methoxylated phenolic compounds	28-1,000	[12, 14, 15]	1,200-1,500	[13]
Other substituted Aromatic Compounds	5.0-120,000	[12, 14, 16, 17]	110-3,600	[13, 15]
Polycyclic Aromatic Hydrocarbons (PAHs) and substituted PAHs	5.1-32,000	[12, 14-17]	43.4-355	[13, 15]
Sugar Derivatives	1.4-12,600	[14, 16, 17]		
Coumarins and Flavonoids	0.71-12	[14]		
Phytosteroids	1.7-34.0	[12, 14]		
Resin Acids and Terpenoids	1.7-41,000	[12, 14, 16, 17]	21-430	[15]
Other Compounds	1.2-120	[14]	20-600	[13, 15]
Unresolved complex mixture (UCM)	300-1,130,000	[14, 16, 17]		

Woodsmoke particles are generally smaller than 1 μm, with a peak in the size distribution between 0.15 and 0.4 μm [21, 22]. This is consistent with the observation that most of the mass is formed by condensation processes as the smoke cools. Fine particles in this size range efficiently evade the mucociliary defense system and are deposited in the peripheral airways where they may exert toxic effects. Particles in this size range are not easily removed by gravitational settling and therefore can be transported over long distances [23]. The transport of biomass combustion particles over hundreds of kilometers has been extensively documented [24]. Haze layers with elevated concentrations of carbon monoxide (CO), carbon dioxide (CO_2), ozone (O_3), and nitric oxide (NO) have been observed. During transport, many of the gaseous species are converted to other gases or into particles. The "black carbon" from biomass emissions is now thought to contribute to regional and global climate change in some parts of the world [25, 26].

Although approximately 5-20% of woodsmoke particulate mass consists of elemental carbon, the composition of the organic carbon fraction varies dramatically with the specific fuel being burned and with the combustion conditions. Detailed analysis of organic woodsmoke aerosol

were conducted by Rogge et al. (1998), who measured nearly 200 distinct organic compounds, many of them derivatives of wood polymers and resins [12]. Since profiles of specific PAHs, of concern for their potential carcinogenicity, are likely to be variable, many measurements have focused on benzopyrene (BaP), a probable human carcinogen.

A number of toxic or carcinogenic compounds are present in biomass smoke, including free radicals, polycyclic aromatic hydrocarbons (PAHs) and aldehydes, as shown in Table 1 [9, 13, 27].). Organic extracts of ambient particulate high in woodsmoke are 30 fold more potent than extracts of cigarette smoke condensate in a mouse skin tumor induction assay [28], and are mutagenic in the *S. typhimurium* microsuspension and plate incorporation assays [29]. Few if any reports exist in which the toxicity of smoke from different biomass sources was compared and related to differences in the chemical composition of each smoke type.

A number of chemicals are enriched in woodsmoke relative to other sources of air pollution. Examples include potassium, methoxyphenols, levoglucosan, retene, and specific resin acids (e.g. abietic acid). [12-14, 30-32]. Many of these chemicals have been used either individually, or in multivariate analyses, to quantify woodsmoke emissions [30, 33, 34].

Levoglucosan is an anhydrous sugar derived from the pyrolysis of the major wood polymer cellulose. Levoglucosan is one of the most abundant organic compounds associated with particles in woodsmoke [14, 31]. It is stable in the environment and has been used extensively to estimate woodsmoke levels in ambient PM samples [33, 35, 36]. Levoglucosan is present in other biomass smoke samples including smoke from tobacco, grasses and rice straw [11, 37]. Under conditions in which woodsmoke dominates the biomass smoke contribution to ambient aerosol, however, levoglucosan can be considered a unique tracer for woodsmoke.

Methoxyphenols are a class of chemicals derived from the pyrolysis of the wood polymer lignin. This class of chemicals spans a range of volatilities from relatively volatile (e.g. guaiacol) to exclusively particle associated (e.g. sinapinaldehyde). These chemicals are relatively abundant in woodsmoke, albeit the most abundant compounds are predominantly in the vapor phase [13, 38]. Accurate chemical analysis of the methoxyphenols, however, has proved to be an analytical challenge, and many of the methoxyphenols were found to be chemically reactive – a property that would undermine their suitability as tracers for biomass smoke [39]. Smoke from hardwood versus softwood burning can be distinguished by the relative proportions of substituted guaiacols compared to syringols. Methoxyphenols have been used as woodsmoke tracers in multivariate source apportionment models to determine the proportion of urban fine PM derived from wood burning [33].

The organic chemical composition has been used to distinguish smokes from different biomass fuels. Smoke from combustion of softwood is enriched in vanillyl-type methoxylated phenols whereas hardwood combustion produces both vanillyl and syringyl type methoxylated phenols [16, 17, 33, 38]. Mono- and di-methoxyphenols are also present in small amounts in grass and grain smokes, however the major phenolic compounds in grass smoke a p-coumaryl derivatives [19]. Diterpenoids (e.g. dehydroabietic acid) are abundant in smoke from gymnospems (conifers) compared to angiosperms [40]. Certain chemicals may even be tree-species-smoke specific (e.g. juvabione from balsam fir), although the atmospheric stability of such compounds

and their hence their utility as source-specific exposure markers has not been established [16, 31].

Emission factors for fine particles are highly dependent on the fuel characteristics and burn conditions (smoldering vs. flaming). Similarly, emission factors for specific organic chemicals are influenced by fuel moisture content and burn conditions, although the relationships may not parallel those observed for fine particles [30, 41].

3 TOXICOLOGICAL EFFECTS OF WOODSMOKE EXPOSURE

Although studying the effects of air pollutants directly on humans offers a number of advantages, epidemiologic and controlled clinical studies are often limited by societal concerns, ethical and legal issues, as well as cost. Because of these difficulties, predictive health assessments associated with inhaled woodsmoke need to include information gained from animal exposure studies and, in some cases, *in vitro/ex vivo* assay systems. Furthermore, animal studies also have the potential to help uncover information concerning the mechanisms of toxicity and relative toxicity of different mixtures and sources. The discussion below summarizes the principal published toxicologic studies of woodsmoke.

3.1 *In Vivo* Inhalation Studies

Although woodsmoke can be delivered to animals by a variety of methods including open-chest ventilation, only those studies that employed exposure routes most relevant to the human woodsmoke experience (i.e., nose-only/whole body inhalation in conscious animals) are reviewed herein.

3.1.1 Single Woodsmoke Exposures

3.1.1.1 Acute Lung Injury

Because of increasing interest in the mechanisms of damage in fire-fighting and other high-exposure situations, the majority of toxicologic studies reviewed for this document fell within this category. As the exposure levels used for these studies are greater than those to which the public are exposed, these studies will serve primarily to demonstrate the effects that could occur as a result of lower-level, repeated exposures. In 1982, an inhalation study by Thorning et al. (1982) described the effects of inhaled woodsmoke on pulmonary lung cell injury (as determined by changes in lung morphology) [42]. Two combustion products (i.e., CO and organic aldehydes) were monitored as a measure of woodsmoke exposure. Total aldehyde concentrations in the chamber ranged from 285 – 1,707 ppm for the 25-45 min exposures. Smoke-exposed rabbits exhibited necrotizing tracheobronchial epithelial cell injury that peaked by 24 hr post-exposure. The authors concluded that the acute lung injury and early reactions to smoke damage observed in rabbits closely resembled those pulmonary lesions seen in smoke-injured victims and that these injuries could affect pulmonary host resistance. Particle-adsorbed aldehydes were suggested to play the major role in such injuries.

In another study using rabbits, Loke et al. (1984) investigated the effects of a 60 min inhalation of Douglas fir-generated woodsmoke (mean COHb level = 16.4%) on alveolar macrophage (Mø) response and tracheobronchial morphology [43]. Smoke injury to the proximal tracheal lining cells was severe with major epithelial cell loss observed in exposed animals. In addition, mucociliary escalator dysfunction was also observed. Similar to the cellular alterations observed in long-term cigarette smokers [44], alveolar Mø from woodsmoke-exposed rabbits were flatter and contained less surface ruffling (a marker of cell activation) than their unexposed counterparts. In addition, Mø numbers were increased within the alveoli, suggesting an inflammatory response in smoke-exposed rabbits.

In a study by Fick et al. (1984), rabbits were exposed acutely (i.e., 30 – 120 min) to smoke from the pyrolysis of Douglas fir wood and effects on Mø functional activity were examined

immediately following exposure [45]. At a smoke concentration yielding a COHb level of 7% and no evidence of thermal injury, pulmonary Mø -mediated bacterial phagocytosis and intracellular killing of the Gram-negative bacterial pathogen, *Pseudomonas aeruginosa* was dramatically reduced. Although an inflammatory response was not observed, smoke-exposed animals demonstrated a significantly greater lavageable cell yield than the unexposed controls. This investigation provided the first evidence that low levels of woodsmoke could produce sub-clinical effects and alter lung properties in the absence of any acute lung injury. This well executed toxicologic study employed the most current inhalation procedures of their time and evaluated effects in a dose-response manner based on increasing COHb levels. The authors concluded that inhalation of woodsmoke, at a relatively low level, had the potential to alter host pulmonary immune defense mechanisms in such a way as to lead to an increased susceptibility to infectious lung disease.

Woodsmoke-induced alterations in phagocyte-mediated oxidative stress response and antioxidant status were studied in a rat model designed to simulate an inhalation injury as might be encountered by firefighters and/or burn victims [10]. In this study, rats, either previously scalded or sham-burned, were exposed for approximately 16 minutes to clean air or smoke generated from the burning of western bark wood (i.e., fir and pine); smoke exposure was assessed by measuring COHb levels in exposed hosts (19 ± 2%). At different time points post-exposure, animals were sacrificed and either their lungs lavaged for evaluation of total protein, airway cellular/lung water content, or tissues recovered for measurements of antioxidant enzyme activities and lipid peroxidation (as measured by thiobarbituric acid reactive substances {TBARS}). Similar to that seen in other smoke injury studies, Dubick et al. (2002) observed that acute inhalation of woodsmoke produced areas of tracheal erosion resulting in the loss of epithelium [10]. At 24 hr post-exposure, lipid peroxidation was increased 2-3-fold above control values in the smoke only and burn/smoke exposure groups; TBARS declined after 48 hours in the smoke-only group. Other investigators have also reported oxidative stress as a result of woodsmoke exposure [46-48]. Minor changes in lung antioxidant enzyme activities were also observed in this study. However, in contrast to the dramatic inflammatory response observed in previous studies examining acute smoke-induced lung injury [49, 50], pulmonary immune cell infiltration was not observed. Given that leukocyte infiltration appears to "track" with woodsmoke-induced pulmonary injury, a lack of neutrophil influx was not surprising given the modest level of lung injury produced in the aforementioned study.

In addition to tracheobronchial alterations and changes in immune cell morphology, acute inhalation (30 min) of Douglas fir-generated woodsmoke has been shown to diminish the ventilatory response of exposed guinea pigs. In a study by Wong et al. (1984), animals were exposed for 30 min to increasing woodsmoke concentrations generated by burning different amounts of wood chips [51]. Just prior to exposure and at various time points post-exposure, guinea pigs challenged with CO_2 were placed in a whole-body plethysmograph and lung compliance (elastic properties of the lungs) were measured. Exposure to mid-levels of smoke increased baseline ΔP (change in lung relaxation pressure), but only 0.5 hr after exposure and none of the exposure groups varied significantly from controls after this early time point. Reduction in pulmonary compliance has also been shown in woodsmoke-exposed dogs [52] and in human victims of smoke inhalation [53]. Wong et al. (1984) concluded that acute inhalation of

woodsmoke can alter lung function, but that recovery occurs within several days after exposure [51].

Woodsmoke-induced alterations in airway responsiveness to bronchoconstrictor challenge have also been observed more recently in other studies using guinea pigs. A particulate matter (PM) concentration of 25 mg/m^3 produced airway hyperreactivity in response to challenge with the bronchoconstrictors substance P, capsaicin and prostaglandins [54-58]. Despite the "artificial" exposure route used to deliver woodsmoke in these studies, results provide convincing evidence regarding the adverse effects of woodsmoke on airway responsiveness and demonstrate the need for further studies using more relevant smoke exposure routes.

The same investigative team has carried out studies in rats to evaluate the role of sensory receptors and nerve fibers in pulmonary ventilation following woodsmoke exposure [59-69]. For these studies, rats were exposed to either particle-free (i.e., gas only) or whole woodsmoke effluents. Results from these studies demonstrated that (among other things) woodsmoke-induced slowing of respiration is a reflex resulting from stimulation of bronchopulmonary C-fiber nerve endings (unmeylinated sensory neurons that conduct nerve impulses slowly) induced by the woodsmoke gas phase. Moreover, that increased hydroxyl radical ($^{\cdot}$OH) burdens following smoke exposure was actively involved in evoking the acute irritant effects of woodsmoke. Ho and Kou (2002), using anesthetized rats (exposed via nasal exposure), also demonstrated that woodsmoke exposure increases nasal airway resistance and airway reactivity [70].

In a thorough, well-executed inhalation study by Matthew et al. (2001), exposure of mice to high doses of woodsmoke (COHb level of 50% immediately after exposure) altered pulmonary histology, induced an inflammatory response and increased static lung compliance, lavageable cytokine levels and cell counts [71]. Given that smoke inhalation damage is mediated in part via an up-regulated immune response, increased numbers of lavageable immune cells is consistent with the observed lung pathology. The authors speculated that smoke-associated particulates, with or without adhered noxious gases, were likely responsible for the majority of observed lung pathologies. This notion is supported by the studies of Thomas and Zelikoff (1999) who demonstrated diminished immunotoxicity of inhaled woodsmoke effluents in rats following removal of the particulate smoke phase [72].

Refer to Appendix 1 for a detailed listing of the acute lung injury studies mentioned above.

3.1.2 Repeated and Sub-Chronic Woodsmoke Exposures

3.1.2.1 Pulmonary and Systemic Effects

Inhalation studies utilizing scenarios other than acute single exposure regimes are extremely limited. Long-term investigations are desperately needed that more closely reflect smoke exposures associated with domestic pollution (i.e., home heating and cooking). Lai et al. (1993) examined the hematologic and histopathologic response of rats exposed repeatedly to smoke generated from the combustion of wood dust. Despite the: primitive nature of the exposure system; lack of information concerning smoke concentration and wood type; and, lack of data regarding thermal burn, many of the same smoke-induced pathologies observed in this study

(i.e., epithelial lining cell desquamation, pulmonary edema, and peribronchiolar and perivascular infiltration of polymorphonuclear neutrophils (PMN) have also been reported in the studies of acute smoke exposures [73]. Bronchiolitis, parenchymatous blood vessel congestion, hyperplasia of lymphoid follicles (enlargement of the lymphoid follicles caused by increased cell numbers and mild emphysema were also observed after 15 d of exposure. Although the emphysematic changes remained constant over time, other pulmonary lesions worsened dramatically with increasing exposure duration. In addition, marginal alterations in hemoglobin levels, sedimentation rate, packed cell volume and total and differential leukocyte counts from animals exposed to smoke for 15 d, were observed. Eosinophilia (increased numbers of esosinophils in the blood) was also observed, but only in rats exposed for 30 and 45 d. Results of this study demonstrated that woodsmoke-induced pulmonary lesions are progressive with subsequent smoke exposures. Moreover, given that domestic woodsmoke pollution has been associated with chronic obstructive pulmonary disease (i.e., chronic bronchitis and emphysema) in developing countries and that emphysematic changes were observed in this toxicologic study, the authors concluded that the rat model of acute lung injury may prove useful for assessing the toxicologic impact and human health outcomes of inhaled woodsmoke.

Recent studies from this group have demonstrated that repeated short-term (1 hr/d for 4 d) nose-only inhalation exposure of rats to woodsmoke, generated from the burning of red oak wood (i.e., 750 μg PM/m^3, < 2 ppm CO, 3 ppb NO_x and 1.5 ng total PAH/m^3), inhibited pulmonary clearance of intratracheally instilled *Staphylococcus aureus*. This effect was observed in the absence of any lung pathology, lung cell damage or inflammation [74]. The lack of pulmonary injury and/or inflammation was most likely due to the relatively low woodsmoke concentration used for these studies. The well-described generation system [75] (**Figure 1**) produced a smoke concentration reflective of that found indoors during operation of a poorly operating fireplace or non-EPA certified wood burning device.

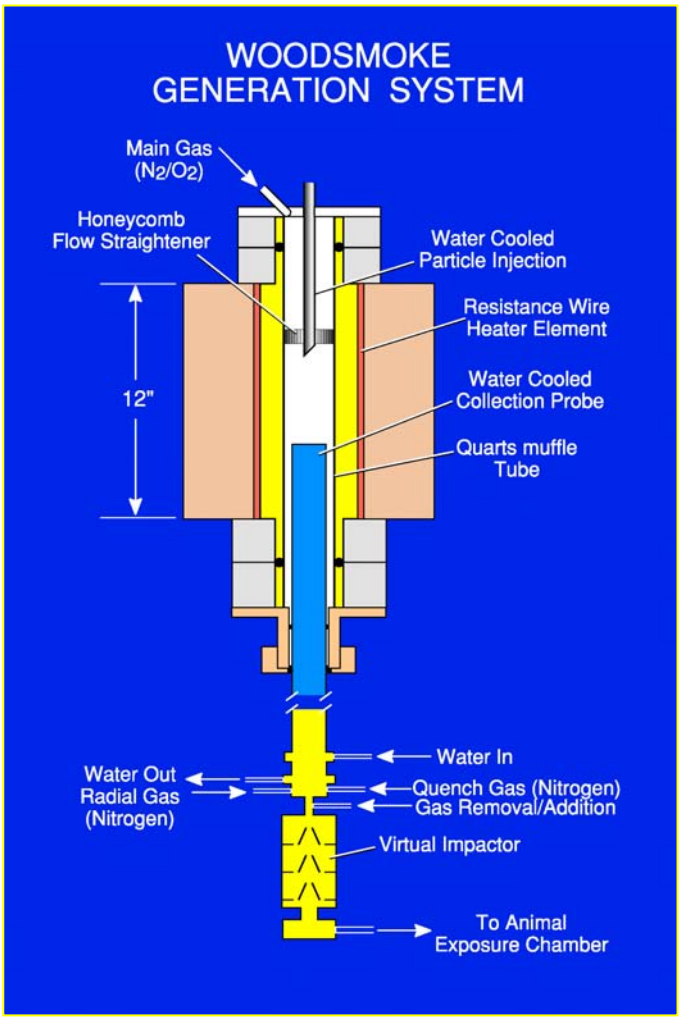

Figure 1: Woodsmoke generation system used to generate smoke from the burning of red oak wood (Source: [75]).

In these studies, suppressed bacterial clearance began as early as 3 hours post-exposure and persisted for almost two weeks [72]. Interestingly, similar dramatic effects on lung clearance were not observed for rats exposed to particle-free woodsmoke effluents. This response demonstrates the importance of the woodsmoke -associated particulates for bringing about the observed time-related effects on pulmonary host resistance. In the same study, woodsmoke exposure also suppressed pulmonary Mø-mediated superoxide anion (O_2^-) production, a reactive oxygen species (an oxygen molecule with an additional electron that makes it unstable) critical for the intracellular killing of *Staphylococcus aureus*. The authors suggested that reduced production of O_2^- might in part be responsible for the observed woodsmoke-induced decrease in pulmonary host resistance against bacterial pathogens. Taken together with results from earlier studies, they concluded that short-term inhalation of woodsmoke can compromise pulmonary immune defense mechanisms. Moreover, that the lung Mø (if you use the abbreviations, you will save space- Mø is previously defined-see above) represents a sensitive target for the toxic effects of inhaled woodsmoke.

A study of health effects arising from sub-chronic exposure to woodsmoke has recently been published [76]. In this investigation, rats were exposed to woodsmoke (1 or 10 mg/m^3) generated from the burning of *pinus edulis* (i.e., pinon wood) for 4 or 12 wk. In the absence of any effects on lung-associated T- lymphocyte proliferation or lavageable cytokine levels, smoke inhalation produced a small (non-statistically significant) exposure-related reduction in lung function, as well as mild chronic inflammation and squamous cell metaplasia (this is a pre-cancerous condition) in the larynx of exposed rats. Themucous cell metaplasia observed after 30 d of exposure was transient and resolved after 90 days. The severity of alveolar Mø hyperplasia and pigmentation increased with smoke concentration and length of exposure. While this laboratory disagrees with the Authors conclusion regarding the insensitivity of the immune system for assessing the health impacts of inhaled woodsmoke, the study was well executed, demonstrated a dose response relationship for some endpoints, employed a well-described generation/exposure system, and, incorporated extensive chemical characterization of the woodsmoke effluents.

3.1.2.2 Lung Cancer

A field study was carried out in which mice and rats were placed for 15 or 19 months, respectively in an indoor environment to inhale either air (without burn products) or smoke generated from burning wood or coal [77]. Burning was carried out in round shallow pits of individual rooms This and the quantity of burned material (241 kg/mo), pattern of burning (12 hr smoke: 12 hr no smoke), and structure of the rooms matched those of villagers in Xuan County, China. Although smoke generated from both coal and wood contained similar TSP levels (i.e., 14.4 vs. 14.9 mg/m^3, respectively), BaP concentration in the wood exposure room was approximately 47 times higher than that measured in the air control environment, although substantially less than in the coal-using rooms. In the woodsmoke room, CO, SO_2 and H_2SO_4 levels were 80 mg/m^3, 0.05 mg/m^3 and 0.27 mg/m^3, respectively. Following exposure, animals were immediately sacrificed and the incidence of non-malignant and malignant lung tumors evaluated. Tumors were histologically differentiated into several groups including adenomas, adenocarcinomas (AC), adenosquamous carcinomas (ASC) and squamous cell carcinomas (SCC). Control mice demonstrated a lung cancer incidence of 17%, while that for mice exposed long-term to either wood or coal smoke was 45.8% and 89.5%, respectively. Although all lung cancer types were observed in coal smoke exposed mice, those exposed to woodsmoke demonstrated far fewer SCC and ASCs. Lung cancer incidence for rats exposed to ambient air or smoke from burning wood or coal was 0, 0 and 67.2%, respectively.

The authors concluded that woodsmoke proved to be only a weak carcinogen compared to coal smoke. Differential effects between wood and coal have also been observed in a mouse skin tumorigenicity study that compared the effects of organic extracts from smoky coal and wood combustion ([78]). In this case, particle extracts from smoky coal combustion proved to be a potent complete carcinogen, whereas that from woodsmoke proved relatively inactive in this capacity.

Refer to Appendix 2 for a detailed listing of the relevant repeated and sub-chronic *in vivo* woodsmoke exposure studie, including those studies mentioned above.

3.2 *Ex Vivo*/*In Vitro* Woodsmoke Exposure Studies

Although toxicologic studies using routes of exposure other than inhalation were not the primary focus of this review, several *ex vivo/in vitro* studies are briefly discussed inasmuch as they might contribute to a better understanding of the potential health impacts of woodsmoke. Bhattacharyya et al. (1998) examined the effects of pine woodsmoke exposure for 5 – 20 minutes on rabbit tracheal explants [79]. Exposure of explants for 20 minutes led to degeneration of the mucociliary epithelial sheath, although shorter exposures (i.e., 10 minutes) retained tissue integrity, but altered epithelial morphology. Similar woodsmoke-associated pathologies have been observed *in vivo* following acute inhalation exposure.

Exposure of cultured eye lenses to Woodsmoke condensates for 10 minutes resulted in woodsmoke metabolite-induced opacification ([80]). Histological analyses of smoke-exposed lenses revealed distinct morphological changes including hyperplasia, hypertrophy and multi-layering of epithelial cells. The authors concluded that exposure to woodsmoke could contribute to progressive eye lens opacification.

Leonard et al. (2000) examined the effects pine- and Douglas fir-generated liquefied woodsmoke on cultured mouse Mø free radical generation, DNA damage, NFκB activation (signal transduction factor) and tumor necrosis factor-α release and demonstrated that exposure in combination with hydrogen peroxide (H_2O_2) resulted in hydroxyl radical ($^{\cdot}OH$)-induced DNA damage, and that co-exposure to a $^{\cdot}OH$ radical scavenger or a metal chelator inhibited the observed genotoxicity [9]. The authors concluded that free radicals generated by woodsmoke through the reaction of iron with H_2O_2 could produce genetic and cellular damage. Moreover, such free radicals could also play a role in the development of woodsmoke-induced pulmonary fibrosis.

A number of genetic toxicology studies have evaluated the mutagenicity of woodsmoke condensates [81-84]. In all cases, woodsmoke extracts were mutagenic in bacterial systems. A number of factors including heating conditions, type of wood-burning device, wood origin, and PAH concentration seemed to play an important role in overall mutagenic activity.

Refer to Appendix 3 for a detailed list of the *ex vivo* and *in vitro* woodsmoke exposure studies mentioned above.

3.3 Toxicology Summary

These toxicology studies provide biologic plausibility for the epidemiologic evidence suggesting that exposure to woodsmoke emissions adversely affects human health. In addition, results of these animal studies contribute to a better understanding of the possible mechanisms(s) by which woodsmoke, and its associated particulates, may act to bring about increased pulmonary morbidity in exposed individuals.

It appears clear from the toxicologic studies that short-term inhalation of woodsmoke can compromise pulmonary immune defense mechanisms important for maintaining host resistance against pulmonary infections. Moreover, that lung macrophage, the primary defense of the deep lung that provides a link between the nonspecific and specific defense systems of the respiratory

tract, is the likely target for woodsmoke-induced immunotoxicity. Whether similar effects occur following long-term exposure, a scenario more reflective of the human situation, remains to be seen. These studies lend support to the notion that inhaled woodsmoke contributes to the increased incidence of infectious respiratory disease reported in children living in developing nations and at or near homes heated by woodburning devices.

Making interspecies comparisons between humans and other mammalian species is complicated and needs to be approached with caution due to such things as differences in breathing rates, metabolism and particle deposition. However, results of these studies lend support to the applicability of laboratory animals as a model to predict woodsmoke-induced alterations in pulmonary health in humans. The observed similarities/differences in the effects of woodsmoke exposure between humans and laboratory models underscore the importance of comparative studies as a basis for extrapolation modeling. While more toxicologic studies are needed to determine the effects of long-term exposure (at environmentally-relevant levels, and the particular woodsmoke constituents that may be responsible for the observed toxicities, it appears clear that inhalation of woodsmoke can have a significant impact upon pulmonary homeostasis and/or exacerbations of ongoing lung disease processes.

4 BIOMASS SMOKE EXPOSURE ASSESSMENT

Accurate assessments of population and individual exposures to biomass smoke are critical for epidemiological studies into the health effects of biomass smoke and for targeting regulation efficiently. Exposure assessment for biomass smoke is inherently challenging, however, because smoke concentrations typically exhibit greater temporal and spatial variability than is true for other major sources of PM; highest smoke concentrations typically occur in rural areas which have limited infrastructure for monitoring smoke levels; and, for much of the world, the majority of smoke exposures comes from indoor sources (i.e. residential stoves and fireplaces). Nevertheless, it is clear that biomass exposure is important in many developed countries as well, including Canada and the US. In the US alone, more than 100,000 people annually are exposed to elevated biomass smoke levels from wildfires, prescribed burns and agricultural field burning. Another 70,000 – 80,000 people involved in wildland fire fighting also receive substantial occupational exposure to woodsmoke [85]. Many millions more are subjected to indoor and neighborhood biomass pollution from household use of fireplaces and wood heating stoves during cold periods.

Researchers have used a variety of approaches to *estimate* human exposure to biomass smoke particulate, ranging from simple dichotomous exposure variables to sophisticated continuous exposure measurements and microenvironmental exposure modeling. Unfortunately, many of the techniques used to estimate biomass smoke exposure may not accurately capture the full range of personal exposures to biomass smoke, and exposure misclassification is likely to be important. Some of the limitations include: lack of specificity for biomass smoke, poor correlation between personal and central monitor exposure metrics, inability to account for modifying effect of personal protective equipment or behavior on personal exposures, and failure to capture differences in personal exposure versus absorbed dose.

In this section, the various methods that have been used to assess exposure in epidemiological studies investigating health effects of biomass smoke are critically reviewed, focusing on those used in developed countries. The literature cited is intended to be illustrative of the range of biomass smoke exposure assessment methodologies, rather than being a comprehensive review of all investigations of woodsmoke exposures. Advantages and limitations of the various exposure assessment methods are discussed, and where appropriate suggestions made about the types of exposure situation where each method may be reliably employed.

A variety of metrics have been used to estimate exposure to biomass smoke, as summarized in Table 5.. Most epidemiological studies have used indirect measures of exposure to biomass smoke: examples include occupation, type of fuel used for domestic cooking/heating, housing characteristics. Some studies have deduced biomass smoke exposures based on physical measurements performed on samples collected at regional monitoring sites, outside residences, or at home indoor locations. The level of biomass smoke may be estimated from discrete chemicals (for example, levoglucosan mass), or from a source apportionment model using data for a number of different chemicals. In studies that focused specifically on exposure assessment, the tendency has been to use personal monitors that collect either integrated or continuous data.

Table 5: Summary of Published Studies That Assessed Biomass Smoke Exposures, Grouped by Exposure Assessment Method.

Exposure assessment metric	Population	Health Endpoints measured	Reference
Exposure assessment based on occupation:			
Dichotomous categorization: cross-shift and cross-seasonal	Forest Firefighters	Measured changes in lung function	[86]
Dichotomous categorization: hours spent firefighting	Forest Firefighters	N/A	[91]
Categorization based on years of service, age at hire	Structural Firefighters	Mortality	[92]
Firefighters vs. non-occupationally exposed controls	Structural Firefighters	Sister chromatid exchange frequency	[90]
Dichotomous categorization: cross-shift	Structural Firefighters	Spirometry, specific serum proteins	[89]
Firefighters vs. non-occupationally exposed controls	Structural Firefighters	Spirometry, specific serum proteins	[94]
Dichotomous categorization: cross-shift	Charcoal production workers	Spirometry	[211]
Exposure assessment based on questionnaire:			
Dichotomous categorization: Homes with and without woodstove, Awareness of fumes from combustion device	Adult/Child pairs in Quebec city, Canada	Respiratory symptoms and respiratory illnesses	[100]
Categorized by years spent using wood for cooking	Honduran women	Cervical neoplasia	[103]
Dichotomous categorization: Homes with and without woodstove, participated in smoke curing of foods	Aboriginal women, NWT, Canada	Respiratory symptoms	[101]
Dichotomous categorization: Homes with and without woodstove	Indian population	Asthma prevalence	[99]
Frequency of use of domestic WS	Asthmatic adults	**Asthma severity & healthcare utilization for asthma**	[102]
Exposure assessment based on physical measurements:			

Integrated PM10 in counties downwind of Montana wildfires	Wildfire smoke-exposed population	Respiratory and circulatory hospital admissions	[106]	
Integrated PM10 on tribal reservation	Wildfire smoke-exposed population	Visits to health clinic for respiratory symptoms	[146]	
Integrated regional PM10 before and after wildfire episodes	Army recruits	Peripheral white blood cell counts	[212]	
Regional monitoring: Integrated PM, CO, formaldehyde, acrolein, benzene	Population of Theobroma Village, Rhondia, Brazil	N/A	**[107]**	
Indoor monitoring: Continuous PM, CO, size distributions	Villages in rural Costa Rica	N/A	[111]	
Indoor monitoring: Continuous and Integrated PM	Villages in rural Mexico	N/A	[110]	
Personal monitoring: PM, CO, PAHs, Si, aldehydes	Forest Firefighters	N/A	[96]	
Personal monitoring: PM, CO, formaldehyde, acrolein	Forest Firefighters	Spirometry	[97]	
Personal monitoring: PM, CO, formaldehyde, acrolein, benzene	Forest Firefighters	N/A	[93]	
Personal monitoring: PM, volatile organics, PAHs, aldehydes, CO, HCN, NO_2, SO_2	Structural Firefighters	N/A	[98]	
Indoor PM10, NO2, CO and formaldehyde	Adult/Child pairs in Quebec city, Canada	Respiratory symptoms and respiratory illnesses	[100]	
Measured methoxyphenols and levoglucosan in personal filter samples	Adults and children in Seattle, WA	Spirometry, exhaled NO, heart rate variability, serum markers of inflammation	[109, 114]	
Personal PMF based source apportionment derived from measurements of EC, OC and trace metals in integrated personal samples. Verified with personal levoglucosan measurements	Adults and children in Seattle, WA	Spirometry, exhaled NO, heart rate variability, serum markers of inflammation	[36, 109]	
Micro environmental model based on continuous PM monitoring	Rural communities in Kenya	Respiratory infection	[108]	

Micro environmental model using source apportioned indoor, residential-outdoor and ambient PM concentrations	Population of Boise, Idaho	Estimated cancer risk	[28]
Exposure assessment based on biomarkers:			
Carboxyhemoglobin	US population (NHANES II)	Body mass, triceps skinfold thickness	[121]
Carboxyhemoglobin	Structural Firefighters	Spirometry, specific serum proteins	[89]
Urinary PAH metabolites (1-hydroxypyrene)	Structural Firefighters	N/A	[88]
Urinary PAH metabolites (1-hydroxypyrene)	Structural Firefighters	N/A	[87]
PAH-DNA adducts in peripheral blood lymphocytes	Structural Firefighters	Sister chromatid exchange frequency	[90]
PAH-DNA adducts in peripheral blood lymphocytes	Forest Firefighters	N/A	[91]
Urinary methoxyphenols	Campfire exposure, 1 adult	N/A	[119]

4.1 Exposure Assessment Based on Categorical Variables

Several epidemiological studies have assessed woodsmoke exposure in wildland and structural firefighters. The simplest approach has been to categorize exposure amongst firefighters as a dichotomous variable e.g. pre and post-shift, pre-and post season [86-89], or to perform case-control studies with non-occupationally exposed controls [90]. More refined exposure metrics designed to better capture exposure intensity include hours spent firefighting [91], years of service [92], visual assessment of smoke intensity [93, 94] and job classification [93, 94]. Potential confounders (e.g. cigarette smoking) were typically controlled for as dichotomous variables also [88]. Notably, authors that did collect personal exposure measurement on both wildland and structural firefighters observed substantial (up to100-fold) variation in measured PM exposures between subjects fighting the same fire – both within and between job classification [93, 95-98]. Consequently, the questionnaire approach is poorly suited to capturing short-term spatial/temporal variability in exposure intensity. One might argue that this short-term inter-subject variability in smoke exposures averages out over longer time periods, and this limitation to the questionnaire approach to exposure assessment would therefore be less of an issue in studies concerned with chronic health effects resulting from long term exposures to biomass smoke. Few data exist, however, that address the extent to which short-term measurements of smoke exposures are reflective of longer term smoke exposures.

Categorical exposure variables derived from questionnaire responses have been used to study the health consequences of exposure to indoor-generated biomass smoke. [99-103]. For example, Eisner et al assessed woodsmoke exposure amongst asthmatics in Northern California by means of telephone interviews. Subjects were asked about the frequency with which they used a wood stove or fireplace to heat their living space. Exposure was categorized as “no woodstove use”, “lower use” (1-4 times/week) and “higher use” (≥ 5 times/week). The study concluded that “there was no clear evidence that domestic woodsmoke exposure had adverse effects on adults with asthma”, but the reliability of the exposure metrics used in this study was not evaluated. Levesque et al noted little difference in the frequency distribution of PM_{10} for houses that did have a woodstove, compared to houses that did not [100]. The questionnaire approach is relatively inexpensive, can be applied to analysis of opportunistic data, and has frequently been sufficiently effective to demonstrate adverse health consequences of biomass smoke exposures. The questionnaire approach has limitations, however, including: potential susceptibility to recall bias, failure to adequately capture exposure intensity, and failure to adequately capture the full range of biomass smoke exposures due to the spatial and temporal variability in biomass smoke concentrations.

4.2 Exposure Assessment Based on Physical Measurements

Woodsmoke exposure has also been assessed via collection and analysis of physico-chemical data. These measurements cover a wide range of sophistication, from regional measurements of index pollutants (PM, CO) integrated over time (24 hr or annual average) through multi-location and personal sampling, continuous monitoring, use of biomass-specific tracers, development of mircoenvironmental models and source-apportioned exposures derived from multivariate receptor models.

Comprehensive exposure monitoring affords the opportunity to minimize exposure misclassification. Thus, although several questionnaire-based studies have used the presence or use of a woodstove in the home as their indictor of woodsmoke exposure [99, 101], measurements in homes with and without woodstoves were equivocal whether woodstove use increases indoor cumulative-exposures and dose [100]. A variety of metrics have been used to assess exposures to biomass smoke including measurements of PM mass, carbon monoxide (CO) organic carbon (OC) and potassium [30, 104, 105]. Several organic tracers have also been used, including methyl chloride, PAHs, levoglucosan and methoxylated phenolic compounds (methoxyphenols) [30, 33, 34].

To date, most exposure assessment studies have relied on integrated PM measurements (e.g. 8hr workshift, 24-hr sample) [93, 100, 106, 107]. The samples have been collected at regional monitoring sites, outside residential locations, inside homes and personal samples. Collection of integrated samples facilitates subsequent chemical analysis for biomass smoke and other source markers to facilitate quantification of the biomass smoke specific contribution to PM. Integrated samples collected at fixed monitoring locations often fail to adequately reflect personal exposures due to the inherent spatial and temporal variability in woodsmoke concentrations. Collection of integrated personal samples resolves the issue of exposure estimation, and clearly highlights the variability in individual exposures amongst subjects who would typically be grouped into a single category if categorical methods of exposure assessment were used [95]. However, the aggregation of exposure data over space and time that is inherent with integrated exposure metrics ignores spatial and temporal patterns of exposure, and limits a predictive assessment of the impacts of various intervention strategies on individual exposure [108].

An alternative approach has been to use monitors capable of collecting continuous or semi-continuous air pollution data [108-111]. Continuous data provides important temporal information about air pollutant sources. The temporal data facilitates estimation of infiltration from recursive models [112] and aids development of microenvironmental exposure models [108]. The most commonly used continuous monitors are devices that use light scattering (nephelometry) to monitor fine particulate levels. A limitation of these devices is that the light scattering coefficient that relates optical density to particle mass is dependent on the physico-chemical properties of the aerosol being measured (size distribution and absorptivity of the particles). The light-scattering devices have to be calibrated against gravimetric methods for each type of aerosol being studied, and as the aerosol properties change (for example, due to changing source contributions), the calibration relationship will change also.

Continuous monitors that respond to other components of biomass smoke are available and may be used for exposure assessment of biomass smoke also. Examples include CO monitors, photoionization monitors that respond to particle bound PAHs, and aethelometers (semi-continuous monitors that respond to black carbon or elemental carbon). These monitors are useful to the extent that in the specific samples analyzed, the chemicals of interest are uniquely or predominantly derived from biomass smoke. Reinhardt et al. (2000) demonstrated that PM exposure in wildland firefighters was correlated with CO exposure, and developed regression equations to estimate PM exposure based upon CO measurements recorded with a continuous CO datalogger [93]. While CO appeared to be a good surrogate for PM exposure at project wildfires (r^2=0.79), the correlation was weaker at initial attack wildfires (r^2=0.46), and the

correlation may be expected to be weaker still in situations where multiple sources of CO and PM exist (e.g. urban settings) and where lower biomass smoke levels are present.

In a recent report measurements of levoglucosan in filter samples were used to validate the assignment of the woodsmoke feature in a PMF based source apportionment in indoor, outdoor and personal PM samples [105]. It has been reported that typical cooking conditions do not generate the temperatures required to produce appreciable amounts of levoglucosan ([113]). In a Seattle panel study, however, it was observed that for several residences, the 24hr average levoglucosan concentrations were higher indoors than outdoors in a number of samples [114]. Levoglucosan is formed in substantial amounts during the thermal alteration of starches and sugars (caramelization) [115], so the possibility that residential cooking may produce levoglucosan-containing aerosols cannot be discounted. Therefore, the possibility remains that indoor sources of levoglucosan exist that would confound its use as a tracer for ambient biomass smoke penetrating indoors.

Novel multivariate statistical methods including UNMIX, positive matrix factorization (PMF) and the multilinear engine (ME) are being applied to multivariate air pollution data to identify specific sources and quantify source contributions [36, 105, 116]. PMF and ME are statistical models that adopts the least-squares approach to solve the factor analysis problem without requiring source profiles as the input variable [117]. These approaches promise to provide a powerful approach for assessing the health effects from specific air pollutant sources in a complex mixture[4] [118].

4.3 Biomarkers of Woodsmoke Exposure

Obtaining accurate measures of personal exposure and, more importantly, absorbed dose, for particulate air pollution is inherently difficult. This is due to the substantial spatial and temporal variation in pollutant levels, coupled with people's habit of constantly moving between different microenvironments. Thus, traditional fixed-site monitors fail to capture the full variability in exposures experienced by individuals. Although active personal monitors are effective in accurately monitoring personal exposures, it is impractical and expensive to implement active personal monitoring on a large scale. Furthermore, external personal monitors fail to account for substantial differences in breathing volume, and hence inhaled dose, due to physical exertion. An alternative approach to exposure assessment, which addresses many of the limitations noted above, is biomonitoring. Biomonitoring of exposure biomass smoke involves measurement of biomass smoke-derived chemicals in biological media such as human urine, blood, or hair, and the chemical so monitored is called a biomarker.

Ideally, a biomarker for biomass smoke should possess the following characteristics:
(1) it should be uniquely derived from the biomass smoke;

[4] An important caveat, however, is that the error structure of the predicted exposures from the multivariate models is complicated and not clearly defined. Initial studies using PMF and Monte Carlo simulation indicate that the naïve use of imputed source-specific exposure estimates in health effects models may cause substantial bias in subsequent health effect estimates. This bias is amplified by the rotational ambiguity of the imputed source-specific exposure estimates (Liu, H. and T. Lumley, 2004) 118.Liu, H. and T. Lumley, *On source apportionment and health effects models*, Univ. Washington, Editor. 2004. One possible strategy to minimize the rotational ambiguity involves incorporating known source-specific tracers into the speciated data.

(2) it should be a relatively abundant constituent of biomass smoke, such that ambient exposure levels generate sufficiently high biomarker levels to be measured reproducibly;
(3) the parent marker should be chemically stable in the environment, and the compound (or its metabolites) should be chemically stable in biological samples; and,
(4) the excretion kinetics in the media of interest (e.g. urine, blood etc) should be suited to the exposures and/or health endpoints of interest.

Several classes of chemicals have been proposed as biomarkers for smoke exposures, including: carboxyhemoglobin, PAH metabolites, levoglucosan and methoxyphenols [88-91, 119, 120].

Exposure to CO present in biomass smoke leads to formation of carboxyhemoglobin. Elevated carboxyhemoglobin have been measured in active and passive cigarette smokers [121], and in tunnel workers and loggers (references in [89]). Carboxyhemoglobin levels correlated with decrements in lung function (FEV1) amongst structural firefighters [89]. Carboxyhemoglobin may be a useful biomarker of smoke exposure where biomass smoke is the dominant source of CO, however in many situations this marker will be confounded by cigarette smoking and non-biomass combustion emissions.

The polycyclic aromatic compound pyrene is abundant in biomass smoke. Upon uptake by humans, pyrene is extensively modified to 1-hydroxypyrene, and excreted in the urine and faeces. Feunekes et al. measured 1-hydroxypyrene in urine from firefighters [88]. They reported a positive association between urinary 1-hydroxypyrene and smoke exposure. Liou et al. found higher PAH-adducts in peripheral blood cells in wildland firefighters exposed to woodsmoke vs. non-exposed controls [90]. In contrast, Rothman et al. reported no association between PAH-DNA adducts in peripheral blood from wildland firefighters, and smoke exposures [91]. PAH-DNA adducts were, however, associated with charbroiled beef consumption. In both studies, the subject's exposure to woodsmoke was estimated based upon number of hours spent fighting fires. PAHs are by no means specific to biomass smoke; they are a component of incomplete combustion and are present in a variety of PM sources including vehicle exhaust, gas and coal combustion and cooking fumes [40, 122, 123]. Furthermore, for non-occupationally exposed non-smokers, the major portion of PAH dose is taken up through the diet [124, 125]. Therefore, PAH biomarkers in urine or blood are only likely to be significantly associated with biomass smoke exposures when the exposures are very high.

There is one report describing measurement of levoglucosan in human urine [120]. Dorland et al. used an approach combining TLC and GC/MS, and reported 0 to 0.8 mg levoglucosan per mL of urine [120]. Since levoglucosan is one of the most abundant particle phase organic compounds in biomass smoke, this report suggests that measurement of levoglucosan levels in urine samples may be a viable method of assessing biomass smoke exposure and dose. The finding of Dorland et al, however, should be replicated using modern techniques combining high-resolution chromatography and mass spectrometry since the analytical methods used in the original study have limited specificity. The upper range of the urinary levoglucosan levels reported by Dorland et al are higher than would be achieved from inhalational exposure to woodsmoke at ambient levels, and the possibility exists of a substantial dietary contribution of levoglucosan from caramelized sugars [115].

Another group has described the use of methoxyphenols as potential biomarkers of biomass smoke exposure [119]. Sensitive and specific methods based on GC/MS analysis for determination of approximately 12 methoxyphenols in human urine have been developed. Multiple methoxyphenols are present in the urine of individuals with no known elevated exposure to biomass smoke, and a substantial increase in urinary methoxyphenol excretion was reported subsequent to inhalation of woodsmoke from a campfire [119]. It was also noted that ingestion of food items containing woodsmoke flavoring (e.g. smoked salmon) caused a substantial increase in urinary methoxyphenol excretion [119]. The utility of methoxyphenols as a biomarker for woodsmoke exposure at ambient levels was evaluated in a panel study in Seattle WA. Multiple methoxyphenols were detected in all urine samples, and a dynamic range up to 1000-fold between lowest and highest reported concentrations was observed [126]. No obvious association between personal exposure to woodsmoke-derived PM and urinary methoxyphenols was observed, however. In this situation, it appears that dietary sources of methoxyphenols overwhelm the contribution from inhalational exposures to woodsmoke. Thus while urinary methoxyphenols may well have a role in assessment of biomass smoke exposures in high exposure situations (e.g. occupational or developing-world exposures), it seems unlikely that this specific biomarker will be useful for monitoring population exposures to biomass smoke levels below about 30 $\mu g/m^3$ $PM_{2.5.}$

5 FOREST FIRE AND AGRICULTURAL BURNING: EXPOSURE AND HEALTH STUDIES

In contrast to the large amount of information relating particles in urban air to health impacts, there are only a limited number of studies directly evaluating the community health impacts of air pollution resulting from the burning of vegetation. Several reviews have discussed the health impacts and pollutants associated with woodsmoke air pollution [104, 127-129]. Although the emphasis of these reviews was on community exposures resulting from burning of wood in fireplaces and wood stoves, many of the conclusions are relevant to the broader understanding of vegetation fire air pollution. The World Heath Organization has published a document describing Health Guidelines for Vegetation Fire Events[5], which also contains a review of evidence linking air pollution from vegetation fires with human health effects. Specific information relating agricultural and forest/brush burning with human health effects is summarized below and presented in Table 6.

Smoke from African and Brazilian savanna fires contained substantial quantities of fine particles [23, 130] Mass concentrations ranged from 30 $\mu g/m^3$ in areas not affected by biomass burning to 300 $\mu g/m^3$ in large areas (2 million km^2) with intense burning. Additional studies of fine particle (<2 μm) composition associated with biomass burning in the Amazon Basin was reported by Artaxo et al. (1994) who found 24-hour average PM_{10} and $PM_{2.5}$ mass concentrations as high as 700 and 400 $\mu g/m^3$, respectively [131]. In one of the few measurements of rural community air pollution associated with large tropical forest fires, Reinhardt and Ottmar measured formaldehyde, acrolein, benzene, CO and respirable particulate matter ($PM_{3.5}$) in Rondonia, Brazil during the peak of the 1996 biomass burning season [107]. Of the species measured, respirable particle levels were elevated 5-10 times above background levels, with mean levels of 190 $\mu g/m^3$ and levels as high as 250 $\mu g/m^3$ measured during several of the 12-hour sampling periods. The mean CO level was 4 ppm, which is similar to levels measured in moderately polluted urban areas and below the level expected to be associated with acute health impacts. Benzene levels (11 $\mu g/m^3$ average) were higher than those measured in other rural areas and were comparable to those measured in cities.

Measurements from Southeast Asia also indicate that particles are the main air pollutant elevated during periods of vegetation fire-related air pollution [132]. For example, during a 2-3 month period in 1994, PM_{10} levels up to 409 $\mu g/m^3$ were recorded in Kuala Lumpur [133], and levels ranged from 36 – 285 $\mu g/m^3$ in Singapore [134]. In 1997, PM_{10} levels as high as 930 and 421 $\mu g/m^3$ were measured in Sarawak and Kuala Lumpur, respectively, while levels in Singapore and southern Thailand were somewhat lower [135]. Closer to the fire source in Indonesia, PM_{10} concentrations as high as 1,800 $\mu g/m^3$ were measured [136]. In February - May 1998 a more limited vegetation fire episode affected regions of Borneo. In Brunei, PM_{10} levels as high as 440 $\mu g/m^3$ were measured during this period [132].

[5] www.who.int/docstore/peh/Vegetation_fires/Health_Guidelines_final_3.pdf

Table 6: Summary of Selected Epidemiological Studies of Large-Scale Vegetation Fires

Population	Study design	Endpoints measured	Results	Reference
All ages	Longitudinal	Emergency room visits	Increased respiratory visits in communities exposed to fire smoke	[144]
All ages	Longitudinal	Emergency room visits, hospital admissions	Increased emergency room visits and hospital admissions for asthma and bronchitis during fire period relative to same period in previous year	[145]
Adults with airways obstruction	Prevalence	Symptoms	42% of population reported increased or worsened symptoms during episode of exposure to agricultural burning emissions. 20% reported breathing trouble	[213]
Adults with COPD	Longitudinal	Symptoms	Significant increase in symptom index (dyspnea, cough, chest tightness, wheezing, sputum production) on two days of elevated pm2.5 (65 ug/m3) relative to control days (14 ug/m3). Days of elevated pm attributed to fire smoke by satellite imaging	[147]
All ages	Longitudinal	Acute respiratory distress hospital visits	Increase in acute respiratory distress inhalation therapy visits associated with indirect measure (sedimentation) of air pollution during sugar cane burning season in Brazil	[165]
Children	Longitudinal	Respiratory hospital admissions	Increased pediatric respiratory hospital admissions associated with increased biomass smoke markers (potassium and black carbon) during sugar cane burning season in Brazil	[164]
Adults	Cross-sectional	Asthma medication, lung function, asthmatic and other respiratory symptoms	Increased prevalence of respiratory symptoms and various asthma indicators, decreased lung function post rice stubble burning period relative to period prior to burning in 3 communities in Iran	[160]

5.1 Wildland Fire Fighters

In general, wildland fire-fighters experience greater exposure from forest fire smoke than members of the general public. Patterns of exposure can be intense in initial fire-suppression efforts or in situations involving thermal inversions. Workshifts are frequently 12 to 18 hours and can last for more than 24 hours. In large fires, prolonged work shifts can last for many days. In wildland firefighting, it is not feasible to use a self-contained breathing apparatus; often the only respiratory protection used is a cotton bandana tied over the nose and mouth. Moreover, many of the tasks in wildland firefighting are physically demanding, and require elevated pulmonary ventilation rates, which can result in substantial doses of various smoke constituents to the respiratory tract. Off-shift smoke exposures may occur as well, depending on the location of the base camp (where firefighters eat and sleep) in relation to the fire and the prevailing meteorology. With the intensity of smoke exposures, it is not surprising that respiratory problems accounted for about 40% of all medical visits made by wildland firefighters during the Yellowstone firestorm of 1988 [137].

There have been several investigations of both exposures and health impacts of smoke exposure among wildland firefighters. Exposure assessment of wildland firefighters can represent a major logistical challenge, considering that the work often takes place in remote locations, on steep terrain, and may involve extreme physical exertion. In addition, woodsmoke contains thousands of chemical substances, both gases and particles; thus, exposure assessment must of necessity be limited to relatively few substances. By extension, the few health studies that have been undertaken have not involved concurrent exposure assessment, but have focused on cross-shift or cross-seasonal respiratory effects.

Reinhardt and Ottmar (2000) undertook an exposure assessment of breathing-zone levels of acrolein, benzene, carbon dioxide, carbon monoxide, formaldehyde and respirable particulate matter among firefighters at 21 wildfires in California between 1992 and 1995 [95]. Interestingly, exposures to the gases were generally well below time-weighted average occupational health standards. However, some of the fires resulted in high-level peak exposures to heavy smoke. Respirable particle (PM3.5) exposures on multi-day fires averaged 0.72 mg/m^3 on the fireline, and 0.5 mg/m^3 over the work shift, with peak concentrations of 2.3 and 2.93 mg/m^3. The corresponding exposures to carbon monoxide were 4.0 and 2.8 ppm, with peak (2-hr TWA) exposures of 38.8 ppm and 30.5 ppm. The particle concentrations are about 10 to 30 times higher than 24-hr average ambient air quality standards for fine particulate matter (currently 65 $\mu g/m^3$ in the U.S.).

Materna et al. (1992) also found extremely high particle exposures among California wildland firefighters during the 1987 – 1989 fire seasons [96]. Table 7 presents their data on particulate exposures. These investigators sampled for 12 PAHs and found all below 1 $\mu g/m^3$. The highest CO levels were associated with tending gasoline-powered pumping engines rather than from smoke exposure, per se. An aldehyde screen detected formaldehyde, acrolein, furfural, and acetaldehyde. Most levels were well below occupational exposure limits; however, formaldehyde (which was detected in all samples) in several instances exceeded such limits (max TWA [226 min] = 0.42 mg/m^3). In general, these studies demonstrate that, of the various

measured constituents of smoke, particulate matter tends to be the most consistently elevated during wildland firefighting in relation to health-based exposure standards.

Table 7: Personal TWA Particle Exposures among California Wildland Firefighters (Source: [96]).

Particle metric	Site/Activity	Mean (mg/m^3)	Range (mg/m^3)
TSP	Base camp/waiting in staging area	3.27	1.80-4.40
TSP	Fireline/mop-up	9.46	2.70-37.4
Respirable	Fireline/mop-up	1.75	0.327 – 5.14
Respirable	Prescribed burn	1.15	0.235 – 2.71

In the first report of cross-seasonal changes in respiratory symptoms and lung function in wildland firefighters, Rothman et al. (1991) examined 69 Northern California firefighters who were nonsmokers or former smokers who had not smoked in at least six months [138], There were significant cross-seasonal increases in reported cough, phlegm, wheeze, and eye and nasal irritation. Only eye irritation, however, was significantly associated with firefighting activity ($r = 0.$ $p<0.001$), while the association of wheeze with firefighting in the last two weeks of the study was of borderline significance ($r = 0.$ $p = 0.07$). There were small, but statistically significant, declines in several measures of pulmonary function across the season, with the strongest relationships for the highest cumulative exposure category for the final week preceding the follow-up spirometry. The associations became weaker and less significant with the progressive inclusion of additional weeks prior to the spirometry. Across the eight-week study, several lung function metrics exhibited significant declines, including FEV1 (-1.2% 95% CI -0.5,-2.0 %), FEV1/FVC (-0.006, 95% CI -0.001, -0.01), and although FVC also declined, this change was not significant. For those in the highest category for hours worked in the week preceding spirometry experienced larger decrements in lung function (FEV1 = -2.9% (130 mL) and FVC = - 1.9% (101 mL)). These changes were not affected by adjustment for potential confounders (not specified). The use of a cotton bandana for respiratory protection was not associated with any measurable protection.

Liu et al. (1992) examined cross-season changes in pulmonary function and airways hyperresponsiveness in 63 wildland firefighters in Northern California and Montana in 1989 [139]. They were tested before the start of the fire season and within two weeks of discharge from service. Though pre- and post-season spirometric measurements were within the normal range for all participants, there were significant cross-seasonal declines in FVC, FEV1, and FEF25-75 of 0.09 L, 0.15 L, and 0.44 L/sec, respectively. There was no significant relationship with any of the covariates measured, including smoking status, history of allergy, asthma, or upper/lower respiratory symptoms, specific firefighting crew membership, or seasonal vs. full-time employment. Airways responsiveness to methacholine increased significantly across the fire season, which was not affected by gender, history of smoking, allergy, full-time versus seasonal employment, or crew membership. This study suggests that, in addition to persistent cross-seasonal changes in lung function, fire-fighting may also be associated with increased airway hyperresponsiveness, although the effect was not-significant.

Letts et al. (1991) conducted a health survey of 78 wildland firefighters in Southern California [140]. There were no changes in symptom prevalence cross-seasonally, nor were there any significant associations with exposure (defined as low, medium and high based on hours of work weighted by visual estimates of smoke intensity). There were small, nonsignificant changes in FEV1 and FVC. The decrements in FEF25-75 and FEV1/FVC, however, were both significant (-2.3%, 95% CI -4.2, -0.5% and -0.5%, 95% CI -1.0, -0.1%). The changes in FEF25-75 showed a non-significant exposure-response trend ($p = 0.08$) of: 0.5%, -1.9%, and -4.7% for the low-, medium-, and high-exposure groups, respectively. Interestingly, however, there were no associations with the number of seasons of fire fighting, days since the last fire, or age. Although these investigators concluded that there was limited evidence of cross-seasonal effects of firefighting on lung function, they indicated that the season in which their survey was conducted involved an atypically low number of firefighting hours. Moreover, the baseline was established in June, reportedly "before significant smoke exposure occurred," though the extent of firefighting preceding the initial measurement was not documented.

In addition to examining cross-seasonal lung function changes, Betchley et al. (1997) also examined cross-shift changes among forest firefighters in the Cascade Mountains of Oregon and Washington [86]. Among 76 workers examined at the beginning and immediately after prescribed burns, mean declines in FVC, FEV1, and FEF25-75 were 0.065 L (1%), 0.150 L (3%), and 0.496 L/sec (6%), respectively. These changes were significant even after adjusting for respiratory infections in the preceding four weeks, smoking status, any "lung condition", and allergy. Examining cross-seasonal changes in 53 firefighters, the values for these same measures were 0.033 L, 0.104 L, and 0.275 L/sec, respectively. The changes for FEV1 and FEF25-75 were significant, and remained so even after adjustment for the same potential confounders and effect modifiers. There were no significant cross-seasonal changes in respiratory symptoms. The cross-seasonal lung function measurements and symptom reports were taken on average 78 days after the last occupational fire-fighting activities of the season.

Investigators in Sardinia compared lung function among 92 wildland firefighters with a "control" group of policemen [141]. The testing was undertaken in late spring, just prior to the onset of the principal fire season. The two groups had identical mean values for FVC and TLC[6], and showed no significant differences for FRC[7], DLCO, or DLCO[8]/TLC. The firefighters, however, demonstrated modestly lower lung function test results for FEV1, FEV1/FVC, FEF50, FEF25, and RV[9]. Although there were significant differences in age and height between the two groups (the firefighters were older and shorter, both of which would favor lower mean lung function for this group), the significant differences in lung function remained after multivariate control for age, height, smoking status, and pack-year history for current smokers. The investigators found no relationship between pulmonary function and years of service or with the number of fires extinguished over their careers. Cough and expectoration were more common among firefighters, but these differences were not significant.

[6] Total lung capacity (TLC) is the volume of air contained in the lung after a full inhalation.
[7] Functional residual capacity (FRC) is the lung volume after a normal expiration.
[8] Carbon monoxide diffusing capacity (DLCO) is a measure of how efficiently oxygen can be transferred into blood.
[9] Residual volume (RV) is the volume of the lung remaining after maximal exhalation.

Wildland firefighting can involve intermittent prolonged exposures to high concentrations of respirable particles, which consist of mixtures unique to each situation. Exposures to elevated levels of CO and respiratory irritants such as formaldehyde also occur, but respirable particles probably represent the principal exposure of concern. The few health studies conducted on such workers have documented cross-seasonal decrements in lung function, increased airway hyperresponsiveness, and increased prevalence of respiratory symptoms. Rothman et al. (1991) showed that recent cumulative exposures were more strongly associated with greater changes in lung function than were more remote exposures [138]. At least one study has also shown acute cross-shift spirometric changes as well [139]. There has been no long-term follow-up of the respiratory health of wildland firefighters, however. Among municipal firefighters, chronic pulmonary dysfunction may result from repeated smoke exposure, particularly among those who do not use respiratory protective devices [142, 143]. It is unknown whether cessation of exposure among wildland firefighters during the off-season may allow for recovery and reversibility of effects, in contrast to municipal firefighters who can be exposed year-round. In any case, the relatively small effects demonstrated among firefighters cannot be quantitatively extrapolated to non-occupational exposures, as the demands of the job require a degree of physical fitness and resilience far beyond that found in most of the general population.

5.2 Forest and Brush Fires

Several studies in North America have evaluated the health impacts associated with forest and brush fires. In the first study examining the effect of wildfire smoke on the general population, Duclos and colleagues evaluated the impact of a numerous large forest fires on emergency room (ER) visits to 15 hospitals in six counties in California [144]. The authors calculated observed-to-expected ratios of ER visits, based on the numbers of visits during two reference periods. During the approximately 2½-week period of observation, ER visits for asthma and chronic obstructive pulmonary disease increased by 40% ($p<0.001$) and 30% ($p=0.02$), respectively. Significant increases were also observed for bronchitis (Observed (O)/Expected (E) = 1.2, $p=0.03$), laryngitis (O/E = 1.6, $p=0.02$), sinusitis (O/E = 1.3, $p=0.05$), and other upper respiratory infections (O/E = 1.5, $p<0.001$). Exposure assessment was problematic, however, as few PM_{10} or other monitors were located downwind of the fires. The highest PM_{10} concentration measured was 237 $\mu g/m^3$. In contrast, several measurements of total suspended particles (TSP) exceeded 1,000 $\mu g/m^3$; the highest recorded was 4,158 $\mu g/m^3$. Exposure to forest fire smoke can be unpredictable, changing with wind direction, intensity of the fire, precipitation, and other variables. The few air quality measurements available to these investigators could not serve to reliably characterize population exposures, which is a general limitation of all wildfire studies. In addition, this study was subject to other typical limitations of ER analyses related to behavioral and economic factors (e.g., perceptions of illness severity, access to other health care providers, and availability of health insurance, with the latter more problematic in the U.S. than elsewhere).

Although no air pollutant concentrations were reported, the impact of wildfires in Florida on ER visits to eight hospitals in 1998 were compared to visits during the same five-week period during the previous year. From 1997 to 1998, ER visits increased substantially for asthma (91%), bronchitis with acute exacerbation (132%), and chest pain (37%), while visits decreased for painful respiration (27%) and acute bronchitis (20%). Though based on smaller numbers, there were modest changes in the number of hospital admissions (increases of 46% for asthma, and

24% for chest pain) ([145]. Although this study suggests that wildfire smoke exposure resulted in increased ER visits for respiratory disease and symptoms, no firm conclusions are possible. There was only one reference period selected, which might not provide a stable basis for comparison and no statistical testing was undertaken.

In a retrospective evaluation of the health impacts of a large wildfire in a northern California Indian reservation, visits to the local medical clinic for respiratory illness increased by 52% over the same period the prior year [146]. During the ten weeks that the fire lasted, PM10 levels exceeded 150 $\mu g/m^3$ (24-hr average) 15 times, and on two days the levels exceeded 500 $\mu g/m^3$. Weekly concentrations of PM_{10} were strongly correlated with weekly visits for respiratory illness during the fire year ($r = 0.74$), but not in the prior year ($r = -0.63$). In a community survey of 289 respondents, more than 60% reported respiratory symptoms during the smoke episode; 20% reported symptoms persisting at least two weeks after the smoke cleared. Individuals with pre-existing cardiopulmonary diseases reported significantly more symptoms before, during, and after the fire than those without such illnesses. The investigators also retrospectively evaluated the efficacy of several public health interventions in symptom reduction: (1) filtered and unfiltered masks distributed free of charge; (2) vouchers for free hotel accommodations in towns away from the smoke to assist evacuation efforts; (3) high efficiency particulate air (HEPA) cleaners distributed for residential use; and, (4) public service announcements (PSAs) about exposure reduction strategies. Mott and colleagues found that increased duration of use of a residential HEPA air cleaner was associated with decreased odds of reporting increased symptoms (OR = 0.54, 95% CI (0.32 - 0.89)), with an inverse trend of symptom reporting with increasing duration of use. Similarly, ability to accurately recall a PSA was also associated with reduced odds for respiratory symptoms. In contrast, there was no detectable beneficial effect of evacuation from smoky areas or of the use of masks. However, the timing and duration of evacuation were not optimal: on the days with the highest recorded smoke concentrations, over 80% of the subjects had not evacuated. That mask use was not protective is unsurprising; the masks were distributed without fit-testing and had variable filtration efficiencies. Moreover, none of the interventions was randomized, and in fact individuals with smoke-related health effects or a prior diagnosis of respiratory or cardiovascular disease were given priority to receive hotel vouchers and HEPA air cleaners. Finally, due to the retrospective nature of the investigation, recall bias may have affected the results based on the survey.

More recently, Sutherland and colleagues reported an increase in an index of respiratory symptoms (dyspnea, cough, chest tightness, wheezing, and sputum production) amongst a panel of 21 subjects with COPD associated with two days of elevated ambient particle levels resulting from a forest fire near Denver. On the two days in which symptom scores were increased, average $PM_{2.5}$ concentrations increased to 63 $\mu g/m^3$ relative to an average of 14 $\mu g/m^3$ on control days [147].

During 1994, bush fires near Sydney, Australia led to elevated PM_{10} levels (maximum hourly values of approximately 250 $\mu g/m^3$) for a 7-day period. Two studies of asthma emergency room visits during the bushfire smoke episode failed to detect any association with air pollution [148, 149]. The report by Copper et al. (1994) was in the form of a letter to The Lancet, with few details were provided. The investigators examined only three inner-city hospitals, preferring to avoid the influence of "patients who presented with direct effects of smoke inhalation," which

might have occurred had they included hospitals with catchment areas closer to the fires. They compared the numbers of asthma ER visits for the week before the bushfires (Jan 1-8), the fire period (Jan 9-20), and afterwards (Jan 21-31), and found no difference among the three periods. These comparisons were based on relatively small numbers, however, with fewer than 100 visits for asthma during the entire month for all three hospitals. The report by Smith et al. (1996) involved a comparison of the proportions of asthma to total ER visits to seven hospitals during the week of high smoke levels compared to the same week the prior year. There was no difference in these proportions, nor was there a relationship between the maximum daily nephelometric particle measurement and the number of asthma ER visits in multi-week regression models. Although it appears that the bushfire smoke did not have an impact on asthma ER visits, this study is limited by the use of a single reference period. In addition, the regression analysis is likely to have had very limited statistical power, with relatively few days of observation.

A recent analysis of these same fires and lung function (measured as peak expiratory flow rate [PEFR]) did not detect any association between either PM_{10} levels or an indicator variable representing the fire period and evening PEFR in 25 asthmatic children, although 20 children without airway hyperreactivity showed a significant decrease in PEFR with increasing same-day PM_{10} concentrations. [150]. Whether this represents a true lack of association or an artifact of experimental design is difficult to ascertain. Thirty-two children in this analysis were recruited the week of the fire. There did not seem to be any examination of whether there was a learning period for these children (during which the initial PEFR measurements might have been more variable), nor was there any discussion of the quality control for recording the measurements, or even what the PEFR protocol was. Of the 32 children (mean age = 9.2 years), 25 had a doctor's diagnosis of asthma; however, only 12 of the 32 had evidence of airway hyperresponsiveness, which is considered a hallmark of asthma. Although the regression model included indicator variables for use of asthma medications, there could nonetheless still have been residual confounding by medication use. In other words, the use of asthma medications might still have had enough of an effect on lung function to obscure a relationship between PEFR and smoke exposure, despite the attempt to control for this influence statistically. Finally, the children were recruited during the week of the fires; therefore it is not clear how many child-days of observation during the fires actually contributed to the analysis. The reported data suggest that this study is likely to have had very limited statistical power.

The results from these studies appear to conflict with those conducted in North America. As noted above, however, all have significant limitations that suggest caution in generalizing the results. It is also possible that there is less respiratory toxicity from bushfire smoke than forest fire smoke due to chemical and physical differences between the two. A recent study undertaken at the only hospital at Darwin (northwestern Australia) evaluated the association between daily asthma emergency room visits (adjusted for influenza and day-of-week effects) and measured PM_{10} over a 7-month period, which included two bushfire smoke episodes. Bushfires represent the principal regional source of significant levels of air pollution in Darwin during the dry season in which this investigation took place. Increased asthma visits were associated with PM_{10} concentrations, especially for days in which PM_{10} concentrations exceeded 40 $\mu g/m^3$ [151]. The adjusted rate ratio per 10 $\mu g/m^3$ increase in PM_{10}, was 1.20 (95% CI, 1.09–1.34; $P < 0.001$). The largest association was observed for a five-day lag, comparing days when PM_{10} exceeded 40

$\mu g/m^3$ with those on which PM_{10} was less than 10 $\mu g/m^3$ (adjusted rate ratio = 2.56 (95% C.I., 1.60 – 4.09)). Unlike the prior studies of smoke conducted in Australia, this investigation clearly had adequate statistical power to detect an association between particulate matter and asthma visits. Though the time-series analysis did not control for pollen or mold, which are not routinely monitored in Darwin, the investigators considered it "extremely unlikely" that either of these would vary systematically with bushfire smoke. This assessment by the authors is probably true; however, without analyzing the smoke for these bioaerosols, it is not possible to state definitively that they did not confound the results.

Major regional episodes of air pollution from vegetation fires in Southeast Asia have been the subject of several investigations and surveillance programs. An analysis of emergency room visits for asthma in Singapore during a 1994 episode of regional pollution resulting from forest and plantation fires reported an association between PM_{10} and emergency room visits for childhood asthma. During the "haze" period, mean PM_{10} levels were 20% higher than the annual average. Although a time series analysis was not conducted, the authors suggest that the association remained significant for all concentrations above 158 $\mu g/m^3$ [152].

Reports from surveillance monitoring activities conducted during major Southeast Asian episodes in 1997 and 1998 also indicated effects on health care utilization. In Singapore, for example, there was a 30% increase in hospital attendance for "haze-related" illnesses and a time series analysis indicated that a PM_{10} increase of 100 $\mu g/m^3$ was associated with 12%, 19% and 26% increases in cases of upper respiratory tract illness, asthma and rhinitis, respectively. It is not clear why rhinitis constituted a separate diagnostic category in this investigation, rather than being included with upper respiratory illness. This analysis did not observe any significant increases in hospital admissions or mortality [153]. Similar findings were also observed in Malaysia [135, 154].

Preliminary results from a study of 107 Kuala Lumpur school children found statistically significant decreases in lung function between pre-episode measurements in June-July 1996 and measurements conducted during the haze episode in September 1997 [155]. A convenience sample questionnaire survey conducted in Indonesia during the 1997 haze episode also suggested acute impacts on respiratory and cardiovascular symptoms [136]. Of 539 interviewees, 91% reported respiratory symptoms (cough, sneezing, runny nose, sputum production, or sore throat), 44% reported shortness of breath on walking, 33% reported chest discomfort, and 23% reported palpitations. Although the numbers were small, respondents with asthma or heart disease tended to experience a greater proportion of moderate and severe symptoms relative to those without pre-existing disease. Despite these findings, however, the cross-sectional nature of the sampling and reporting as well as the absence of an unexposed reference population weaken any inference of a causal relationship between the smoke and these symptoms.

In another study of the 1997 Southeast Asia haze episode, Tan and colleagues (2000) obtained blood samples at weekly intervals from 30 Singaporean military recruits who followed standardized outdoor routines during the episode. The mean 24-hr PM_{10} level during the episode was 125.4 $\mu g/m^3$. Analyzing the numbers of immature PMNs (polymorphonuclear leukocytes – see toxicology section) in the subjects' blood in relation to daily measures of several pollutants, these investigators found the strongest relationship with same-day PM_{10}, though a one-day lag of

this metric was also statistically significant. Although these results are insufficient to establish a causal relationship, they suggest that smoke inhalation stimulated the bone marrow to eject immature PMNs into the circulation.[10]

Only one study has evaluated the impacts of air pollution from vegetation fires on mortality. Sastry evaluated the population health effects in Malaysia of air pollution generated by a widespread series of fires that occurred mainly in Indonesia between April and November of 1997. The results show that the haze from these fires had a deleterious effect on population health in Malaysia and were in general agreement with the mortality impacts associated with particles in urban air [156]. A 10 $\mu g/m^3$ increase in PM_{10} measured in Kuala Lumpur was associated with 0.7 % (all ages) and 1.8% (ages 65-74) increases in adjusted relative risk of non-traumatic mortality. Visibility-based estimates of particulate matter concentrations in Kuching, a city closer to the fire sources, were also associated with increased mortality.

With the exception of three of the Australian bushfire investigations, all of which have significant structural limitations, the epidemiological studies of indoor and community exposure to biomass smoke indicate a generally consistent relationship between exposure and increased respiratory symptoms, increased risk of respiratory illness, including hospital emergency room visits, and decreased lung function. The majority of these studies are not focused solely on children. Several studies that asthmatics are a particularly susceptible subpopulation with respect to smoke exposure, which is consistent with the results of many studies of the impacts of ambient air pollution. The effects of community exposure to biomass air pollution from wildfires on hospitalizations or mortality have not been sufficiently studied to support general conclusions.

5.3 Agricultural Burning

There have been relatively few studies of the health effects of agricultural burning, despite growing concern about its health effects [157]. In one Canadian study, however, 428 middle-aged subjects with slight-to-moderate airway obstruction were surveyed about respiratory symptoms during a 2-week period of exposure to straw and stubble combustion products. During the exposure period, 24-hour average PM_{10} levels increased from 15-40 $\mu g/m^3$ to 80-110 $\mu g/m^3$. One-hour levels of carbon monoxide and nitrogen dioxide reached 11 ppm and 110 ppb, respectively. Total volatile organic compound levels increased from 30-100 $\mu g/m^3$ before the episode to 100-460 $\mu g/m^3$ during the episode. While 37% of subjects were not bothered by smoke at all, 42% reported that several respiratory symptoms (cough, wheezing, chest tightness, shortness of breath) developed or became worse due to the air pollution episode and 20% reported that they had breathing trouble. Subjects with asthma and chronic bronchitis were more likely to be affected, and women appeared to be more susceptible than men for several symptoms (cough, shortness of breath, nocturnal awakening) [158]. In contrast, current cigarette smokers reported significantly fewer symptoms than the former smokers constituting the rest of the study population. This study indicates that, besides woodsmoke, biomass air pollution from

[10] In a subsequent toxicological examination involving rabbits, these same investigators found that repeated PM_{10} instillations into the respiratory tract resulted in increased production of PMNs in the bone marrow and an acceleration of their release into the blood, both of which were associated with the numbers of particles ingested by the animals' alveolar macrophages (Mukae et al. 2001)

agricultural burning is associated with increased respiratory symptoms among a susceptible population with pre-existing lung disease.

A time-series study in California suggested that agricultural burn smoke was associated with serious exacerbations of asthma. The association between asthma hospital admissions and the burning of rice field stubble and waste rice straw was examined in Butte County, California over a ten-year period [159]. Although burning was not associated with any measurements of major air pollutants (probably because monitors were not sited to provide optimal measurement of burn smoke), burn acreage was significantly associated with an increased risk of asthma hospitalization and showed an exposure-response trend. The greatest risk of hospitalization was observed on days when 500 or more acres were burned (RR = 1.23; 95% CI 1.09-1.39).

Field burning of straw and stubble near Winnipeg in 1992 exposed residents of that city to 24-hour integrated concentrations ranging from 20-80 $\mu g/m^3$ from July through September. The 24-hour concentrations during the burning episode ranged from 80 to 200 $\mu g/m^3$. One hour maximum CO during the burning episode was as high as 11 ppm and hourly NO_2 also reached 11 ppm. Forty-two percent of a sample of residents reported increased respiratory symptoms. Twenty percent said they had trouble breathing [158].

A recent cross-sectional study in three rural villages in Iran also evaluated the relationship between rice stubble burning and respiratory morbidity, especially asthmatic symptoms [160]. During a burning period lasting several weeks, PM_{10} concentrations doubled. Based on responses to a physician-administered survey before and after this episode, the investigators reported significant increases in the prevalence of asthma attacks, use of asthma medications, the occurrence of nocturnal sleep disturbances, and other respiratory symptoms among 994 residents of an agricultural region. Several measures of pulmonary function also decreased significantly.

The relationship of rice stubble burning with asthma was also studied in Niigata prefecture, Japan [161]. In this study, measured PM_{10} concentrations were associated with monthly asthma hospital admissions in a region where rice straw burning emissions led to high particle concentrations during the September – October burning season. Although PM_{10} levels were not associated with monthly emergency room visits for asthma, the investigators reported a significantly higher number of asthma emergency room visits on days when rice straw burning occurred and the following two days (7.1 ± 3.9) versus other days (4.5 ± 3.3). The latter comparison would have better time resolution than an analysis of monthly average of asthma exacerbations, and should probably be accorded greater weight. Although this investigation also involved a parental questionnaire suggesting more asthma exacerbations in children during the rice burning season than at other times of the year, an autumn peak in asthma flares is also common in other parts of the world where rice burning does not occur. In general, multiple findings in this investigation are suggestive of a rice smoke effect on asthma, but several limitations of the study design limit both causal inference and the generalizability of the findings.

A metric commonly used as a surrogate of exposure to biomass smoke is the amount of agricultural land burned, for example rice fields [159]. This study reported an association with asthma hospitalizations when the number of acres burned was greater than 499. Norris (1998) evaluated the association between acres of grass seed residues burned around Spokane, WA and

visits to local emergency departments for asthma [162]. During one burning event, peak PM_{10} concentrations in Spokane reached 100 μg/m^3 during a four- hour period (Figure 2). Using a bivariate indicator (20 days with > 499 acres burned) for the exposure surrogate, an association with increased emergency department visits for children was observed (relative risk = 1.30 (955 CI, 1.08-1.58

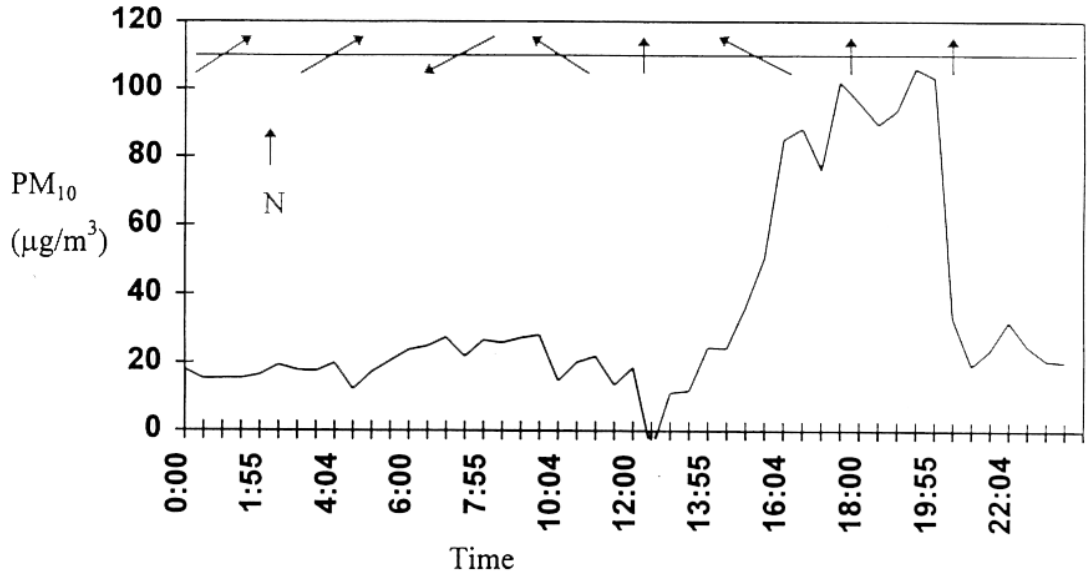

Figure 2: Grass burning event measured in Spokane, WA at the Rockwood residential monitoring site (9/6/94) (Source: [162]).

Together, these epidemiological studies suggest that exposure to products of agricultural burning, specifically the burning of rice stubble/straw, may be associated with exacerbation of asthma. In a chamber study of smoke generated by controlled burning of rice stubble straw, Solomon and colleagues exposed 13 adults with allergic rhinitis (age range 24-55) at rest to filtered air, rice straw smoke (RSS) at 200 μg/m^3 or at 600 μg/m^3 for 30 minutes, or RSS at 200 μg/m^3 on three consecutive days. Bronchoalveolar lavage (BAL) was conducted six hours post-exposure. Of a variety of cell types and cytokine measured in BAL fluid, the investigators found a near doubling of epithelial cells only after the 3-day exposure, but no difference from filtered air exposures in total white blood cells, macrophages, PMNs, lymphocytes, eosinophils, or interleukin-8 under any of the RSS exposure conditions. Interestingly, this effect was not observed at a higher concentration (600 μg/m^3) delivered over a shorter time interval, suggesting that repeated exposures may be necessary, at least among individuals with allergic rhinitis [163].

A very limited number of studies have specifically examined air pollution and resultant health effects associated with the burning of sugar cane. Cancado and colleagues examined the relationship between particulate matter components and pediatric respiratory hospital admissions in a region of Brazil where sugar cane cultivation is common [164]. Analyses were performed during both burning and non-burning seasons. The main elemental particle components in the

PM mass were potassium and black carbon, which are both known to be generated during biomass combustion. The investigators reported significantly increased risks of pediatric respiratory hospital admissions associated with the concentrations of both of these particle components. These risks were three times greater in the burning season relative to the non-burn season, suggesting the importance of sugar cane combustion in this association. In previous work in Brazil, daily indirect measurements (sedimentation of particle mass) of air pollution during the sugar cane burning season in 1995 were associated with the number of patients visiting hospitals for inhalation therapy for acute respiratory distress [165]. The relative risk of such a hospital visit associated with an increase of 10 mg in the sediment was 1.09 (1-1.19); this association displayed an exposure-response relationship as well. Boopathy and colleagues presented a descriptive analysis of asthma hospital visits to a medical center in Houma, Louisiana during 1998-99 [166]. The area served by this medical center accounted for approximately 27% of Louisiana's sugar cane cultivation during this period. Although no air pollution measurements were available, asthma hospital visits increased dramatically during the October-December sugar-cane burning season. As noted above, however, an autumn peak in asthma exacerbations is common, and respiratory infections (the main precipitating factor for severe asthma attacks) also typically increase in frequency during this time. Therefore, it would be inappropriate to infer a causal relationship between sugar cane burning and asthma hospital visits based on this descriptive study.

Several studies have also reported an increased risk of lung cancer and mesothelioma among sugar cane workers, although specific job activities were not evaluated and exposure measurements were not made [167, 168]. A study in India found an increased risk of lung cancer in sugar cane workers associated with post-harvest burning [169]. It has been suggested that this association may be due to the liberation of asbestos-like biogenic silica fibers in sugar cane smoke. Boeniger and co-workers (1991) conducted an exposure assessment of smoke during sugar cane harvesting in Hawaii in 1987 [170]. They gathered both area and personal samples. The concentration of PM increased by at least 20 and up to 70 times the measured background levels at the sampling sites chosen (Table 8). A consequent study of Hawaiian sugar cane workers, however, reported no elevated morbidity or mortality rates or decreased lung function [171].

Table 8: Area Air Concentrations of PM During Sugar Cane Field Burning (Source: [170]).

Sample number	Minute	PM mass (mg/m3)
1	420	< 0.01
2	380	0.01
3	29	0.5
4	100	0.7
5	70	<0.2
6	35	<0.4
7	57	N/A
8	60	0.5
9	60	0.3

6 RESIDENTIAL WOODSMOKE USE IN DEVELOPED COUNTRIES: EXPOSURE AND HEALTH STUDIES

6.1 Residential Woodsmoke Emissions[11]

Residential wood burning is common in the Western US and Canada. Source apportionment studies show that woodsmoke is a major source of PM during winter months in several parts of the US, for example. Table 9 shows data from San Jose [172], which indicates that 42% of the PM_{10} during winter months could be attributed to wood burning. Chemical mass balance receptor modeling of fine particles in Fresno and Bakersfield (California) during wintertime identified both hardwood and softwood as sources of PM and organic compounds [33]. It is likely that residential wood burning is the source of this wood combustion. One study of emergency department visits for asthma in the Santa Clara County airshed (which includes San Jose), found an association with wintertime ambient PM_{10} [173]. In this investigation, a substantial fraction of the particulate matter was contributed by residential wood combustion.

Table 9: Chemical Mass Balance (CMB) Results for Aggregated Wintertime High PM10 Samples from San Jose, 11/93 to 1/94 (Source: [172]).

Source	CMB	
	µg/m3	%
Woodburning/cooking	36	42
Motor vehicle	15	17
Geological	12	14
Marine air	2	3
Sulfate	2	3
Nitrate	19	22

A study of wood-burning stoves and acute lower respiratory illnesses (ALRI) in Navajo children from the Public Health Service Indian Hospital at Fort Defiance, AZ included measurements of indoor concentration of respirable particles [174]. Indoor air samples were collected for determination of time-weighted average concentrations of respirable particles (PM_{10}) found peak concentrations $\geq$ to 65 $\mu g/m^3$ (90^{th} percentile). The health results are discussed below.

Christchurch, New Zealand is another city impacted by woodsmoke. It is estimated that more than 90% of particulate matter (PM) air pollution is emitted from wood burners and open fires during winter [175]. Frequent periods of air stagnation compound the air pollution problem by trapping the particles at the surface. Local meteorologists estimate that there is relatively even mixing of particles resulting in homogeneous exposure of the population [175]. Christchurch has a population of approximately 333,000 people, allowing sufficient power for health outcome estimation. One study examined the association between hospital admissions and PM_{10} for the period 1988-1998. PM_{10} exposure during the study period averaged 25 $\mu g/m^3$, and a maximum of 283 $\mu g/m^3$. The results were stratified into total cardiac and total respiratory admissions. The

[11] A thorough summary of emissions from woodsmoke was published several years ago 104.Larson, T.V. and J.Q. Koenig, *Woodsmoke: emissions and noncancer respiratory effects.* Annu Rev Public Health, 1994. **15**: p. 133-56.(Larson and Koenig, 1994).

estimated percentage increase for all age groups was 1.26 (CI: 0.3-2.2) for cardiac admissions and 3.37 (CI: 2.3-4.4) for respiratory admissions [175].

Data from Seattle also indicate that use of residential wood stoves contributes substantially to outdoor PM in that airshed. Data from three years of IMPROVE (Improvement of Microphysical Parameterization through Observational Verification Experiment) sampling in Seattle were analyzed for sources using positive matrix factorization (PMF) [105]. The PMF analysis found that vegetative burning contributed 34% to the total sources of PM in Seattle over three years (Figure 3). It must be noted that PM is only one of the toxic components of woodsmoke. Health effects research in Seattle shows associations between $PM_{2.5}$ and lung function decrements in children [8], visits to emergency departments for asthma [176], hospitalizations for asthma [177] and, increases in asthma symptoms in children [178], as well as increases in exhaled nitric oxide [179, 180]. Since vegetative burning is the primary source of fine particles in the Seattle air shed, the health effects studies suggest a causal relationship.

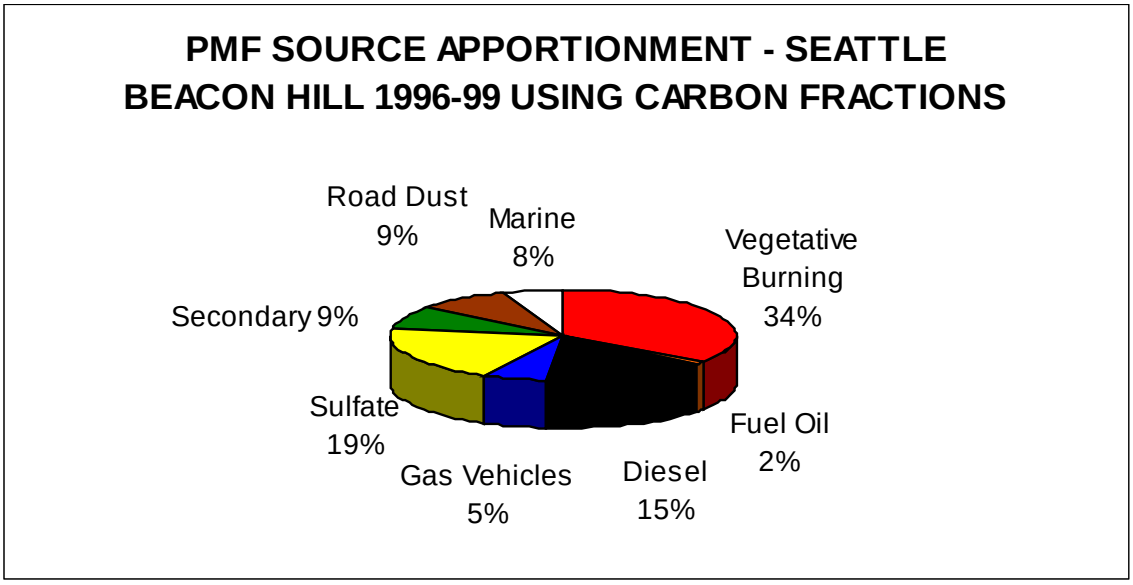

Figure 3: PMF source apportionment results for Beacon Hill, Seattle, 1996-1999 (Source: [105]).

Another study utilized a large data set from a 2-year exposure assessment and health effects panel study in Seattle during September 2000 through May 2001. Indoor, outdoor, personal and fixed-site PM monitoring data were available. The samples were analyzed for elements using XRF. Positive matrix factorization was used with these data to apportion sources [36]. The investigators identified five sources contributing to indoor and outdoor samples: vegetative burning, mobile emissions, secondary sulfate, a chlorine source, and a crustal-derived source. Vegetative burning contributed the largest fraction of PM mass in all the samples (35%, 49% and 62% in indoor, outdoor, and personal mass, respectively). These data indicate that significant exposure to vegetative burning sources exists in urban settings in some regions of developed countries. Statisticians are attempting to derive models that will allow source apportionment data to be added to health endpoint analysis without creating undue bias [181]. Existence of these models will help apportion specific health outcomes to specific sources.

Early studies of woodsmoke health effects often used the presence or absence of a wood stove in the home as the indicator of exposure. Due to penetration of woodsmoke particles indoors, these exposures may not be due exclusively to indoor sources of woodsmoke. It has been shown in a woodsmoke-impacted community, that particles readily penetrate inside residences [182]. The contribution from outdoor-generated particles to indoor and personal exposure in Seattle residences has been estimated using a recursive model [112, 183]. Nonlinear regression was used to estimate particle penetration, particle decay rate, and particle infiltration. Estimates of particle infiltration agree well with those derived from sulfur-tracer methods ($R^2 = 0.78$).) [184]. In a sample of 44 residences, outdoor-generated particles accounted for an average of 79 $\pm$ 17% of the indoor PM concentration. These data suggest that in epidemiological studies of associations between health outcomes and outdoor PM, much of the exposure to outdoor particles can occur inside the home. Other factors, such as the age of the house, opening of windows, and air conditioning, can affect penetration. In one study, home air conditioning was associated with lower penetration of outdoor particles; moreover, the associations between PM_{10} and hospital admissions were lower in cities with a higher prevalence of air conditioning [185]. These findings imply that even well-operating woodstoves and fireplaces that vent most smoke outside may produce substantial exposures through penetration back into the house.

Although population exposure to emissions from residential wood stoves has been an acknowledged problem in the Western US, recent data show that exposure to woodsmoke is common in other parts of North America as well. Source apportionment data have found factors associated with wood burning in Atlanta [186], Vermont and southwestern Quebec. The estimated percentages of the total mass of fine PM associated with the woodsmoke factor are 11% in Atlanta and 14.5% in Vermont.

One study evaluated the distribution of particle-phase organic compounds in communities with children participating in the Southern California Children's Health Study [187]. Concentrations of levoglucosan, a good tracer for woodsmoke aerosol, were seen in all 12 CHS communities (Figure 4). The average concentration increased in the winter, as would be expected for woodsmoke emissions. The concentrations of levoglucosan were highest at the inland Atascadero site, which is about 15 miles inland. Earlier, these investigators identified two additional sugar anhydride tracers of woodsmoke (galactosan and mannosan) in a study of urban sites in the San Joaquin Valley, California [188]. These data may allow an estimation of the effects of woodsmoke exposure on health outcomes among the participants in the Children's Health Study.

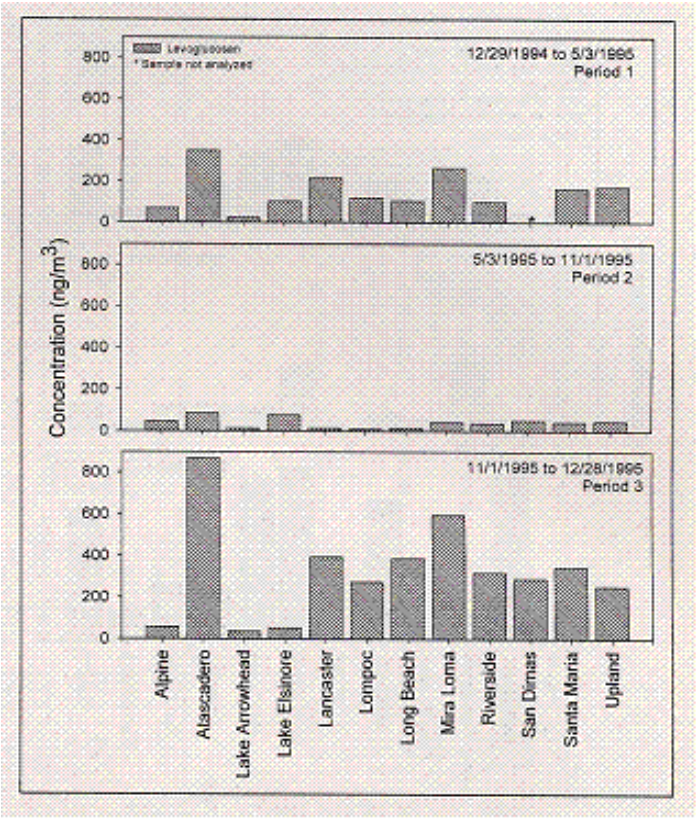

Figure 4: The seasonal and spatial distribution of levoglucosan measured at the Southern California Children's Health Study sampling sites, 1995 (Source: [187]).

This short summary of studies documenting exposure to vegetative burning in a part of the developed world (North America) indicates that sufficient exposure exists for public health concern about adverse effects in exposed populations. Table 10 summarizes the data on concentrations and constituents of woodsmoke from several studies. These data demonstrate millions of individuals across the US and Canada may be exposed to smoke from combustion of wood and other types of biomass. In some communities the use of residential wood burning appears to be increasing [129].

Table 10: Woodsmoke Concentrations Measured at Various Locations in Developed Countries.

Location	**Woodsmoke Concentration**	**Source**
Santa Clara Co. CA	42% of CMB	[172]
Seattle Indoor	35% of total PM2.5 mass	Larson et al, In press
Seattle Outdoor	49% of total PM2.5 mass	Larson et al, In press
Seattle Personal	62% of total PM2.5 mass	Larson et al, In press
Atlanta Outdoor	11% of total PM2.5 mass	[214]
Vermont	10-18% of PM2.5	[214]
Atascadero, CA	Levoglucosan	[187]
Fort Defiance, AZ	PM2.5 peak concentrations of 65 ug/m^3	[174]

6.2 Health Effects of Residential Wood Burning

A number of epidemiological studies have documented respiratory effects of residential wood burning especially in children. One of the earliest studies was conducted by Honicky et al (1985) in Michigan [189]. These investigators compared respiratory symptoms in 31 children who lived in homes with wood stoves with 31 children who lived in homes without wood stoves. Symptoms were categorized as mild, moderate, and severe. The two groups did not different with respect to mild symptoms but differed significantly for severe symptoms ($p < 0.001$). A similar study was conducted in Boise ID in 1989 by Butterfield et al. (1989) [190]. Respiratory symptoms were tracked in 59 children under the age of 5 1/2 years during a winter season. Symptoms such as wheeze, cough, and nocturnal awakening were associated with woodsmoke exposure.

Morris et al (1990) evaluated the health of children living on the Navajo reservation in Arizona by assessing use of a well-child clinic [191]. All cases of respiratory illness were evaluated for wood stove use in the home. The odds ratio for a lower respiratory tract infection in children with presence of a wood stove was 4.2 ($p < 0.0012$). A more recent case-control study among slightly younger (1-24 months) Navajo children reached similar conclusions [174]. Matched-pair analysis found an increased risk of acute lower respiratory illness in children living in households with wood cook stoves. The odds ratio was 5.0 (95% CI 0.6, 42.8). The low number of cases (45) may have reduced the investigators ability to see a significant association between use of wood-burning devices and health.

In Seattle 326 elementary school children were studied during the heating season of 1988-89 and 1989-90 [8]. Monthly or bimonthly spirometry values were collected during the school year. PM exposure was measured by light scattering using nephelometers. The exposure metric used was 12-hour averages from 7 PM to 7AM to reflect the hours when woodsmoke is most elevated. A random-effects statistical model compared changes in FEV_1 and FVC with changes in the light-scattering coefficient. The 26 children with asthma showed a significant decrement (18 $ml/\mu g/m^3$ $PM_{2.5}$) for both measures of lung function. Children without asthma showed no significant changes in lung function associated with PM values.

Another study looked at otitis media and asthma as health outcomes in a case-control study of home environmental air pollutants in Springville, NY [192]. That study found use of wood-burning stoves was more likely to be present in homes of children with otitis media (1.7 times as likely, 95% CI = 1.03, 2.89, $p < 0.037$). There is at least one study of the effects of woodsmoke exposure in adult subjects aged 18 to 70 years of age [193]. In a study of effects of indoor air pollution including use of wood stoves or fireplaces in adult subjects in Denver, the use of wood fires was associated with an increase in daily moderate or severe shortness of breath (odds ratio = 1.3, 95% CI: 1.1,1.4, $p < 0.0001$). Use of wood stoves or fireplaces was second only to presence of smokers in the home, and more strongly associated with shortness of breath than use of gas stoves or occupational exposures. On the other hand, there are a few studies that have failed to find associations between woodstove emission exposure and respiratory health (e.g. [102]).

Two large epidemiologic studies have been conducted in communities with known woodsmoke sources. The first was conducted in Seattle over a one-year period (September 1989-September 1990) [194], during which there were 2,955 emergency department visits for asthma to eight

hospitals. Twenty four-hour PM_{10} values ranged from 6 to 103 $\mu g/m^3$, with a mean of 29.6. In Poisson regressions controlling for weather, season, time trends, age, hospital, and day of the week, the daily counts of emergency room visits for persons under age 65 were significantly associated with PM_{10} exposure on the previous day (Figure 5) The relative risk for a 30 $\mu g/m^3$ increase in PM_{10} was 1.12 (95% CI 1.04, 1.2). Woodsmoke contributed approximately 85% of the wintertime PM in residential areas during the study period.

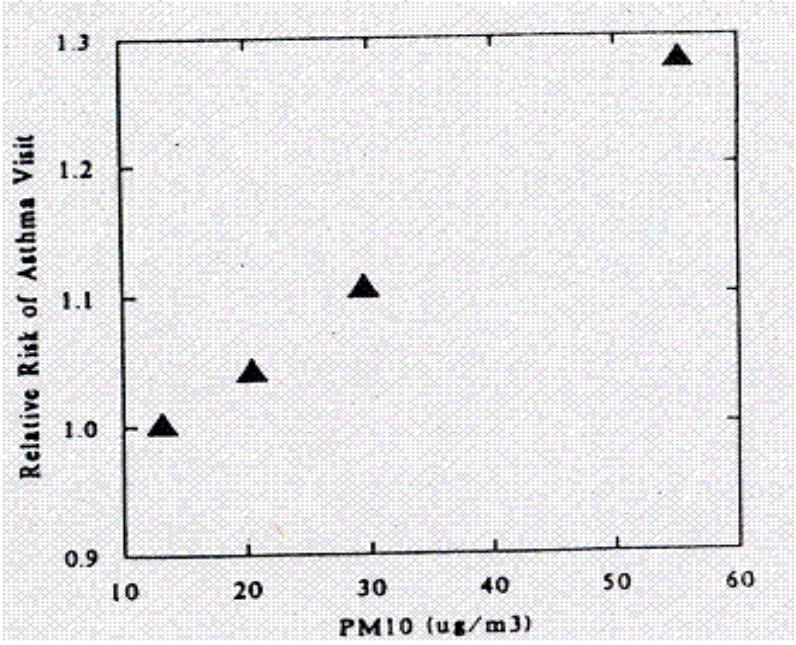

Figure 5: Relative risk of asthma visits (Source [194]).

A more recent study of emergency department visits and PM was conducted in Santa Clara County, California [173]. The investigators found a relative risk for an emergency department visit for asthma adjusted to a 60 $\mu g/m^3$ increase in PM_{10} of 1.4 (95% CI 1.2, 1.7) at 20 º F. The results of an epidemiological study in Christchurch, New Zealand, were discussed earlier [175]. Boman et al (2003) reviewed the literature relating to adverse health effects from ambient exposure to woodsmoke and, comparing the results of studies of acute exposure to those done in areas without much woodsmoke, concluded that there was no reason to think that the adverse impacts of acute woodsmoke exposure would be less than those associated with other sources of ambient particulate matter [129].

These studies do have some limitations. There all are ecologic and none has personal exposure information. Nevertheless there is a coherence to the data that has been described as a strength [195]. Among the studies cited for documentation of adverse health effects associated with residential wood burning there are differences in proportional seriousness. The indicators of adverse effects run from increases in respiratory symptoms to lung function decreases to visits to emergency departments and finally hospitalization. It is highly unlikely that this pyramid of adverse effects could be built if the associations reported were not real. In assessing the "strength" of air pollution health effects data, Bates (1992) concluded that the question of coherence is crucial [195]. He goes on to state that such coherence may exist at different levels:

within epidemiological data; and between epidemiological and toxicological data and between epidemiological data and controlled studies. Woodsmoke exposure of residents inside their homes is supported by the infiltration data discussed earlier in this section. Therefore, it is reasonable to conclude that exposure to the concentrations and durations of woodsmoke associated with residential wood burning is likely to cause a variety of adverse respiratory health effects. The conclusion is supported by the wealth of data on health effects of biomass burning in developing countries reviewed elsewhere in the document.

7 BIOMASS USE IN DEVELOPING COUNTRIES

7.1 Indoor air pollution from household fuels

As control of fire is often used as the defining distinction between pre-human and human evolution, it is literally true that woodsmoke exposure is as old as humanity itself. Throughout most of that history, as today, the largest exposures occurred in households through use of wood and other forms of biomass as sources of cooking, drying, and space-heating energy. Even today, such uses probably account for the majority of human exposure to respirable particulates worldwide because of the continued high dependence on such household fuels [196]. As shown in Figure 6, for example, about half of the world's households are still thought to cook with solid fuels on a daily basis [197]. Of this, about 95% consists of wood and agricultural residues. Household use of mineral coal for cooking, which makes up the remainder, is mainly confined to China.

In simple devices, like the household stoves commonly used in developing countries, biomass fuel does not combust cleanly. Systematic emissions studies in India and China, for example, have generally validated the so-called "energy ladder" concept with regard to the emissions of

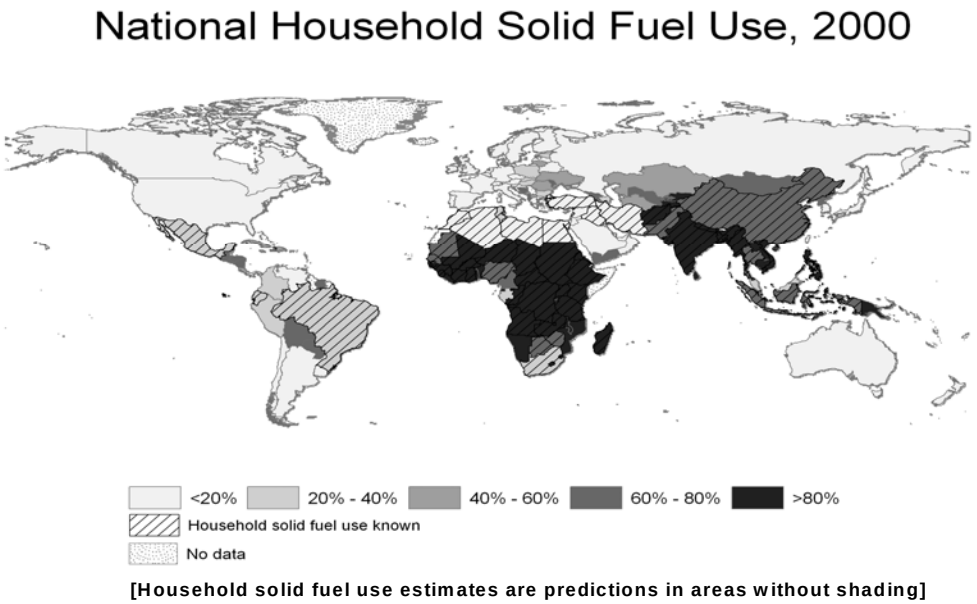

Figure 6: Map of solid fuel use (Source [197]).

household fuels [198]. As shown in Figure 7, the energy ladder starts at the bottom with low-quality biomass fuels such as cow dung, moves up through crop residues, to wood. Further up lie liquid and then gaseous fuels (kerosene and LPG), with electricity being at the top (least

emissions) at the stove. Nominal combustion efficiency (percent of fuel carbon emitted as CO_2) is as low as 80% for the poorer fuels and reaches more than 99% with gaseous fuels [199]. In addition to cleanliness, cost, complexity, and ease of household use generally increase as one moves up the ladder [200]. Broadly, as average household income increases in societies, usage tends to move up the ladder, although not always to the last rung (electricity). This is shown by econometric studies at the national level [201]. In individual communities, however, the situation is often more complicated, particularly during transition phases when household may straddle several rungs of the ladder at once by using multiple fuels depending on prices, seasons, availability, etc [202].

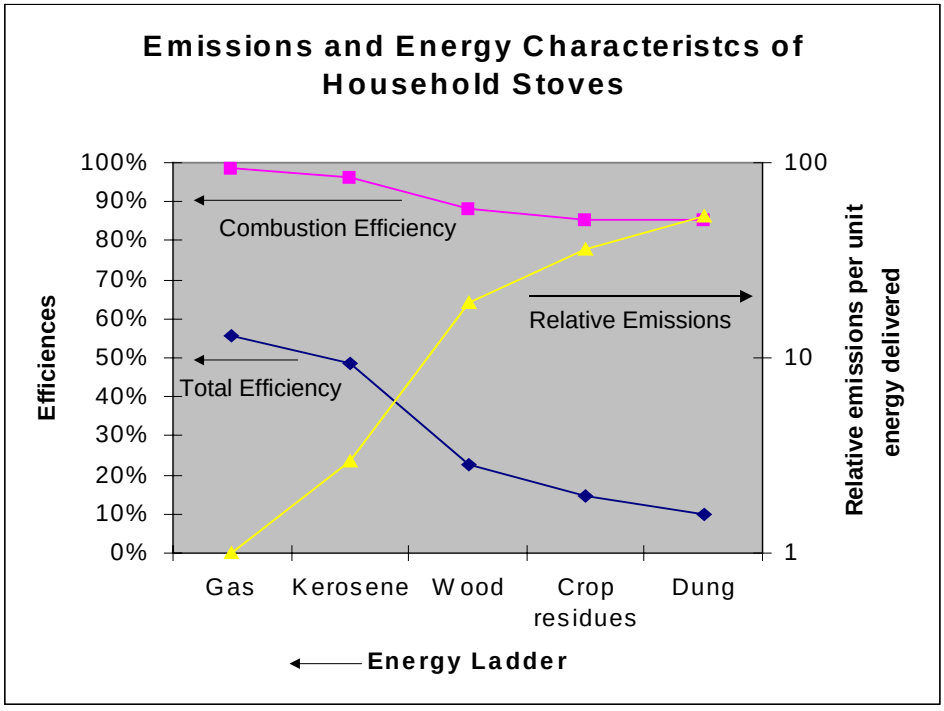

Figure 7: Emissions and energy characteristics of typical Indian cookstoves. Note improvement in combustion and total efficiency moving from solid to liquid and gaseous fuels and great reduction in emissions per unit energy delivered. (Source [119])

As noted elsewhere in this report, poor combustion efficiency creates high emission factors for wood and other biomass across a wide range of health-damaging pollutants. High emissions, however, do not necessarily lead to high exposures unless they reach human breathing zones. Unfortunately, however, conditions in hundreds of millions of third world households are nearly ideal to optimize exposures from emissions. A large, but unknown, fraction of daily cooking is done in unvented stoves, i.e., in which the emissions are released directly into the living area and not vented through a chimney or hood. Although there are not systematic surveys in developing-country settings, about 200 studies of indoor air quality (IAQ) measurements in households using solid fuels have been published, more than half from China.[12] Table 11 shows a summary of

[12] See the Chinese IAQ database 203.Sinton, J.E., et al., *Indoor Air Pollution Database for China*, in *Human Exposure Assessment Series*. 1996, World Health Organization, United Nations Environment Program: Geneva. and

these studies for the two most widely measured pollutants, particulates and carbon monoxide.These studies have been published between 1968 and March 2003. The studies in South Asia were mainly conducted in Nepal and in India, with only one reported from Bangladesh. The studies in Africa come mainly from Kenya, Gambia, and South Africa. Most of the Latin American studies have been conducted in Guatemala and Mexico.

Table 11: Summary of indoor air quality studies: household pollution levels in developing countries outside of China. (Source: [204]).

	Number of Studies	Total Number of Samples	TSP (mg/m^3)		PM_{10} (mg/m^3)		$PM_{2.5}$ (mg/m^3) [*3]		CO (mg/m^3)	
			Meal[*4]	Daily[*5]	Meal[*4]	Daily[*5]	Meal[*4]	Daily[*5]	Meal[*4]	Daily[*5]
Bangladesh	1	53							15-26	
Bolivia	1	169			3700					
Brazil	1	23			530					
Burundi	1	2								42
Ethiopia[*1]	1	N/A							48	
Gambia	1	12				1600-2200				
Ghana	1	21			590				9	
Guatemala	7	768		280-840		190-1200	450-27000	97-1900	2-149	1.2-17
India	13	1009	646-16000		900-1100	506	110-2100	1300-1500	5-216	
Kenya	4	199	3776			1800-3900		630-3500		5-60
Malaysia	1	10		300						3
Mexico	5	191			280-1200	97-290		890	10-22	
Mozambique	1	114			1200				48	
Nepal	5	127	710-8800	4600-8420	4700		1700-5700		14-360	14-52
New Guinea	1	9		360-670						13-24
Nigeria[*2]	1	28		108					1076	
South Africa	1	20		1725					79-180	92
Zambia	1	89			890				10	
Zimbabwe	1	34	1357							

[*1] Unspecified averaging time for CO measurements
[*2] A few minutes averaging time for CO measurements
[*3] PM_5, PM_4, $PM_{3.5}$, and $PM_{2.5}$ have been lumped into one category ($PM_{2.5}$)
[*4] Daily column corresponds to samples taken for 12+ hours
[*5] Meal column corresponds to samples taken for less than 3 hours, and in some cases less than 8 hours

the non-Chinese IAQ database 204.Saksena, S., L. Thompson, and K.R. Smith, *Indoor Air Pollution and Exposure Database*. 2003, School of Public Health, University of California: Berkeley.) both available at http://ehs.sph.berkeley.edu/krsmith/

Most of these studies were conducted in rural settings and attempted to characterize the distribution of concentration levels in the kitchens, with the earlier studies reported from the highlands in different parts of the world. Also, there is little information available on seasonal affects or differences across the various meals cooked in a day. Meal cooking time varied from study to study and this time ranges anywhere between 30 minutes to 3 hours with one study reporting up to 8 hours. Although several studies made comparisons between the traditional and the improved stove, in this summary table that distinction has not been made.

A highly polluting source releasing pollution indoors at times and places when people are always present (household cooking) has a potential to produce high exposure. Put another way, the associated *intake fraction (*fraction of material released that is actually inhaled by someone) is orders of magnitude higher for indoor than for outdoor sources of air pollution [205]. Although the uncertainties are large, the available evidence would indicate that the total exposure to combustion-derived small particles from indoor solid fuel use is larger than that from all outdoor sources of pollution in the world [196].[13]

Even in communities where most households use chimneys, however, the intake fraction can be substantially higher than for typical outdoor sources since the smoke may sit in the area among the houses in what is called "neighborhood pollution." Such pollution may not be fully reflected by ambient monitoring data, but nevertheless substantially influence local exposures [198]. This same phenomenon exists in developed countries as well, for example from household fireplaces as discussed in Section 5. Because of their almost universal role as household cook, the highest exposures from household use of solid fuels, however, seem usually to occur to women and their youngest children who are with them during cooking, although significant exposures can accrue to other household members as well [206].

Although few studies have actually linked IAQ measurements to ill-health, a growing number of epidemiological studies have found significant risks of various exposure indicators and ill-health in developing-country biomass-using households. Such exposure indicators include use of solid or "dirty" fuel versus liquid/gas "clean" fuel; using a stove with flue or without; and, for babies, being carried on their mother's back while cooking or not. Taking advantage of the increasing number of such studies, the recent global Comparative Risk Assessment (CRA) managed by the WHO included indoor as well as outdoor air pollution among the 26 risk factors examined [207].

The available epidemiologic evidence was divided into three categories, as shown in Table 12. Considered sufficient in quantity and quality to justify inclusion in the global CRA was evidence only in the top category: acute lower respiratory infections (ALRI - pneumonia) in young children, chronic obstructive pulmonary disease (COPD) in adults, and lung cancer in adults (for coal smoke only). The odds ratios shown in the table are the result of meta-analyses of the data in published studies that met the criteria for inclusion [197].

A number of epidemiologic studies have also been published for these populations in relation to other important diseases, however. As shown in the second category ("Moderate" evidence) are simple means of odds ratios in available studies showing significant impact of use of solid fuels

[13] See also Table 3, which shows the estimate for total global health effects for indoor and outdoor air pollution.

Table 12: Health effects of use of household fuels in developing countries (Source: [197]).

Disease	Population affected	Relative risk (95% confidence interval)	Strength of Evidence
COPD	Females>15 years	3.2 (2.3, 4.8)*	Strong
	Males > 15 years	1.8 (1.0, 3.2)*	Intermediate
ALRI	Children < 5 years	2.3 (1.9,2.7)*	Strong
Lung cancer	Women >15 years	1.9 (1.1,3.5)*	Strong
	Men >15 years	1.5 (1.0,2.5)*	Intermediate
Blindness (cataracts)	Females > 15 years	1.3-1.6**	Intermediate
Tuberculosis	Females > 15 years	1.5-3.0**	Intermediate

* Based on formal meta-analyses
** Range of results in published studies

on tuberculosis and cataracts. Although consistent, the number and character of these studies was not considered sufficiently persuasive to include these diseases in the CRA. Similarly, although studies of outdoor air pollution, ETS, and other sources of particle exposure indicate impacts on asthma and heart disease, no good studies are yet available in the populations of interest for household solid fuel use.

Table 13: Global Burden of Disease and Pre-Mature Death Due to Major Environmental Risk Factors in 2000 (Source: [208]).

	Poor Countries	Mid-Income Countries	Rich Countries	World
Population (million)	2343	2424	1358	6125
DALYs (million disability-adjusted lost life years)	846	406	214	1,467
Unsafe water, sanitation and hygiene	5.5%	1.8%	0.4%	3.7%
Indoor smoke from solid fuels	3.6%	1.9%	0.3%	2.6%
Occupational risks	1.1%	2.4%	1.5%	1.5%
Lead exposure	0.7%	1.4%	0.6%	0.9%
Urban air pollution	0.3%	1.0%	0.5%	0.5%
Climate change	0.6%	0.1%	0.0%	0.4%
Total Environmental Burden of Disease, % of total for region	11.8%	8.5%	3.4%	9.7%
Deaths (thousand)	26700	16000	13000	55700
Unsafe water, sanitation and hygiene	1538	172	20	1730
Indoor smoke from solid fuels	1039	558	22	1619
Occupational risks	2393	640	176	3209
Lead exposure	93	69	72	234
Urban air pollution	220	426	154	800
Climate change	148	5	0	153
Total Environmental Mortality in region (thousand)	5431	1870	444	7745

Using only the "Strong" evidence category in Table 12 and the distribution of solid fuel use shown in Figure 6, the total impact of IAQ from solid fuel use calculated in the CRA is shown and compared to other environmental risk factors in the CRA in Table 13.

At 1.6 (0.8-2.4) million deaths and 2.6% of the global burden of disease (as measured in lost life-years), IAQ ranks second only to poor water/sanitation/hygiene among environmental health risk factors. Among all major policy-relevant risk factors, indoor air pollution from solid fuel is tenth globally, and fourth in least-developed countries.[14] See Figure 8.

7.2 Biomass smoke in LDC Cities

Ambient air pollutants come primarily from combustion of fossil fuel. But in many cities and rural areas in developing countries, residential space heating and cooking with solid fuels, mostly biomass and coal, can also contribute significantly to the ambient pollution. Several studies have been conducted in developing countries quantifying the contribution of biomass smoke in cities. Begum et al. (2004) report contribution from biomass combustion to be 12% in Dhaka and 50% in Rajshahi, Bangladesh [209]. From a study conducted by Zheng et al. (in preparation) in three sites in Hong Kong, they find that 9-10% of the organic carbon in $PM_{2.5}$ comes from biomass smoke. Chowdhury et al report contributions from biomass in three Indian cities by season as seen in Table 14([210]).

Table 14: Contribution of Biomass Smoke in $\mu g/m^3$ to the Ambient $PM_{2.5}$ in India (Source: [210]).

	Spring	% of $PM_{2.5}$	Summer	% of $PM_{2.5}$	Autumn	% of $PM_{2.5}$	Winter	% of $PM_{2.5}$
Delhi	25	22%	5	10%	33	21%	66	29%
Mumbai	5	13%	N/A	N/A	14	21%	11	13%
Kolkata	10	19%	6	24%	14	32%	51	17%

Based on 5-7 24-h samples per season.

High concentration of biomass smoke in the colder months compared to the warmer months can be explained by the regional meteorology where the monsoon rains are dominant in the summer months followed by the dry winter when pollutants are trapped inside the inversion layer. Also, during the winter months, there is a tendency to use biomass fuel for heating purposes leading to the higher concentrations in Delhi and Kolkata, 66/51 $\mu g/m^3$ $PM_{2.5}$, representing 29/17% of the total $PM_{2.5.}$ Although a bit higher in absolute concentrations, this winter pattern is also found in developed- country urban areas using wood fuels.

[14] The summary results of the CRA were released in the World Health Report 207.Ezzati, M., et al., *Selected major risk factors and global and regional burden of disease.* Lancet, 2002. **360**(9343): p. 1347-60, 208.World Health Organization, *World Health Report: Reducing Risks, Promoting Healthy Life*. 2002, World Health Organization: Geneva. and were published in detail in 197.Smith, K.R., S. Mehta, and M. Maeusezahl-Feuz, *Indoor Smoke from Household Solid Fuels*, in *Comparative Quantification of Health Risks: Global and Regional Burden of Disease due to Selected Major Risk Factors*, M. Ezzati, et al., Editors. 2004, World Health Organization: Geneva. p. 1437-1495.

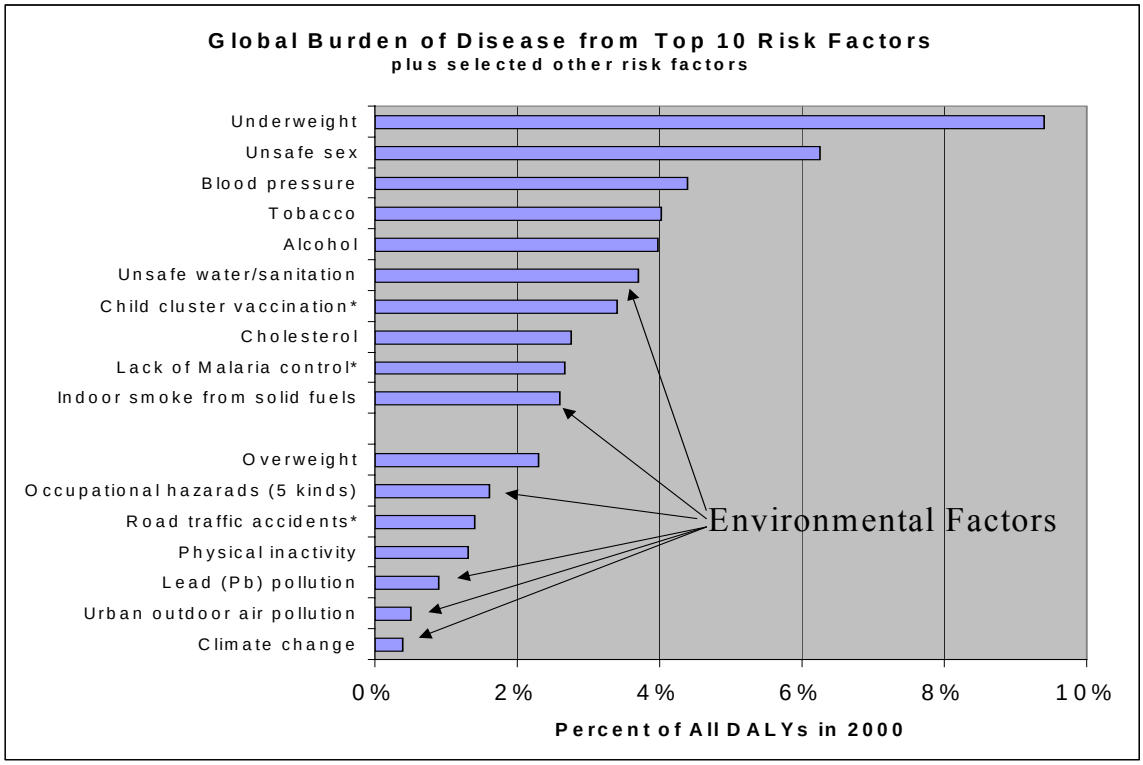

Figure 8: Estimated burden of disease in 2000 measured as lost healthy life years (DALYs) from major preventable risk factors. Note importance of indoor smoke from solid fuels, which are mostly biomass. Those marked * are based on outcomes in the Global Burden of Disease database. See Footnote 14.

8 SUMMARY, RECOMMENDATIONS, AND CONCLUSION

Even though woodsmoke is natural, it is not benign. Indeed, there is a considerable and growing body of epidemiologic and toxicologic evidence that both acute and chronic exposures to woodsmoke in the developed world are associated with adverse health impacts.

SUMMARY

Chemical composition. Woodsmoke contains thousands of chemicals, many of which are known to be toxic, including criteria air pollutants such as fine particles, carbon monoxide, and nitrogen oxides; cilia-toxic respiratory irritants such as phenols, cresols, acrolein, and acetaldehyde; carcinogen volatile organics such as benzene, formaldehyde, and 1,3 butadiene; as well as many carcinogenic cyclic compounds such as polyaromatic hydrocarbons. It contains at least five chemical groups classified as known human carcinogens by the International Agency for Research on Cancer (IARC) and others categorized by IARC as probable or possible human carcinogens. Among the currently regulated pollutants in woodsmoke, fine particles (PM2.5) are the most elevated with regard to existing standards as well as serve as the best exposure metric in most circumstances.

Toxicology. Available animal studies indicate that total woodsmoke has significant impacts on the respiratory immune system and, at high doses, can produce long-term or permanent lesions in lung tissues. Based on relatively few studies, these effects seem most strongly associated with the particle phase of woodsmoke. Woodsmoke is also mutagenic and carcinogenic in laboratory and animal studies, but less so than typical coal smoke. Unfortunately, not enough is currently known to reliably distinguish the effects of different types of biomass smoke (e.g., smokes from wood versus crop wastes)

Exposures. Measured in the form of fine particles, significant woodsmoke exposures, mostly in winter, occur indoors and outdoors in areas of North America where wood is used for residential heating or in fireplaces. Woodsmoke often contributes a significant fraction of ambient particle levels in such areas, both on a daily and an annual basis. Woodsmoke-specific chemical tracers provide the potential for increased understanding of the contribution of ambient woodsmoke concentrations to indoor and personal woodsmoke exposures, in settings where multiple sources of fine particles are present. Relatively high levels of biomass smoke exposures from wildland fires and agricultural burning occur infrequently. Occupational exposures can be extremely high for firefighters.

Epidemiology: Wildland and Agricultural Burning. Although rarely combined with individual exposure assessment, a number of studies have found associations of wildfires with emergency room visits for both upper and lower respiratory tract illnesses, respiratory symptoms (including asthma), and decreased lung function. In one study, wildfire smoke resulting from a major episode in Southeast Asia was associated with increased cardiopulmonary mortality, although this is the only study to have specifically evaluated mortality as an outcome in relation to wildfire smoke. Though less well documented, exposures to smoke from agricultural burning may also be linked with adverse respiratory outcomes, particularly exacerbations of asthma. Only one study has examined the efficacy of various interventions to reduce morbidity among the general population during wildfires. Thus, there are few data on which to base

recommendations to the general population on effective measures to reduce exposures. Additional research in this area is needed to inform the guidance offered by public health officials. Several studies have documented cross-seasonal effects of wildland fire smoke exposure on firefighters' lung function. The long-term consequences of repeated occupational exposures to such extraordinarily high concentrations of vegetation smoke have not been investigated, however.

Epidemiology: Residential Woodburning. Surprisingly few studies have been done in developed countries of the health impacts of woodsmoke, partly due to the difficulty of separating out the woodsmoke portion of the health effects of mixtures of particles in the environment. In addition, most available studies are ecologic in design, limiting the ability to derive strict causality. Those that have been done, however, indicate that exposure to the smoke from residential wood burning is associated with a variety of adverse respiratory health effects, which are no different in kind and, with present knowledge, show no consistent difference in magnitude of effect from other combustion-derived ambient particles. Only one study seems to be available focusing on cardiovascular effects, which shows a somewhat smaller effect than those related to general ambient particles. No studies seem to be available related to cancer endpoints.

Exposures and Epidemiology: Developing Countries. Exposures to biomass smoke are common in nearly half the households in the world that use wood, crop residues, or animal dung for cooking and heating. Although poorly characterized overall, such biomass smoke exposures are substantially higher than those in developed countries. In more than a dozen studies each, two important diseases, chronic obstructive lung diseases and acute lower respiratory infections have been strongly associated with these household exposures, leading to an estimate by WHO of some 1.3 million premature deaths per year globally. Multiple studies have also shown relationships with tuberculosis, cataracts, adverse birth outcomes, and asthma. Biomass smoke is also an important part of outdoor air pollution in many developing-country cities, although no studies seem to have been done to separate out their impacts from other particles.

Hundreds of studies have examined the relationships between outdoor pollutants and disease around the world. To the extent that woodsmoke contributes to individual airborne chemicals, such as carbon monoxide, nitrogen and sulfur oxides, and benzene, the conclusions of these studies can be applied to those same chemicals in woodsmoke because, being specific molecules, they do not vary by source. It is less clear, however, whether woodsmoke-associated particles are differentially toxic relative to particles from other sources, which have been dominant in most ambient epidemiologic studies. Since particles are probably the single most important health-related constituent of woodsmoke,[15] an assessment of their hazard is crucial to evaluation the overall hazard of the woodsmoke mixture.

Perhaps because of long human association with woodsmoke particles and the consequent perception that they are natural and thus somehow less hazardous than particles from more modern sources, such as fossil fuels, there has been some reluctance to treat them equally, for example in emissions standards. This effectively constitutes a decision that, in contrast to diesel particles for example, woodsmoke particles are actually less hazardous per unit concentration

[15] In terms of health impact. The largest constituent of woodsmoke, in terms of mass, is carbon monoxide.

than "average" ambient particles. Given the current state of the science, however, this perception is not based on any compelling evidence.

Although the depth and relevance of evidence in each of the areas reviewed in this report is not as extensive as would be desirable to pin down the health effects in a thorough and quantitative manner for the exposures of interest in developed countries, the cumulative impact of the evidence of chemical and physical composition, animal and *in vitro* toxicology, the extent of human exposures; public and occupational epidemiology of vegetation fires; and the epidemiology related to household wood combustion make a strong case that woodsmoke particles are hazardous to human health. Since source apportionment studies show that woodsmoke is a major contributor to PM in many communities, it is very likely that woodsmoke exposure plays a role in the spectrum of adverse effects linked to PM exposure.

The quite large effects seen at higher exposures in the developing world bolster the case against woodsmoke. Although not reviewed here, the health impacts of the most well studied smoke from burning biomass, tobacco smoke, also strongly supports this conclusion, particularly the well documented ill-health from passive smoking (environmental tobacco smoke or secondhand tobacco smoke), for which exposure levels are much closer to those typically experienced for woodsmoke in developed countries than exposures experienced by active smokers.

RECOMMENDATIONS FOR FURTHER RESEARCH

Several lines of research need to be encouraged in order to improve knowledge about the health effects of smoke from combustion of wood and other biomass:

Chemical composition. Better understanding of the similarities and differences of smokes from important categories of biomass including wood and major crop residues under typical combustion conditions
Utilization of recent advances in analytical chemistry (e.g. LC/MS techniques) to identify and quantify a wider range of chemicals in biomass smoke.

Toxicology. More long-term animal inhalation studies at relevant indoor and outdoor woodsmoke concentrations, as well as comparative studies of particle phase woodsmoke and particles from other sources.

Controlled Human Exposures.
More chamber studies to elucidate the acute effects of high exposure to major types of biomass smokes.

Exposure Assessment. More source and exposure apportionment studies to determine the degree to which woodsmoke contributes to particle exposures from outdoor and household sources
Work should continue to develop reliable biomarkers for biomass smoke exposure.

Epidemiology. Better ways to combine source and exposure apportionment assessments in epidemiologic studies designed to determine the proportion of particle health effects due to woodsmoke (and other major sub-categories of particles)

Take advantage of local situations where epidemiology can be done with populations exposed primarily to woodsmoke particles, at least seasonally.
Conduct studies focused on cardiovascular effects to compare with risks from general ambient particles.
Consider case-control or other study designs to assess cancer risks.

Exposures and Epidemiology: Developing Countries. Joint work by researchers from the developed and developing world to address the potential health benefits of practical interventions to control exposures to biomass smoke, a problem common to both parts of the world.

CONCLUSION

Recognizing the limitations of current knowledge and need for additional information, we nevertheless offer preliminary answers to the questions we raised in the introduction:

The hazards of woodsmoke as a mixture. Because woodsmoke is made up of a plethora of different chemicals it is difficult to accurately assess the health effects by simply adding up the effects of individual constituents. Moreover, the effects of toxic mixtures are often times not reflected by the toxic potential of each individual mixture constituent. Particularly in high exposure situations with fresh woodsmoke, as with occupational exposures or vegetation fire episodes, there may be need to derive indices of exposure that take into account a range of toxic endpoints so that regulations can be adequately protective.

Woodsmoke particles. There is no persuasive evidence that woodsmoke particles are significantly less dangerous for respiratory disease than other major categories of combustion-derived particles in the same size range. There is too little evidence available today to make a judgment about the relative toxicity of woodsmoke particles for cardiovascular or cancer effects.

9 APPENDICES

Appendix 1: Acute lung injury

Authors	Animal Model	Wood Type	Route of Exposure	Dose; Duration of exposure; Post-exposure timepoint	Endpoints	Outcome
[42]	Rabbits	White pine	Head only inhalation	30.4cm long x 1.9cm square (wt = 30 ± 0.6g); 25, 30, 40 or 45 min; 6, 24 or 72 hr	Lung pathology	Tracheobronchial epithelial cell injury
[43]	Rabbits	Douglas fir	Whole body inhalation	COHb = 16.4%; ~60 min; 24 hr	Lung pathology and cellular morphology	Tracheobronchial epithelial cell inju loss; mucociliary escalator disorganization; altered alveolar macrophage morphology and cell number
[45]	Rabbits	Douglas fir	Nose only Inhalation	COHb = 0 –>20%; 30 – 120 min; Immediately	Alveolar macrophage Function and cell number	Reduced macrophage phagocytosis and intracellular killing; increased lavageable cell numbers
[10]	Rats	Douglas fir and pine	Whole body inhalation (subsequent to scald or sham burn)	COHb = 19%; 16.25 min; 1, 12, 24, 48, 96 hr	Tracheo-bronchial pathology; airway cellular/lung water content; antioxidant enzymes; lipid peroxidation; total lavageable protein	Tracheal erosion and epithelial Cell loss; oxidative stress; reduced (albeit, not significantly) antioxidant levels and increased lavageable protein levels
[51]	Guinea pigs	Douglas fir	Whole body inhalation	Chamber CO=1000–9000ppm (15, 18, 23, 30, 40, 50, 60 g of wood chips); 30 min; 0.5 and 3 hr, 1, 2, 3, 5, 7, 9, 11, 13, 17, 21,25, 29, 36, 43, 50, 57 d	Lung tidal volume and respiratory frequency during breathing and following CO_2 challenge	Transient decrease in lung function in response to CO_2 challenge

[54-56, 58, 68]	Guinea pigs	Luan wood	Anesthetized and artificially (mechanically)-ventilated	PM conc. = 25 mg/m^3 (WS delivered in 10mL volumes);*	Airway responsiveness	Airway hyperresponsiveness to bronchonstrictor challenge (i.e., substance P, capsaicin and prostaglandins)
[59]	Rats	Luan Wood	Tracheostomy or Open chest ventilation	Particle-free or whole WS effluents (6 ml),*	Ventilatory response	WS both slowed and augmented respiration as a result of effects on bronchopulmonary C-fiber nerve endings and vagal bronchopulmonary afferent nerves
Nieman et al., (1980 – 1995)	Dogs, rabbits	Douglas fir (plywood sawdust moisten -ed with kerosene)	Artificial ventilation, open-chest or isolated lung exposure	COHb = ~20%,*	Static lung compliance lung water; blood gas measurements; pulmonary vascular resistance; pulmonary surface tension [@]	Reduced pulmonary compliance and increased lung surface tension and edema
[71]	Mice	Pine wood	Whole body Inhalation	0 – 17 mg wood/ g body wt. (COHb =10–50%); 15 - 20 min (depending upon mouse species); 24, 48, 72 hr	Mortality; lung pathology; lavageable cell counts; TNFα levels; static lung compliance	Lower airways pathology (Type II cell hypertrophy, Type I cell hyperplasia, cytoplasmic blebbing and vacuolization and sloughing of the bronchiolar epithelium); increased lavageable cell count; macrophage and lymphocyte infiltration into the lungs

* Variable depending upon specific study
# See additional citations by these same Authors in references
[@] Specific endpoints dependent upon individual study

Appendix 2: Repeated and sub-chronic *in vivo* woodsmoke exposure

Authors	Animal Model	Wood Type	Route of Exposure	Dose; Duration of exposure; Post-exposure timepoint	Endpoints	Outcome
[73]	Rats	Not defined	Whole body Inhalation	1 g wood/15 min; 75 min/d (15 min each, 5 times/d), 6d/wk for 15, 30 or 45 d; 15, 30 and 45 d	Body weight; hematological parameters; lung histology	Lung pathology (i.e., bronchiolits; hyperplasia and hypertrophy of bronchiolar epithelial lining cells; blood vessel congestion; edema; PMN influx, mild emphysematic changes; alveolar septae thickening; marginal hematologic alterations; eosinophilia
[72, 74, 7	Rats	Red oak	Nose-only inhalation	750 µg PM/m^3 (< 2 ppm CO; 3 ppb NO_x; 1.5 ng total PAH/m^3); 1 hr/d for 4 d; 3, 24, 72, 120, 264 hr	Clearance of *Staphylococcus aureus* from the lungs; macrophage -mediated oxyradical production	Decreased host resistance against pulmonary bacterial infection; reduced $O_2^{\cdot -}$ production
[76]	Rats	Pinon wood	Whole body inhalation	1 or 10 mg PM/m^3; 4 or 12 wk (3 h/d, 5 d/wk); Immediate	Pulmonary function; lung histology; lavageable markers of lung cell damage; lymphocyte proliferation; body weight	Minor reduction in lung function; chronic inflammation; laryngeal squamous metaplasia; transient mucous cell metaplasia
[77]	Mice and rats	Not defined	Whole body inhalation of WS generated from open pit burning	14.9 mg/m^3 TSP (80 mg/m^3 CO; 0.05 mg/m^3 SO_2; 0.27 mg/m^3 H_2SO_4); 15 mo for mice and 19 mo for rats (12hr smoke:12 hr none)	Malignant and non -malignant lung tumor incidence	Increased lung cancer incidence in mice

Appendix 3: *Ex vivo* and *in vitro* woodsmoke exposure studies

Authors	**Animal Model**	**Wood Type**	**Route of Exposure**	**Dose; Duration of exposure; Post-exposure timepoint**	**Endpoints**	**Outcome**
[79]	Rabbit	Pine wood	Tracheal Explants	Unknown; 5, 10, 15, 20 min; 48 and 96 hr	Tissue histology (based on cytological and biochemical criteria)	Degeneration of the mucociliary epithelial sheath; altered epithelial cell morphology
[80]	Not defined	Red oak	Organ cultured eye lenses	100 μL (1:20 dilution) smoke condensate generated from burning 2 g of wood chips; 10 min; 24 hr	Physiological integrity and morphology of organ cultured lenses	Eye lens opacification; hyperplasia; hypertrophy; multilayering of epithelial cells
[9]	Mouse	Douglas fir and pine wood	Cultured mouse macrophage (RAW 264.7)	50 - 100 μL (1:7 dilution) of liquefied smoke generated from 100 g wood bark; 0, 5, 10, 15, 20 min; 24 hr	Free radical generation; DNA damage; NFκB activation; TNFα release	DNA damage; increased lipid peroxidation; TNFα production and NFκB activation

10 REFERENCES

1. UNDP, *World Energy Assessment*. 2004, United Nations Development Programme: New York City.
2. Koopmans, A., *Trends in Energy Use*. 1999, RWEDP, Food and Agriculture Organization of the United Nations: Rome and Bangkok.
3. IEA, *Energy Balances of OECD Countries*. 2004, International Energy Agency: Paris.
4. Smith, K.R., Biofuels, Air Pollution, and Health: A Global Review. 1987, New York: Plenum.
5. Tuthill, R., *Woodstoves, Formaldehyde, And Respiratory Disease.* American Journal of Epidemiology, 1984. 120(6): p. 952-955.
6. Traynor, G., et al., *Indoor air pollution due to emissions from wood-burning stoves.* Env Sci Technol, 1987. 21: p. 691-697.
7. Koenig, J. and P. WE, Air pollutants and the respiratory system: Toxicity and pharmacologic interventions. Clin. Toxicol., 1991. 29: p. 401-411.
8. Koenig, J., et al., Pulmonary function changes in children associated with fine particulate matter. Environ Res, 1993. 63(1): p. 26-38.
9. Leonard, S.S., et al., Woodsmoke particles generate free radicals and cause lipid peroxidation, DNA damage, NFkappaB activation and TNF-alpha release in macrophages. Toxicology, 2000. 150(1-3): p. 147-57.
10. Dubick, M., et al., Indices of antioxidant status in rats subjected to woodsmoke inhalation and/or thermal injury. Toxicology, 2002. 176(1-2): p. 145-57.
11. Simoneit, B.R.T., et al., Levoglucosan, a tracer for cellulose in biomass burning and atmospheric particles. Atmospheric Environment, 1998. 33(2): p. 173-182.
12. Rogge, W.F., et al., Sources of fine organic aerosol. 9. Pine, oak and synthetic log combustion in residential fireplaces. Environmental Science and Technology, 1998. 32(1): p. 13-22.
13. Schauer, J.J., et al., Measurement of emissions from air pollution sources. 3. C1-C29 organic compounds from fireplace combustion of wood. Environ Sci Technol, 2001. 35(9): p. 1716-28.
14. Fine, P.M., G.R. Cass, and B.R. Simoneit, Chemical characterzation of fine particle emissions from the fireplace combustion of woods grown in the Southern United States. Environ Sci Technol, 2002. 36(7): p. 1442-51.
15. McDonald, J.D., et al., *Fine Particle and Gaseous Emission Rates from Residential Wood Combustion.* Environmental Science and Technology, 2000. 34(11): p. 2080-2091.
16. Oros, D.R. and B.R.T. Simoneit, Identification and emission factors of molecular tracers in organic aerosols from biomass burning. Part 1. Temperate climate conifers. Applied Geochemistry, 2001. 16(13): p. 1513-1544.
17. Oros, D.R. and B.R.T. Simoneit, Identification and emission factors of molecular tracers in organic aerosols from biomass burning. Part 2. Deciduous trees. Applied Geochemistry, 2001. 16(13): p. 1545-1565.
18. Lee, S. and K. Baumann, 2005.
19. Simoneit, B.R.T., et al., Lignin pyrolysis products, lignans, and resin acids as specific tracers of plant classes in emissions from biomass combustion. Environmental Science and Technology, 1993. 27(12): p. 2533-41.

20. Rinehart, L.R., et al., Characterization of PM2.5 associated organic compounds of emission sources collected during the California regional PM10/PM2.5 Air Quality Study., A.A.f.A. Research, Editor. 2002: Charlotte, NC.
21. Kleeman, M.J., J.J. Schauer, and G.R. Cass, Size and Composition Distribution of Fine Particulate Matter Emitted from Wood Burning, Meat Charbroiling, and Cigarettes. Environ Sci Technol, 1999. 33(20): p. 3516-3523.
22. Hays, M.D., et al., Speciation of gas-phase and fine particle emissions from burning of foliar fuels. Environ Sci Technol, 2002. 36(11): p. 2281-95.
23. Echalar, F., et al., Aerosol emission by tropical forest and savanna biomass burning : characteristic trace elements and fluxes. Geophysical Research Letters, 1995. 22(22): p. 3039-3042.
24. Andrae, M., et al., *Biomass burning emissions and associated haze layers over Amazonia.* Journal of Geophysical Research, 1988. 93(D2): p. 1509-1527.
25. Venkataraman, C., et al., Residential biofuels in South Asia: Carbonaceous aerosol emissions and climate impacts. Science, 2005. 307: p. 1454-1456.
26. Koch, D. and J. Hansen, Distant origins of Arctic black carbon: A Goddard Institute for Space Studies ModelE experiment. J of Geophysical Research, 2005. 110: p. 4204-.
27. Pryor, W.A., Biological effects of cigarette smoke, woodsmoke, and the smoke from plastics: the use of electron spin resonance. Free Radic Biol Med, 1992. 13(6): p. 659-76.
28. Cupitt, L.T., W.G. Glen, and J. Lewtas, Exposure and risk from ambient particle-bound pollution in an airshed dominated by residential wood combustion and mobile sources. Environ Health Perspect, 1994. 102(S4): p. 75-84.
29. Claxton, L.D., et al., A comparative assessment of Boise, Idaho ambient air fine particle samples using the plate and microsuspension Salmonella mutagenicity assays. Sci Total Environ, 2001. 275(1-3): p. 95-108.
30. Khalil, M.A.K. and R.A. Rasmussen, *Tracers of woodsmoke.* Atmospheric Environment, 2003. 37(9-10): p. 1211-1222.
31. Fine, P.M., G.R. Cass, and B.R.T. Simoneit, Chemical characterization of fine particle emissions from fireplace combustion of woods grown in the Northeastern United States. Environ Sci Technol, 2001. 35(13): p. 2665-2675.
32. Hawthorne, S.B., et al., PM-10 high-volume collection and quantitation of semi- and nonvolatile phenols, methoxylated phenols, alkanes, and polycyclic aromatic hydrocarbons from winter urban air and their relationship to woodsmoke emissions. Environmental Science and Technology, 1992. 26(11): p. 2251-62.
33. Schauer, J.J. and G.R. Cass, Source apportionment of wintertime gas-phase and particle-phase air pollutants using organic compounds as tracers. Environ Sci Technol, 2000. 34(9): p. 1821-1832.
34. Larsen, R.K., 3rd and J.E. Baker, Source apportionment of polycyclic aromatic hydrocarbons in the urban atmosphere: a comparison of three methods. Environ Sci Technol, 2003. 37(9): p. 1873-81.
35. Katz, B., et al., Estimating the contribution of woodsmoke to PM2.5 at an urban IMPROVE site in Seattle, WA. in prep., 2004.
36. Larson, T., et al., Source apportionment of indoor, outdoor, and personal PM2.5 in Seattle, Washington, using positive matrix factorization. J Air Waste Manag Assoc, 2004. 54(9): p. 1175-87.

37. Sakuma, H. and T. Ohsumi, Studies on cigarette smoke. Part VIII. Particulate phase of cigarette smoke. Agric. biol. Chem, 1980. 44(3): p. 555-561.
38. Hawthorne, S.B., et al., Collection and quantiation of methoxylated phenol tracers for atmospheric pollution from residential wood stoves. Environ Sci Technol, 1989. 23(4): p. 470-475.
39. Simpson, C.D., et al., Determination of methoxyphenols in ambient atmospheric particulate: tracers for wood combustion. Environ. Sci. Technol, 2005. 39(2): p. 631-637.
40. Schauer, J.J., et al., Measurement of Emissions from Air Pollution Sources. 2. C1 through C30 Organic Compounds from Medium Duty Diesel Trucks. Environmental Science and Technology, 1999. 33(10): p. 1578-1587.
41. Guillen, M.D. and M.L. Ibargoitia, Influence of the moisture content on the composition of the liquid smoke produced in the pyrolysis process of Fagus sylvatica L. wood. J Agric Food Chem, 1999. 47(10): p. 4126-36.
42. Thorning, D.R., et al., Pulmonary responses to smoke inhalation: morphologic changes in rabbits exposed to pine woodsmoke. Hum Pathol, 1982. 13(4): p. 355-64.
43. Loke, J., et al., Rabbit lung after acute smoke inhalation. Cellular responses and scanning electron microscopy. Arch Surg, 1984. 119(8): p. 956-9.
44. Rasp, F.L., et al., Reversible impairment of the adherence of alveolar macrophages from cigarette smokers. Am Rev Respir Dis, 1978. 118(6): p. 979-86.
45. Fick, R.B., Jr., et al., Alterations in the antibacterial properties of rabbit pulmonary macrophages exposed to woodsmoke. Am Rev Respir Dis, 1984. 129(1): p. 76-81.
46. Demling, R.H. and C. LaLonde, Moderate smoke inhalation produces decreased oxygen delivery, increased oxygen demands, and systemic but not lung parenchymal lipid peroxidation. Surgery, 1990. 108(3): p. 544-52.
47. Demling, R., et al., Changes in lung and systemic oxidant and antioxidant activity after smoke inhalation. Shock, 1994. 1(2): p. 101-7.
48. Lalonde, C., et al., Lung and systemic oxidant and antioxidant activity after graded smoke exposure in the rat. Circ Shock, 1994. 42(1): p. 7-13.
49. Traber, D.L. and D.N. Herndon, Pathophysiology of smoke inhalation, in Respiratory Injury: Smoke Inhalation and Burns. 1986. p. 61-71.
50. Hubbard, G.B., et al., *The morphology of smoke inhalation injury in sheep.* J Trauma, 1991. 31(11): p. 1477-86.
51. Wong, K.L., et al., Evaluation of the pulmonary effects of woodsmoke in guinea pigs by repeated CO2 challenges. Toxicol Appl Pharmacol, 1984. 75(1): p. 69-80.
52. Stephenson, S.F., et al., *The pathophysiology of smoke inhalation injury.* Ann Surg, 1975. 182(5): p. 652-60.
53. Garzon, A.A., et al., *Respiratory mechanics in patients with inhalation burns.* J Trauma, 1970. 10(1): p. 57-62.
54. Hsu, T.H., Y.L. Lai, and Y.R. Kou, Acetylcholine and tachykinin receptor antagonists attenuate woodsmoke-induced bronchoconstriction in guinea pigs. Eur J Pharmacol, 1998. 360(2-3): p. 175-83.
55. Hsu, T.H., Y.L. Lai, and Y.R. Kou, Smoke-induced airway hyperresponsiveness to inhaled woodsmoke in guinea pigs: tachykininergic and cholinergic mechanisms. Life Sci, 1998. 63(17): p. 1513-24.
56. Hsu, T.H. and Y.R. Kou, Airway hyperresponsiveness to bronchoconstrictor challenge after woodsmoke exposure in guinea pigs. Life Sci, 2001. 68(26): p. 2945-56.

57. Lin, Y.S. and Y.R. Kou, Acute neurogenic airway plasma exudation and edema induced by inhaled woodsmoke in guinea pigs: role of tachykinins and hydroxyl radical. Eur J Pharmacol, 2000. 394(1): p. 139-48.
58. Lin, Y.S., et al., Alleviation of woodsmoke-induced lung injury by tachykinin receptor antagonist and hydroxyl radical scavenger in guinea pigs. Eur J Pharmacol, 2001. 425(2): p. 141-8.
59. Kou, Y.R. and C.J. Lai, Reflex changes in breathing pattern evoked by inhalation of woodsmoke in rats. J Appl Physiol, 1994. 76(6): p. 2333-41.
60. Kou, Y.R., C.Y. Wang, and C.J. Lai, Role of vagal afferents in the acute ventilatory responses to inhaled woodsmoke in rats. J Appl Physiol, 1995. 78(6): p. 2070-8.
61. Wang, C.Y., C.J. Lai, and Y.R. Kou, Inhibitory influence of lung vagal C-fiber afferents on the delayed ventilatory response to inhaled woodsmoke in rats. Chin J Physiol, 1996. 39(1): p. 15-22.
62. Lin, Y.S. and Y.R. Kou, Reflex apneic response evoked by laryngeal exposure to woodsmoke in rats: neural and chemical mechanisms. J Appl Physiol, 1997. 83(3): p. 723-30.
63. Kou, Y.R., et al., Involvement of hydroxyl radical in the immediate ventilatory responses to inhaled woodsmoke in rats. Respir Physiol, 1997. 107(1): p. 1-13.
64. Lai, C.J. and Y.R. Kou, Stimulation of vagal pulmonary C fibers by inhaled woodsmoke in rats. J Appl Physiol, 1998. 84(1): p. 30-6.
65. Lai, C.J. and Y.R. Kou, Inhibitory effect of inhaled woodsmoke on the discharge of pulmonary stretch receptors in rats. J Appl Physiol, 1998. 84(4): p. 1138-43.
66. Lai, C.J. and Y.R. Kou, Stimulation of pulmonary rapidly adapting receptors by inhaled woodsmoke in rats. J Physiol, 1998. 508 (Pt 2): p. 597-607.
67. Kou, Y.R., et al., Neonatal capsaicin treatment alters immediate ventilatory responses to inhaled woodsmoke in rats. Respir Physiol, 1999. 116(2-3): p. 115-23.
68. Lin, Y.S., et al., Laryngeal C-fiber afferents are not involved in the apneic response to laryngeal woodsmoke in anesthetized rats. Life Sci, 2000. 66(18): p. 1695-704.
69. Ho, C.L. and Y.R. Kou, Protective and defensive airway reflexes evoked by nasal exposure to woodsmoke in anesthetized rats. J Appl Physiol, 2000. 88(3): p. 863-70.
70. Ho, C.Y. and Y.R. Kou, Mechanisms of woodsmoke-induced increases in nasal airway resistance and reactivity in rats. Eur J Pharmacol, 2002. 436(1-2): p. 127-34.
71. Matthew, E., G. Warden, and J. Dedman, *A murine model of smoke inhalation.* Am J Physiol Lung Cell Mol Physiol, 2001. 280(4): p. L716-23.
72. Thomas, P.T. and J.T. Zelikoff, *Air pollutants: modulators of pulmonary host resistance against infection*, in *Air Pollution and Health*, S.T. Holgate, et al., Editors. 1999, Academic Press: San Diego. p. 357-359.
73. Lal, K., et al., Histomorphological changes in lung of rats following exposure to woodsmoke. Indian J Exp Biol, 1993. 31(9): p. 761-4.
74. Zelikoff, J.T., *Woodsmoke, kerosene heater emission, and diesel exhaust.*, in *Pulmonary Immunotoxicology*, M.D. Cohen, J.T. Zelikoff, and R.B. Schlesinger, Editors. 2000, Kluwer Academic Publishers: Boston. p. 369-386.
75. Zelikoff, J.T., et al., *The Toxicology of Inhaled Woodsmoke.* Journal of Toxicology and Environmental Health, 2002. 5: p. 269-282.
76. Tesfaigzi, Y., et al., Health effects of subchronic exposure to low levels of woodsmoke in rats. Toxicol Sci, 2002. 65(1): p. 115-25.

77. Liang, C.K., et al., Natural inhalation exposure to coal smoke and woodsmoke induces lung cancer in mice and rats. Biomed Environ Sci, 1988. 1(1): p. 42-50.
78. Mumford, J.L., et al., Mouse skin tumorigenicity studies of indoor coal and wood combustion emissions from homes of residents in Xuan Wei, China with high lung cancer mortality. Carcinogenesis, 1990. 11(3): p. 397-403.
79. Bhattacharyya, S.N., et al., Smoke-induced inhalation injury: effects of retinoic acid and antisense oligodeoxynucleotide on stability and differentiated state of the mucociliary epithelium. Inflammation, 1998. 22(2): p. 203-14.
80. Rao, C.M., et al., Effect of smoke condensate on the physiological integrity and morphology of organ cultured rat lenses. Curr Eye Res, 1995. 14(4): p. 295-301.
81. Hytonen, S., I. Alfheim, and M. Sorsa, *Effect of emissions from residential wood stoves on SCE induction in CHO cells.* Mutat Res, 1983. 118(1-2): p. 69-75.
82. Alfheim, I. and T. Ramdahl, Contribution of wood combustion to indoor air pollution as measured by mutagenicity in Salmonella and polycyclic aromatic hydrocarbon concentration. Environ Mutagen, 1984. 6(2): p. 121-30.
83. Alfheim, I., et al., Short-term bioassays of fractionated emission samples from wood combustion. Teratog Carcinog Mutagen, 1984. 4(6): p. 459-75.
84. Asita, A.O., et al., Mutagenicity of woodsmoke condensates in the Salmonella/microsome assay. Mutat Res, 1991. 264(1): p. 7-14.
85. Harrison, R., B. Materna, and N. Rothman, *Respiratory health hazards and lung function in wildland firefighters.* Occup Med, 1995. 10(4): p. 857-870.
86. Betchley, C., et al., *Pulmonary function and respiratory symptoms in forest firefighters.* American Journal of Industrial Medicine, 1997. 31: p. 503-509.
87. Moen, B.E. and S. Ovrebo, Assessment of exposure to polycyclic aromatic hydrocarbons during firefighting by measurement of urinary 1-hydroxypyrene. J Occup Environ Med, 1997. 39(6): p. 515-9.
88. Feunekes, F.D., et al., Uptake of polycyclic aromatic hydrocarbons among trainers in a fire-fighting training facility. Am Ind Hyg Assoc J, 1997. 58(1): p. 23-8.
89. Burgess, J.L., et al., *Adverse respiratory effects following overhaul in firefighters.* J Occup Environ Med, 2001. 43(5): p. 467-.
90. Liou, S.H., et al., Biological monitoring of fire fighters: sister chromatid exchange and polycyclic aromatic hydrocarbon-DNA adducts in peripheral blood cells. Cancer Research, 1989. 49(17): p. 4929-4935.
91. Rothman, N., et al., Contribution of occupation and diet to white blood cell polycyclic aromatic hydrocarbon-DNA adducts in wildland firefighters. Cancer Epidemiol Biomarkers Prev, 1993. 2(4): p. 341-7.
92. Haas, N.S., et al., *Latent health effects in firefighters.* Int J Occup Environ Health, 2003. 9(2): p. 95-103.
93. Reinhardt, T.E., R.D. Ottmar, and A.J.S. Hanneman, Smoke Exposure Among Firefighters at Prescribed Burns in the Pacific Northwest, in United States Dept. of Agriculture Research Paper. 2000.
94. Burgess, J.L., et al., Serum pneumoproteins: a cross-sectional comparison of firefighters and police. Am J Ind Med, 2003. 44(3): p. 246-53.
95. Reinhardt, T.E. and R.D. Ottmar, Smoke exposure at western wildfires, in US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Research Paper. PNW-RP-525, 2000. 2000.

96. Materna, B.L., et al., *Occupational Exposures in California Wildland Fire Fighting.* American Industrial Hygiene Association Journal, 1992. 53(1): p. 69-76.
97. Slaughter, J.C., J.Q. Koenig, and T.E. Reinhardt, *Association between lung function and exposure to smoke among firefighters at prescribed burns.* J Occup Environ Hyg, 2004. 1(1): p. 45-9.
98. Bolstad-Johnson, D.M., et al., *Characterization of firefighter exposures during fire overhaul.* Aihaj, 2000. 61(5): p. 636-.
99. Mishra, V., Effect of indoor air pollution from biomass combustion on prevalence of asthma in the elderly. Environ Health Perspect, 2003. 111(1): p. 71-8.
100. Levesque, B., et al., *Wood-burning appliances and indoor air quality.* Sci Total Environ, 2001. 281(1-3): p. 47-62.
101. Guggisberg, M., et al., Respiratory symptoms and exposure to woodsmoke in an isolated northern community. Can J Public Health, 2003. 94(5): p. 372-6.
102. Eisner, M.D., et al., Exposure to indoor combustion and adult asthma outcomes: environmental tobacco smoke, gas stoves, and woodsmoke. Thorax, 2002. 57(11): p. 973-8.
103. Velema, J.P., et al., Burning Wood in the Kitchen Increases the Risk of Cervical Neoplasia in HPV-Infected Women in Honduras. International Union Against Cancer, 2002. 97: p. 536-541.
104. Larson, T.V. and J.Q. Koenig, *Woodsmoke: emissions and noncancer respiratory effects.* Annu Rev Public Health, 1994. 15: p. 133-56.
105. Maykut, N.N., et al., Source apportionment of PM2.5 at an urban IMPROVE site in Seattle, Washington. Environ Sci Technol, 2003. 37(22): p. 5135-42.
106. Gwynn, C., et al., Respiratory and Circulatory hospital admissions associated with forest fires - Montana, July-September, 1999 & 2000. Centers for Disease Control, Atlanta. GA, 2001.
107. Reinhardt, T.E., R.D. Ottmar, and C. Castilla, *Smoke impacts from agricultural burning in a rural Brazilian town.* J Air Waste Manag Assoc, 2001. 51(3): p. 443-50.
108. Ezzati, M. and D.M. Kammen, The health impacts of exposure to indoor air pollution from solid fuels in developing countries: knowledge, gaps, and data needs. Environ Health Perspect, 2002. 110(11): p. 1057-68.
109. Liu, L.J., et al., Exposure assessment of particulate matter for susceptible populations in Seattle. Environ Health Perspect, 2003. 111(7): p. 909-18.
110. Brauer, M., et al., Assessment of particulate concentrations from domestic biomass combustion in rural Mexico. Environmental Science and Technology, 1996. 30(1): p. 104-109.
111. Park, E. and K. Lee, Particulate exposure and size distribution from wood burning stoves in Costa Rica. Indoor Air, 2003. 13(3): p. 253-9.
112. Allen, R., et al., Use of real-time light scattering data to estimate the contribution of infiltrated and indoor-generated particles to indoor air. Environ Sci Technol, 2003. 37(16): p. 3484-92.
113. Simoneit, B.R.T., J. Schauer James, and e. al., 1999.
114. Simpson, C.D. and e. al, *unpublished data.* 2004.
115. Ratsimba, V., et al., Qualitative and quantitative evaluation of mono- and disaccharides in D-fructose, D-glucose and sucrose caramels by gas-liquid chromatography-mass spectrometry Di-D-fructose dianhydrides as tracers of caramel authenticity. Journal of Chromatography, B: Biomedical Sciences and Applications, 1999. A 844(1 + 2): p. 283-293.
116. Kim, E., et al., *Factor Analysis of Seattle Fine Particles.* Aerosol Science and Technology, 2004. in press.

117. Paatero, P., *Least squares formulation of robust non-negative factor analysis.* Chemometrics and Intelligent Laboratory Systems, 1997. 37(1): p. 23-35.
118. Liu, H. and T. Lumley, *On source apportionment and health effects models*, U.o. Washington, Editor. 2004.
119. Dills, R.L., X. Zhu, and D.A. Kalman, Measurement of urinary methoxyphenols and their use for biological monitoring of woodsmoke exposure. Environ Res, 2001. 85(2): p. 145-58.
120. Dorland, L., et al., 1,6-Anhydro-beta-D-glucopyranose (beta-glucosan), a constituent of human urine. Clin Chim Acta, 1986. 159(1): p. 11-6.
121. Istvan, J.A. and C. TW, Smoking rate, carboxyhemoglobin and body mass in the second National Health and Nutritional Examination Survey (NHANES II). Journal of Behavioural Medicine, 1992. 15(6): p. 559-72.
122. Rogge, W.F., et al., *Sources of fine organic aerosol. 1. Charbroilers and meat cooking operations.* Environmental Science and Technology, 1991. 25(6): p. 1112-25.
123. Rogge, W.F., et al., *Sources of fine organic aerosol. 5. Natural gas home appliances.* Environmental Science and Technology, 1993. 27(13): p. 2736-44.
124. Vyskocil, A., et al., *Assessment of multipathway exposure of small children to PAH.* Environ. Toxicol. Pharmacol., 2000. 8(2): p. 111-118.
125. Chuang, J.C., et al., *Polycyclic aromatic hydrocarbon exposures of children in low-income families.* J Expo Anal Environ Epidemiol, 1999. 9(2): p. 85-98.
126. Simpson, C.D., et al., Use of methoxyphenols in biological monitoring of woodsmoke exposure. Epidemiology, 2002. 13: p. 136.
127. Pierson, W., J. Koenig, and E.J. Bardana, *Potential adverse health effects of woodsmoke.* West J Med, 1989. 151(3): p. 339-342.
128. Vedal, S., Health effects of woodsmoke. Report to the Provincial Health Officer of British Columbia. 1993, The University of British Columbia: Vancouver, BC.
129. Boman, B.C., A.B. Forsberg, and B.G. Jarvholm, Adverse health effects from ambient air pollution in relation to residential wood combustion in modern society. Scand J Work Environ Health, 2003. 29(4): p. 251-60.
130. Artaxo, P., et al., Case Study of Atmospheric Measurements in Brazil: Aerosol Emission from Amazon Basin Fires, in Global biomass burning: Atmospheric climatic and biospheric implications, J. Levine, Editor. 1991: Cambridge, MA.
131. Artazo, P., et al., Fine mode aerosol composition at three long-term atmospheric monitoring sites in the Amazon Basin. Journal of Geophysical Research, 1994. 99(D11): p. 22857-22868.
132. Radojevic, M. and H. Hassan, *Air quality in Brunei Darussalam during the 1998 haze episode.* Atmospheric Environment, 1999. 33(22): p. 3651-3658.
133. Hassan, M., et al. Damage costs of the 1991 and 1994 haze episodes in Malaysia. in Climate and life in the Asia Pacific. 1995. Brunei Darussalam: Universiti Brunei Darussalam.
134. Nichol, J., *Bioclimatic impacts of the 1994 smoke haze event in Southeast Asia.* Atmospheric Environment, 1997. 31(8): p. 1209-1219.
135. Brauer, M., *Health Impacts of Biomass Air Pollution*. 1998, World Health Organization, Regional Offices for Southeast Asia and the Western Pacific: Kuala Lumpur.
136. Kunii, O., et al., The 1997 haze disaster in Indonesia: its air quality and health effects. Arch Environ Health, 2002. 57(1): p. 16-22.
137. US Department of Agriculture, F.S.I.R.S., The effects of forest fire smoke on fire fighters; a comprehensive study plan. 1989: Missoula, MT.

138. Rothman, N., et al., Pulmonary function and respiratory symptoms in wildland firefighters. J Occup Med, 1991: p. 1163-1167.
139. Liu, D., et al., The effect of smoke inhalation on lung function and airway responsiveness in wildland fire fighters. Am Rev Respir Dis, 1992. 146: p. 1469-1473.
140. Letts, D., et al., Health hazard evaluation prepared for the US Department of the Interior, National Park Service, Southern California. Report, in HETA 91-152-2140. 1991.
141. Serra, A., F. Mocci, and F.S. Randaccio, *Pulmonary function in Sardinian firefighters.* Am J Ind Med, 1996. 30: p. 70-78.
142. Tepper, A., G.W. Comstock, and M. Levine, *A longitudinal study of pulmonary function in fire fighters.* Am J Ind Med, 1991. 20: p. 307-316.
143. Sparrow, D., et al., The effect of occupational exposure on pulmonary function: a longitudinal evaluation of fire fighters and nonfire fighters. Am Rev Respir Dis, 1982. 128: p. 319-322.
144. Duclos, P., L.M. Sanderson, and M. Lipsett, *The 1987 Forest Fire Disaster in California: Assessment of Emergency Room Visits.* Archives of Environmental Health, 1990. 45(1): p. 53-58.
145. Sorenson, B., et al., Surveillance of Morbidity During Wildfires -- Central Florida, 1998. MMWR, 1999. 48(4): p. 78-9.
146. Mott, J.A., et al., Wildland forest fire smoke: health effects and intervention evaluation, Hoopa, California, 1999. West J. Med, 2002. 176: p. 157-162.
147. Sutherland, E.R., et al., *Wildfire smoke and respiratory symptoms in patients with chronic obstructive pulmonary disease.* Journal of Allergy and Clinical Immunology, 2005. 115(2): p. 420-422.
148. Copper, C., et al., *Acute exacerbations of asthma and bushfires.* The Lancet, 1994. 343(June 11): p. 1509.
149. Smith, M., et al., Asthma presentations to emergency departments in western Sydney during the January 1994 Bushfires. Int J Epidemiol, 1996. 25(6): p. 1227-1236.
150. Jalaludin, B., et al., Acute effects of bushfires on peak expiratory flow rates in children with wheeze: a time series analysis. Aust N Z J Public Health, 2000. 24(2): p. 174-7.
151. Johnston, F.H., et al., *Exposure to bushfire smoke and asthma: an ecological study.* Med J Aust, 2002. 176(11): p. 535-8.
152. Chew, F., et al., *Singapore's haze and acute asthma in children.* The Lancet, 1995. 346(Nov. 25): p. 1427.
153. Emmanuel, S.C., Impact to lung health of haze from forest fires: the Singapore experience. Respirology, 2000. 5(2): p. 175-82.
154. Leech, J., et al., *The Sarawak September haze episode.* Am J Resp Crit Care Med, 1998. 157(3): p. A260.
155. Hisham-Hashim, J., et al., Respiratory function of elementary school children exposed to the 1997 Kuala Lumpur haze. Epidemiology, 1998. 9(4 (S1)): p. S1.
156. Sastry, N., Forest fires, air pollution, and mortality in southeast Asia. Demography, 2002. 39(1): p. 1-23.
157. Tenenbaum, D.J., *A burning question: do farmer-set fires endanger health?* Environ Health Perspect, 2000. 108(3): p. A117-A118.
158. Long, W., et al., Respiratory symptoms in a susceptible population due to burning of agricultural residue. Chest, 1998. 113(2): p. 351-7.

159. Jacobs, J., R. Kreutzer, and D. Smith, *Rice burning and asthma hospitalizations, Butte County, California, 1983-1992.* Environ Health Perspect, 1997. 105(9): p. 980-5.
160. Golshan, M., et al., Early effects of burning rice farm residues on respiratory symptoms of villagers in suburbs of Isfahan, Iran. Int J Environ Health Res, 2002. 12(2): p. 125-31.
161. Torigoe, K., et al., Influence of emission from rice straw burning on bronchial asthma in children. Pediatr Int, 2000. 42(2): p. 143-50.
162. Norris, G.A., Air pollution and exacerbation of asthma in an arid, western US city. PhD Thesis, University of Washington, 1998.
163. Solomon, C., et al., Effect Of Exposure To Smoke From Rice Straw On Airway Inflammation In Individuals With Allergic Rhinitis. American Thoracic Society 2002 Conference Abstract (A15), 2002.
164. Cancado, J., et al., The effect of sugarcane-burning air pollution on children respiratory diseases. Epidemiology, 2002. 13(4): p. S103.
165. Arbex, M., et al., Assessment of the effects of sugar cane plantation burning on daily counts of inhalation therapy. J. Air and Waste Manage. Assoc., 2000. 50: p. 1745-1749.
166. Boopathy, R., B.R. Asrabadi, and T.G. Ferguson, *Sugar cane (Saccharum officinarum L) burning and asthma in Southeast Louisiana, USA.* Bull Environ Contam Toxicol, 2002. 68(2): p. 173-9.
167. Rothschild, H. and J. Mulvey, An increased risk for lung cancer mortality associated with sugar cane farming. J. Natl Cancer Inst, 1982. 68: p. 755-60.
168. Brooks, S., H. Stockwell, and P. Pinkham, *Sugar cane exposure and the risk of lung cancer and mesothelioma.* Environ Res, 1992. 58: p. 195-203.
169. Amre, D., et al., Case-control study of lung cancer among sugar cane farmers in India. Occup Environ Med, 1999. 56: p. 548-552.
170. Boeniger, M.F., et al., Exposure assessment of smoke and biogenic silica fibers during sugar cane harvesting in Hawaii. Appl Occup Environ Hyg, 1991. 6: p. 59-66.
171. Miller, W., D. Reed, and J. Banta, *Sugar cane workers: morbidity and mortality.* Hawaii Med J, 1993. 52: p. 300-306.
172. Fairley, D., The Relationship of Daily Mortality to Suspended Particulates in Santa Clara County, 1980-1986. Environmental Health Perspectives, 1990. 89: p. 159-168.
173. Lipsett, M., S. Hurley, and B. Ostro, *Air pollution and emergency room visits for asthma in Santa Clara County, California.* Environ Health Perspect, 1997. 105(2): p. 216-22.
174. Robin, L.F., et al., *Wood-burning stoves and lower respiratory illnesses in Navajo children.* Pediatr Infect Dis J, 1996. 15(10): p. 859-65.
175. McGowan, J.A., et al., Particulate air pollution and hospital admissions in Christchurch, New Zealand. Aust N Z J Public Health, 2002. 26(1): p. 23-9.
176. Norris, G., et al., An association between fine particles and asthma emergency department visits for children in Seattle. Environ Health Perspect, 1999. 107(6): p. 489-93.
177. Sheppard, L., et al., Effects of ambient air pollution on nonelderly asthma hospital admissions in Seattle, Washington, 1987-1994. Epidemiology, 1999. 10(1): p. 23-30.
178. Yu, O., et al., Effects of ambient air pollution on symptoms of asthma in Seattle-area children enrolled in the CAMP study. Environ Health Perspect, 2000. 108(12): p. 1209-14.
179. Koenig, J.Q., et al., Measurement of offline exhaled nitric oxide in a study of community exposure to air pollution. Environ Health Perspect, 2003. 111(13): p. 1625-9.
180. Koenig, J., et al., Pulmonary effects of indoor- and outdoor-generated particles in children with asthma. Environ Health Perspect-in-Press, 2005. Online.

181. Lumley, T. and H. Liu, How can source apportionment and receptor modelling data be used in epidemiology? AAAR abstract, 2003: p. 87.
182. Anuszewski, J., T. Larson , and J. Koenig *Simultaneous indoor and outdoor particle light-scattering measurements at nine homes using a portable nephelometer.* Journal of Exposure Analysis and Environmental Epidemiology, 1998. 8(4): p. 483-493.
183. Allen, R., et al., Estimated hourly personal exposures to ambient and nonambient particulate matter among sensitive populations in Seattle, Washington. J Air Waste Manag Assoc, 2004. 54(9): p. 1197-211.
184. Sarnat, J.A., et al., *Using sulfur as a tracer of outdoor fine particulate matter.* Environ Sci Technol, 2002. 36(24): p. 5305-14.
185. Janssen, N.A., et al., Air conditioning and source-specific particles as modifiers of the effect of PM(10) on hospital admissions for heart and lung disease. Environ Health Perspect, 2002. 110(1): p. 43-9.
186. Kim, E., P.K. Hopke, and E.S. Edgerton, *Source identification of atlanta aerosol by positive matrix factorization.* J Air Waste Manag Assoc, 2003. 53(6): p. 731-9.
187. Manchester-Neesvig, J.B., J.J. Schauer, and G.R. Cass, The distribution of particle-phase organic compounds in the atmosphere and their use for source apportionment during the Southern California Children's Health Study. J Air Waste Manag Assoc, 2003. 53(9): p. 1065-79.
188. Nolte, C.G., et al., Highly polar organic compounds present in woodsmoke and in the ambient atmosphere. Environ Sci Technol, 2001. 35(10): p. 1912-9.
189. Honicky, R.E., J.S. Osborne, 3rd, and C.A. Akpom, Symptoms of respiratory illness in young children and the use of wood-burning stoves for indoor heating. Pediatrics, 1985. 75(3): p. 587-93.
190. Butterfield, P., et al., *Woodstoves and Indoor Air: The effects on preschoolers' upper respiratory systems.* Journal of Environmental Health, 1989. 52(3): p. 172-173.
191. Morris, K., et al., Wood-burning stoves and lower respiratory tract infection in American Indian children. Am J Dis Child, 1990. 144(1): p. 105-8.
192. Daigler, G.E., S.J. Markello, and K.M. Cummings, *The effect of indoor air pollutants on otitis media and asthma in children.* Laryngoscope, 1991. 101(3): p. 293-6.
193. Ostro, B.D., et al., *Indoor air pollution and asthma. Results from a panel study.* Am J Respir Crit Care Med, 1994. 149(6): p. 1400-6.
194. Schwartz, J., et al., Particulate air pollution and hospital emergency room visits for asthma in Seattle. Am Rev Respir Dis, 1993. 147(4): p. 826-31.
195. Bates, D.V., Health indices of the adverse effects of air pollution: the question of coherence. Environ Res, 1992. 59(2): p. 336-49.
196. Smith, K.R., Fuel combustion, air pollution exposure, and health - the situation in developing countries. Annual Review of Energy and the Environment, 1993. 18: p. 529-566.
197. Smith, K.R., S. Mehta, and M. Maeusezahl-Feuz, Indoor Smoke from Household Solid Fuels, in Comparative Quantification of Health Risks: Global and Regional Burden of Disease due to Selected Major Risk Factors, M. Ezzati, et al., Editors. 2004, World Health Organization: Geneva. p. 1437-1495.
198. Smith, K.R., et al., *Air pollution and the energy ladder in Asian cities.* Energy, 1994. 19(5): p. 587-600.
199. Smith, K.R., et al., *Greenhouse implications of household fuels: An analysis for India.* Annual Review of Energy and Environment, 2000b. 25: p. 741-763.

200. Office of Technology Assessment, *Fueling Development: Energy Technologies for Developing Countries*. 1992, Congress of the United States: Washing, D.C. p. 324.
201. Mehta, S., Characterizing Exposures to Indoor Air Pollution From Household Solid Fuel Use, in Environmental Health Sciences. 2003, University of California: Berkeley.
202. Sinton, J.E., et al., *As assessment of programs to promote improved household stoves in China.* Enegy for Sustainable Development, 2004. 8(3): p. 33-52.
203. Sinton, J.E., et al., *Indoor Air Pollution Database for China*, in *Human Exposure Assessment Series*. 1996, World Health Organization, United Nations Environment Program: Geneva.
204. Saksena, S., L. Thompson, and K.R. Smith, *Indoor Air Pollution and Exposure Database*. 2003, School of Public Health, University of California: Berkeley.
205. Bennett, D.H., et al., *Defining intake fraction.* Environ Sci Technol, 2002. 36(9): p. 207A-211A.
206. Balakrishnan, K., et al., Exposure assessment for respirable particulates associated with household fuel use in rural districts of Andhra Pradesh, India. J Expo Anal Environ Epidemiol, 2004. 14 Suppl 1: p. S14-25.
207. Ezzati, M., et al., Selected major risk factors and global and regional burden of disease. Lancet, 2002. 360(9343): p. 1347-60.
208. World Health Organization, *World Health Report: Reducing Risks, Promoting Healthy Life*. 2002, World Health Organization: Geneva.
209. Begum, B.A., et al., Investigation of sources of atmospheric aerosol at urban and semi-urban areas in Bangladesh. Atmospheric Environment, 2004. 38(19): p. 3025-3038.
210. Chowdhury, Z., et al., Source apportionment and detailed speciation of ambient fine organic carbon particles in Delhi, Mumbai, Kolkata, and Chandigarh. Environ. Science & Technol, 2005 (in press).
211. Tzanakis, N., et al., Short-term effects of woodsmoke exposure on the respiratory system among charcoal production workers. Chest, 2001. 119(4): p. 1260-5.
212. Tan, W.C., et al., *The human bone marrow response to acute air pollution caused by forest fires.* Am J Respir Crit Care Med, 2000. 161(4 par 1): p. 1213-7.
213. Tanner, R.L., et al., Impact of the 1998 Central American Fires on PM_2.5 Mass and Composition in the Southeastern United States. Atmospheric Environment, 2001. 35(36): p. 6539-6547.
214. Polissar, A.V., P.K. Hopke, and R.L. Poirot, *Atmospheric aerosol over Vermont: chemical composition and sources.* Environ Sci Technol, 2001. 35(23): p. 4604-21. check