The Economics of Household Air Pollution

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Abstract

Traditional energy technologies and consumer products contribute to household well-being in diverse ways but also often harm household air quality. We review the problem of household air pollution at a global scale, focusing particularly on the harmful effects of traditional cooking and heating. Drawing on the theory of household production, we illustrate the ambiguous relationship between household well-being and adoption of behaviors and technologies that reduce air pollution. We then review how the theory relates to the seemingly contradictory findings emerging from the literature on developing country household demand for clean fuels and stoves. In conclusion, we describe an economics research agenda to close the knowledge gaps so that policies and programs can be designed and evaluated to solve the global household air pollution problem.

1. INTRODUCTION

Approximately 80% of the air that humans breathe during their lifetime is indoors—at home, work, or school. Decisions about cooking and heating fuels, furnishings and consumer technologies, and building materials and configurations can therefore impact human health (Huang et al. 2013). Furthermore, most inhalation of poor-quality air occurs inside dwellings because people spend many of their living hours inside their homes (Sundell 2004, Bureau of Labor Statistics 2014). On a global scale, household air pollution (HAP) poses the most important indoor air quality challenges because of the number of people affected, the range of contaminants involved, and the severity of the risks involved (**Table 1**). The harmful health impacts of poor indoor air quality include acute and chronic disease risks such as asthma, respiratory infections, cardiovascular disease, and cancer.

This review focuses on the economics of the HAP problem. Because households have a say over housing design and technologies, an economic conception of the problem begins from the idea that individuals make choices—about home design and the use of indoor technologies—that account for the private impacts (both positive and negative) that these choices generate. Not all factors are controllable, however, and poor outdoor air quality, for example, can constrain attempts to avoid HAP. For example, in developing country urban centers, such as Beijing, Dakar, and Cairo, average annual concentrations of PM_{10} [particulate matter (PM) less than 10 μm in diameter] are more than five times the average annual concentration (20 $\mu g/m^3$) recommended as healthful by the World Health Organization (WHO) (**Figure 1**). Levels in Karachi, Kabul, and Delhi are 10–15 times the recommended level, and some cities have even greater concentrations. These levels contrast with those of most cities in Europe, the United States, and Japan, which are below or near the guideline (WHO 2006, 2014a). Stepping outdoors in lower-income countries therefore certainly does not guarantee a breath of fresh air. In addition, other environmental hazards, such as poor water quality and chronic food insufficiency, may make people more vulnerable to diseases caused by HAP.

HAP occurs in all regions of the world and at all income levels. Still, as we discuss below, its effects are most acute among households living in regions where use of modern fuels (i.e., gas and electricity) for cooking and heating is limited. Modern fuels tend to generate limited HAP because they either (*a*) burn efficiently and completely when used indoors, as in the case of biogas or liquefied petroleum gas (LPG), or, (*b*) in the case of electricity, are generated through combustion (e.g., coal) or other processes (e.g., wind or hydropower) that take place outside the home. As of 2013, approximately three-fifths of the global population used gas or electricity for cooking (IEA 2012, Smith et al. 2013). The rates of use of such cleaner-burning household fuels show a strong positive association with indicators of socioeconomic status, both within and across countries. This observation explains our primary focus on the HAP challenges in low- and lower-middle-income countries and our lack of attention to other issues related to indoor air (e.g., occupational health). To further focus this article, we also omit discussion of environmental tobacco (i.e., secondhand) smoke (see Chaloupka & Warner 2000 for a review of the economics of smoking).

We also note that HAP is typically coproduced, with two major nonhealth costs that are external to the household and are generated at two different scales: (*a*) the degradation of local and regional forests and air quality and (*b*) global warming because of the climate forcing caused by the black carbon that is emitted from incomplete burning of biomass. The nature and size of these externalities are discussed elsewhere (Venkataraman et al. 2005, Ramanathan & Carmichael 2008, Bailis et al. 2014). However, considerations of these externalities add urgency to understanding how to induce households to reduce HAP, which will in turn deliver positive regional and global externalities.

Table 1 Major indoor air contaminants

Contaminant	Typical sources
Particulate matter	Outdoors; combustion sources such as cigarettes, wood stoves, and candles; cooking; cleaning; general activity
Polycyclic aromatic hydrocarbons	Vehicle exhaust, cigarette smoke, cooking, wood smoke, pesticides, commercial and residential application of insecticides and herbicides, treated wood products
Nitrogen dioxide	Combustion sources, particularly unvented gas or kerosene appliances
Volatile organic compounds	Cleaning agents, aerosol sprays, pesticides, paints, solvents, building materials, combustion sources, glues
Formaldehyde	Composite wood products such as particleboard, furnishings, combustion sources, environmental tobacco smoke, cosmetics, paints
Environmental tobacco smoke	Cigarettes, cigars, pipes
Biological contaminants (e.g., house dust mites, animal dander, mold, cockroaches)	Dampness, moisture, floor dust, bedding, insects, pets, pests
Radon	Soil and bedrock under homes, groundwater

Adapted with permission from Franklin (2007).

The remainder of the review is structured as follows. In Section 2, we discuss the magnitude and range of health impacts of HAP. Although we do not review all contaminants or discuss all literature, we argue that the most important HAP problems in the world today stem from use of solid fuels and inefficient stoves by approximately 3 billion people in low-income countries. We therefore orient our subsequent discussion primarily around HAP concerns in poor countries. In Section 3, we present a stylized model that illustrates how a household might make choices that generate potentially dangerous levels of HAP. We use the model to highlight the important role of biophysical constraints (e.g., the link between HAP and health), household income, prices of polluting technologies, information and knowledge, markets and institutions, and preferences and social norms related to the behaviors that generate HAP. Section 4 then reviews the empirical literature to which the model speaks. We focus on a number of issues that have been overlooked in economics. The article concludes in Section 5 with a brief summary of our findings and of the knowledge gaps that help define a future research agenda for economists.

2. BACKGROUND

This section discusses the range of HAP issues that have received attention in the published literature. We begin with a broad overview of the main problems, but reviewing all contaminants of concern is beyond the scope of this article. Nonetheless, to cover a diversity of pollutants of widespread popular interest and for the sake of comparison, we offer brief detailed discussions of three contaminants that have received significant attention in rich countries: mold, radon, and formaldehyde. Given the clear differences in magnitudes of the health concerns posed by different sources of HAP, we ultimately narrow our focus to the effects of household use of solid fuels.

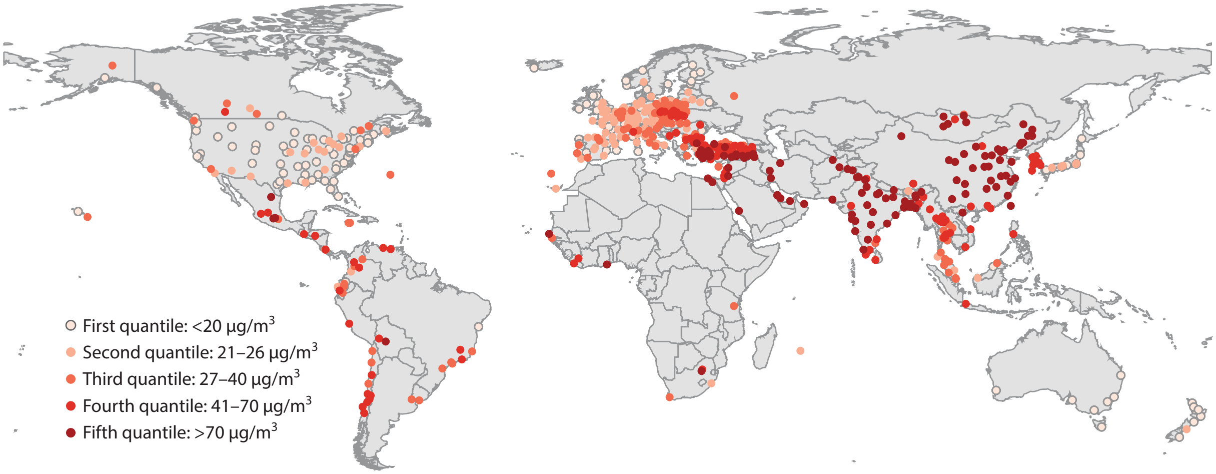

Figure 1

Ambient air pollution levels in cities worldwide. Data from WHO cities ambient air pollution data for 2008–2013 (WHO 2014a).

2.1. Overview of Household Air Pollution Issues

A wide variety of household air pollutants have been identified as posing significant threats to human health (**Table 1**). Some of these (e.g., formaldehyde, volatile organic compounds) come mainly from consumer products or materials used in home construction. Others (e.g., PM, nitrogen dioxide, polycyclic aromatic hydrocarbons) are generated primarily from combustion processes occurring within a household, for example, from cooking or heating. Finally, a third category of contaminants encompasses natural sources (e.g., radon) or biological sources that are seen around the home, for example, mold, insect, or other animal sources.

With the notable exception of the by-products of in-home combustion, which we address further below, the effects of most of these contaminants have been studied primarily in higher-income settings.[1] The literature documents clear associations between various contaminants and a range of illnesses, particularly in children and other vulnerable populations. The most significant evidence among the contaminants unrelated to combustion pertains to the health effects of exposure to mold, radon, and formaldehyde.

Mold in the home is caused by dampness and may affect people via transmission through the air. Mold exposures are very common around the world because indoor dampness is quite common, in some regions rising to 60% (Jaakkola et al. 2013). Rayner (1996), for example, notes that 20% of the UK housing stock has significant dampness and mold, whereas Howden-Chapman et al. (2005) report that 35% of their New Zealand respondents indicate that they have mold in their homes. This contaminant is thought to contribute to several common health conditions such as asthma and rhinitis,[2] and the Centers for Disease Control have concluded that excessive exposure to mold can have negative effects regardless of the type of mold (Weinhold 2007). Clear causality has been difficult to show, however, because studies documenting

[1]This is not to say that such contaminants are not also a problem in less-developed countries; however, they have hardly been studied in those contexts.

[2]Rhinitis, for example, has been estimated to affect between 10% and 40% worldwide, whereas asthma and environmental allergies affect 6% and 20% of Americans, respectively (Fisk 2000).

associations between mold and health impacts often rely on respondent recall and visual and/or smell tests for mold presence (Zock et al. 2002, Bellanger et al. 2009, Rabinovitch 2012).

Several recent studies, however, utilize more sophisticated mold measurement methods or have implemented randomized control trials (RCTs) of mold control interventions, allowing for better causal inference. Two studies are particularly noteworthy. First, papers from the Cincinnati Childhood Asthma and Air Pollution Study tighten the link between mold exposure during infancy and childhood asthma by taking mold samples rather than relying on self-reports of mold presence (Cho et al. 2006; Vesper et al. 2006, 2007; Reponen et al. 2011). Researchers followed newborn children until the age of 7, taking baseline mold samples shortly after birth. Reponen et al. (2011) report that 24% of sampled children in the greater Cincinnati area had asthma and that infant exposure to three particular species of molds was positively associated with asthma at age 7. The magnitude of the effect of mold is unclear, however. Second, Burr et al. (2007) conduct an RCT of mold control within a group of asthma patients. In treatment households, indoor mold was removed, fungicide was applied, and a fan was installed in the attic. Burr et al. conduct surveys and measure peak respiratory flow at baseline, after 6 months, and after 1 year. They conclude that, "although there was no objective evidence of benefit, symptoms of asthma and rhinitis improved and medication use declined following removal of indoor mould. It is unlikely that this was entirely a placebo effect."

Radon is a naturally occurring, odorless, and radioactive gas that originates from uranium found in soils and rocks. Most studies of radon exposure risk focus on those exposed to high concentrations, such as underground miners and people living near mines. This research offers clear evidence that exposure to radon can cause lung cancer, which is the most deadly form of cancer (Tracy et al. 2006, Sainz et al. 2009). In fact, there is believed to be no concentration level that does not elevate lung cancer risks (Pacheco-Torgal 2012). The WHO has therefore identified an action level of 250 Bq/m^3, which can generally be reached only indoors, and a limit of 100 Bq/m^3 to minimize health risks (WHO 2009). Average indoor radon concentrations measured in select countries are presented in **Table 2**; these average levels suggest that radon exposure may be an important HAP problem in many buildings and homes and particularly in basements.

Radon exposure is believed to be the second-leading cause of lung cancer after smoking, causing an estimated 21,000 US deaths out of the approximately 157,000 total US lung cancer deaths per year, which is also similar to the ratio in Canada (Tracy et al. 2006, Lantz et al. 2013, US EPA 2014b). What is perhaps underappreciated in the popular discussion about radon, however, is that radon-related lung cancer and smoking are highly correlated, which suggests that there may be important disease-causing synergies between smoking and radon exposure (Lantz et al. 2013). Indeed, 86% of US radon-related lung cancer deaths occurred in smokers, and 90% of Canadian radon deaths were among smokers (Tracy et al. 2006, Lantz et al. 2013). In the United States, there are only approximately 2,900 annual radon-related lung cancer deaths among those who have never smoked (US EPA 2014b). **Table 3** presents estimated excess mortality for smokers and never-smokers.

Because children rarely smoke, focusing on children eliminates an important potential factor that could confound the relationship between radon and cancer. Tong et al. (2012) conduct a comprehensive review of the empirical literature on radon exposure and childhood leukemia. They conclude that the literature generally finds a positive association, although there have been relatively few large-scale studies and radon measurement methods vary across the literature, potentially confounding results. In contrast, a recent cohort study of almost 1.3 million Swiss children finds no association between radon concentration and malignancies of any kind (the median concentration was 77.7 Bq/m^3, and the ninetieth-percentile concentration was 139.9 Bq/m^3) (Hauri et al. 2013). This collective body of evidence suggests that radon likely does have

Table 2 Average indoor radon concentrations in select OECD countries (in Bq/m^3)

Country	Arithmetic mean	Geometric mean	Geometric standard deviation
United States	26	25	3.1
Canada	28	11	3.9
Germany	49	37	2.0
Finland	120	84	2.1
Mexico	140	90	NA
Sweden	108	56	NA
United Kingdom	20	14	3.2
France	89	53	2.0
Worldwide	39		

Data from WHO (2009). Note that 100 Bq/m^3 = 2.7 P/CL.

negative consequences for health but that these make up a relatively small fraction (perhaps 10% at most) of the 1 million annual global lung cancer deaths.

Finally, formaldehyde is a naturally occurring compound that is present in the ambient environment at approximately 1 $\mu g/m^3$. In outdoor urban environments with heavy vehicle traffic, concentrations can reach 100 $\mu g/m^3$ (Nielsen & Wolkoff 2010), however. It is often found in high concentrations indoors as well because formaldehyde is used in pressed wood products, such as plywood, that require resins in their manufacture and that are commonly used in home construction, cabinetry, and furniture. Formaldehyde is also in flooring and carpeting, as well as in numerous consumer products, such as deodorizers, mothballs, deodorants, facial moisturizers, and hair conditioners (Hun et al. 2010, Huang et al. 2013).

Formaldehyde is considered to be a potent respiratory irritant, and the US Environmental Protection Agency (US EPA) classifies it as a probable human carcinogen (US EPA 2014a). Duong et al. (2011) conduct a meta-analysis of 18 studies and find some evidence of a linkage between formaldehyde exposure by pregnant women and child development. This chemical is the subject of a variety of guideline levels worldwide; for example, the state of California has set strict chronic reference levels at 9 $\mu g/m^3$ (Hun et al. 2010), and the WHO has established a guideline value of 100 $\mu g/m^3$ for 30-min indoor exposures. Reviews of scientific and dose-response studies point to levels ranging from 98 to 123 $\mu g/m^3$ as preventative for respiratory irritation and carcinogenic effects in indoor environments (Nielsen & Wolkoff 2010, Golden 2011). (The US Department of Housing and Urban Development has a maximum allowable concentration for wood products of 300 ppb.) In general, such concentrations are considered to be unlikely in most ordinary settings, although they may occur where highly formaldehyde intensive construction materials are used.[3]

Thus, contaminants such as mold, radon, and formaldehyde can have significant negative effects on health. If the numbers are put in perspective, though, radon would appear to contribute at most 10% of the burden of disease related to lung cancer, which itself ranks sixteenth on the list of causes contributing to the global burden of disease (GBD) (Lozano et al. 2012), and perhaps to

[3]For example, consider the United States prior to the 1982 ban on urea foam formaldehyde insulation. Shortly after the ban, studies of condominiums in the mid-1980s found formaldehyde concentrations of 80–90 ppb, whereas studies in the 2000s found concentrations of 15–36 ppb in newly manufactured homes constructed after the ban (CDC 2014).

Table 3 Radon-related excess lung cancer mortality for smokers and never-smokers

Radon concentration	Lung cancer risk/1,000 population	
	Smokers	Never-smokers
20 pCi/L	260	36
10 pCi/L	150	18
8 pCi/L	120	15
4 pCi/L	62	7
2 pCi/L	32	4
1.3 pCi/L*	20	2
0.40 pCi/L**	3	0

Data from US EPA (2014b). The single asterisk refers to average indoor concentration, and the double asterisk refers to average outdoor concentration.

other cancers. Mold clearly aggravates asthma, which ranks forty second on the list, whereas the effects of formaldehyde are difficult to quantify but appear to be geographically limited. In contrast, the health effects of solid-fuel combustion, which we review below in Section 2.2, are felt by billions of people worldwide.

2.2. The Challenge of Household Use of Solid Fuels

Approximately 1.3 billion people, living mostly in low-income countries, do not have access to household electricity. These and many more people—globally approximately 2.8 billion (0.5 billion in urban areas), or 40% of the world population—often find commercial fuels to be too expensive or too irregularly supplied to use for cooking and heating. Instead, they rely on solid fuels like coal, fuelwood, dung, and charcoal that are combusted inside their homes to meet their needs (Grieshop et al. 2011, Jeuland & Pattanayak 2012, Smith et al. 2013). Approximately 52% of the world population that uses solid fuels today lives in India and China, and another 21% lives in sub-Saharan Africa (Smith et al. 2013). Without dramatic changes in policies, the global number of such people is projected to remain roughly constant through 2030 at 2.7 billion people, or one-third of the world's population (IEA 2012). Most of the projected continued reliance on solid fuels is due to increases in usage in the lowest-income countries in sub-Saharan Africa and Asia, even as solid-fuel use in higher-income countries declines (**Figure 2**).

Solid fuels tend to be collected by households or, if purchased, are typically cheaper than cleaner-burning commercial (or modern) fuels. Solid fuels are easy to use in the traditional stoves that were developed specifically to handle such fuels. As a result, those who live in rural areas of low- and lower-middle-income countries rely heavily on solid fuels (Bluffstone & Toman 2014). The particular fuels, of course, vary across locations. For example, coal is commonly used in China and some parts of India, whereas charcoal is burned in urban areas of East Africa, and dung and fuelwood are used in much of India and Nepal (Smith et al. 2013). Yet even among households with access to commercial fuels, in many settings there is continued substantial use of solid fuels in cooking and heating, due to their relative cost advantage, user preferences, and unreliable stove or fuel availability (Masera et al. 2000, Heltberg 2004).

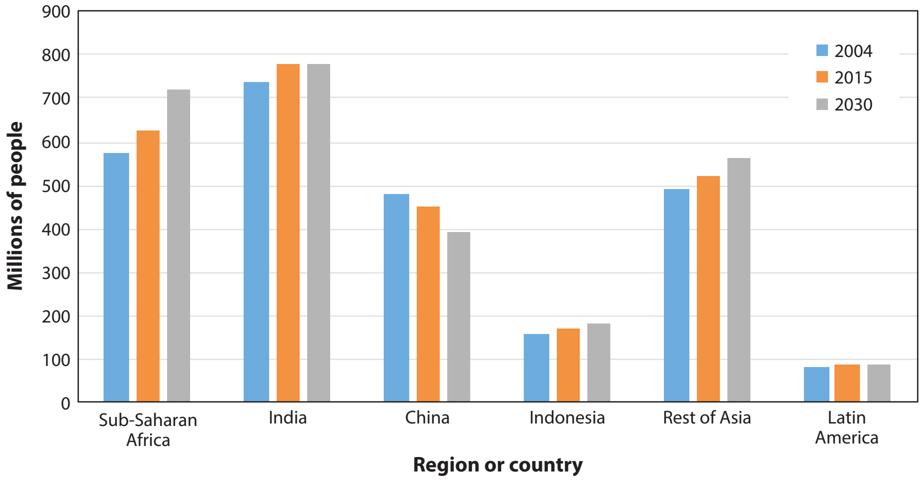

Figure 2

The number of people relying or projected to rely on solid fuels for cooking by major region and country. Regions that are not shown have very small populations using solid fuels. Data from IEA (2006), p. 431.

Table 4 presents average household-level use of solid fuels in eight countries using World Bank LSMS data. The table illustrates the well-known correlation between higher income and lower use of solid fuels but also highlights that the transition to clean-burning commercial fuels is typically incomplete (Heltberg 2003, 2004). Fuels and the technologies that use them therefore tend to be stacked, with households mixing technologies and fuels. For example, an urban household often has and regularly uses biomass, electric, and LPG stoves (Masera et al. 2000).

Combustion of solid fuels in traditional or even higher-efficiency cookstoves is incomplete and can generate high levels of HAP. The pollutants released include particulates; carbon monoxide; nitrogen oxide; and organic air pollutants such as benzene, formaldehyde, and polycyclic aromatic hydrocarbons (American Lung Association 2011, Smith et al. 2013). Alarmingly, particulate concentrations in developing country kitchens where wood or other biomass is burned can reach 10–30 mg/m^3 (Eisner et al. 2010). The WHO PM_{10} guideline for acute exposures is 50 μg/m^3 (WHO 2006).

When inhaled, the pollutants emitted during biomass burning cause various diseases, including lower respiratory infections (LRIs) such as pneumonia, chronic obstructive pulmonary disease (COPD), cardiovascular disease, and cancers. Exposures typically start in utero and continue via respiratory pathways through childhood and into adulthood. Thus, cumulative lifetime exposures can be very high, especially for women, who tend to be more heavily involved in cooking.

The research suggests that the effects of HAP from solid-fuel combustion are substantial, but there are major unknowns related to specific consequences. Most evidence comes from observational studies (Bruce et al. 2000, Dherani et al. 2008), which raises the possibilities of confounding by omitted variables or selection on unobservables, each of which may bias estimates of impacts up or down (Mueller et al. 2011). The negative impacts of $PM_{2.5}$ (PM less than 2.5 μg in diameter) and carbon monoxide on birth weight, child respiratory health [e.g., acute lower respiratory infection (ALRI) and pneumonia in particular], and mortality are perhaps best documented (Smith et al. 2000, Mishra et al. 2004, Edwards & Langpap 2012, Gajate-Garrido 2013), whereas effects on long-term cognitive and physical development remain uncertain. With respect

Table 4 Household characteristics and reliance on solid fuels in eight low- and middle-income countries

	Characteristics		Solid fuels				
	Per capita expenditure ($/day)	% urban	Fuelwood	Coal or charcoal	Dung	Straw, leaves, or twigs	Any solid fuel
Brazil	$15.1	80.7%	16%	0%			16%
South Africa	$6.1	53.3%	31%	8%	1%		38%
Guatemala	$2.70	43.1%	74%	12%			82%
Nicaragua	$2.0	56.7%	66%	1%			67%
Ghana	$1.80	36.7%	62%	46%			96%
Vietnam	$0.60	24.1%	67%	18%		60%	89%
India	$0.50	27.3%	72%	3%	37%		78%
Nepal	$0.30	7.3%	78%	1%	28%	32%	96%

Data from Heltberg (2003, 2004). Blank cells denote categories not present in some surveys.

to chronic impacts, a number of studies use spirometry to demonstrate the association between biomass fuel combustion and the development of chronic bronchitis and COPD in women, evidence that is supported by exposure-response experiments (Eisner et al. 2010). The evidence for cardiovascular disease (Baumgartner et al. 2011) and lung cancer (Zhang & Smith 2007, Smith et al. 2014) is somewhat more limited. In addition, few studies explicitly consider the interactions between ambient and household air quality, and these studies often fail to find significant differences from such interactions (Lewis et al. 2014b).

Recent GBD calculations, based exclusively on the impacts of particulates for which the best evidence exists, are used to argue that approximately 3.5 million premature deaths are caused each year by HAP stemming from the indoor combustion of solid fuels (Lim et al. 2013).[4] An additional 0.5 million deaths are attributable to the particle emissions that migrate from homes into the outdoor environment, where such emissions represent 16% of total outdoor concentrations (Smith et al. 2013). Thus, the WHO estimates total premature deaths due to HAP at 4.3 million, which is more than the 3.7 million total premature deaths attributable to ambient air pollution (WHO 2014b). All but 20,000 of these deaths are in low- and middle-income countries, and the GBD of disability-adjusted life years (DALYs) per capita due to outdoor air pollution (OAP) pales in comparison to that attributable to indoor air (WHO 2007) (**Figure 3**). Approximately 3.6 million premature deaths occurred in Asia and the western Pacific and 580,000 in Africa. Among the diseases linked to harmful HAP, LRI (not all attributable to HAP) is believed to cause an annual loss of 147 million DALYs (or 6% of total GBD), which is second only to ischemic heart disease.[5] In 2000 and 2011, LRI was the primary cause of reduced DALYs worldwide (WHO 2013, 2014b).

[4]The mortality and burden of disease numbers are therefore almost surely underestimates of the health consequences of HAP, given that other pollutants in HAP affect health (and the environment) in ways that are only beginning to be understood.

[5]The DALY is a standard way of quantifying the effects of diseases on human well-being. The first component of a DALY is the estimated mortality effect of disease, which is referred to as years of life lost. The second component of disease impact is years lost due to disability, which captures the morbidity and infirmity associated with disease. These two components, when added together, make up the DALY burden of disease (WHO 2013).

3. A CONCEPTUAL MODEL FOR THE PRODUCTION OF HOUSEHOLD AIR QUALITY

3.1. Basic Formulation

In this subsection, we apply a largely micro-level perspective to help (*a*) explain patterns observed in the global data on household exposure to HAP and its associated health burden and (*b*) motivate more nuanced thinking about the effects of interventions to reduce this exposure. This approach accommodates a focus on the production of improved air quality and health as an individual or household decision that is nonetheless affected by external factors and agents. Building on more fundamental work in health and environmental economics (Grossman 1972, Pattanayak & Pfaff 2009), our conceptual model starts from the idea that the decision to invest in preventive health or environmental improvements involves a trade-off with consumption of other goods and leisure. In the model, individuals or households maximize utility (u) by allocating resources—time and money—to these separate domains. Therefore, initial endowments of these resources constrain behavior and influence the extent of investment in environmental quality (which requires a mix of inputs) versus spending on consumption.

In mathematical terms, we start with modifications to the Lagrangian ($\mathcal{L}$) corresponding to the basic utility maximization problem for the case of binding time and health production constraints that is described in Pattanayak & Pfaff (2009) (henceforth P&P):

$$\begin{aligned} \mathcal{L} = \max u[\theta, l, c, a, s(a, A, G, e), e(a, c, A, G, E)] - \lambda[f(a, t, m, k)] - \gamma[g(a, c, t, m, k)] \\ + \mu[y - c - pm - rk + w(24 - s - l - t)], \end{aligned} \tag{1}$$

where θ represents a set of preferences that affect the concavity and shape of the utility function; l is leisure; c is consumption; a represents risk-averting behavior; s represents time spent sick; and e is household environmental quality, including, most specifically, air quality. Sickness s (produced by the health production function f) is decreasing in household environmental quality e and household averting behavior, as well as in aggregate community averting behavior A and government action to reduce pollution G. In addition, a, A, and G, plus ambient environmental quality E and consumption c, collectively influence household environmental quality through the production function for environmental quality g. Household environmental quality is increasing in a, A, G, and E but is decreasing in c because we assume that consumption generates pollution through channels such as harmful cooking emissions or the use of building or other materials that release toxic chemicals (e.g., formaldehyde) into a household's living space. Both the health and environmental quality production functions are assumed to be twice differentiable, continuous, and convex.

Regarding the constraints facing households, potential averting behavior is restricted by (and increasing in) inputs of time t, material m, and knowledge k. The allocation of these inputs is subject to typical time and money budget constraints. The income budget, made up of exogenous income y and wages obtained through work hours compensated at a wage rate w, is devoted to consumption; to the purchase of averting materials at price p; and to the acquisition of knowledge about the efficacy of averting behavior or the negative effects of pollution, which has unit cost r. In reality, the wage rate itself likely decreases in sickness s, reflecting the relationship between good health, and human capital development and worker productivity (Graff-Zivin & Neidell 2013); for simplicity, we do not include this complication in the mathematical treatment of model implications presented below but rather discuss it qualitatively. Finally, the 24-hour time budget is allocated to leisure, to time spent on risk-averting behavior, and to time spent sick.

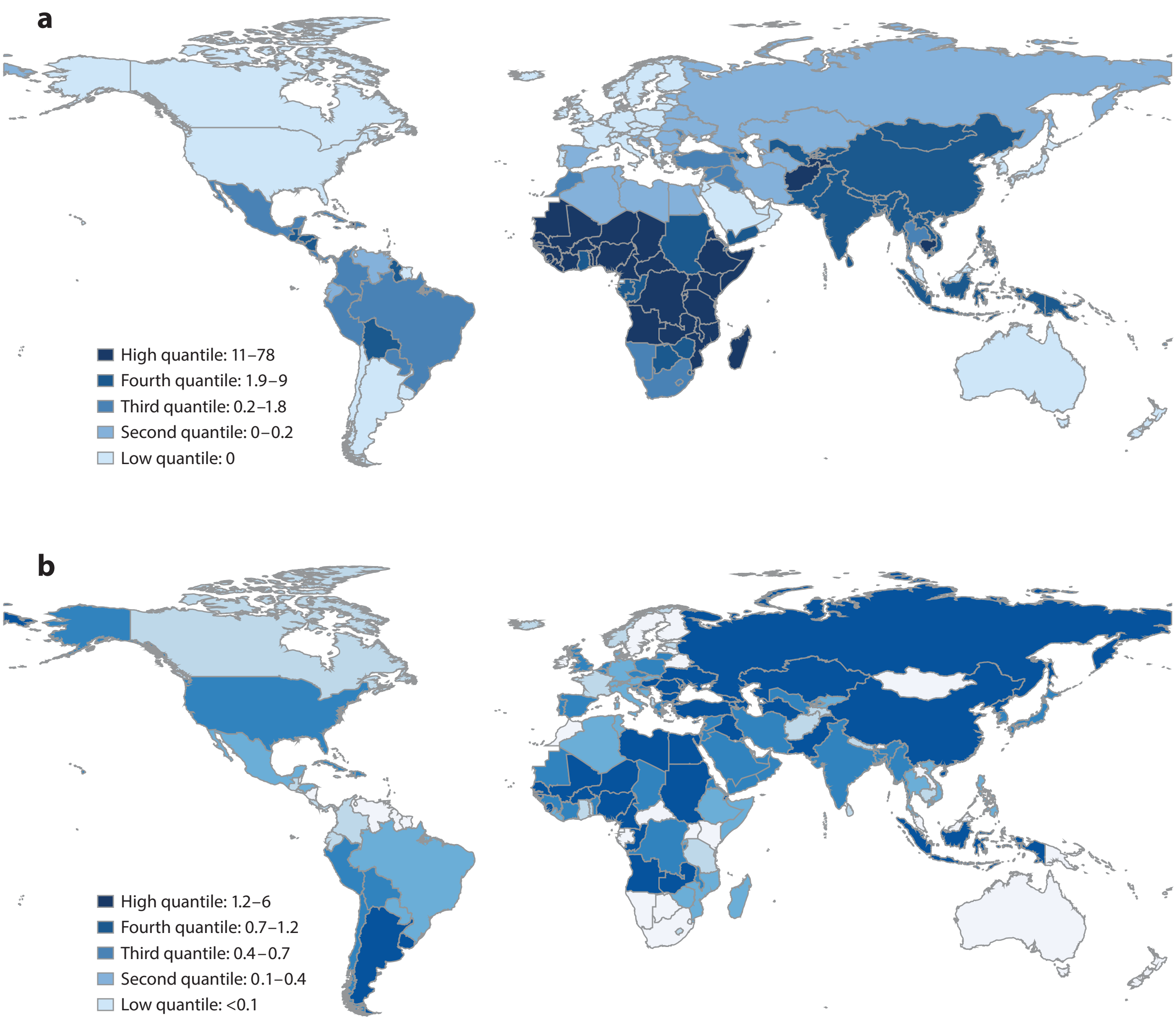

Figure 3

Global burden of disease in disability-adjusted life years (DALYs) per 1,000 people per year due to (*a*) indoor air pollution and (*b*) outdoor air pollution. Note the differences in scales between the two panels. Data from WHO 2004 environmental burden of disease data (WHO 2007).

3.2. The Model as It Relates to the Household Air Pollution Problem

The model reflects a set of issues that are important for understanding the basic challenges associated with household air quality. We discuss these issues in more detail in this section before turning to the implications of the model.

First, the model includes an explicit link between household environmental quality and health, on the one hand, and community (e.g., ambient) environmental quality on the other hand, a link that is established through both behavioral and physical mechanisms. For example, ambient air quality—influenced by a mix of industrial sources, nonindustrial sources, and natural sources such as radon—physically affects household air quality (and vice versa) because home-building materials are often porous (Baumgartner et al. 2014). Behaviors are also critical, however, because householders may react to poor air quality inside the home by spending more time outdoors, opening windows to increase ventilation, or sealing their homes more completely, thereby affecting exposures. This link also highlights the important and recent emphasis in the exposure science literature on the difficulty of separating indoor and outdoor air quality in many real-world settings (Smith et al. 2014).

Second, the model allows for a very general connection between environmental quality and disease risks. More specifically, poor environmental quality that generates health risks (e.g., poor sanitation that leads to diarrheal diseases) that seem unrelated to air quality may in fact render those air quality risks more severe, if these other diseases generally decrease household resilience to health risks. Faced with multiple serious disease risks, a household may choose low averting investment if it is unable to sufficiently reduce the whole set of risks to deliver good health (Yarnoff 2011). The timing of the health effects of risks may also be important. Given that most of the negative consequences of air pollution do not manifest immediately (except perhaps for young children with ALRI), a household may prioritize reduction of risks that have shorter-term impacts. Alternatively, household averting behaviors (or community averting behaviors) that successfully reduce health risks may lead to reduced investment in future prevention due to the prevalence elasticity of demand for health risk reductions (Philipson 2000, Pattanayak et al. 2006).

Third, averting behavior enters the utility function directly as well as through improved environmental quality and reduced illness. This characteristic of averting behavior stems from the joint production aspects of activities that emit air pollution, as well as the potential psychic benefits of averting behaviors. For example, many important social interactions among householders may occur around activities such as cooking and eating, so some types of averting behavior may thus decrease exposures but harm utility. Smoke emissions also generate both costs and benefits that are unrelated to health, such as fouling household goods and assets (e.g., house walls), driving out insects, or producing valuable (or possibly uncomfortable) heat (Parikka 2004, Biran et al. 2007, Jeuland & Pattanayak 2012). Similarly, households often find the taste of certain foods to be better if these are cooked over an open flame (Bhojvaid et al. 2014), or they may prefer the physical appearance or other aspects of goods that release greater amounts of toxic compounds into the household environment. Averting behaviors that change the production of these benefits and costs will therefore also affect utility.

Fourth, by treating knowledge as a costly input, this formulation highlights the important role that is often played by a lack of awareness of averting solutions. Constraints on knowledge about the effectiveness of prevention behaviors in improving environmental quality, and on the health or other benefits that these behaviors may deliver, receive consistent mention in the literature (Pattanayak & Pfaff 2009, Ashraf et al. 2013, Orgill et al. 2013). Conversely, higher levels of education are often found to be positively related to the adoption of averting behaviors.

Fifth, the model acknowledges the role of preference parameters θ in influencing behavior in the production of household air quality and health. These preference parameters may relate to a household's relative weighting of immediate versus long-term benefits (i.e., time preferences). Time preferences will influence whether households make up-front investments in preventive health behaviors or technologies that deliver benefits only gradually or at some date far in the future, for example, investments in avoiding the many chronic respiratory disease conditions that potentially affect older adults (Speizer et al. 2006, Atmadja et al. 2014). Time preferences will also affect how households perceive the trade-off between technologies or interventions that cost more initially (e.g., efficient and advanced stoves, investment in mold removal) and those with higher operating costs (e.g., inefficient traditional stoves, installation of fans that run on electricity).

Given that sickness is not a certain outcome of poor environmental quality and that the efficacy of preventive technologies and the cost of any episode of illness are probably not fully known to households, risk and ambiguity preferences will also influence averting behavior (Courbage & Rey 2006, Finkelstein & McGarry 2006). Risk-averse households will typically seek out options that help insure them against poor outcomes, including averting and defensive expenditures. If the effectiveness of these preventive behaviors is unknown, however, risk and ambiguity aversion may lead to the opposite situation in which a household does not invest (Treich 2010). This negative influence on adoption may be especially relevant if the supply of clean fuels is unreliable, as is the case for the supply of gas and electricity in many low-income countries.

Sixth, the model includes a formal link between both sickness and environmental quality on the one hand and government policy on the other hand. Environmental quality clearly increases with effective government regulation of the negative externalities, including the community-wide pollution discussed above. Perhaps less obviously, government action can also influence the quasi-public goods that are household and community averting behaviors. More specifically, averting behaviors are quasi-public goods because they typically have effects that are external to households and therefore are not fully considered in private decisions (e.g., positive health spillovers on neighbors or larger-scale impacts, for example, on global warming). They may also require complementary supply-side investments (e.g., electrification infrastructure) that correct for market inefficiencies that arise from high fixed costs. Thus, government may stimulate private averting behavior by offering subsidies or mandating adoption of certain technologies or behaviors (e.g., testing for radon at the time of purchase of a new home) (Andalón 2013). The motivation for such policies could be to improve efficiency (by reducing negative spillovers on others), but need not be. Distributional concerns may motivate policies to decrease the cost of averting behavior for specific segments of the population, such as the poor. In addition, behavioral nudges aiming to correct common failings of private decision making may also be warranted in some cases (Loewenstein et al. 2007). Subsidies can also take the form of supports for the supply chain or complementary investments that make prevention technologies available—for example, rural electrification that allows for wider use of electric stoves and heaters in the place of biomass-burning technologies. Of course, such supports may also lead to greater generation of ambient pollution when the production of such complements generates harmful emissions or when there is substantial crowding out of private averting behavior.

3.3. Implications for Private Averting Behavior

As discussed in P&P, the model in Equation 1 points to a number of economically relevant concepts for understanding the nature of the HAP problem. In particular, the solution of the utility maximization problem represented in Equation 1 equates marginal opportunity costs (in terms of materials, knowledge, and time) with the marginal benefit produced by increasing consumption, leisure, and household environmental quality on the one hand and reducing sickness on the other. If we extend the basic result from P&P, the reduced form of the first-order condition for optimal averting behavior is

$$u_a + u_s \cdot (s_a + s_e \cdot e_a) + u_e \cdot e_a - \mu \cdot w \cdot s_a = \lambda \cdot f_a + \gamma \cdot g_a. \tag{2}$$

If the time and income constraints are assumed to bind, and if Equation 2 is combined with the other first-order conditions to the maximization problem, specifically with respect to the other decision variables *m, k,* and *t*, this expression simplifies to

$$u_a + u_s \cdot (s_a + s_e \cdot e_a) + u_e \cdot e_a - \mu \cdot w \cdot s_a = w \cdot a_t + p \cdot a_m + r \cdot a_k, \tag{3}$$

where the left-hand side represents the marginal benefit of averting behavior. This benefit includes the marginal utility produced directly by averting behavior (term 1, which may in some cases be a net marginal cost, as discussed above), reduced pain and suffering due to illness (term 2), an improved aesthetic environment (term 3), and fewer lost work days (term 4). The right-hand side expression then pertains to the costs of this averting behavior in the acquisition of time, material, and knowledge, e.g., the defensive or averting expenditures. These expenditures may involve locational sorting and migration into areas with better environmental quality (Tan-Soo 2015); in the environmental economics literature, such behaviors have typically been studied using hedonic property valuation models applied to the case of responses to outdoor air quality (Smith & Huang 1995).

One of the functions of this model is to organize our understanding of how households value marginal improvements in air quality, that is, to arrive at a value of their marginal willingness to pay. Starting with the result in Harrington & Portney's (1987) seminal article, this type of model has repeatedly been used to derive a microeconomic measure of the value of improvements in environmental quality. In particular, four economic avoided-cost concepts taken together—averting costs, costs of illness, opportunity costs of lost work days, and monetary value of pain and suffering—indicate the value of a better environment (Pattanayak et al. 2005).

The expression in Equation 3 also provides the basis for exploring the implications of the model by using comparative statics (Pattanayak & Pfaff 2009). Specifically, reductions in the prices of inputs should increase demand for averting behaviors. Increases in perceptions of the direct (joint production) benefits of averting behaviors should similarly increase demand, as should increases in the effects of averting behaviors on aesthetics and on health. These changes could be facilitated by a variety of interventions for which we consider the empirical evidence more carefully in Section 4.2, including subsidies on materials, relaxation of liquidity constraints that preclude large up-front investments, provision of new and useful information, technological changes that improve the efficiency or aspirational (i.e., status-enhancing) value of averting behaviors, and social marketing that moves perceptions of the value of averting behaviors. Meanwhile, reduced income and productivity, tighter budget constraints, and exogenous changes to the environment that improve health tend to decrease demand.

3.4. Some Complications

The idea that interventions to reduce the marginal costs of averting behaviors should increase averting behaviors and thus reduce sickness may seem obvious, but this notion is unfortunately overly simplistic for a number of reasons. For one, reduced prices generate a positive income effect for households. This effect will lead to a shift toward greater consumption and leisure, which will at least partially offset the substitution effect induced by lower prices. How these income and substitution effects change investments in health versus more consumption and leisure is, of course, an empirical question. The empirical effect will depend partly on the shapes of the indifference curves for each of these utility-generating goods. In addition, averting investments depend on their relative returns, which may be low with existing technologies (i.e., materials) and knowledge. In particular, if averting behavior directly contributes to utility through reduced sickness $(u_s \cdot s_a \gg 0)$ or improved environmental quality $(u_s \cdot s_e \cdot e_a \gg 0; u_e \cdot e_a \gg 0)$, then changes in prices will have a relatively stronger effect on averting behavior, all else being equal. Conversely, P&P discuss a case in which free testing to inform households about the presence of a contaminant may be insufficient if general knowledge about the risks of that contaminant are not understood (which corresponds to the question of how $s_e \cdot e_a$ affects utility).

However, when averting behavior has a direct negative effect on utility $(u_a < 0)$, perhaps due to aesthetic preferences, then there may be little to no shift in such behaviors from reduced prices. This may be particularly true if there are diminishing marginal benefits of reduced sickness and increasing marginal costs of these negative aspects of behavior change.

Second, we should reconsider interactions between various averting inputs. For example, a household may choose to offset better materials with less learning or to decrease time spent on averting behavior. Both of these choices will indirectly increase consumption through greater wage income from decreased time spent on averting behavior or lower expenditures on costly knowledge inputs. Similar effects can be seen for responses to other changes in averting input costs, and the total effect will again depend both on the shapes of the production relationships for sickness and environmental quality and on the trade-offs across goods in the utility function. Perhaps equally important, the degree of substitution that is possible across averting inputs seems critical. For example, if markets for clean stoves and fuels are missing and the health production function requires these materials, then subsidized knowledge will be insufficient.

Third, from the main model, we can observe that, even when averting behavior increases, if $u_c \gg 0$, there will be increased demand for consumption despite the negative effect that this consumption has on environmental quality. This polluting effect of consumption may thus cancel out health and environmental benefits from increased averting behavior. In other words, given that $e_c < 0$, the increased consumption induced through the income effect may lead to greater sickness. This is the mechanism behind the concept of induced demand [more commonly discussed in the transportation literature (Hymel et al. 2010)], whereby households may respond to cleaner cooking technologies by increasing the amount of cooking they do, which has clear implications for health benefits and fuel savings (Chaudhuri & Pfaff 2003).

Fourth, a variety of complex connections between averting behavior and the environment occur through broader community effects. P&P discuss the fact that one household's averting behaviors—perhaps induced by lower prices for chimney construction, for example—may in some cases decrease community environmental quality ($E_a < 0$) and lead to increased downwind health impacts due to porous home construction or time spent outdoors. Other types of behavior (e.g., adopting cleaner stoves, where $E_a > 0$) may, in contrast, induce positive spillovers for the health of neighbor households. In addition, when community averting behavior increases due to reduced prices, the health benefits produced by this community behavior may reduce the marginal benefits of private averting behavior because demand is prevalence elastic ($s_A < 0$). That is, as the air gets cleaner and the perceived prevalence of the disease decreases, the interest in averting behavior declines. The same logic also applies when government policy G improves household environmental quality.

Finally, as noted above in the discussion of the main model, the health and human capital literature suggests that wage is endogenous to health status. If this relationship is strong, both the marginal benefits of additional work days (due to improved productivity) and the marginal costs of time inputs will increase with additional averting behavior that improves health. The net effect of these changes will depend on the relative balance of these marginal improvements in productivity versus the need for increasingly costly time inputs. Given these various complications, examining the empirical evidence on the economics of HAP seems appropriate. We turn to this topic next.

4. EMPIRICAL EVIDENCE ON THE ECONOMICS OF HOUSEHOLD AIR QUALITY

This section reviews the empirical evidence related to household investment in averting behavior as described in the model presented above. We focus primarily on this evidence as it relates to household stove and fuel use, because such use is by far the most significant contributor to the

GBD from HAP, as discussed in Section 2. We first consider the evidence from observational studies and then turn to the results of experimental or quasi-experimental studies.

4.1. The Production of Household Air Pollution: Evidence from Observational Studies

We discuss findings for three patterns that emerge from observational studies aimed at understanding the economic dimensions of HAP: (*a*) the determinants of exposure to HAP (especially from solid-fuel use), (*b*) valuation of the economic costs of HAP, and (*c*) the effectiveness of private averting behavior for mitigating these negative consequences.

With regard to the first of these issues, the empirical literature on biomass fuel use by households—in the fields of exposure science, epidemiology, and economics—helps to explain why harmful emissions are generated inside the home. In this regard, Larson & Rosen (2002) first apply a household production framework to study the demand for improved household air quality. Findings from a range of studies of the determinants of adoption largely mirror those from the wider literature on environmental health behaviors in other domains, e.g., water-related disease and malaria prevention (Lewis & Pattanayak 2012). In particular, adoption of cleaner technologies is correlated with household-level demographic and socioeconomic factors such as higher income, access to credit and liquidity, increased education and awareness of the negative effects of air pollution, and gender of the head of household (Jack 2004, Gupta & Köhlin 2006, Farsi et al. 2007, Papineau et al. 2009, Gebreegziabher et al. 2012, Jeuland et al. 2014a, Bensch et al. 2015). Many of these same factors are identified in the literature on demand for radon mitigation (Wang et al. 1999, Riesenfeld et al. 2007). Several recent studies also apply discrete choice experiments to explore the degree of heterogeneity in household demand for different features of improved cookstoves (ICSs) (Jeuland et al. 2014a, van der Kroon et al. 2014).

This literature on household solid-fuel use also highlights the role of supply-side influences, including the availability or prices of clean alternatives like LPG and the prices, ease of use, and adaptability of ICSs for traditional food preparations (Gupta & Köhlin 2006, Akpalu et al. 2011, Ruiz-Mercado et al. 2011, Venkataramani & Fried 2011, Alem et al. 2013). Some studies consider how the adoption curve for clean stoves evolves over time (Beyene & Koch 2013) and discuss the striking lack of development of a supply chain for alternatives to traditional stoves (Lewis et al. 2014a). Lewis et al. (2014c) conduct a macro-scale quantitative appraisal of global ICS sales by using multivariate regression analysis of a unique dataset on product and organization features of more than 200 organizations across the world. They find that stove sales rose from 970,000 in 2008 to 2,800,000 in 2010 and that greater sales were associated with (*a*) stove testing; (*b*) low prices; and (*c*) large organizations, especially governments. They confirm that, although organizations are located in countries with high levels of respiratory illnesses and biomass fuel use, sales levels are correlated only with the extent of biomass fuel use, and not with health.

With regard to the second issue—valuation of the economic costs of HAP—research to date is surprisingly limited. Although the recent epidemiological literature is rich with findings on the ill effects of solid-fuel burning for a variety of health endpoints (as discussed in Section 2), the majority of valuation studies for improved indoor air quality come from middle- or upper-income countries (e.g., Carlsson & Johansson-Stenman 2000, Chau et al. 2008). Furthermore, most of these studies relate to occupational issues, applying the hedonic property valuation method (Addae-Dapaah et al. 2010) or focusing on the link between office air quality and work productivity (Wargocki et al. 2000, Wyon 2004, Fisk & Seppanen 2007). With regard to HAP, a few studies use data from household surveys to determine the economic damages to health from the use of solid fuels, applying valuation concepts such as cost of illness and the value of a statistical life (Arcenas et al. 2010, Pant 2012). A small set of cost-benefit analyses of improved technologies also incorporate environmental cobenefits in terms of reduced forest degradation and global climate damages (Hutton et al. 2007, Jeuland & Pattanayak 2012).

The third aspect of the HAP problem identified above concerns the effectiveness of behaviors for mitigating the negative consequences of biomass burning. In this regard, there is fairly good evidence that use of cleaner stoves and fuels is associated with lower time spent cooking and collecting fuel. Brooks et al. (2015), for example, find that rural LPG stove owners consume less biomass and spend less time cooking and collecting fuel than do nonowners, after one accounts for community characteristics and observed differences across households. Nepal et al. (2011) offer contrasting evidence, however, showing that some ICS owners have higher firewood consumption than do traditional stove users. If ownership of multiple stoves increases cooking activity and fuel consumption through an income effect, fuel use and pollution may also increase.

There is a growing literature on the importance of fuel and stove choice in determining household and individual exposures to air pollution (Smith 1993, Ezzati et al. 2000). For example, both Pant (2012) and Lewis et al. (2014b) find evidence of lower exposures among users of clean technologies after controlling for various household-level confounders. A more limited and inconclusive set of studies explore the effects of home design and behavioral responses that improve ventilation or decrease exposures, including keeping doors and windows open during cooking (Pitt et al. 2005, Dasgupta et al. 2006). For example, Dasgupta et al. (2006) find that structural features greatly influence air pollution levels, whereas Pitt et al. (2005) argue that the primary response for coping with poor air quality is in terms of intrahousehold allocation of time and cooking tasks. In particular, deducing the link between air quality and outcomes may be challenging, given that women with worse health have greater exposure to smoke, whereas those with younger children have lower exposures.

With regard to the health impacts of adopting cleaner cooking technologies, Mueller et al. (2011) conduct one of the few studies that control for differential selection into clean stove ownership and find that cleaner stoves do improve health outcomes. In general, though, the lack of rigorous evidence on the link between adoption of clean technologies and health improvements is best explained by a collective set of facts and challenges, including (*a*) the nonlinearity of the exposure–health response function, (*b*) low levels of adoption of cleaner technologies in many settings and the potential for confounding of impacts by correlated unobservables, and (*c*) importance of behavioral responses to ownership of cleaner technologies.

Indeed, one of the most important recent findings from the environmental health literature on stove emissions relates to the shape of the relationship between exposures and health risks. Decades of work have contributed to a broad consensus that fine particulate emissions ($PM_{2.5}$) from biomass burning must reach extremely low levels to deliver a significant reduction in the risk of ALRI (Ezzati & Kammen 2001), which is the most readily observable short-term health impact of averting behavior. Framed in terms of the household production model presented in Section 3, sickness is highly nonlinear in air quality. The health production curve stays flat and at very low levels over a wide range of low environmental qualities, and the curve rises (steeply) only once a high level of environmental quality has been achieved (Burnett et al. 2014). Achieving health benefits—at least with respect to PM—therefore requires a very significant level of household averting behavior that is complemented by a relatively clean ambient environment.

In rural environments in low-income countries, where ambient air quality is often relatively good, households tend to be poor and to have low education and limited awareness of the potential negative impacts of smoke. They may also have fairly ready access to biomass fuel and limited access to alternative energy supplies (Gebreegziabher et al. 2012, Lewis et al. 2014a). Budget and information constraints and relatively low biomass fuel costs thus discourage investment in pollution-averting behavior, and household air quality is low and dominated by pollution from inefficient biomass cooking. It is unclear whether providing cleaner alternatives in such settings will result in sufficient adoption and reduction of pollution to produce measurable health impacts. In contrast, higher-income and better-educated households in urban areas have greater demands for averting technologies and often face lower net prices for defensive expenditures (due to the higher cost of biomass fuel in urban areas) (Gundimeda & Köhlin 2008). Yet ambient air quality in the urban environments of lower-income countries may be poor due to higher population density and other sources of pollution, and any improvements in household air quality may thus be offset by low outdoor ambient air quality (Papineau et al. 2009).

In fact, the lack of documented effectiveness of averting behavior for delivering health improvements through reductions in HAP is not limited to solid-fuel use alone. With radon, for example, there is evidence that information can change risk perceptions (Smith et al. 1990) but that timely household adoption of recommendations for mitigation following testing is often low (Ford & Eheman 1997). There is little to no published evidence that household behavior to avoid radon has any impact on health, and the cost-effectiveness of policies to reduce exposures to this type of contaminant has also been controversial. For example, Gray et al. (2009) find that radon prevention is cost effective in the United Kingdom only if conducted at the time of construction of new homes, due to the high cost of remediation once a house has been constructed.

4.2. The Production of Household Air Pollution: Evidence from Analyses of Interventions and Policies

The literature on evaluation of policies and interventions to reduce OAP is fairly rich; see, for example, Portney (1990) for discussions of the value of amendments to the US Clean Air Act in the early 1990s, Stavins (1998) on lessons from US SO_2 emissions trading policies, and Greenstone & Hanna (2014) for a recent analysis of the value of air pollution regulations enacted in India. Interventions to address HAP, in contrast, have received much less attention. This lack of attention is in part because of the lack of clear evidence that clean household energy technologies cause measurable health improvements.

There are many possible reasons for this relative lack of evidence in support of interventions to decrease HAP. First, the idea of intervening in this environmental health domain—in contrast to a longer tradition of foreign donor assistance activity in water and sanitation or malaria control—is fairly new; the Global Alliance for Clean Cookstoves (GACC), for example, was formed only in 2010. A second contributing factor may be that the problems of cooking technology adoption have only recently been highlighted as major issues worthy of study on their own. This lack of attention to the demand side of the intervention equation may partly explain why previous top-down efforts, for example, the National Program on Improved Chulha, met with limited success and achieved only a low rate of uptake of favored technologies (Kishore & Ramana 2002).

Policy interest in these questions is now changing, however, and there are today increasing efforts to promote a variety of cleaner technologies across a range of low-income settings. These efforts are allowing for greater use of experimental or quasi-experimental designs developed to answer questions that are specifically about adoption, in addition to the more traditional focus on impacts.

Contributing to the evidence on demand for improved cooking technologies, several studies use randomized designs to assess the different roles of prices, financing, preferences, and information in affecting purchasing decisions. For example, Pattanayak et al. (2014) use experimental data to show that demand for ICSs in rural Uttarakhand (India) is highly price elastic such that modest subsidies can have a large effect on purchases. This finding is consistent with those on demand for other preventive health technologies in low-income settings. Moreover, preferences for the improvements promised by ICS technologies clearly affect the likelihood of purchasing an ICS, the choice of an ICS, and the extent to which a household uses (and therefore benefits from) an ICS (Jeuland et al. 2014b). These issues have obvious implications for stove promotion programs, which generally do not allow beneficiaries to choose between several technologies. In another setting, households in Uganda appeared to consider an ICS to be a risky investment such that rent-to-own models or sales approaches that allowed payment over time substantially boosted adoption (Levine et al. 2013, Beltramo et al. 2015b). Finally, there is recent evidence on the role of neighbor and decision-leader preferences in affecting purchasing decisions (Beltramo et al. 2015a, Miller & Mobarak 2014). Taken together, the studies by Beltramo et al. (2015a) and Miller & Mobarak (2014) appear to indicate that such influences may have an asymmetric effect on purchases; i.e., negative signals about stoves reduce purchases, whereas positive signals have little effect.

Yet even with this new focus on demand, technological challenges continue to impede the design of effective interventions and policies aimed at reducing the health impacts of solid-fuel combustion. Much hope has been placed on improved-efficiency biomass stoves because these would not require a large-scale change in the supply of fuel (e.g., electricity or gas). Nonetheless, evidence of improved air quality from such biomass stove interventions is limited, with only a few intervention trials showing modest reductions in individual exposures to particulates (Smith et al. 2011, Hartinger et al. 2013, Rosa et al. 2014). Similarly, only two experimental evaluations show evidence of improvements in household health from such technologies (Smith et al. 2011, Bensch & Peters 2015). Both Smith et al. (2011) and Bensch & Peters (2015) note improvements in self-reported health, but Smith et al. find only statistically insignificant reductions in diagnoses of pneumonia cases from use of a ventilated biomass ICS. In a quasi-experimental study, Yu (2011) combines a difference-in-difference methodology with propensity-score matching techniques to show that both ICSs and behavioral interventions in China contributed to reduced ALRI. On the negative side, Hanna et al. (2012) conduct a long-term randomized evaluation of biomass ICSs in Orissa, India, and fail to find any evidence of health improvements. Collectively, these results are consistent with the idea that efficient biomass stoves may not sufficiently reduce exposures to deliver measurable health benefits. The null results in Hanna et al. (2012) are probably also related to breakage and low sustained use of the ICS model that was promoted in the intervention.

The evidence on solid-fuel savings from randomized field experiments of efficient biomass stoves is also limited but is less ambiguous than that for improved health (Gebreegziabher et al. 2014, Bensch & Peters 2015). Such evidence lends credibility to the results from observational studies (described above) that indicate that such updated technologies do reduce fuel expenses.

Importantly, there has been only one evaluation of the impact of an intervention to promote a technology that uses cleaner commercial fuels, probably because ensuring supplies of such alternative fuels in most relevant settings (predominantly rural and low income) requires major complementary investments in the supply chain for fuels. Pattanayak et al. (2014) find that households who were subjected to a stove sales pitch and received subsidies in rural India use less biomass fuel than do control households, although they continue to use their traditional stoves alongside the new stove. Work to assess the impacts of these stoves on air quality and health is ongoing.

5. CONCLUSIONS

Traditional energy technologies and consumer products contribute to household well-being in diverse ways but often damage household air quality and human health. Although we begin above by discussing the global distribution and scale of HAP emissions, we note that the negative effects of HAP arise predominantly from cooking and heating. Drawing on household production theory, we illustrate the potentially ambiguous relationship between household utility and adoption of behaviors and technologies that decrease HAP.

With regard to the empirical literature, five key observations emerge. First, most research examines how demand for HAP reduction varies with income, education, and liquidity. A smaller literature argues for more attention to factors that affect supply, such as pricing plans, the appropriate technology, the supply chain, complementary infrastructure (e.g., roads, banks), and local institutions. Unfortunately, most of this work relies on convenient cross-sectional samples and therefore remains merely correlational.

Second, economic valuation of HAP reduction benefits is surprisingly limited. Although the recent epidemiological literature finds that solid fuels impair health, very little of this research is coupled with behavioral or economic data to allow for estimation of monetized benefits. Furthermore, most valuations of HAP reduction have been derived for middle- or upper-income countries and focus on occupational health.

Third, household behavioral adaptations (averting behaviors) can reduce fuelwood use and, to some extent, HAP exposures. However, these gains do not always translate into improvements in health outcomes, possibly due to some combination of (*a*) nonlinearity in the exposure–health response function, (*b*) low adoption of clean technologies, and (*c*) behavioral responses to ownership of cleaner technologies that undermine reductions in HAP exposure.

Fourth, most knowledge about effective policies and programs comes from studies of OAP in high-income countries, not from careful evaluations of actual policies to reduce either OAP or HAP carried out in poor regions of the tropics and subtropics. A small and growing experimental literature is attempting to fill this gap, but generalizing from these limited findings would be premature.

Fifth, technological optimism remains the Achilles heel of the HAP conundrum. The existing improved biomass cookstove technologies are simply not clean enough, especially at prices that will allow scaling up to serve 3 billion people around the world. Unfortunately, there appears to be no promising technological pipeline for developing and deploying sufficiently clean biomass cookstoves (Sovacool 2012).

These findings and challenges point to a set of important knowledge gaps. Research and evidence gathered to date have been extremely limited in several domains. Therefore, we believe that it is vitally important to build a research program that addresses the issues discussed below.

First, we need to better understand how improved biomass-burning stoves can reduce HAP burdens in low-income countries. In part because they do not require a large change in the supply of fuel, such stoves have received significant attention in recent years. Yet it is important to recognize that biomass-burning ICSs have been heavily promoted in the past at great cost and with little success, for example, as early as the 1980s (Manibog 1984, Gill 1987, Barnes et al. 1993). It is particularly critical for economic research to apply rigorous impact evaluation methodologies, including RCTs and quasi-experimental approaches, to better understand household demand for, and benefits obtained from, such technologies. Rather than simply assuming the superiority of the latest innovative ICS model, such evaluations should do more to leverage insights gained from recent studies that point to the importance of incorporating user preferences into intervention designs (Gebreegziabher et al. 2014, Jeuland et al. 2014b, Bensch & Peters 2015).

Future evaluations should also better anticipate the multitude of potential household adjustments to cooking behavior. For example, positive income effects due to fuel savings may induce greater cooking and therefore increase HAP (Chaudhuri & Pfaff 2003). Alternatively, a new stove may induce changes in diet if the relative prices of different food preparations change with technology design. Cooking technology may also influence people's allocations of time spent in locations with varying pollution levels (e.g., inside the home, outside, or at work), with important implications for overall individual exposures and health benefits. Finally, a change in stove technology may influence investment in alternative interventions such as water and sanitation services or bednets, depending on whether interventions to address different health impacts are seen as complements or substitutes (Dow et al. 1999).

Second, it is important to value the full economic benefits of a transition toward cleaner options and HAP reductions. Such valuation includes not only the private health costs (or benefits) of inefficient (or improved) stoves to households, an area about which considerable uncertainty remains, but also the valuation of environmental externalities (e.g., pressure on local forests and loss of ecosystem services) and health externalities associated with such technologies. For the valuation of private benefits, studies have focused primarily on the demand for specific technologies; there is likely an opportunity to study whether individuals are willing to pay for a cleaner home environment by applying hedonic models to study variation in property values and variation in home infrastructures or designs. Two relevant and related questions that economists have ignored concern (*a*) the connection between ambient air quality [the more traditional domain of interest to economists working on air pollution (Pearce 1996)] and a household's own emissions and (*b*) the ways in which this connection may modify incentives for private adoption of cleaner technologies. Finally, the extent to which costs and benefits vary across space and time—which is of vital importance for the design of incentives that better achieve socially desirable levels of investment in pollution reduction—deserves greater attention (Jeuland & Pattanayak 2012).

Third, perhaps because of challenges related to designing research studies that would rigorously test the effects of varying supply-side inputs, little is known about the extent to which these inputs are complementary to incentives for averting behaviors. Such supply-side factors include roads and market connectivity, maintenance and servicing of stoves, local institutional involvement and capacities, and other vital infrastructure. Many of these complementary inputs are quasi-public goods that are chronically undersupplied in low-income settings and that have the potential to fundamentally change household calculations of costs and benefits. For example, a recent intervention to promote stoves in the Indian Himalayas effectively solved supply chain constraints by providing stoves at the doorstep of the potential consumer (Pattanayak et al. 2014).

Fourth, the importance of these quasi-public goods broadly reminds us about the widespread phenomena of thin, incomplete, and/or missing markets for many inputs and outputs in these settings. Missing markets (and associated transaction costs) can imply that households face effective shadow prices that are greater (or less) than observed "market" prices for, to provide one example, material inputs (which had to be subsidized in the Himalayan case). The prevailing reality of high shadow prices also implies a high failure rate for interventions designed under the strictly neoclassical assumptions of rational agents making choices in complete market settings. Under such circumstances, incentives (e.g., subsidized information or reduced stove prices) may be insufficient because they are dwarfed by nonmarket signals (e.g., local norms or ethnic politics). Economists can play an especially important role here by applying well-tested analytical tools to model the size, sign, and drivers of the wedge between market prices and shadow prices (Pattanayak 1997). For example, if road or nongovernmental organization (NGO) quality changes the effective price paid by households, we can first hypothesize and then field test how households in communities with differential road or NGO quality will respond to sales campaigns.

Finally, the complementarity of the supply- and demand-side constraints discussed above points to a bigger methodological concern. The dominant evaluation approach in the literature on intervention impacts (e.g., RCTs) takes a monocausal view of the problem—the focus is less on how specific variables impact behavior and outcomes collectively and more on isolating a single cause. Thus, researchers typically design and conduct impact evaluations in locations with a strong enabling environment that will allow isolation of the influence of the particular variable of interest. Choosing locations with strong enabling environments likely implies bias toward locations with a relatively high supply of similar quasi-public goods. Therefore, we cannot apply and scale up the findings from such locations and experiences. Indeed, attempts to utilize findings from such studies in global assessments of the net benefits of different strategies to promote HAP are likely optimistic (Jeuland & Pattanayak 2012, Whittington et al. 2012). The academic and practitioner communities must devise creative ways to study multiple institutional, sociopsychological, economic, and geographical drivers of behavior change so that we can develop appropriate policies and strategies for reducing the global negative effects of HAP.

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