The Economics of Household Air Pollution

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Table of contents

Abstract

Traditional energy technologies and consumer products contribute to household well-being in diverse ways, but often also harm household air quality. This paper reviews the problem of household air pollution (HAP) generation at a global scale, focusing particularly on the negative effects of traditional cooking and heating. Drawing on the theory of household production of improved health, we illustrate the ambiguous relationship between household utility and adoption of behaviors and technologies that decrease air pollution. We then review how the theory relates to the seemingly contradictory findings emerging from the literature on 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 this critical global problem.

Keywords: Air quality, household cooking, respiratory illness, health behavior, household production

JEL Codes: D13, I12, I31, O13, Q40, Q53

1. Introduction

Approximately 80% of the air humans' breathe during their lifetime occurs indoors – at home, work or school. Decisions about cooking and heating fuels, furnishings and consumer technologies, and building materials and configurations therefore can have consequences for human health (Huang et al., 2013). Furthermore, much of this inhaled air occurs inside dwellings because people spend many of their living hours inside the home (Sundell, 2004, Bureau of Labor Statistics, 2014). On a global scale, household air pollutants (HAP) pose 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 negative health impacts of indoor air quality include acute and chronic disease risks such as asthma, respiratory infection, cardiovascular disease, and cancer.

[Table 1 about here]

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 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 PM_{10} concentrations are more than five times the average annual concentration (20 μg/m^3) recommended as healthful by the World Health Organization (Figure 1). Levels in Karachi, Kabul and Delhi are 10 – 15 times the recommended level and some cities have even greater concentrations. This contrasts with most cities in Europe, United States and Japan, which are below or near the guideline (WHO, 2006, WHO, 2014a); stepping outdoors in lower income countries therefore certainly does not guarantee a breath of fresh air. In addition, other outdoor environmental hazards, such as poor water quality and conditions of 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 will see, its effects are most acute among households living in regions where use of commercial fuels (i.e., gas and electricity) for cooking and heating is limited. Commercial fuels tend to generate limited HAP because they either: 1) burn efficiently and completely when used indoors, as in the case of biogas or LPG; or 2) 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 about three-fifths of the global population used gas or electricity for cooking (Smith et al., 2013, IEA, 2012). The rates of use of such cleaner-burning household fuels show a strong positive association with indicators of socio-economic status, both within and across countries. This observation serves to motivate 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 paper, we also omit discussion of environmental tobacco (i.e. “second-hand”) smoke; the economics of smoking in general are reviewed elsewhere (Chaloupka and Warner, 2000).

We also note that HAP is typically co-produced, at least in the case of half the world’s population, generating two major non-health externalities at two different scales: (i) the degradation of local and regional forests and air quality; and (ii) global warming because of the climate-forcing caused by the black carbon that is emitted from incomplete burning of biomass.

These impacts are discussed elsewhere (Ramanathan and Carmichael, 2008, Bailis et al., 2014, Venkataraman et al., 2005), but add to the urgency of understanding how to induce household cooperation to reduce production of HAPs that will in turn deliver regional and global benefits.

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 about 3 billion people in low income countries. We therefore primarily orient our subsequent discussion around HAP concerns in poor countries. In Section 3, we present a stylized model that serves to illustrate 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), income and prices of polluting technologies, information and knowledge, markets and institutions, and preferences and social norms related to the behaviors that generation 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 paper concludes in Section 5 with a brief summary of our findings and of the knowledge gaps that help define a future research agenda.

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 problems, but reviewing all contaminants of concern is beyond the scope of this paper. 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.

2.1. Overview of HAP 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., particulate matter (PM), nitrogen dioxide, polycyclic aromatic hydrocarbons) are primarily generated from combustion processes occurring within a household, for example from cooking or heating. Finally, a third category of contaminants comes from natural sources (e.g., radon) or biological sources that occur 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 has documented clear associations between various contaminants and a range of illnesses, particularly among 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.

[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.

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, while Howden-Chapmen et al. (2005) report that 35% of their New Zealand respondents indicate 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 associations between mold and health impacts often rely on respondent recall and visual and/or smell tests for mold presence (Bellanger et al., 2009, Zock et al., 2002, Rabinovitch, 2012).

Several recent studies have, however utilized more sophisticated mold measurement methods or have implemented randomized control trials (RCTs) of mold control interventions, allowing for better causal inference. Two 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 (Reponen et al., 2011, Vesper et al., 2006, Vesper et al., 2007, Cho et al., 2006). 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

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

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 applied and a fan installed in the attic. They conduct surveys and measure peak respiratory flow at baseline, after 6 months and one year after baseline, and 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, 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 (Sainz et al., 2009, Tracy et al., 2006). In fact, there is believed to be no concentration level that does not elevate lung cancer risks (Pacheco-Torgal, 2012). The World Health Organization has therefore identified an action level of 250 Bq/m^3, which generally can only be reached 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.

[Table 2 about here]

Radon exposure is believed to be the second most important cause of lung cancer after smoking, causing an estimated 21,000 US deaths per year out of the approximately 157,000 total US lung cancer deaths, which is also similar to the ratio in Canada (Lantz et al., 2013, USEPA, 2014a, Tracy et al., 2006). What is perhaps under-appreciated 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 (Lantz et al., 2013, Tracy et al., 2006). In the US there are only approximately 2900 annual radon-related lung cancer deaths among those who have never smoked (USEPA, 2014a). Table 3 presents estimated excess mortality for smokers and never-smokers.

[Table 3 about here]

Because children rarely smoke, focusing on children therefore 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, though there have been relatively few large-scale studies and radon measurement methods vary across the literature, potentially confounding results. On the other hand, a recent cohort study of almost 1.3 million Swiss children found no association between radon concentration and malignancies of any kind (median 77.7 Bq/m3 and 90th percentile was 139.9 Bq/m3) (Hauri et al., 2013). This collective body of evidence suggests that radon likely does have negative consequences for health, but that these likely 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 natural environment at about 1 μg/m3. In outdoor urban environments with heavy vehicle traffic, concentrations can reach 100 μg/m3 (Nielsen and Wolkoff, 2010), however, and it is often found at high concentrations indoors as well. This is because formaldehyde is used in press 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 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 USEPA classifies it as a probable human carcinogen (USEPA, 2014b). Duong et al. (2011) conduct a meta-analysis of 18 studies that finds 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), while the World Health Organization has established a guideline value of 100 μg/m3 for 30 minute indoor exposures. Reviews of scientific and dose-response studies point to levels ranging from 98 to 123 μg/m3 as preventative for respiratory irritation and carcinogenic effects in indoor environments (Nielsen and Wolkoff, 2010, Golden, 2011). In general, such concentrations are considered unlikely in most settings, although they may occur where highly formaldehyde-intensive construction materials are used.[3]

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

Thus, it seems clear that contaminants such as formaldehyde, radon, and mold can have significant negative effects on health. Putting the numbers in perspective, it would seem that radon might contribute to at most 10% of the burden of disease related to lung cancer, which itself ranks 16th on the list of causes listed in the global burden of disease (Lozano et al., 2012), and perhaps to other cancers. Mold clearly aggravates asthma, which ranks 42nd on the list, while the effects of formaldehyde are difficult to quantify but would appear to be geographically limited. This is less true for the case of combustion of solid fuels, which affects billions of people worldwide, and is the issue we consider in more detail in the section below.

2.2. The challenge of household use of solid fuels

Approximately 1.3 billion people, mostly living in low-income countries, do not have access to household electricity. These and many more – globally about 2.8 billion people (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 (Jeuland and Pattanayak, 2012, Grieshop et al., 2011, Smith et al., 2013). About 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 1/3 of the world population (IEA, 2012). Most of the projected continued reliance on solid fuels is due to increases in the lowest-income countries in Sub-Saharan Africa and Asia even as solid fuel use in higher income countries declines (Figure 2).

[Figure 2 about here]

Solid fuels tend to be self-collected or more affordable than cleaner-burning commercial fuels, and are easy to use in the traditional stoves that were developed specifically to handle solid fuels. As a result, those who live in rural areas of low and lower-middle income countries rely heavily on solid fuels (Bluffstone and Toman, 2014). The particular fuels, of course, vary across locations. For example, coal is commonly used in China and some parts of India, while 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 (Heltberg, 2004, Masera et al., 2000). Table 4 presents average household-level use of solid fuels in 8 countries using World Bank LSMS data. It 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, Heltberg, 2004). Fuels and the technologies that use them therefore tend to be "stacked", with households mixing technologies and fuels. For example, an urban household will often have and regularly use biomass, electric and LPG stoves (Masera et al., 2000).

[Table 4 about here]

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 (PAHs) (Smith et al., 2013, American Lung Association, 2011). Alarmingly, particulate concentrations in developing country kitchens where wood or other biomass is burned have been found to be 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 are known to cause various diseases, including lower respiratory infections (LRI) such as pneumonia, chronic obstructive pulmonary disease (COPD), cardiovascular disease, and cancers. Exposures typically start in utero and continue through childhood and into adulthood, which implies that cumulative lifetime exposures can be very high. This may be especially true 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 possibility of confounding by omitted variables or selection on unobservables, and bias of impact estimates up or down (Mueller et al., 2011). The negative impacts of PM2.5 and carbon monoxide on birth weight, child respiratory health (e.g. acute lower respiratory illness (ALRI) and pneumonia in particular) and mortality are perhaps best documented (Edwards and Langpap, 2012, Smith et al., 2000, Gajate-Garrido, 2013, Mishra et al., 2004), while effects on long-term cognitive and physical development remain uncertain. With respect to chronic impacts, a number of studies have used 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 and 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 often fail to find significant differences (Lewis et al., 2014b).

Recent global burden of disease (GBD) calculations, based exclusively on the impacts of particulates for which the best evidence exists, argue that about 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 they represent 16% of total outdoor concentrations (Smith et al., 2013). Thus the WHO estimates total 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 global burden of 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, lower respiratory infection (LRI) (not all attributable to HAP) is believed to cause an annual loss of 147 million DALYs (or 6% of total global BOD), which is second only to ischaemic heart disease.[5] In 2000 and 2011, LRI was the primary cause of reduced DALYs worldwide (WHO, 2014b, WHO, 2013).

[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 disability-adjusted life year (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 (YLL). The second component of disease impact is years lost due to disability (YLD), which captures the morbidity and infirmity associated with disease. These two components when added together comprise the DALY burden of disease (WHO, 2013).

3. A conceptual model for the production of household air quality

3.1. Basic formulation

In this paper, we apply a largely micro-level perspective to help 1) explain patterns observed in the global data on household exposure to HAP and its associated health burden and 2) motivate more nuanced thinking about the effects of interventions to reduce these. 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, our conceptual model starts from the idea that the decision to invest in preventive health or environmental improvements involves a tradeoff with consumption of other goods and leisure (Grossman, 1972, Pattanayak and Pfaff, 2009). 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, and 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:

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

where l is leisure, c is consumption, a represents risk averting behavior, s represents time spent sick, e is household environmental quality, and θ represents a set of preferences that affect the concavity and shape of the utility curve. Sickness s (produced by the health production function f) is decreasing in household environmental quality e and household averting, as well as aggregate community averting A and government action to reduce pollution G. In addition, the latter three factors – 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 decreasing in c, since 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.

Turning to the constraints facing households, potential averting 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 and wages obtained through work hours compensated at a wage rate w, is devoted to consumption, purchase of averting materials with price p, and acquisition of knowledge, which has unit cost r. The 24-hour time budget is allocated to leisure, time spent on risk averting, and time spent sick.

3.2. The model as it relates to the HAP problem

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

First, it includes an explicit link between household environmental quality and health on the one hand, and community (e.g., ambient) environmental quality, on the other, a link that is established through both behavioral and physical mechanisms. For example, ambient air quality – influenced by a mix of industrial, non-industrial sources and natural sources such as radon – affects household air quality (and vice versa) because home building materials are often porous; this constitutes a direct physical connection (Baumgartner et al., 2014). Behaviors are also critical, however, since householders may react to poor air quality inside the home by spending more time outdoors or open windows to increase ventilation, or alternatively may seal 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 could in fact render the latter more severe, if these other diseases 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). Alternatively, averting (or community averting) that successfully reduces health risks may lead to reduced investment in future prevention due to the prevalence elasticity of demand (Ahituv et al., 1996, Pattanayak et al., 2006).

Third, averting enters the utility function directly as well as through improved environmental quality and reduced illness. This is important because of joint production aspects of activities that emit air pollution, as well as potential psychic benefits of averting. For example, many important social interactions among householders may occur around activities of cooking and eating; some types of averting may thus decrease exposures but harm utility. Smoke emissions also generate both benefits and costs 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 (Jeuland and Pattanayak, 2012, Parikka, 2004, Biran et al., 2007). 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 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 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 may deliver, receive consistent mention in the literature (Orgill et al., 2013, Pattanayak and Pfaff, 2009, Ashraf 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 upfront investments in preventive health behavior or technologies that deliver benefits only gradually or at some date far in the future, for example in avoiding the many chronic respiratory disease conditions that potentially affect adults (Speizer et al., 2006, Atmadja et al., 2014). Time preferences will also affect how households perceive the tradeoff between technologies or interventions that cost more initially (e.g., efficient and advanced stoves, or investment in mold removal) versus those with higher running costs (e.g. inefficient traditional stoves, or 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 (Finkelstein and McGarry, 2006, Courbage and Rey, 2006). Risk averse households will typically seek out options that help insure them against poor outcomes, including averting/defensive expenditures. If the effectiveness of these preventive behaviors is unknown, however, risk and ambiguity aversion may lead to the opposite situation where a household does not invest (Treich, 2010).

Sixth, the model includes a formal link between both sickness and environmental quality on the one hand, and government policy on the other. Environmental quality clearly increases with effective government regulation of the negative externalities associated with pollution. Perhaps less obviously, government action can also influence the quasi-public goods that are household and community averting via subsidy 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 pro-poor concerns, or paternalistic motivations aiming to correct common failings of private decision-making may also apply (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 crowd out of private averting.

3.3. Implications for private averting behavior

As discussed in P&P, this model points to a number of economically relevant concepts for understanding the nature of the household air pollution problem. In particular, the solution of the utility maximization problem represented in equation 1 equates marginal opportunity costs (in terms of material, 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. Extending from P&P, the reduced form of the first order condition for optimal averting is:

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

Using the other first order conditions to the maximization problem, 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. This benefit includes the marginal utility produced by direct averting (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 lost work days (term 4). The right-hand side expression pertains to the costs of this averting, in time, material and knowledge acquisition, which are often referred to as defensive or averting expenditures. It is worth noting that these expenditures may involve sorting and migration into locations with better environmental quality (Tan Soo, 2014); in the environmental economics literature such behaviors have typically been studied using property hedonic models applied to the case of responses to outdoor air quality (Smith and Huang, 1995).

One of the implications of this model is to organize our understanding of how households value improvements in air quality, or their marginal willingness-to-pay, mWTP. Starting with the result in Harrington and Portney's (1987) seminal article, this type of model has repeatedly been used to derive a micro-economic measure of the value of improvement in environmental quality. In particular, four economic 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 implications of the model using comparative statics (Pattanayak and Pfaff, 2009). Namely, reductions in the prices of inputs should increase demand for averting. Increases in perceptions of the direct (joint production) benefits of averting should similarly increase demand, as will increases in its effects on aesthetics and on health. These changes could be facilitated by a variety of interventions for which we will consider the empirical evidence more carefully in Section 4.2, including subsidies on materials, relaxing liquidity constraints that preclude large upfront investments, provision of new and useful information, technological changes that improve the efficiency or aspirational value of averting, or 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, will tend to decrease demand.

3.4. Some complications

The idea that interventions to reduce the marginal costs of averting behaviors should increase averting and thus reduce sickness may seem obvious, but it is unfortunately overly simplistic for a number of reasons. For one, reduced prices generate a positive income effect for households. This will lead to a shift towards 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 vs. more consumption and leisure is of course an empirical question, which depends 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 directly contributes to utility through reduced sickness ($u_s \cdot s_a$ >> 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, all else being equal. Conversely, P&P discuss a case where 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 how $s_e \cdot e_a$ affects utility).

On the other hand, when averting behavior has a direct negative effect on utility ($u_a < 0$), 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 re-consider interactions between various averting inputs. For example, a household may choose to offset better materials with less learning or decrease time spent on averting. These will both indirectly increase consumption, through greater wage income or lower expenditures. 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 tradeoffs 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 increases, if $u_c >> 0$, there will be increased demand for consumption despite the negative effect this consumption has on environmental quality. This polluting effect of consumption could thus cancel out health and environmental benefits from increased averting. In other words, given that $e_c < 0$, the increased consumption induced through the income effect may in fact lead to greater sickness. This is the mechanism behind the idea that households might respond to cleaner cooking technologies by increasing the amount of cooking they do, which has clear implications for health benefits and fuel savings (Chaudhuri and Pfaff, 2003).

Fourth, there are a variety of complex connections between behavior and the environment that 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 $E_a > 0$) might in contrast induce positive spillovers for health. In addition, when community averting increases due to reduced prices, this could reduce the marginal benefits of private averting 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 declines. The same logic also applies when government policy G improves household environmental quality. Given these various complications, it seems appropriate to examine the empirical evidence on the economics of HAP. This is the topic to which we next turn.

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 this is by far the most significant contributor to the global burden of disease 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 HAP: Evidence from observational studies

We discuss findings on three aspects that emerge from observational studies aimed at understanding the economic dimensions of HAP: 1) the determinants of exposure to HAP (especially from solid fuel use); 2) valuation of the economic costs of HAP; and 3) the effectiveness of private averting behavior for mitigating these negative consequences.

Turning to the first of these issues, the empirical literature on biomass fuel use by households – in exposure science, epidemiology, and economics – helps to explain why harmful emissions are generated inside the home. In this regard, Larson and Rosen (2002) first applied 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 behavior in other domains, e.g., water-related disease, or malaria prevention (Lewis and Pattanayak, 2012). In particular, adoption of cleaner technologies is correlated with household-level demographic and socio-economic factors including higher income, access to credit / liquidity, increased education and awareness of the negative effects of air pollution, and gender of the head of household (Jeuland et al., 2014a, Jack, 2004, Gupta and Köhlin, 2006, Farsi et al., 2007, Gebreegziabher et al., 2012, Papineau et al., 2009, Bensch et al., 2014). 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 have also applied discrete choice experiments to explore the heterogeneity in household demand for different features of improved cook stoves (ICS) (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, or the prices, ease of use, and adaptability of ICS for traditional food preparations (Akpalu et al., 2011, Gupta and Köhlin, 2006, Venkataramani and Fried, 2011, Ruiz-Mercado et al., 2011, Alem et al., 2013). Some studies consider how the adoption curve for clean stoves evolves over time (Beyene and Koch, 2013), and the striking lack of development of a supply chain for alternatives to traditional stoves (Lewis et al., 2014a). Recently, Lewis et al. (2014c) conducted a macro-scale quantitative appraisal of global ICS sales 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) testing stoves, (b) low prices, (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 only correlated with the extent of biomass fuel use and not health.

Turning to the second issue, valuation of the economic costs of household air pollution, research to date is surprisingly limited. While the recent epidemiological literature is rich with findings on the ill effects of burning of solid fuels 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., Chau et al. (2008); Carlsson & Johansson-Stenman (2000)). Furthermore, most of these 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 (Wyon, 2004, Fisk and Seppanen, 2007, Wargocki et al., 2000). With regards to HAP, a few studies have used data from household surveys to determine the economic damages to health from 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 have also incorporated environmental co-benefits –

in terms of reduced forest degradation and global climate damages (Jeuland and Pattanayak, 2012, Hutton et al., 2007).

The third aspect of the HAP problem identified above concerns the effectiveness of behaviors to mitigate 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. (2014) for example find that rural LPG stove owners consume less biomass, and spend less time cooking and collecting fuel than non-owners, after accounting for community characteristics and observed differences across households. Nepal et al. (2011) offer contrasting evidence, showing that some ICS owners have higher firewood consumption than traditional stove users. If ownership of multiple stoves increases cooking 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, Pant et al. (2014) and Lewis et al (2014b) both 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 (Dasgupta et al., 2006, Pitt et al., 2006). For example, Dasgupta et al. (2006) find that structural features greatly influence air pollution levels, whereas Pitt et al. (2006) argue that the primary response for coping with poor air quality is in terms of intra-household allocation of time and cooking tasks. In particular, women with worse health have greater exposure to smoke, while those with younger children have lower exposures.

Turning to the health impacts of adopting cleaner cooking technology, 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 this question is best explained by a collective set of facts and challenges, including (i) the nonlinearity of the exposure-health response function, (ii) low levels of adoption of cleaner technologies in many settings and potential for confounding of impacts by unobservables, and (iii) 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 particulate emissions (PM2.5) from biomass burning must reach extremely low levels to deliver a significant reduction in the risk of ALRI (Ezzati and 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 environmental quality, and only rises (steeply) once a high level of environmental quality has been achieved (Burnett et al., 2014). Achieving health benefits – at least with respect to particulate matter – therefore requires a very significant level of household averting 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, have low education and limited awareness of the negative impacts of smoke. They also may 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 limit 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 observe health impacts. In contrast, the higher-income and better-educated households in urban areas have greater demand for averting technologies, and often face lower net prices for defensive expenditures (due to the higher cost of biomass fuel in urban areas) (Gundimeda and Köhlin, 2008). Yet ambient air quality in urban environments of lower-income countries may be poor due to higher population density and other sources of pollution, and household air quality could thus be compromised by low ambient air quality (Papineau et al., 2009).

In fact, the lack of effectiveness of averting behavior for delivering health improvements through reductions in household air pollution 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 household adoption of recommendations for mitigation following testing is often low (Ford and Eheman, 1997). There is little to no published evidence that household averting behavior has any impact on health, and the cost effectiveness of policies to reduce exposures to these contaminants has also been controversial. For example, Gray et al. (2009) find that radon prevention is only cost effective in the UK 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 HAP: 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, or Greenstone and Hanna (2014) for a recent analysis of the value of air pollution regulations enacted in India. Interventions to address household air pollution, in contrast, have received much less attention, and perhaps only partly because of the lack of clear evidence that clean technologies cause measurable health improvements.

There are likely many 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 donor activity in water and sanitation or malaria control – is fairly new; the GACC for example was only formed 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 Programme on Improved Chulhas, met with limited success and achieved only low uptake of favored technologies (Kishore and Ramana, 2002).

The momentum on 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 have used randomized designs to assess the role of prices, financing, preferences, and information in affecting purchasing decisions. For example, Pattanayak et al. (2014) use experimental data from rural northern India to show that demand for ICS (like many other preventive health goods) is highly price elastic in the same locations, such that modest subsidies have a large effect on purchases. Moreover, preferences for the improvements promised by ICS technology 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., 2014b). Finally, there is recent evidence on the role of neighbor and decision-leader preferences in affecting purchasing decisions (Miller and Mobarak, 2013, Beltramo et al., 2014a). Taken together, these two studies appear to indicate that such influences may have an asymmetric effect on purchases, in that negative signals about stoves reduce purchase, while positive ones have little effect.

Yet even with this new focus on demand, technological aspects continue to challenge 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 fuel supply (e.g., to 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 (Hartinger et al., 2013, Rosa et al., 2014, Smith et al., 2011). Similarly, only two experimental evaluations have shown evidence of improvements in household health from such technologies (Smith et al., 2011, Bensch and Peters, 2014). Both of the latter studies noted improvements in self-reported health, but Smith et al. (2011) found only statistically insignificant reductions in diagnoses of pneumonia cases from use of a ventilated biomass ICS. In a quasi-experimental study, Yu (2011) combined a difference-in-difference methodology with matching techniques to show that ICS and behavioral interventions in China both contributed to reduced ALRI. On the negative side, Hanna et al. (2012) conducted a long-term randomized evaluation of biomass ICS in Orissa, India, and failed to find any evidence of health improvements. Collectively, these results are consistent with the idea that efficient biomass stoves may not reduce exposures to levels sufficient to achieve health benefits, and the null results in Hanna et al. (2012) are probably also related to breakage and low sustained use of the ICS that was promoted in the intervention.

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

Importantly, there has only been one impact evaluation 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) found that households who were subjected to a stove sales pitch and received subsidies in rural India use less biomass fuel than control households, though 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. We began this review with a discussion of the generation of HAP at a global scale, but noted that the negative effects of HAP predominantly arise from cooking and heating. Drawing on the theory of household production of improved health, we illustrated the ambiguous relationship between household utility and adoption of behaviors and technologies that decrease air pollution.

Turning to the empirical literature, five generalities emerge. First, most research has examined how demand for HAP reduction varies by income, education, and liquidity. A smaller literature has argued for more attention to supply drivers such as pricing plans, appropriate technology, supply chain, complementary infrastructure (roads, banks), and local institutions. Unfortunately, most of this work relies on convenient cross-sectional samples and therefore remains correlational.

Second, economic valuation of the HAP reduction benefits is surprisingly limited. While the recent epidemiological literature finds that solid fuels impair health, very little of that is coupled with behavioral or economic data to allow estimation of benefits. Further, most valuations of HAP reduction come from middle or upper-income countries and focus on occupational health.

Third, household behavioral adaptations (averting and coping 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 (i) nonlinearity in the exposure-health response function, (ii) low adoption of clean technologies, and (iii) behavioral responses to ownership of cleaner technologies that undermine health benefits.

Fourth, most knowledge about effective policies and programs come from studies of OAP in high income countries, not from careful evaluations of policies to reduce either OAP or HAP carried out in poor regions of the tropics and sub-tropics. There is a small and growing experimental literature that attempts to fill this gap, but it would be premature to generalize these findings.

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 is no promising pipeline for developing and deploying sufficiently clean biomass cookstoves (Sovacool, 2012).

These findings and challenges point to a set of important knowledge gaps that are critical to better understanding the economics of household air pollution. 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 following issues:

First, we need a better understanding of 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 ICS have been heavily promoted in the past at great cost and with little success, for example as early as the 1980s (Barnes et al., 1993, Manibog, 1984, Gill, 1987). It is particularly critical for economic research to apply rigorous impact evaluation methodologies, including randomized control trials 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 also do more to leverage learning from recent studies that point to the importance of incorporating and accounting for user preferences into intervention designs (Bensch and Peters, 2014, Gebreegziabher et al., 2014, Jeuland et al., 2013).

Future evaluations should also better anticipate the multitude of household cooking adjustments. For example, positive income effects due to fuel savings may induce greater cooking and therefore increase HAP (Chaudhuri and Pfaff, 2003). Or a new stove may induce changes in diet if the relative prices of different food preparations change with technology design. It may also influence the allocation of time spent in locations with varying levels of pollution (e.g., inside the home, outside, or at work), with important implications for overall exposures and health benefits. Finally, it may influence investment in water and sanitation services or bednets, depending on whether interventions to address different health impacts are seen as complements or substitutes (Dow et al., 1995).

Second, it is important to value the full economic benefits of a transition towards cleaner options and HAP reductions. This 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 (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 primarily focused 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 value and variation in home infrastructure or designs. One relevant and related question that has been ignored by economists concerns 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 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 design of incentives that better achieve socially desirable levels of investment in pollution reduction – deserves greater attention (Jeuland and Pattanayak, 2012).

Third, perhaps because of challenges related to study design, little is known about the extent to which incentives for averting behaviors, and the policies that could create such incentives, vary with complementary supply-side factors, such as 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 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 remind 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 'market' prices, for material inputs for example (which had to be subsidized in the Himalayan case). It also implies that if the intervention is designed assuming strictly neo-classical assumptions of rational agents making choices in complete market settings, the market signals (e.g., in the form of subsidized information) could be insufficient because they are dwarfed by non-market 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 and shadow prices (Pattanayak, 1997). For example, if road or 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 a sales campaign.

Finally, the complementarity of supply and demand-side constraints discussed above point to a bigger methodological concern. The dominant evaluation approach (e.g., RCTs) takes a mono-causal view of the problem – not so much in asserting that the focus is on a sufficient variable that impacts behavior, but on isolating one cause. Thus, researchers typically design and conduct impact evaluations in locations with a strong enabling environment (or relatively high supply of such quasi-public goods) for obvious reasons, but the applicability of such findings and experiences to a broader scaling-up of similar activities is questionable. Indeed, studies of the global cost-benefits and cost-effectiveness of different strategies to promote prevention investment that utilize findings from such studies are likely optimistic (Whittington et al., 2012, Jeuland and Pattanayak, 2012). The academic and practitioner communities must devise creative ways to study multiple drivers of behavior change so that we can inform policies and strategies that can avoid coordination failures.

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Figure captions

Figure 1. Ambient air pollution levels in cities worldwide (WHO, 2014).

Figure 2. People relying on solid cooking fuel by major region and country (Data from IEA, 2006; regions that are not shown have very small populations using solid fuels).

Figure 3. Global burden of disease in DALYs per 1000 people per year due to (Top panel) indoor and (Bottom panel) outdoor air pollution (Source: Data from WHO, 2007). [Note the difference in scales]

Tables

Table 1. Major Indoor Air Contaminants (Adapted from Franklin (2007)).

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 / 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, ETS, cosmetics, paints
Environmental tobacco smoke (ETS)	Cigarettes, cigars, pipes
Biologicals (e.g. house dust mite, animal dander, mold, cockroaches)	Dampness, moisture, floor dust, bedding, insects, pets, pests
Radon	Soil and bedrock under homes, ground water

Table 2. Average Indoor Radon Concentrations in Select OECD Countries Bq/m^3

Country	Arithmetic Mean	Geometric Mean	Geometric Std. Dev.
USA	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
UK	20	14	3.2
France	89	53	2.0
Worldwide	39		

Source: WHO (2009)
Notes: 100 Bq/m^3 = 2.7 P/CL

Table 3. Radon-Related Excess Lung Cancer Mortality for Smokers and Never-Smokers

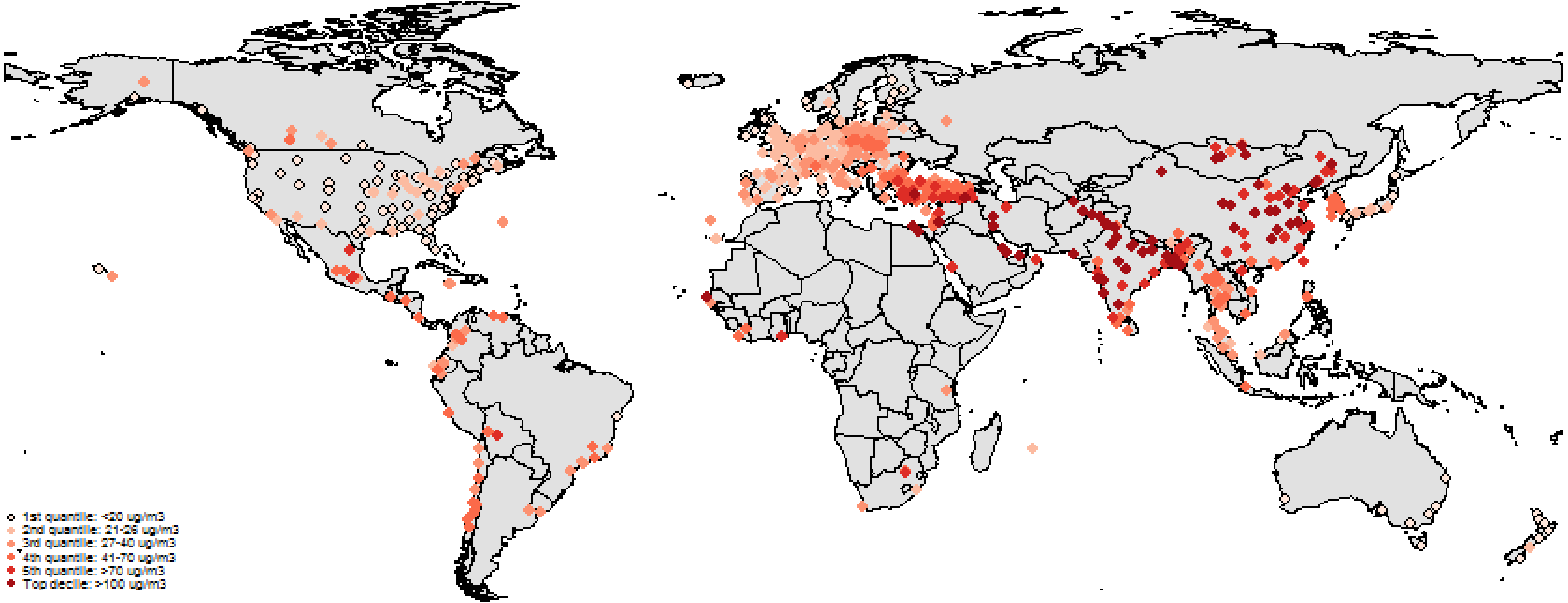
	Lung Cancer Risk/1000 Pop	
Radon Concentration	Smokers	Never-Smokers
20 p Ci/L	260	36
10 p Ci/L	150	18
8 p Ci/L	120	15
4 p Ci/L	62	7
2 p Ci/L	32	4
* 1.3 p Ci/L	20	2
** 0.40 p Ci/L	3	0

Source: USEPA (2014)
Notes: * Average Indoor Concentration ** Average Outdoor Concentration

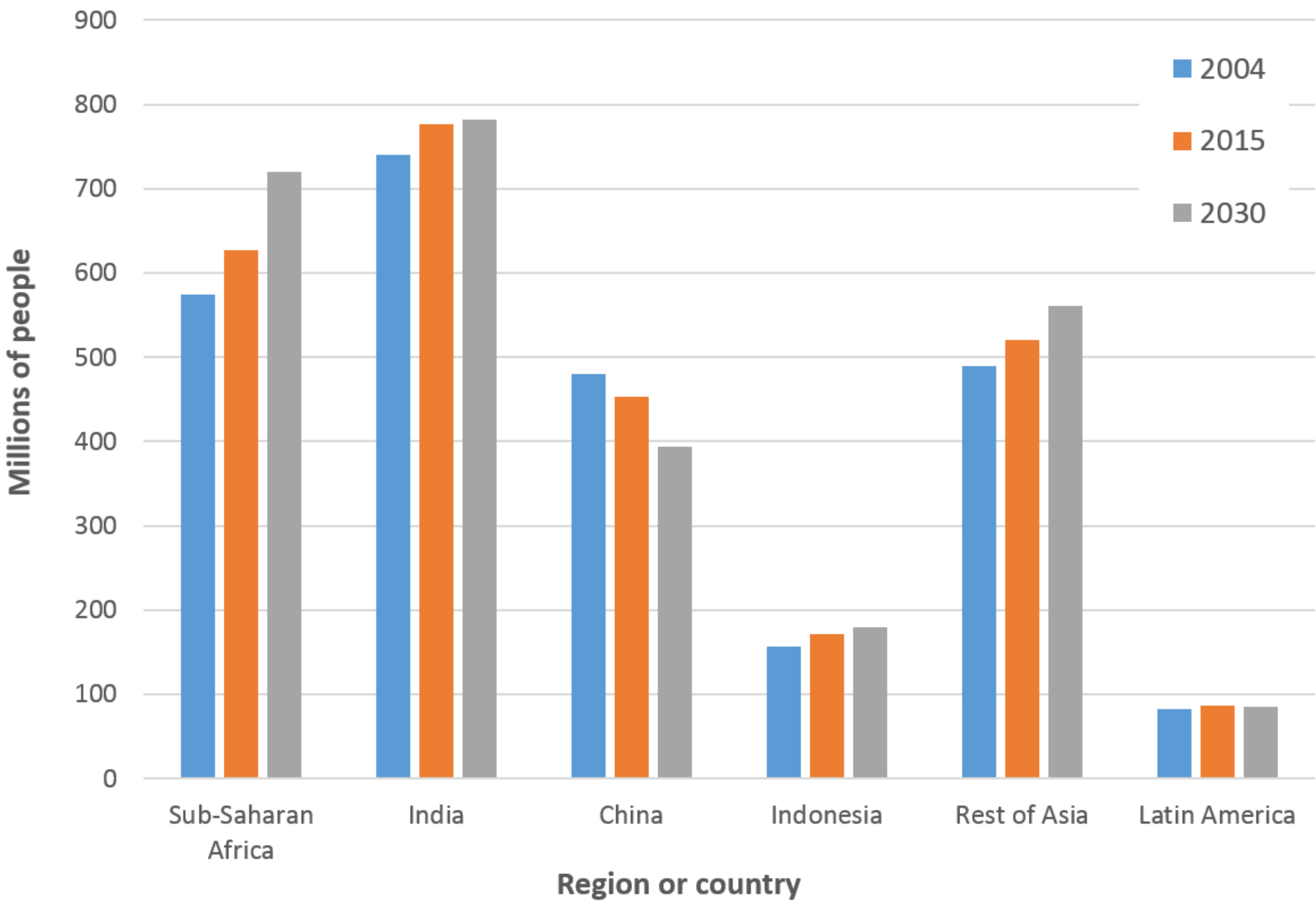
Table 4. Household Characteristics and Reliance on Solid Fuels in 8 Low and Middle Income Countries

	Characteristics		Solid Fuels				
	Per Capita Expenditure ($/Day)	% Urban	Fuelwood	Coal/ Charcoal	Dung	Straw/ leaves/ 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%

Source: Heltberg (2003; 2004).



Date Source: WHO Cities Ambient Air Pollution Data for 2008-2013



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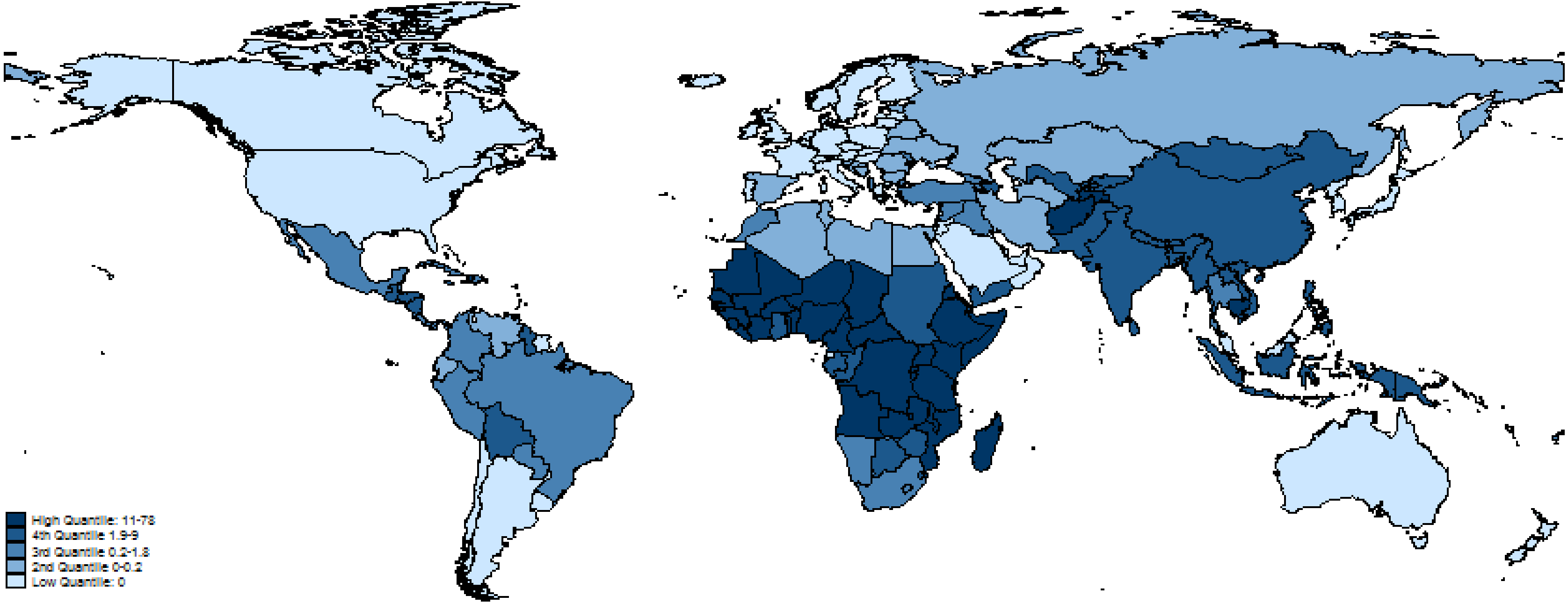
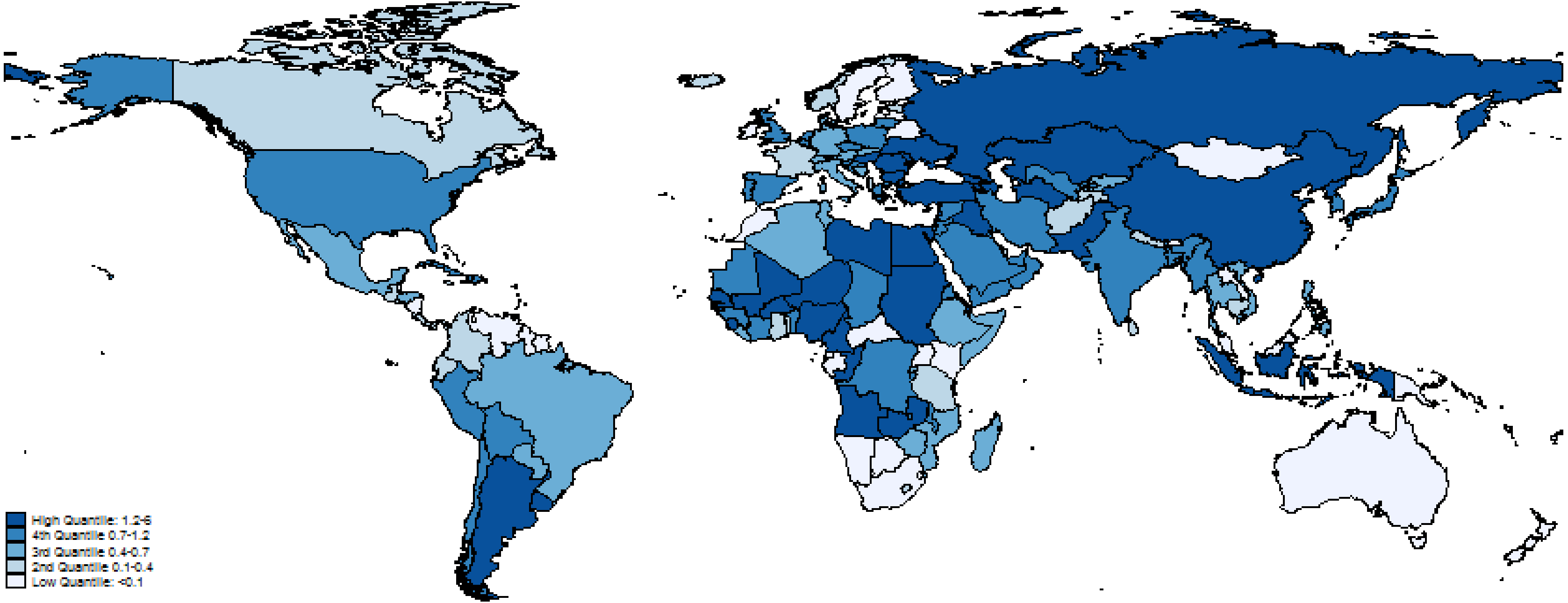


Date Source: WHO 2004 Environmental Burden of Disease Data



Date Source: WHO 2004 Environmental Burden of Disease Data