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## Health and climate benefits of cookstove replacement options

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## ABSTRACT

The health and climate impacts of available household cooking options in developing countries vary sharply. Here, we analyze and compare these impacts (health; climate) and the potential co-benefits from the use of fuel and stove combinations. Our results indicate that health and climate impacts span 2 orders of magnitude among the technologies considered. Indoor air pollution is heavily impacted by combustion performance and ventilation; climate impacts are influenced by combustion performance and fuel properties including biomass renewability. Emission components not included in current carbon trading schemes, such as black carbon particles and carbon monoxide, can contribute a large proportion of the total climate impact. Multiple ‘improved’ stove options analyzed in this paper yield roughly equivalent climate benefits but have different impacts on indoor air pollution. Improvements to biomass stoves can improve indoor air quality, which nonetheless remains significantly higher than for stoves that use liquid or gaseous hydrocarbons. LPG- and kerosene-fueled stoves have unrivaled air quality benefits and their climate impacts are also lower than all but the cleanest stoves using renewable biomass.

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## 1. Introduction

Solid fuels such as wood, coal and dung dominate household energy provision in developing countries. Globally, 3 billion people use solid fuels for cooking; only around a quarter of those – mostly in China – use some form of improved cookstoves (Legros et al., 2009). Efforts to upgrade household energy services for rural and urban residents of less developed countries (LDCs) have a long history. Early efforts aimed to reduce fuel use and the associated human labor and environmental degradation (Kammen, 1995). The past few decades have seen increasing interest in reducing the health impacts of indoor air pollution (IAP) from solid fuel use (Smith, 1993; Bruce et al., 2006). Recently, the emissions of long- and short-lived climate forcing agents by cooking fires have brought renewed attention to household energy provision in LDCs (Venkataraman et al., 2005; Bond et al., 2004b). Household cookstove interventions may

provide cost-efficient ways to mitigate carbon-equivalent emissions (Kandlikar et al., 2010; Smith and Haigler, 2008). Climate change mitigation as a rationale for cookstove interventions creates opportunities for program funding via carbon finance mechanisms.

This paper analyzes health and climate impacts of household cooking energy options that replace ‘traditional’ cooking means with some form of improved stove. The performance of improved stoves varies widely and is evolving rapidly as new technologies and approaches are developed. Implementation models and opportunities for leveraging carbon finance are also changing rapidly. There is a consequent need for a simple, extensible analytical framework that helps compare stove options across multiple impacts including climate and health. Here, we develop a broadly applicable framework for such analysis and apply it to a range of existing stove technologies with the expectation that it will be applied in assessing future stove designs. We begin with a brief overview of stove interventions’ potential for fuel savings, improving human health, and mitigating climate-change (Section 2). We then present the properties of a representative selection of stove–fuel combinations (Section 3.1). Next, we describe our method for estimating exposure and inhalation intake of indoor air pollution, based on ‘intake fraction’ (Section 3.2). We quantify climate impacts using the global warming commitment (GWC) framework (Section 3.3). We then apply these methods to our selection of stove technologies and

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explore the associated tradeoffs and co-benefits and discuss implications and limitations of our approach (Section 4). We conclude that the relative ranking of stove/fuel combinations in most cases depends on whether one emphasizes health or climate impacts.

## 2. Background: stove interventions to save fuel, lives and the climate

The term ‘improved’ is used to describe a wide range of replacements for traditional cooking methods, with a correspondingly large variation in performance. Traditional methods as defined here typically rely on a clay ‘U’ or three stones to support cooking vessels over an open fire, and do little to control combustion or optimize heat transfer, and thus are highly inefficient in their use of fuel. This inefficiency results in excess fuel use and the release of products of incomplete combustion (PICs), including gaseous (e.g., non-methane hydrocarbons [NMHC] and carbon monoxide [CO]) and particulate (e.g., black carbon [BC] and organic carbon [OC]) substances with documented health and climate impacts. Traditional stoves can be improved in three ways: (1) increasing thermal efficiency, (2) reducing specific emissions and (3) increasing ventilation. A stove’s thermal efficiency is the product of combustion efficiency (the proportion of fuel chemical-potential energy converted to thermal energy) times heat transfer efficiency (the portion of thermal energy that is transferred to the intended point, e.g., a cooking pot). Increasing thermal efficiency will generally reduce fuel requirement for a given activity but not necessarily reduce PIC emissions. Specific emissions, which are the emissions per activity (e.g., per fuel used or per food cooked), can be reduced without necessarily reducing fuel use—for example, if combustion efficiency is improved at the expense of heat transfer efficiency. Increased ventilation aims to remove stove emissions from the cooking area, thereby reducing exposure concentration (either via increased room air exchange rate or via active venting using a chimney); impacts on emissions and fuel use may vary. Importantly, stove programs with different goals might focus on ‘improving’ stoves differently.

Many early stove programs focused on improving thermal efficiency to address fuel shortages and reduce fuel use and its associated environmental impacts (largely, deforestation) and viewed reduced emissions/exposures as a secondary benefit. These efforts emphasized improved heat transfer efficiency via an enclosed combustion chamber and enhanced contact between hot gases and the cooking vessel; in some cases chimneys were also included. Chinese stove programs, operating in some form since the 1980s, are a good example of this type of program (Sinton et al., 2004). The Chinese program was successful in introducing improved stoves (> 100 million) and in reducing the pressure on biomass fuel sources, though it did not focus on reducing pollutant emissions or exposures. These programs generally introduced vented stoves with varying emissions and associated exposures but did not discourage the adoption of unvented coal stoves with high exposures and health impacts (and climate forcing emissions) (Edwards et al., 2004; Sinton et al., 2004). The Indian National Programme on Improved Chulhas (NPIC), which ran through 2002, was less successful owing to low stove uptake and poor stove durability and performance (Barnes, 1994; Venkataraman et al., 2010). A new Indian effort, the National Biomass Cookstove Initiative (NCI), has recently been announced with a goal of wide dissemination of biomass stoves with high efficiency and low emissions (Venkataraman et al., 2010).

The health impacts of indoor air pollution (IAP) are driven by indoor exposures during cooking, which disproportionately affect women and small children. Health endpoints commonly associated with IAP include stillbirth and low birth weight in infants

(Pope et al., 2010), acute lower respiratory infections (ALRI, typically pneumonia) in small children, and chronic obstructive pulmonary disease (COPD), tuberculosis and lung cancer in adults (Bruce et al., 2006). Globally, more than 1.6 million premature mortalities and 38 million disability adjusted life years (DALYs) in 2000 were attributed to IAP from solid fuel use; more than half the deaths, and 80% of the DALYs, are in children under 5 years old and are due to ALRI (Bruce et al., 2006). Assessments of the impact of improved stove interventions have found reductions in indoor and personal concentrations (Naeher et al., 2000a; Northcross et al., 2010; Zuk et al., 2006) and in respiratory symptoms (Ezzati and Kammen, 2001; Smith-Sivertsen et al., 2009).

Climate forcing emissions are another reason to address unimproved cookstoves (Smith, 1994). The reduction of CO<sub>2</sub> and some PIC emissions (CH<sub>4</sub> and N<sub>2</sub>O) from improved stove programs is now used as the basis for carbon offset projects (e.g., J. P. Morgan ClimateCare, 2010). Properly assessing and verifying projects is difficult; capabilities to do so effectively are being developed (Johnson et al., 2009). Recently, as the dramatic global and regional impacts of black carbon (BC) particulate matter (PM) emissions have become more clear (Ramanathan and Carmichael, 2008), addressing cookstove emissions has garnered renewed interest as a means to mitigate this potent short-lived climate forcing agent (Bond, 2007; Bond et al., 2004b; Grieshop et al., 2009). The short and complex atmospheric lifecycle and varied climatic effects of BC, and to a lesser extent other short-lived climate forcers such as CO and non-methane hydrocarbons (NMHCs), has led to their exclusion from current climate agreements (e.g., the Kyoto Protocol) and offset schemes. However, the acute climate impacts of BC – potentially including the disruption of monsoon rainfalls and the rapid melting of glaciers – and the strong evidence that limiting emissions from household fuel use provides a low-risk means by which to mitigate these impacts (Aunan et al., 2009; Bond, 2007; Kandlikar et al., 2010) give additional impetus to improve stoves.

Several factors impact the potential health and environmental benefits of a stove intervention project. For example, program design and implementation must consider and respect local cooking styles, food types, fuel supplies and cooking roles. Successful stove uptake involves, e.g., empowering women and local institutions and businesses and emphasizing market research and development. An effective technology by itself is not a solution (Barnes, 1994; Ezzati and Kammen, 2002). However, technology choices remain a central part of any effort to improve household energy provision, especially considering multiple performance criteria such as efficiency, indoor air pollution and climate forcing.

## 3. Methods: assessment of the relative climate and health impacts of stove interventions

Here we develop a method to compare and rank the health and climate impacts of stove–fuel combinations. Stove and fuel properties can impact usage. For example, gaseous fuels can be easily lit and re-lit throughout a day, while lighting a solid-fuel stove takes time and effort; as a result, solid-fuel users may sometimes keep stoves alight all day. Appropriate controlling for variations in usage across a wide range of stoves and fuels in different geographical locations with different usage patterns is challenging. Below, we compare stove impacts ‘per day’ of usage, assuming the provision of a fixed daily energy service. We feel this approach is justified for the comparisons made here, but recognize that it may underestimate differences among stoves, especially between solid- and non-solid-fuel stoves. Below, we quantify variability among stoves via central-tendency, high and low values for emission factors.

### 3.1. Stove activity and emissions

Cooking energy demand may vary by an order of magnitude or more across households, stoves and cultures. For example, cooking practices, family size and fuel availability influence ‘at the cookpot’ energy requirements. Estimates for average per-household energy demand include 11 MJ d<sup>-1</sup> in India (Venkataraman et al., 2010) and ~20 MJ d<sup>-1</sup> (which was compared to literature estimates of 10–20 MJ d<sup>-1</sup>) in Nepal (Pokharel, 2004). Estimates of *per-capita* cooking energy usage compiled by the World Energy Council included values ranging from 11 to 49 MJ d<sup>-1</sup> person<sup>-1</sup>; for comparison, per-capita values of only 2–3 MJ d<sup>-1</sup> person<sup>-1</sup> were estimated for modern devices and fuels and eating habits that include purchasing partially cooked foods (World Energy Council and Food and Agriculture Organization, 1999).

To facilitate comparisons across stoves on a consistent basis, we employ here a single value for energy consumption. We choose 20 MJ d<sup>-1</sup> stove<sup>-1</sup> as a representative value for the cooking energy demands per household. Employing another value would shift the absolute results proportionally; relative results (rankings) would not change.

A household’s annual fuel use (AFU, units: kg year<sup>-1</sup>) can be calculated from the ‘at-the-cookpot’ annual energy consumption (AEC; here, assumed 7300 MJ year<sup>-1</sup>):

$$\text{AFU} = \frac{\text{AEC}}{\eta_{\text{th}} H_{\text{fuel}}} \quad (1)$$

Here,  $\eta_{\text{th}}$  is the thermal efficiency (MJ delivered to pot per MJ chemical-potential in fuel) for a stove–fuel combination, and  $H_{\text{fuel}}$  is the fuel energy content (MJ chemical-potential per kg). Fuel use values are multiplied by fuel-based emission factors (EF; units: mass emitted per mass fuel burned) to estimate emissions of health- and climate-active pollutants. Emission factors applied here were synthesized from the literature, which predominantly contains data from standardized tests for measuring stove efficiency such as the Water Boiling Test (WBT). Efficiencies, estimated fuel usage and pollutant emission factors may differ in-home vs. in-laboratory (Ezzati and Kammen, 2002; Johnson et al., 2008; Roden et al., 2009), with performance often worse in-home (actual) than in-laboratory (Johnson et al., 2008). In-home emissions may vary with stove age and condition (Roden et al., 2009). Because in-use emission data are unavailable for many stove-types, central values employed here are from laboratory tests. Emission factor ranges per stove-type include in-use measurements where available.

This analysis can be updated to reflect the in-use performance of stoves as such data becomes available and as new stoves enter the market.

### 3.2. Stoves and fuels

Hundreds of stove designs are in use; a full accounting of all options is beyond the scope of this analysis. The options selected vary among attributes such as fuel type, stove type, and venting status. The stoves evaluated here (Table 1) represents a range of unimproved and improved options currently in use, and include unvented and vented stoves using renewable and non-renewable fuels. The unimproved options are wood used in an open fire or traditional clay pot support (abbreviation: ‘W-Tr-U’) and an unvented metal coal stove (‘Coal-U’) used in China (Zhang et al., 2000). Improved stoves considered include earlier-generation improved models tested over 10 years ago: a metal coal stove (‘Coal-V’) and a brick wood stove (‘W-Im-V’), both with chimneys and from China (Zhang et al., 2000), and an unvented stand-alone metal stove marketed in India (‘W-Im-U’) (Smith et al., 2000). More recently developed stoves were also considered: a ‘build in place’ vented masonry model with a large open cooking surface developed and used in Mexico over the past 7 years (Patsari stove; ‘W-Pat-V’) (Berrueta et al., 2008; Johnson et al., 2008; Zuk et al., 2006); a small (~20 cm diameter × 30 cm tall), cylindrical top-fed ‘gasification’ stove marketed in India (Karve Gasifier, ‘W-Gas-U’); and, a similarly sized stand-alone stove that uses a battery-powered fan to enhance combustion performance (Phillips Stove, ‘W-Fan-U’). Finally, we included a basic charcoal stove (‘Char-U’) widely used in India and elsewhere and free-standing stoves of the Kerosene wick-type (‘Ker-U’) and using Liquefied Petroleum Gas (LPG; ‘LPG-U’).

Table 2 provides emission factors synthesized from the literature for the eleven stove–fuel combinations selected. Emission factor values for different technologies can range widely; central values and ranges given in Table 2 span available measurements for similar technology types and are consistent with other literature sources. For example, emission factors for biomass stoves listed here generally lie within the ranges given for biofuel burning in the review of Andreae and Merlet (2001); EFs from the gasifier and fan stoves considered here are lower than those given by Andreae and Merlet (2001). It is important to note that particle emissions from biomass stoves (Kleeman et al., 1999; Li et al., 2009) and for fossil fuel and charcoal stoves (Chen et al., 2009) are

**Table 1**  
Stove–fuel combinations considered in this analysis.

Description – venting – material	Code	Fuel <sup>e</sup>	$\eta_{\text{th}}$ (%)	Estimated fuel use <sup>f</sup> (t year <sup>-1</sup> )
Wood stove (traditional) – unvented – open fire or mud stove	W-Tr-U	Wood	18 <sup>a</sup>	2.69
Indian wood stove (improved) – unvented – metal	W-Im-U	Wood	23 <sup>a</sup>	2.07
Chinese wood stove (improved) – vented – brick	W-Im-V	Wood	24 <sup>b</sup>	2.02
Mexican wood ‘Patsari’ stove – vented – masonry	W-Pat-V	Wood	24 <sup>c</sup>	2.06
Indian wood Karve ‘Gasifier’ stove – unvented – metal	W-Gas-U	Wood	32 <sup>d</sup>	1.53
Wood ‘Phillips Fan’ stove – unvented – metal	W-Fan-U	Wood	40 <sup>d</sup>	1.21
Indian charcoal stove (use+production) – unvented – metal/mud	Char-U	Charcoal	18 <sup>a</sup>	1.58
Chinese coal stove – unvented – metal	Coal-U	Coal	14 <sup>b</sup>	1.87
Chinese coal stove – vented – metal	Coal-V	Coal	17 <sup>b</sup>	1.54
Indian kerosene wick stove – unvented – metal	Ker-U	Kerosene	50 <sup>a</sup>	0.34
Indian LPG stove – unvented – metal	LPG-U	LPG	54 <sup>a</sup>	0.30

Sources: <sup>a</sup>Smith et al., 2000; <sup>b</sup>Zhang et al., 2000; <sup>c</sup>Berrueta et al., 2008; <sup>d</sup>Jetter and Kariher, 2009; MacCarty et al., 2008.

<sup>e</sup> Assumed fuel energy densities: wood 15 MJ kg<sup>-1</sup>, charcoal 26 MJ kg<sup>-1</sup>, kerosene 43 MJ kg<sup>-1</sup>, LPG 46 MJ kg<sup>-1</sup> (Smith et al., 2000), coal 27 MJ kg<sup>-1</sup> (Zhang et al., 2000).

<sup>f</sup> Calculated assuming given stove efficiency and fuel energy density and energy consumption (at the cookpot) of 20 MJ d<sup>-1</sup> (7300 MJ year<sup>-1</sup>).

**Table 2**  
Emission factors\* (EFs) of stove–fuel combinations.

	CO <sub>2</sub> (g C kg <sup>-1</sup> )	CO (g C kg <sup>-1</sup> )	CH <sub>4</sub> (g C kg <sup>-1</sup> )	NMHC (g C kg <sup>-1</sup> )	OC (g C kg <sup>-1</sup> )	EC (g C kg <sup>-1</sup> )	SO <sub>2</sub> (g kg <sup>-1</sup> )	PM <sub>2.5</sub> (g kg <sup>-1</sup> )	Sources
W-Tr-U	418 (400–440)	35 (29–41)	4.8 (2.5–7.1)	3.2 (0.5–7)	4 (6–0.7)	1.5 (0.2–1.8)	0.01 (0–0.27)	8.5 (6–10)	a–d
W-Im-U	419 (398–440)	29 (14.5–43.5)	2.9 (1.3–4.8)	8.5 (1.7–15.3)	1 (5.1–0.6)	1.2 (0.3–1.9)	0.01 (0–0.27)	2.9 (2–9)	e, f
W-Im-V	415 (394–435)	29.1 (14.6–43.7)	2.9 (1.3–4.7)	5.6 (1.1–10)	1.5 (3.2–0.7)	2.1 (0.5–3.4)	0.01 (0–0.27)	4.1 (2.6–5.6)	a, f
W-Pat-V	441 (463–420)	8 (3–19)	0.9 (0.2–1.9)	0.3 (0–1.9)	1.1 (4.4–0.8)	0.8 (0.1–1.1)	0.01 (0–0.27)	2.1 (1.6–5.4)	d
W-Gas-U	394 (374–414)	12.2 (9.7–14.6)	1.2 (0.6–2.4)	4.1 (2–8.1)	0.4 (0.9–0.2)	0.3 (0.1–0.6)	0.01 (0–0.27)	1.1 (0.5–1.5)	b, g
W-Fan-U	462 (439–485)	2.6 (2.1–3.1)	0 (0–0.5)	1.6 (0.8–3.3)	0.1 (0.1–0)	0.1 (0.1–0.2)	0.01 (0–0.27)	0.5 (0.2–1)	b, g
Coal-U	685 (650–719)	30.3 (15.2–45.5)	7.7 (3.9–11.6)	2.4 (1.2–3.6)	3.1 (4.7–1.6)	4.4 (2.2–6.5)	0.15 (0.3–0.07)	8.7 (4.4–13.1)	a, h
Coal-V	736 (581–810)	40.9 (24.6–71.1)	2.6 (0.6–5.3)	1.3 (0.5–2.6)	1.5 (3–0.7)	2.1 (1–4.1)	0.88 (1.75–0.44)	4.1 (2.1–8.3)	a, h
Char-U	1113 (887–1338)	205.7 (173.1–238.3)	46.5 (33.4–59.6)	63.5 (47.5–79.4)	8 (11.2–4.9)	2.3 (1.4–3.4)	0.01 (0–0.27)	0.4 (0.2–0.9)	h, i, j
Ker-U	825 (802–850)	7.6 (4–26.6)	0.2 (0.1–0.8)	12.8 (6.4–19.2)	0.1 (0.2–0)	0.5 (0.3–1)	0.03 (0.07–0)	0.5 (0.3–1)	a, e, h
LPG-U	841 (799–883)	6.4 (5.1–7.7)	0 (0–0.1)	14.1 (7.1–21.2)	0.1 (0.1–0)	0.2 (0.1–0.4)	0 (0–0)	0.5 (0.3–1)	a, e, h

**General notes:** range is  $\pm 1$  standard deviation when from a single study or range when multiple studies are used; PM<sub>2.5</sub> is assumed to be the same as total suspended particulate (TSP) or PM<sub>10</sub> mass for biomass stoves (Kleeman et al., 1999; Li et al., 2009) and for fossil fuel and charcoal stoves (Chen et al., 2009); Organic matter (OM) to OC ratio=1.4 for conversion of PM to OC (OM+EC=PM); estimates for SO<sub>2</sub> EF from biomass sources average of biomass sources from (Ref. a) and Andreae and Merlet (2001) (high).

Sources: (a) Zhang et al., 2000; (b) MacCarty et al., 2008; (c) Roden et al., 2006; (d) Johnson et al., 2008; (e) Smith et al., 2000; (f) Roden et al., 2009; (g) Jetter and Kariher, 2009; (h) Bond et al., 2004a; (i) Bailis et al., 2003; (j) Pennise et al., 2001.

**Source notes:** **W-Tr-U:** in-use CO<sub>2</sub>, CO, CH<sub>4</sub>, NMHC from (a) and (d); in-use OC, EC, PM<sub>2.5</sub> from (b) and (c); **W-Im-U:** WBT CO<sub>2</sub>, CO, CH<sub>4</sub>, NMHC from (e); WBT/in-use PM<sub>2.5</sub> from (e) OC, EC fractions from (c); **W-Im-V:** WBT CO<sub>2</sub>, CO, CH<sub>4</sub>, NMHC from (a); WBT/in-use PM<sub>2.5</sub> from (e) OC, EC fractions from (c). **W-Pat-V:** In-use/WBT tests from (d). **W-Gas-U** and **W-Fan-U:** WBT from (b) except PM<sub>2.5</sub> from (g). **Coal-U** and **Coal-V:** WBT from (a) with OC, EC fractions from (h). **Char-U:** Stove WBT emissions from (i), fuel production emissions from (j) with OC, EC fractions from (h). **Ker-U** and **LPG-U:** WBT from (a) and (e) with OC, EC fractions from (h).

\* Table shows medium, low and high emission-factor estimates. 'Low' and 'high' values, in parentheses, refers to species' impacts rather than EF values (e.g., due to its negative GWP, larger values of OC EF lead to lower GWC).

predominantly in the sub-micron size range, so from an emissions standpoint, PM<sub>10</sub> and PM<sub>2.5</sub> are essentially equivalent, though other indoor sources (e.g., re-suspended dust) may make important contributions to indoor PM<sub>10</sub> levels.

### 3.3. Exposure/health impacts

Smoke and other emissions from solid-fuel combustion contain thousands of gaseous and particulate chemical species (Naeher et al., 2007). PM<sub>2.5</sub> exposure is commonly used to quantify and compare health impacts of combustion sources. Here, we compare the inhalation intake of PM<sub>2.5</sub>, based on the hypothesis that mass inhaled is a better proxy for health impacts than mass emitted. We choose intake because intake fraction estimates for indoor emissions allow us to directly connect emissions and human intake, and because available estimates of intake-based risk factors for PM<sub>2.5</sub> enable the further connection from intake to risk. Intake and intake-fraction provide a logical, convenient, and robust basis for calculations presented here.

Health impacts of PM<sub>2.5</sub> are well studied. Morbidity and mortality health outcomes have been observed based on chronic exposures, acute exposures during pollution episodes, spatial and temporal variations in exposure, and occupational exposures (Pope and Dockery, 2006). A recent review concluded that there is a lack of compelling evidence that wood-smoke PM is any less toxic than other types of PM<sub>2.5</sub> (Naeher et al., 2007).

We employ here the log-linear intake–response relationship given by Pope et al. (2009). Evidence suggests that the mortality dose–response for PM<sub>2.5</sub> is consistent with a linear no-threshold model when considering only ambient concentrations in developed countries (annual average concentrations from  $\sim 5$  to  $\sim 30 \mu\text{g m}^{-3}$ ) (Pope and Dockery, 2006) or environmental tobacco smoke (ETS, 'secondhand smoke'). However, when also including higher concentrations, such as for a tobacco smoker or a person exposed to indoor solid-fuel combustion, the dose–response relationship appears to be log-linear (Pope et al., 2009). The dose–response curve 'flattening' (deviating from linearity) likely occurs for long-term average exposures on the order of  $100 \mu\text{g m}^{-3}$  (intake of  $\sim 2 \text{mg d}^{-1}$ ) or more, and has potentially large implications for understanding and reducing the health impacts of intermediate-

level exposures (Smith and Peel, 2010). Evidence supporting a log-linear dose–response includes studies of IAP exposures from solid fuel use (Ezzati and Kammen, 2001) and urban concentrations in LDC cities (Cropper et al., 1997). Linear dose–response models derived from low-concentration (approximately  $5\text{--}30 \mu\text{g m}^{-3}$ ) studies yield unrealistically high mortality estimates for indoor settings with larger concentrations (i.e., concentrations significantly greater than  $100 \mu\text{g m}^{-3}$ ) (Smith, 2000).

Many estimates of health impacts from indoor solid fuel use employ a 'bottom-up' approach based on population health outcomes and attributable individual disease risks (Smith, 2000). Bottom-up approaches might estimate, for example, mortality rates, years of life lost or disability-adjusted life-years (DALYs). The World Health Organization's Global Burden of Disease (GBD) project, for example, assessed global impacts of indoor air pollution (IAP) from solid fuel use (Smith et al., 2004) and benefits and cost-effectiveness of interventions (Bruce et al., 2006; Mehta and Shahpar, 2004; Smith and Haigler, 2008; Venkataraman et al., 2010).

The GBD approach estimates population exposure to IAP from solid fuel combustion using a binary metric: individuals in a population are either exposed to IAP, or they are not. Thus, detailed consideration of improved stoves' potential influence on exposure is not explicitly included in this approach. To the extent that emissions and exposure are included in GBD calculations, they are encapsulated in a 'ventilation factor', assigned based on analysts' qualitative judgment, that scales the population using solid fuels to account for technologies, ventilation and fuel choices that mitigate exposure. Ventilation factors for an exposed population vary between 1 (unvented use of traditional solid fuels) and 0 (use of 'clean' fuels or fully vented stoves); a ventilation factor of 0.25 was applied to the portion of China's population using improved biomass stoves (Smith et al., 2004). The GBD's coarse classification is useful in estimating the impact of improved stoves or ventilation on reduced disease incidence over a population, but is not designed to assess the differing exposures an individual or household might experience owing to different stove–fuel options. In our work, exposure and intake values are modeled explicitly, thereby allowing direct and physically meaningful comparisons among stove–fuel combinations.

We compare stove options in terms of estimated average personal PM<sub>2.5</sub> exposure and intake owing only to direct emissions of indoor air pollution. This approach considers a smaller portion of the emission–exposure–impact chain than is encompassed in the GBD studies, but carries the advantage of allowing straightforward comparison of technology choices. As illustrated below, if the dose–response relationship is log-linear, then a unit change in logarithm of intake yields a unit change in disease relative risk. We employ an intake fraction approach (Bennett et al., 2002) that links mass emitted and mass inhaled. Specifically, the individual intake fraction ( $iF_i$ ) quantifies the portion of stove emissions that are inhaled by an household individual, indoors:

$$iF_i = \frac{\text{Individual intake}}{\text{Total emission}} = \frac{\bar{C}_{\text{personal}} \cdot Q_b}{EF \cdot AFU} \quad (2)$$

where  $\bar{C}_{\text{personal}}$  is the annual average attributable exposure concentration for an exposed individual ( $\mu\text{g m}^{-3}$ ),  $Q_b$  is the respired volume for that person ( $\text{m}^3 \text{year}^{-1}$ ),  $EF$  is the PM<sub>2.5</sub> emission factor ( $\text{g kg}^{-1}$ ), and  $AFU$  is fuel use ( $\text{kg year}^{-1}$ ; Eq. (1)). Here, we assume that the exposed individual is in the same room as the emissions during all emission activities; prior work includes further exploration of this issue (Klepeis and Nazaroff, 2006; Nazaroff, 2008a, 2008b). Activity data suggest that in many settings women and the infants that they carry spend most (>60%) of their time indoors, much of that in the kitchen, and that much of the time during which fuel is burned is spent adjacent to the stove (Brauer et al., 1996). Our approach estimates annual intakes, which we convert to average daily intake. Exposures are often dominated by short episodes of very high concentrations, depending on stove activity and user location (Ezzati et al., 2000), which may influence health impacts (Ezzati and Kammen, 2001). Intake fraction values developed here employ an estimated average breathing rate for children and women at rest (Klepeis and Nazaroff, 2006), to represent the groups most likely to be highly exposed to cookstove emissions. Therefore, our intake estimates are for the primary stove user and may not be representative for all household members. We do not account for breathing-rate variability by age, gender and activity (Layton, 1993), which will not impact the relative ranking of stoves, though it would impact the actual intake for a specific individual. Finally, our estimates only include exposure and intake inside the house of the stove user; we do not include local, regional or global exposures or intakes.

We use available exposure and emissions data to estimate  $iF_i$  values for indoor use of an unvented stove in the LDC setting. Central tendencies of indoor exposure concentrations (Ezzati et al., 2000) and daily emissions (Bailis et al., 2003) from indoor stove use in Kenya were used in Eq. (2), along with an assumed average daily inhalation rate of  $7.8 \text{ m}^{-3} \text{ d}^{-1}$  for children plus female adults (Klepeis and Nazaroff, 2006). Ezzati et al. (2000) provide a distribution of 24-h average PM<sub>10</sub> indoor and exposure concentrations for multiple demographic divisions in houses using various unvented fuel–stove combinations. Bailis et al. (2003) estimate daily PM emissions ( $\text{g d}^{-1}$ ) from these different devices. Results here reflect weighted averages based on the number of stoves in use during the former study. Applying median values for these parameters yields an  $iF_i$  value of 1300 ppm (parts per million; 1300 mg inhaled per kg emitted) for females between 16 and 50 ('stove users' and the most highly exposed group); other demographic groups have values approximately 2 to 4 times lower. The 1300 ppm value is similar to an  $iF_i$  estimate of 1400 ppm derived for exposure to environmental tobacco smoke, another common IAP source (Klepeis and Nazaroff, 2006). Smith (1993) estimated total intake fraction for indoor cookstoves and secondhand smoke (2400 ppm) and for vented cookstoves (1000 ppm). Recent measurements of  $iF_i$

for cooking emissions in US kitchens, including those with ventilation hoods installed, ranged from 550 to 2300 ppm (Zhang et al., 2010). Well-functioning ventilated wood stoves contribute little to IAP levels in a developed-country context (Canada) (Allen et al., 2009). Estimates of population intake fraction for outdoor emission from residential wood burning in North America are orders of magnitude lower ( $\sim 15$  ppm) than for indoor emissions (Ries et al., 2009).

To address the impact of venting (e.g., use of chimneys) on attributable indoor exposures, the fraction of a stove's emissions retained inside a household is represented using  $f_{\text{unv}}$  ( $f_{\text{unv}}=1$  for unvented stoves). The individual intake fraction Eq. (2) associated with unvented stove use is multiplied by  $f_{\text{unv}}$  to yield an  $iF_i$  value for vented stoves. A 'perfect' chimney ( $f_{\text{unv}}=0$ ) would mean that none of the stove emissions enter that house's indoor air, directly or indirectly (e.g., emissions exiting in the chimney, then returning indoors via in-draft return of self-polluted air). The true value of  $f_{\text{unv}}$  for a vented stove depends on the design, condition and operation of the stove and chimney; similarly, it may vary over time, depending on meteorology, stove operation and combustion conditions. Recent measurements in Honduras document that even for a vented stove indoor concentrations can depend on stove condition (Clark et al., 2010). To estimate a central-tendency and range of values for  $f_{\text{unv}}$ , we compiled data from studies that measured PM concentrations inside and outside homes with unvented and vented stoves, and then applied the following mass-balance relationship:

$$f_{\text{unv}} = \frac{C_{\text{ven}} - C_{\text{out}}}{C_{\text{unv}} - C_{\text{out}}} \quad (3)$$

Here,  $C$  is the measured integrated PM concentration, and the subscripts 'out', 'unv' and 'ven' indicate measurements collected outdoors and in houses with unvented and vented stoves, respectively. In a well-sealed house, Eq. (3) could yield negative values for  $f_{\text{unv}}$ ; however, based on available data from developing countries, we did not identify studies where central-tendency estimates involved  $f_{\text{unv}} < 0$ .

Eq. (3) implicitly assumes that stove emissions and household air exchange rates in the houses with vented and unvented stoves are identical. This approach produces a conservative (low) estimate for  $f_{\text{unv}}$  if vented 'improved' stoves in these studies actually produced PM at a lower rate than the corresponding traditional stoves during a usage period. However, field tests in developing countries generally report that stove efficiencies (Boy et al., 2000; Granderson et al., 2009) and PM emission factors (Roden et al., 2009) are similar for vented biomass stoves as for traditional cooking methods. Therefore, we expect that this method will yield reasonable  $f_{\text{unv}}$  estimates for estimating indoor exposures from vented stoves.

Table 3 summarizes the studies used in the  $f_{\text{unv}}$  calculations. All studies in Table 3 were conducted in Central America, where the most extensive set of stove effectiveness measurements have been collected. Two types of studies are shown: 'intervention' studies, in which concentrations were measured separately in the same houses using traditional and then vented stoves (the *Patsari* stove in Mexico (Zuk et al., 2006) and the *Plancha* in Guatemala (Naeher et al., 2000a)), and 'cross-sectional' studies, in which PM concentrations were measured in many houses using vented stoves or traditional (unvented) cooking methods. The use of intervention studies is preferred when estimating  $f_{\text{unv}}$  because stove use patterns, housing layouts and air exchange rates remain relatively constant across conditions. However, cross sectional studies are less labor-intensive and thus allow larger number of homes and stoves to be sampled, potentially reducing error and bias.

The median value for  $f_{\text{unv}}$  from the studies in Table 3 is 0.18; the median is essentially unchanged ( $f_{\text{unv}}=0.18$ ) if 'intervention'

**Table 3**Studies used to estimate the fraction of PM emissions from vented stoves that remain indoors ( $f_{\text{unv}}$ ).

Source	Measure	Study details	$C_{\text{unv}}$	$C_{\text{ven}}$	$C_{\text{out}}$	$f_{\text{unv}}$
<b>Intervention studies</b>						
Naeher et al. (2000a)	Mean 22 h in-kitchen PM <sub>2.5</sub>	Guatemala, before and after Plancha intervention, 3 tests each in each of 3 houses with different layouts	528	97	56	<b>0.09</b>
Naeher et al. (2000a)	Mean 22 h in-kitchen PM <sub>2.5</sub>	Guatemala, before and after Plancha intervention, 3 tests each in 1 house with combined kitchen/bedroom	636	174	78	<b>0.17</b>
Zuk et al. (2006)	Mean 48 h in-kitchen PM <sub>2.5</sub>	Mexico, before and after Patsari stove intervention in 37 homes	658	255	60	<b>0.33</b>
<b>Cross-sectional studies</b>						
Brauer et al. (1996)	Mean 9 h in-kitchen PM <sub>2.5</sub>	Mexico, 7 homes with open biomass stoves and 3 with vented biomass stoves	555	132	37	<b>0.18</b>
Naeher et al. (2000b)	Mean of 2 min PM <sub>2.5</sub> samples; breakfast preparation	Guatemala, 46 homes with open fires and 27 homes with Plancha stoves	5040	850	290	<b>0.12</b>
Naeher et al. (2000b)	Mean of 2 min PM <sub>2.5</sub> samples; lunch preparation	Guatemala, 42 homes with open fires and 26 homes with Plancha stoves	6560	1330	130	<b>0.19</b>
Naeher et al. (2000b)	Mean of 2 min PM <sub>2.5</sub> samples; dinner preparation	Guatemala, 58 homes with open fires and 28 homes with Plancha stoves	4360	4580	240	<b>1.05</b>
<b>Median</b>						<b>0.18</b>
<b>Median (intervention studies weighted 2 × )</b>						<b>0.18</b>
<b>Mean</b>						<b>0.30</b>
<b>Mean (excluding the extreme largest and smallest value)</b>						<b>0.20</b>

studies are weighted 2 × the ‘cross-sectional’ studies. Our analysis uses  $f_{\text{unv}}=0.18$  as a representative value, implying that intake values are  $\sim 6 \times$  lower in houses with a chimney than without one. Measurements of women’s and children’s integrated exposures during stove intervention studies have found exposure reductions of approximately 20% to 50% when chimneys are used (Naeher et al., 2000a; Northcross et al., 2010). Direct comparison to our value for  $f_{\text{unv}}$  is not possible because integrated exposure measurements combine stove plus non-stove exposures. (If the  $\sim 20\%$  to 50% reduction were to apply specifically to stoves, then the value for  $f_{\text{unv}}$  used here would be low, i.e., we would have overestimated the effectiveness of chimneys.)

Combining  $iF_i$  for unvented stoves (1300 ppm) and  $f_{\text{unv}}$  (0.18) yields an estimated  $iF_i$  for vented stoves of 240 ppm. In Section 4, we explore the influence of uncertainty in  $f_{\text{unv}}$  on the relative ranking of stove–fuel combinations.

### 3.4. Climate emissions

The annual climate impact of a stove–fuel combination was estimated using global warming commitment (GWC) (Smith et al., 2000; Bond et al., 2004b). GWC approximates the CO<sub>2</sub>-equivalent climate impact (positive or negative radiative forcing over a given time horizon,  $t$ ) of a set of emitted substances by weighting emissions of each substance  $i$  with a global warming potential ( $\text{GWP}_{t,i}$ ). GWP is the temporally integrated radiative forcing from a unit pulse emission of a substance, relative to that from a unit pulse of CO<sub>2</sub>, over a specific time horizon. While GWPs are used to account for emissions of greenhouse gases included in the Kyoto protocol (including CH<sub>4</sub>, N<sub>2</sub>O and other long-lived gases), GWPs are not ideal for representing the warming impacts of aerosols and other short-lived climate forcing agents (Bond, 2007). In particular, aerosols have short atmospheric lifetimes (on the order of days or weeks; much shorter than GWP time horizons (decades or centuries)), are not well-mixed globally, and have poorly understood climatic impacts, making comparisons with long-lived gases difficult. However, GWP are commonly in use, conceptually simple, and institutionally entrenched (Fuglestedt et al., 2003) and thus will likely be used to assess

**Table 4**100-Year global warming potential ( $\text{GWP}_{100}$ ) values used in analysis.

Emission	$\text{GWP}_{100}$
<i>‘Kyoto’ emissions</i>	
CO <sub>2</sub>	1 <sup>a</sup>
CH <sub>4</sub>	25 <sup>a</sup>
<i>‘Non-Kyoto’ emissions</i>	
CO	1.9 <sup>a</sup>
NMHC	3.4 <sup>a</sup>
EC	455 <sup>b</sup>
OC	–35 <sup>b</sup>
SO <sub>2</sub>	–76 <sup>c</sup>

Sources: <sup>a</sup> IPCC, 2007; <sup>b</sup> Reynolds and Kandlikar, 2008; <sup>c</sup> Shindell et al., 2009.

tradeoffs in emissions reductions for some time. GWP time horizons of 100 and 20 years are generally considered; the former (100 years) is prescribed by the Kyoto Protocol for evaluating the impact of GHG species.  $\text{GWP}_{100}$  values used in this analysis are shown in Table 4. Values for other time horizons can be found elsewhere (Bond et al., 2004b). Here we calculate annual, per-stove GWC (units of tCO<sub>2</sub>e year<sup>–1</sup> stove<sup>–1</sup>) as

$$\text{GWC}_{100} = \text{EF}_{\text{CO}_2} \cdot f_{\text{NR}} \cdot \text{AFU} + \sum_{i=1}^n \text{EF}_i \cdot \text{AFU} \cdot \text{GWP}_{100,i} \quad (4)$$

where  $f_{\text{NR}}$  is the fraction of fuel that is from non-renewable sources ( $f_{\text{NR}}=1$  for fossil fuels). In biomass applications,  $f_{\text{NR}}$  depends on the source of biomass, and can vary from 0 for renewable biomass sources (e.g., sustainably harvested wood or agricultural residues) to 1 for wood from deforestation. Here, we calculate GWC for  $f_{\text{NR}}$  values of 0.5 (base case) and 1 (renewable case), to explore the sensitivity of the results to this parameter. Actual values of  $f_{\text{NR}}$  have been shown to vary widely over small spatial scales (Johnson et al., 2009).

GWPs were calculated for two sets of emissions (see also Table 4). The first set, called *GWC-Kyoto*, includes the major cookstove emissions recognized under the Kyoto Protocol: CO<sub>2</sub> and CH<sub>4</sub> (N<sub>2</sub>O is

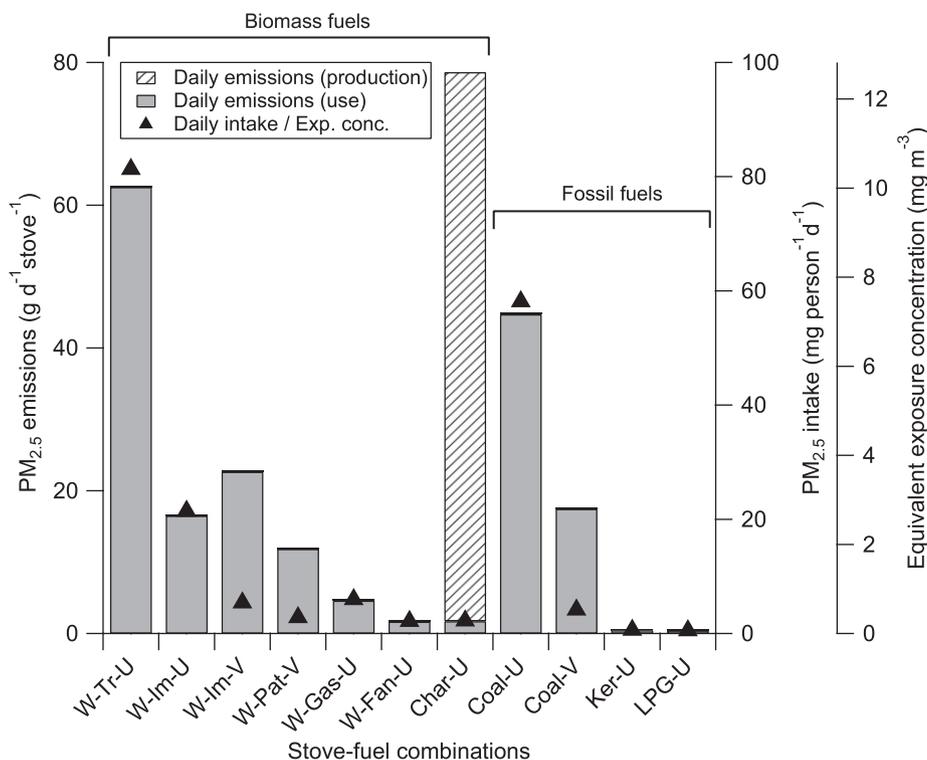


Fig. 1. Central estimates of PM emissions, individual intake and equivalent exposure concentration for analyzed stove options.

included in the Kyoto Protocol and is emitted during combustion but contributes little [e.g., < 0.3% in India (Smith et al., 2000)] to the GWC from cookstove operation.) The second set, *GWC-all*, includes a wider range of species with GWPs that are less well characterized and are not included in the Kyoto Protocol. The emissions included in this set are gaseous ( $\text{CO}_2$ ,  $\text{CH}_4$ , carbon monoxide [CO], total non-methane hydrocarbons [TNMHC],  $\text{SO}_2$  [a precursor for sulfate aerosols]) and particle-phase (black carbon [BC], organic carbon [OC]). The GWP for  $\text{SO}_2$  and OC are negative, reflecting their net cooling effect on the climate. *GWC-all* provides a best estimate of total climate impacts; *GWC-Kyoto* quantifies potentially 'tradable' emissions under current treaties and carbon markets.

## 4. Results

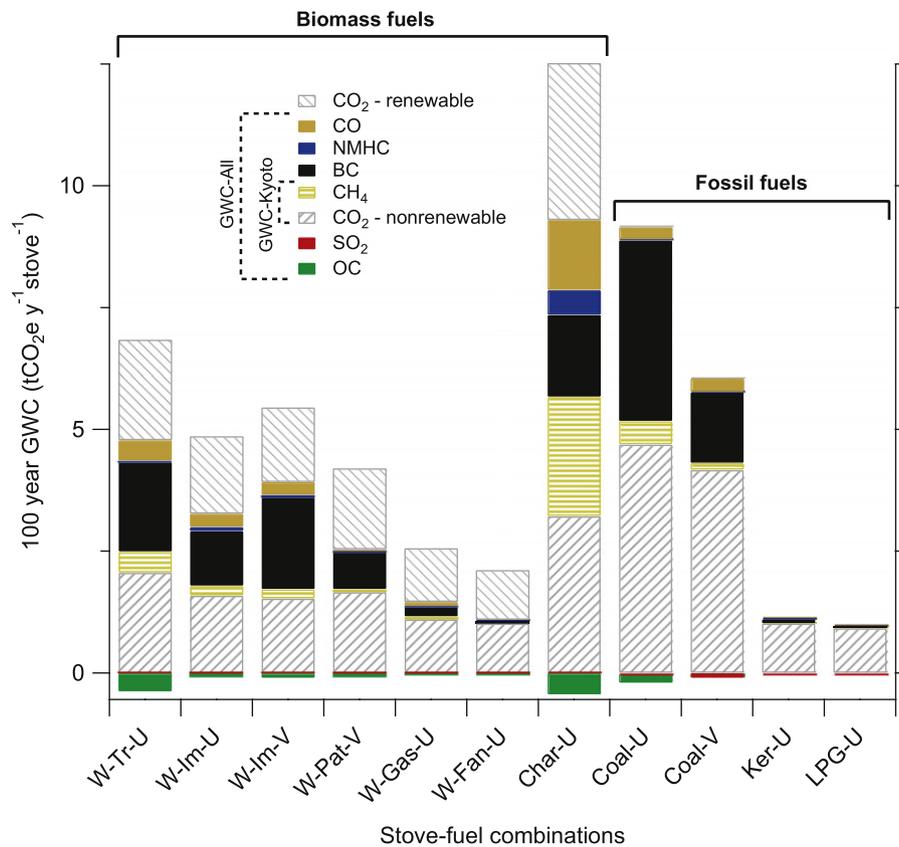
### 4.1. Health and climate impacts

Fig. 1 shows daily PM emission rate and the estimated PM intake for an individual in the same room as the stove. Daily emission rates capture the impact of stoves on urban and regional air pollution, which is an important impact from solid fuel use (Bond et al., 2004a), whereas daily PM intake relates more directly to health risk to the person cooking. In-use emissions and intake levels are notably greater for unimproved wood and coal stoves than for any of the improved options. Considering PM emissions from use and production of charcoal, Char-U yields the highest overall emissions. (Charcoal production emissions are assumed here not to contribute to indoor exposures.) For comparison, the PM emissions from the daily use of a single traditional wood stove are similar to that from a dirty, 'super-emitting' heavy-duty diesel truck in Asia driving 20 km (assuming a truck PM EF of  $\sim 3 \text{ g km}^{-1}$  (Subramanian et al., 2009)) or a clean diesel particulate filter-equipped heavy-duty diesel truck in North America driving 15,000 km (assumption:  $\sim 4 \text{ mg km}^{-1}$  (Biswas et al.,

2009)). While emissions from 'modern' fuels (kerosene and LPG) are comparatively negligible relative to those from unimproved stoves,<sup>3</sup> there is wide variation in the range of emission and intake among other 'improved' options. The basic improved wood burning stoves (W-Im-U, W-Im-V and W-Pat-V) emit a moderate amount of PM, though ventilation can yield large ( $\sim 6 \times$ ) reductions in users' intake.

Also shown in Fig. 1 (secondary right-axis) are equivalent exposure concentrations ( $\bar{C}_{\text{personal}}$ ; concentrations to which a person inhaling  $7.8 \text{ m}^3 \text{ d}^{-1}$  (Klepeis and Nazaroff, 2006) would need to be exposed for 24 h to result in the modeled intake; see Eq. (2)) for each stove option. Considering exposure concentrations enables some comparison of our modeled exposures with field study measurements. Despite the simplifying assumptions of our approach, our estimates are in reasonable agreement with some available data. For example, our estimated  $\text{PM}_{2.5} \bar{C}_{\text{personal}}$  for the W-Tr-U of  $10.4 \text{ mg m}^{-3}$  (range  $7.4\text{--}12.6 \text{ mg m}^{-3}$  due to EF variability, other parameters held constant, see Fig. 3) overlaps with the range of 24 h average  $\text{PM}_{10}$  exposure estimates for Kenyan adult women using various stove types,  $4.8 \pm 3.6 \text{ mg m}^{-3}$  (Ezzati et al., 2000). Ezzati et al. show that single point concentration measurements under-estimated exposure by over 70%. However, other studies have found that integrated kitchen concentration measurements can overestimate personal exposure (Clark et al., 2010; Zuk et al., 2006). For example, a study in Mexico found 48 h kitchen concentrations of  $0.7 \pm 0.3 \text{ mg m}^{-3}$  for houses using an open fire (Zuk et al., 2006), and estimated personal exposures between 0.1 and  $0.3 \text{ mg m}^{-3}$ . This same study found average concentrations of  $0.3 \pm 0.3 \text{ mg m}^{-3}$  for households

<sup>3</sup> While this observation appears to be consistently true for LPG use, kerosene can result in much higher emissions if burned in primitive lamps (Fan and Zhang, 2001) and presumably also in poorly performing stoves. Further, studies have observed associations between kerosene use and significantly higher indoor PM concentrations (Andresen et al., 2005) and tuberculosis risk (Pokhrel et al., 2009).



**Fig. 2.** Central estimates of annual  $GWC_{100}$  from 'Kyoto' (striped) and 'non-Kyoto' (solid) species emitted by stove–fuel combinations calculated with Eq. (4). Brackets in legend indicate the species used to calculate GWC-Kyoto (inner brackets) and GWC-All (outer brackets).  $CO_2$  would have no net contribution to GWC for renewable biomass ( $f_{NR}=0$ ); here we show  $f_{NR}=0.5$  for biomass, implying that for biomass half of the emitted  $CO_2$  contributes to climate change (lower of the two  $CO_2$  bars) while the other half does not (upper  $CO_2$  bar). For fossil fuels, all of the  $CO_2$  contributes to climate change. OC and  $SO_2$  GWC values are negative and thus would be subtracted from overall bar height to calculate GWC-all; summed values are shown in Fig. 3.  $N_2O$  is included in the Kyoto Protocol and is emitted during combustion but contributes negligibly to the GWC from cookstove operation, and so is not included here.

with a Patsari stove (W-Pat-V), which is in line with our estimated  $\bar{C}_{personal}$  of 0.4 (range 0.3–0.9  $mg\ m^{-3}$ ). Our method thus appears to yield exposure estimates that are reasonably consistent with those measured in the field, though may be biased high due to simplifying assumptions. For example, we assume that the kitchen is inhabited by the stove user during all fuel-use, which will tend to overestimate exposure.

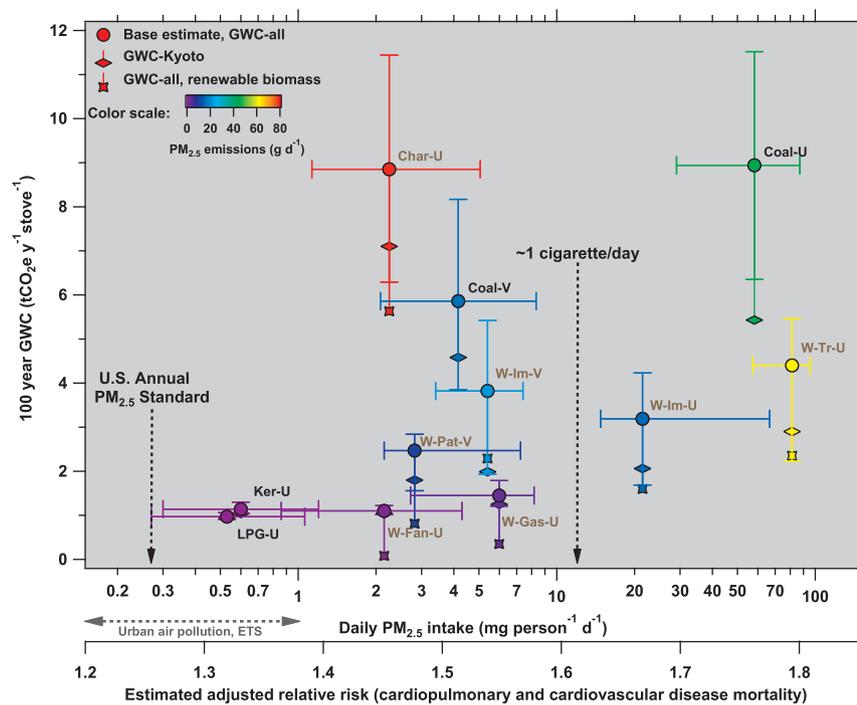
Fig. 2 shows stoves' annual GWCs, indicating the following:

- The GWC-all impact of the traditional and basic 'improved' stoves (and the production and use of charcoal shown by Char-U) shows a large contribution from PIC emissions *not* included in GWC-Kyoto. Much of the climate benefit from replacing these stoves is not included in GWC-Kyoto accounting. At the same time, many traditional stoves (W-Tr-U, Coal-U and Coal-V) have large emissions of Kyoto-included gases, so switching to other options in Fig. 2 would provide GWC-Kyoto benefits.
- GWC-all values vary by  $10\times$  among stove–fuel combinations in Fig. 2. The cleanest-burning biomass stoves (W-Gas-U and W-Fan-U) have  $\sim 4\times$  lower GWC than traditional wood burning. For renewable biomass, these differences grow to  $30\times$  and  $7\times$ , respectively; for non-renewable biomass, these differences shrink to approximately  $3\times$ .
- Fig. 2 reveals GWC differences between improved stoves that rely on chimneys to reduce exposure (W-Im-V) and those that have superior combustion performance (the fan and gasifier stoves). The Patsari vented stove, developed and disseminated in Mexico (Johnson et al., 2009), is a good example of

effectively bridging the exposure and climate goals because it is comparatively more efficient, i.e., has lower emissions and fuel use than other vented stoves (PICs make a small contribution to Patsari GWC-all), while employing a chimney to reduce exposures.

- While charcoal stove *use* releases only small amounts of PM (thus resulting in low intake levels), charcoal *production* emits a large amount of PM (which are purposefully excluded from intake estimates here). Charcoal stoves have a GWC on par with unimproved coal use owing to substantial emissions during use (mostly CO and  $CH_4$ ) and especially during charcoal production (PIC emissions). Charcoal is comparable to unimproved coal combustion from a GWC standpoint and even worse from the standpoint of total PM emissions.<sup>4</sup>
- LPG and kerosene stoves have the lowest GWC other than the cleanest biomass-burning stoves (the 'Fan' and 'Gasifier' stoves) operating on renewable biomass. These stoves (LPG, kerosene) also yield the lowest exposures. Perhaps counter-intuitively, fossil fuel options may be the cleanest options for health and for climate change (Smith, 2002).
- The 'Fan' stove has the best climate performance of the options considered if renewable fuels are used and thus shows the

<sup>4</sup> Though lifecycle emissions from other fuels have not been included, charcoal production has been shown to be especially inefficient and polluting (Pennise et al., 2001) and to play a large role in this fuel's climate impacts (Bailis et al., 2003). Charcoal is an inefficient use of fuel wood, because much of the energy content of the wood is lost during production.



**Fig. 3.** Climate and health impacts of stove–fuel combinations shown in Table 1. Shown are global warming commitment (GWC) values vs.  $PM_{2.5}$  intake for a person using the stove indoors. Biomass fuels are labeled in brown, fossil fuels are labeled in black. Icon colors indicate  $PM_{2.5}$  emissions (bar height in Fig. 1). Error bars represent the range of estimates attributable to variability in stove emission factor. Dashed vertical arrows indicate estimates for PM intake from smoking 1 cigarette  $d^{-1}$  and from breathing air at the US annual  $PM_{2.5}$  standard ( $15 \mu g m^{-3}$ ) for a day (Pope et al., 2009). The secondary x-axis provides estimated adjusted relative risk for cardiopulmonary and cardiovascular disease mortality based on the log-linear dose–response relation suggested by Pope et al. (2009). Note that intake values calculated by Pope et al. (2009) assume a breathing rate of  $18 m^3 d^{-1}$ .

potential of biomass-fueled stoves for climate mitigation. However, its complexity leads to a higher cost and the potential for failure or misuse.

BC emissions make an important contribution to the GWC-all from the stove options discussed here: an average 29% of the net GWC-all of the 12 options comes from BC, increasing to 41% if the ‘cleanest’ options are excluded (LPG-U, Ker-U and W-Fan-U) and to 70% if we include all biomass stoves but assume 100% renewable biomass. Near-term climate impacts of the higher-emitting options are substantially under-stated in our analysis because most climate impacts from BC occur in the first weeks of the 100 year GWP time horizon (Bond, 2007) and important impacts may happen on the regional scale (Ramanathan and Carmichael, 2008). Kyoto-type accounting of emission does not include climate impacts of BC, but climate and health benefits from reductions in solid fuel use provides a strong motivation for doing so (Grieshop et al., 2009). Applying a shorter time horizon for climate calculations (e.g., using  $GWP_{20}$  rather than  $GWP_{100}$ ) would further increase the estimated importance of BC to overall climate impacts.

GWC values calculated here are generally consistent with those calculated in other works. For example, the per-GJ-delivered  $GWC_{100}$  values calculated by Bond et al. (2004b) for a variety of fuel–stove combinations are within 40% of those calculated here for ‘equivalent’ stoves and usage. Bond et al.’s values are generally within our uncertainty bounds (see Fig. 3) but high relative to ours, mainly owing to the lower BC and CO GWP values used here (Table 4) as suggested by recent research (IPCC, 2007; Reynolds and Kandlikar, 2008). Our GWC value for charcoal stoves is over 20% higher than those by Bond et al. (2004b), potentially because of different assumptions about emissions during production or usage of charcoal. GWC due to unimproved coal

combustion estimated by Bond et al. (2004b) is 40% higher than ours, again likely due to GWP values and assumptions about combustion conditions. GWC values here generally agree with those by Smith et al. (2000) because many of our EF and  $\eta_{th}$  values come from that study.

Fig. 3 shows climate (GWC) and health (daily intake) impacts for a range of stove–fuel options. Estimated intake, displayed on a log-axis, varies by more than two orders of magnitude, highlighting the large influence of fuels, stoves and venting.  $PM_{2.5}$  intakes are plotted on a log scale in keeping with the hypothesis of a log-linear dose response relationship (Ezzati and Kammen, 2001; Pope et al., 2009), e.g., that a reduction of intake from 100 to  $10 mg d^{-1}$  would represent the same change in relative risk as a reduction from 10 to  $1 mg d^{-1}$ . The dashed horizontal line below the x-axis indicates the typical intake estimated by Pope et al. (2009) for US outdoor urban air pollution ( $\sim 0.2$ – $0.4 mg person^{-1} d^{-1}$ ) and environmental tobacco smoke (ETS;  $\sim 0.4$ – $0.9 mg person^{-1} d^{-1}$ ); for this range, the dose–response relationship may be linear (Pope et al., 2009). The secondary x-axis in Fig. 3 provides estimated mortality relative risk associated with the  $PM_{2.5}$  intake values.<sup>5</sup> Three ‘strata’ of stove–fuel combinations emerge, vis-à-vis estimated PM intake: (1) unvented traditional (coal and wood) stoves and unvented ‘improved’ stove yielding intake greater than  $10 mg d^{-1}$  (equivalent to smoking at least one cigarette per day); (2) vented coal and biomass stoves and cleaner-burning unvented biomass and charcoal stoves that result in intakes of approximately an order of magnitude less

<sup>5</sup> This axis was generated using the fit for cardiovascular and cardiopulmonary disease mortality risk from Pope et al. (2010) and may not be indicative of other health endpoints, such as ALRI in children. However, ALRI risk for children has also been observed to have a logarithmic dependence on PM exposure (Ezzati and Kammen, 2001).

**Table 5**  
Ranking of stove technologies for daily PM<sub>2.5</sub> intake<sup>a</sup> with different  $f_{\text{unv}}$  values.

Intake rank	Base case $f_{\text{unv}}$	Low $f_{\text{unv}}$	High $f_{\text{unv}}$	
11 (highest intake)	W-Tr-U (81.3)	W-Tr-U (81.3)	W-Tr-U (81.3)	<b>10–100 mg d<sup>-1</sup></b>
10	Coal-U (58.1)	Coal-U (58.1)	Coal-U (58.1)	
9	W-Im-U (21.5)	W-Im-U (21.5)	W-Im-U (21.5)	
8	W-Gas-U (6)	W-Gas-U (6)	<b>W-Im-V (9.7)</b>	<b>1–10 mg d<sup>-1</sup></b>
7	<b>W-Im-V (5.5)</b>	<b>W-Im-V (2.7)</b>	<b>Coal-V (7.3)</b>	
6	<b>Coal-V (4.1)</b>	Char-U (2.2)	W-Gas-U (6)	
5	<b>W-Pat-V (2.8)</b>	W-Fan-U (2.2)	<b>W-Pat-V (5.1)</b>	
4	Char-U (2.2)	<b>Coal-V (2)</b>	Char-U (2.2)	
3	W-Fan-U (2.2)	<b>W-Pat-V (1.4)</b>	W-Fan-U (2.2)	
2	Ker-U (0.6)	Ker-U (0.6)	Ker-U (0.6)	<b>&lt; 1 mg d<sup>-1</sup></b>
1 (lowest intake)	LPG-U (0.5)	LPG-U (0.5)	LPG-U (0.5)	

<sup>a</sup> Central estimates of daily PM<sub>2.5</sub> intake are shown in parentheses. Ventilated stoves are shown in bold text.

(1–10 mg d<sup>-1</sup>, equivalent to less than one cigarette per day but more than the typical intake for exposure to environmental tobacco smoke (Pope et al., 2009)) and (3) stoves using liquid and gaseous fuels that further reduce intakes by another  $5 \times 10 \times (< 1 \text{ mg d}^{-1})$ , or approaching urban pollution levels).

Variation within each of these three bands reveals important patterns:

- The basic unvented ‘improved’ stove (W-Im-U) provides only moderate efficiency and emission improvements; leaving the ‘high exposure’ (intake: greater than 10 mg person<sup>-1</sup> d<sup>-1</sup>) band in Fig. 3 requires large improvements in combustion performance (lower emissions, e.g., W-Fan-U), venting (e.g., W-Im-V), or both (e.g., W-Pat-V).
- The intermediate exposure band (intake: 1–10 mg d<sup>-1</sup>) includes comparatively lower-emitting unvented stoves (W-Fan-U, W-Gas-U) and higher-emitting vented stoves (W-Im-V, W-Pat-V). Since neighborhood-level exposures, especially in dense urban areas, are also important, it is preferable to avoid such emissions rather than just move them to outside the house.
- None of the solid fuel stoves, even the ‘improved’ options are in the low exposure regime (kerosene or LPG stoves). (Biogas stoves, not discussed here, employ gaseous combustion and often are similarly clean as LPG or kerosene (Smith et al., 2000)). Charcoal and fan stoves emit  $4 \times 5 \times$  more PM<sub>2.5</sub> annually than the liquid-/gaseous-fuel stoves. Biomass stoves need substantial further development, or the integration of effective venting in the case of the advanced unvented stoves (e.g., integrating the performance of the ‘Fan’ stove with venting), to reach ‘LPG-like’ (Venkataraman et al., 2010) levels of cleanliness.

The observations above are generally robust to uncertainty in  $iF_i$  values used here. Straightforward variation in  $iF_i$  would alter intake estimates but not the ranking of the stove options (Table 5). Variations in  $f_{\text{unv}}$ , however, affect  $iF_i$  and intake of vented stove emissions and thus the stoves’ relative ranking. Table 5 shows stove ranks using base, low and high  $f_{\text{unv}}$  values (0.18, 0.09 and 0.33, respectively; these values are from the range of realistic values in Table 3). A well-functioning chimney (low  $f_{\text{unv}}$ ) improves the rank of some ventilated stoves (the Patsari; ventilated coal stoves) but does not boost them into the lowest-intake regime, implying that current stove/chimney technologies may not provide truly ‘clean’ (LPG-like) cooking services. Poorly functioning chimneys (high  $f_{\text{unv}}$ ) make ventilated stoves less

appealing and nearly push low-performing chimney stoves (W-Im-V) into the ‘high-intake’ strata, reiterating that interventions should minimize emissions rather than only relying on a chimney. We hypothesize that  $f_{\text{unv}}$  and emission factors may be correlated, because chimney design or condition may be associated with combustion efficiency changes; this issue is not addressed here owing to lack of evidence. This topic may be a useful area for future research.

#### 4.2. Tradeoffs between exposure and climate

Fig. 3 shows that there is not a one-to-one correspondence between the climate and IAP benefits from improved stoves, emphasizing that both dimensions must be considered in technological decisions. Values of GWC-Kyoto can be used to assess the potential for carbon-financing from different stove mitigation options under current frameworks (CDM credits or voluntary offsets). Nearly half of the GWC-all from traditional unvented woodstoves comes from non-Kyoto emissions; GWC-Kyoto is approximately 3 tCO<sub>2</sub>e year<sup>-1</sup> stove<sup>-1</sup>. This value depends on  $f_{\text{NR}}$ , so the relative ‘baseline’ GWC will vary with local fuel supplies. Under base-case assumptions, switching from the traditional wood (W-Tr-U) and coal (Coal-U) stoves to other options offer potential GWC-Kyoto reductions of 1–2 and 1–5 tCO<sub>2</sub>e year<sup>-1</sup> stove<sup>-1</sup>, respectively, with reductions of up to 6 tCO<sub>2</sub>e year<sup>-1</sup> stove<sup>-1</sup> possible (e.g., switching from Coal-U to an efficient biomass stove using renewable biomass). Assuming an offset price of \$15 per tCO<sub>2</sub>e<sup>6</sup> and considering only Kyoto species, these options thus provide potential revenue streams of \$15–\$90 per stove per year. Annual cooking system costs (in year-2000 international dollars, including additional expenses for fuel) have been estimated (depending on region, and with high associated uncertainty) at \$20–\$100 for LPG, \$10–\$20 for kerosene, and \$3–\$24 for improved stoves (Mehta and Shahpar, 2004). There is thus large potential for carbon finance to fund transitions from high-emitting traditional cooking methods to cleaner ones. This case would be even stronger were the non-Kyoto substances and their large short-term impacts considered in this comparison. For example, inclusion of non-Kyoto climate forcers increases by 50–60% the GWC<sub>100</sub> (and thus, potentially, funding) available for switching away from traditional to lower-emitting stoves.

<sup>6</sup> Certified Emission Reductions (CER) are trading for prices in the range of €10–12 per tonne CO<sub>2</sub> (approximately \$14–\$17 per tonne assuming an exchange rate of €1.4=\$1) in March 2011 (Point Carbon, 2011).

Many 'improved' stove options analyzed in this paper yield roughly equivalent GWC-Kyoto mitigation but very different IAP exposure benefits. Holding  $f_{NR}$  constant, moving from traditional stoves (W-Tr-U or Coal-U) to mildly improved biomass stoves (W-Im-U, W-Im-V, W-Pat-V) will result in roughly equivalent GWC-Kyoto reductions ( $\sim 1 \text{ tCO}_2\text{e year}^{-1} \text{ stove}^{-1}$ ); a move to the next category of exposure improvements (W-Gas-U, W-Fan-U, Ker-U or LPG-U) yields another  $\sim 1 \text{ tCO}_2\text{e year}^{-1} \text{ stove}^{-1}$  reduction in GWC-Kyoto. However, within each of these sets (i.e., from W-Im-U to W-Pat-V and W-Gas-U to LPG-U) there is a  $\sim 10 \times$  variation in intake reduction relative to the traditional stoves. For the highest total health benefits, a stove intervention aiming to reduce GHG emissions should select the option with the largest intake reductions. For example, the Patsari offers the potential for a greater PM reduction and health benefit than an unvented improved stove, even though they offer similar GWC-Kyoto reductions, relative to an unimproved stove. Decisions among options must account for regional stove and program costs, fuel availability, extent to which wood being burned is renewable vs. nonrenewable, and the scale of implementation. In a worse-case scenario, climate interventions might reduce GWC while increasing intake, e.g., switching from a ventilated coal stove (Coal-V) to an unvented wood-burning stove (W-Im-U or W-Gas-U).

Similarly, shifting from a traditional wood-burning stove to either a charcoal-burning or fan stove leads to a  $40 \times$  reduction in exposure; the annual climate impact (GWC-Kyoto) per stove would be a  $\sim 2 \text{ tCO}_2\text{e year}^{-1} \text{ stove}^{-1}$  reduction in the case of a fan stove, vs. a  $4 \text{ tCO}_2\text{e year}^{-1} \text{ stove}^{-1}$  increase for a charcoal stove (the latter is all due to emissions from charcoal production).<sup>7</sup> This comparison, and the charcoal case in general, shows that the climate and health-relevant properties of stoves do not always scale together. Among the choices considered here, the only option to reduce the climate impacts of charcoal without causing increased PM exposure is a move 'up the energy ladder' to LPG or kerosene.

Fossil fuels such as kerosene and LPG have the potential to reduce the climate impacts of cooking activities as much as most other options and offer unmatched  $\text{PM}_{2.5}$  exposure reductions. The case to 'leapfrog' improved biomass use and move towards direct use of petroleum fuels has been made on both the basis of health and climate benefits (McDade, 2004; Sagar, 2005; Smith, 2002). Based on the stove cost and carbon offset funding values presented above, an offset program should be able to fully fund the use of kerosene as a solid-fuel replacement and provide substantial support for LPG implementation (e.g., fund initial stove/cylinder purchase and fuel subsidy). However, fuel subsidies have generally been found to be inefficient in getting fuels into the hands of the poor (Bruce et al., 2006); price volatility and supply limits also pose significant challenges for these fuels. Further, kerosene use may increase risks for child poisoning (Bruce et al., 2006) and house fires (McDade, 2004) and its use in primitive or poorly functioning devices may result in higher emissions (see footnote 1). Widespread promotion of LPG may face implementation hurdles such as increased costs and complexities in fuel distribution. Subsidies must be applied carefully in order to promote long-term uptake of technologies (Barnes, 1994) while also encouraging further innovation, but a program using carbon financing could provide an effective framework within which to do so. Opportunities to substitute modern fuels for charcoal and biomass use in urban areas seem an especially appealing option because fuel supply infrastructure limitations

can be more easily overcome in higher-density areas. An additional benefit of modern fuels is that their combustion in stoves is for the most part clean by nature and therefore stove emission performance is robust over long lifetimes, unlike improved biomass stoves whose performance may degrade over time (Clark et al., 2010; Roden et al., 2009).

Our analysis provides a methodology for a simple, technology-oriented comparison of stoves' impacts that required various simplifying assumptions, only some of which have been mentioned. Other important limitations, which can be considered in further analyses, include the following:

- Stove users are assumed to remain indoors with the stove during all fuel use. Cooking behaviors may depend on stove type and can impact exposure.
- Households rarely depend on only one cooking device or fuel for all needs (Venkataraman et al., 2010), as has been assumed here. Factors such as fuel availability, cooking task and work schedules will impact device choice and thus exposure and climate-impacting emissions.
- As noted above, stove performance may vary greatly between laboratory and field testing and among field testing types. Emission factors and efficiencies from the literature (typically, based on water boiling tests) have been used here; data collected under realistic in-use conditions would be preferable when it is available.
- We assumed constant energy demand across technologies, but energy demand may depend on technology (e.g., impacted by attributes such as thermal mass, ease of re-lighting and varying heating levels) and use (changes in cooking practices and diet).

The basic framework presented by this paper could be expanded in many ways:

- Estimating the impact of chimneys on household 'self-pollution' and neighborhood air pollution requires better estimates of individual and population intake fractions in urban and rural LDC settings. Improved individual and population intake fraction estimates would help in characterizing the relative impacts of vented vs. unvented options.
- Understanding the long-term performance of in-use cleaner-burning stoves is a major gap that needs to be filled. Biomass stoves frequently perform worse under field conditions than in controlled lab settings, and performance often degrades over time.
- Incorporating the impact of actual stove-use behaviors (e.g., fast cooking vs. slow simmering; placement of a stove outside during certain times) on fuel use, emissions and exposure would enable the estimation of technologies' impacts in different settings. For example, interactions between stove and user can be a large determinant of overall exposures (Ezzati et al., 2000); intake fraction can capture these dynamics, recognizing that user behaviors may be particular to stove/user/cooking-style combinations.
- Integrating better information about the renewability of biomass fuel sources must be done on a local or regional basis. Applying this information in the framework developed here will allow a full assessment of the climate impact of different options.
- Accounting for uncertainty and variability in the full range of parameters determining exposure and climate impacts would allow their more robust determination; sensitivity analyses over input parameters can be used to prioritize research needs.

<sup>7</sup> A highly efficient charcoal stove operating on charcoal produced in a very clean process would have lower GWC than shown in Char-U, but charcoal in LDCs is typically produced locally in primitive earthen or brick kilns with poor emission performance.

## 5. Conclusions

Health benefits involve reducing emissions *and* exposure. Stoves with chimneys and improved but unvented stoves can provide roughly an order of magnitude reduction in exposure relative to traditional options; these exposures are in turn an order of magnitude higher than for modern fuels (LPG, kerosene). None of the solid-fuel stoves investigated here exhibit emission performance on par with modern fuels (LPG, kerosene). While not investigated here, cookstove change-out programs will need to incorporate local factors (markets, subsidies, cultural issues) to be long-term sustainable; however, our analyses suggest that, if done well, such programs could yield strong climate and health benefits. If performed poorly, change-out programs risk inadvertently worsening one problem (climate or health) while trying to improve the other.

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