Greenhouse Gas Implications of Household Energy Technology in Kenya

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Linkages between household energy technology, indoor air pollution, and greenhouse gas (GHG) emissions have become increasingly important in understanding the local and global environmental and health effects of domestic energy use. We report on GHG emissions from common Kenyan wood and charcoal cookstoves. Data are from 29 d of measurements under the conditions of actual use in 19 rural Kenyan households. Carbon monoxide (CO), particulate matter (PM_{10}), combustion phase, and fuel mass were measured continuously or in short intervals in day-long monitoring sessions. Emissions of pollutants other than CO and PM_{10} were estimated using emissions ratios from published literature. We found that the daily carbon emissions from charcoal stoves (5202 ± 2257 g of C: mean ± SD) were lower than both traditional open fire (5990 ± 1843 g of C) and improved ceramic woodstoves (5905 ± 1553 g of C), but the differences were not statistically significant. However, when each pollutant was weighted using a 20-yr global warming potential, charcoal stoves emitted larger amounts of GHGs than either type of woodstove (9850 ± 4600 g of C for charcoal as compared to 8310 ± 2400 and 9649 ± 2207 for open fire and ceramic woodstoves, respectively; differences not statistically significant). Non-CO_2 emissions from charcoal stoves were 5549 ± 2700 g of C in 20-yr CO_2 equivalent units, while emissions were 2860 ± 680 and 4711 ± 919 for three-stone fires and improved ceramic stoves, respectively, with statistically significant results between charcoal and wood stoves. Therefore in a sustainable fuel-cycle (i.e., excluding CO_2), charcoal stoves have larger emissions than woodstoves. When the emissions from charcoal production, measured in a previous study, were included in the assessment, the disparity between the GHG emissions from charcoal and firewood increased significantly, with non-CO_2 GHG emissions factors (g of C/kg of fuel burned) for charcoal production and consumption 6−13 times higher than emissions from woodstoves. Policy implications and options for environment and public health are discussed.

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Introduction

Between one-third and one-half of the world's population rely on solid biofuels—wood, crop residues, charcoal, and dung—for the majority of their energy needs. Solid-fuel users rely on simple technologies such as open "three-stone" fires and mud, clay, or metal stoves that result in incomplete and inefficient combustion (*1*, *2*), leading to the emission of hundreds of potentially harmful compounds (*3*). Some of these compounds also contribute to global climate change. The health effects of indoor air pollution from biomass fuels in developing countries have been examined in a number of research projects (*4*−*7*). Recent work has shown that greenhouse gas (GHG) emissions from biomass burning may rival or exceed fossil fuel-based GHG emissions in many less-developed countries. For example, the United Nations' Food and Agriculture Organization (FAO) estimates that CO_2 emissions from the production and use of fuelwood and charcoal in Kenya exceeded 30 million ton in 1996, while non-CO_2 GHG emissions exceeded 23 million ton (in CO_2 equivalent units weighted by 20-yr global warming potential (GWP) ton in the same year). In contrast, the World Resources Institute (WRI) report that Kenya's CO_2 emissions from the consumption of fossil fuels and cement production in 1996 was roughly 6.8 million ton (*8*, *9*). They do not report emissions of other GHGs; however, see ref *9* for an overview of Kenya's energy consumption patterns and see refs *10*−*15* for a description of biofuel-based GHG emissions in other contexts.

Under optimal conditions, biomass combustion results almost entirely in the emission of water vapor and carbon dioxide (CO_2). Water vapor, the most prevalent GHG in the atmosphere, is quickly incorporated in the hydrologic cycle with no measurable warming effect, and CO_2, the most common anthropogenic GHG, can be absorbed by new plant growth through photosynthesis. Therefore, if biomass is harvested in a sustainable way so that its long-term stocks are not depleted and burned under ideal combustion conditions, it is effectively GHG neutral.

The issue of sustainable biomass harvesting is important from the perspective of the carbon cycle as well as from the perspective of household welfare in developing countries and has been discussed elsewhere (*16*, *17*). In this paper, we present an empirical analysis of GHG emissions from biomass combustion. We study domestic biomass-burning cookstoves used by an agropastoral community in central Kenya under conditions of actual use, which is characterized by low combustion efficiency. Under these conditions, hundreds of gaseous and aerosolized compounds are emitted in addition to CO_2 and water (*3*, *18*). These include carbon monoxide (CO), methane (CH_4), non-methane hydrocarbons (NMHCs), and particulate matter (PM). CO, CH_4, and NMHCs can affect the radiative balance of the atmosphere to an equal to or greater extent than a molar equivalent amount of CO_2 (*19*). Although CO_2 is the most commonly discussed GHG, non-CO_2 greenhouse gases are more relevant in assessing GHG emissions from biomass combustion because, under a system of sustainable fuel use, CO_2 released by combustion is removed from the atmosphere by future plant growth, while the other compounds remain in the atmosphere until they are removed by different mechanisms (*10*).

Radiative Forcing and Global Warming Potential (GWP). The ability of a chemical compound to trap heat in the atmosphere is termed radiative forcing. To compare this characteristic across different compounds, a GWP is defined, which is a ratio of the radiative forcing of the compound in

10.1021/es026058q CCC: $25.00
Published on Web 00/00/0000

TABLE 1. Global Warming Potential (GWP) of GHGs Commonly Emitted from Biomass Combustion on a Molar Basis (*19*)[a,b]

compd	20-yr GWP	100-yr GWP	500-yr GWP	comment
CO_2	1	1	1	CO_2 GWP is 1 by definition for all time horizons
CO	2−6	0.6−2	0.2−0.6	range of values reported in IPCC (*19*): lower values consider CO effect on OH radicals; higher values also consider ozone (O_3) production
CH_4	22.5	8.4	2.5	from IPCC's third assessment report (*19*, *34*)
NMHC[c]	12	4.1	2.3	from IPCC's first report (*35*); subsequent reports do not offer values for NMHCs due to high degree of uncertainty
N_2O	275	296	156	NO_2 was not measured in this study and is included here for comparison only

[a] The time-dependent behavior of the GWP arises from the atmospheric lifetimes of the compounds and their decay products (*19*). Of the gases listed, only CO_2, CH_4, and N_2O are targeted for emissions limitations and/or reductions in the Kyoto Protocol. CO and NMHCs are not under discussion because of the uncertainty in their effect on climate (*19*). [b] The IPCC does not offer a GWP for PM; hence, we do not include it here. Estimations exist for the cumulative effect of PM on radiative forcing. Airborne PM has a mixed effect on climate, with black carbon particles contributing to climate warming and other carbon particles contributing to climate cooling, but the level of scientific understanding of both effects remains "very low" (*11*, *19*). [c] Following the IPCC (*35*), the molecular weight of NMHCs from biomass combustion is assumed to be 18 g/mol of C. Thus, when using the molar GWP of NMHCs, we are actually considering moles of C rather than moles of a mix of compounds that generally have more than one C atom per molecule. This facilitates comparison with other single C atom compounds such as CO and CH_4.

FIGURE 1. Stoves used in the study area. (a) Three-stone fire with metal grate. (b) Ceramic-lined woodstoves (from left to right): the Upesi, the Kuni Mbili, and the Lira. (c) Charcoal stoves (from left to right): the Kenyan ceramic Jiko (KCJ), a single-walled metal stove, and a double-walled metal stove (Loketo).

question to an equivalent quantity of CO_2 on a mass or molar basis (*19*). Table 1 shows the molar GWP for the most prevalent greenhouse gases contained in typical biomass combustion emissions. Our results are based on the 20-yr GWP. We chose this value in order to be consistent with the work of Smith and co-workers (*12*−*14*). Our choice of GWP has no qualitative effect on our results because the relevant GWPs decrease over time at comparable rates. Only nitrous oxide (N_2O) has an increasing GWP, but N_2O is negligible in our analysis (discussed below) (*19*).

Despite large GWP on a molecular basis for N_2O, the nitrogen content of typical woodfuels is quite small, and only trace amounts of nitrogenous species are released from the fuel itself. Furthermore, the combustion temperatures of household biomass stoves are generally too low to react with atmospheric nitrogen in any appreciable way. Hence, the contribution of N_2O to the GHG emissions and net global warming commitment (GWC) of household-scale woodfuel combustion is negligible (*13*−*15*), and its exclusion from this study does not affect our conclusions.

Methods

Research Location. The study took place at Mpala Ranch and Research Centre, in Laikipia District, central Kenya. Firewood and charcoal (almost entirely of acacia species) are the main fuels in the study households. The stoves tested are shown in Figure 1 and described in Table 2. Firewood was commonly air-dried before use (dryness was confirmed qualitatively on each measurement day). We assumed 20% moisture content (wet basis) and an energy content of 16 MJ (HHV). Charcoal is produced locally, with an assumed energy content of 29 MJ/kg (HHV). (The heat content of air-dried acacia and charcoal are based on the findings of ref *14*.)

TABLE 2. Stove−Fuel Combinations in the Study Group

stove name	material: body	material: liner	fuel	price (US $)
three-stone fire	na[a]	na	firewood	$0
Kuni Mbili	metal	ceramic	firewood	$4−6
Upesi	metal	ceramic	firewood	$4−6
Lira	metal	ceramic	firewood	$4−6
metal Jiko	metal	na	charcoal	$1.5−2
Kenya ceramic Jiko (KCJ)	metal	ceramic	charcoal	$4−6
Loketto	metal	metal	charcoal	$4−6

[a] na, not applicable.

Data Collection. PM was measured with a *personal*DataRAM (PDR) manufactured by MIE, Inc. (Bedford, MA). The PDR uses nephelometric (photometric) monitoring with passive sampling, which minimizes interference with normal activities of the household. The particle size range of maximum response is 0.1−10 μm. Carbon monoxide concentration was measured using Enerac Pocket 100 manufactured by Energy Efficiency Systems, Inc. (Westbury, NY). The instruments were zeroed in clean air outside the village compound every day, and the measurement chamber of PDR was cleaned using pressured air after every 2 d of measurement. The instruments were sent to the factory annually for recalibration of measurement range (span) and replacement of PDR measurement chamber and Enerac sensors. PM_{10} concentration values are relative to factory calibration of the measurement instrument, which is based on light-scattering properties of a standard mixture (dry Arizona road dust) with an uncertainty of 20% for wood smoke. The measurements included both emissions inside the house and contributions

from ambient air including wind-blown dust and smoke from neighboring houses. Because of the extremely low housing density, the latter was negligible.

PM_{10} and CO concentrations were recorded at approximately 0.5 m from the center of the stove, at a height of 0.5 m. PM_{10} concentration was averaged and recorded in 1-min intervals between 06:30 and 20:30. In every day of sampling, the status of the fire was recorded at 5−10-min intervals using the following protocol:

starting: fire being lit by the user (accompanied by high emissions)

burning: vigorously burning fire with extensive flames visible

dying fire: barely burning fire with few flames visible

hot coals: no flames visible but coals visibly glowing

dying coals: coals still hot and possibly used for warming food but largely covered in ash so little or no glow visible

Data collection was performed by two field research assistants, accompanied by a principal researcher for the first 6 months of data gathering, with regular examination of data recording protocol thereafter. Test sessions were conducted, and the protocols were adjusted to ensure minimal interference with household activities and that the classification of fire status was systematic and consistent. PM_{10} concentration data, which were logged automatically by the PDR, were downloaded into a personal computer after every day of monitoring.

A total of 210 d of sampling was conducted in 55 randomly selected houses. The visits were made on random days of the week. Approximately 20% of the households were visited between 6 and 15 times to monitor the intra-household variation in emission concentrations as well as variations in time−activity budgets. Another 25% were visited once, and the remaining households were visited between 2 and 5 times. Data in this analysis come from a subsample of 19 households over 29 measurement d, selected from the larger sample to represent all stove−fuel combinations and village types (15 d for three-stone open fires, 6 for improved woodstoves, and 8 for charcoal).

GHG Estimations. The estimates of carbon-based GHG emissions relied on a carbon-balance calculation in which the carbon content of the fuel minus any unconsumed carbon in char and ash is assumed to equal the sum of carbon contained in the gaseous and aerosolized combustion emissions as shown in eq 1 (C_i is the mass of carbon contained in the ith product of the reaction):

$$C_{\text{fuel}} = C_{\text{CO}_2} + C_{\text{CO}} + C_{\text{CH}_4} + C_{\text{NMHC}} + C_{\text{TSP}} \qquad (1)$$

Dividing both sides of eq 1 by C_{CO} gives a series of emissions ratios with respect to CO as in eq 2. Using CO-based emissions ratios differs slightly from the previous work using CO_2 to define emissions ratios (*14*, *15*). This alternative approach is used in this study because the concentration of CO was measured directly:

$$\frac{C_{\text{fuel}} - (C_{\text{CO}_2} + C_{\text{CH}_4} + C_{\text{NMHC}} + C_{\text{TSP}})}{C_{\text{CO}}} = 1 \qquad (2)$$

Solving eq 2 for C_{CO} provides the mass of CO released in the combustion reaction as a function of fuel carbon and the sum of emissions ratios:

$$C_{\text{CO}} = \frac{C_{\text{fuel}}}{1 + K'}\left[\text{where } K' = \sum\left(\frac{C_i}{C_{\text{CO}}}\right) \text{ for } i = \text{CO}_2, \text{CH}_4, \text{NMHC, and TSP}\right] \qquad (3)$$

The carbon released with each constituent of combustion emissions can then be calculated by a simple cross-multiplication:

$$C_i = \left(\frac{C_i}{C_{\text{CO}}}\right)C_{\text{CO}} = \left(\frac{C_i}{C_{\text{CO}}}\right)\left(\frac{C_{\text{fuel}}}{1 + K'}\right) \qquad (4)$$

Using eq 4, it is possible to determine the emissions factor (EF) for pollutant i during each cooking activity or phase of combustion (labeled with subscript j in eq 5). The emission factor is the rate of pollutant emission with respect to a characteristic of the fuel-like mass or energy consumed during each activity or phase of combustion (j):

$$\text{EF}_{i,j} = \frac{C_i}{H_{\text{fuel},j}}$$

$$\text{where H is the heat content of the fuel consumed during activity } j \qquad (5)$$

where $H_{\text{fuel},j}$ is the heat content of the fuel consumed during activity j. Finally, the GWC of a cooking activity or phase of combustion is defined as the net emissions of GHGs from that activity/phase in carbon mass expressed in CO_2 equivalent units:

$$\text{GWC}_j = \sum_i C_{i,j} \times \text{GWP}_i \qquad (6)$$

Equation 6 can be summed over j to provide a total GWC for the assessment period. In addition, GWC can also be expressed as an emission factor by dividing the result by the mass or energy of fuel consumed.

The variables measured in the field included the mass of fuel input, the concentrations of CO and PM_{10}, and the fire status as described above. To fully account for carbon flows, total suspended particulates (TSP) should be measured rather than PM_{10}. However, 90−95% of particulate mass emitted by biomass combustion consists of particles $<3\,\mu$m in diameter and is included in PM_{10} measurements (*3*). Furthermore, the extremely high indoor concentrations of PM and the heavy blackening of the underside of the thatched roofs and inner walls of the houses indicate that a large fraction of PM does not exit the house. While this is a cause for concern for indoor air quality and public health, PM released indoors in these conditions is not likely to have a measurable impact on climate change (see note in Table 1) and using PM_{10} rather than TSP should not affect our calculations or our policy recommendations.

Calculating K' (the sum of ratios relative to CO) required the ERs of some gases that were not measured directly. These were obtained from the work of Brocard et al. (*15*), who defined ERs relative to CO_2 that were recalculated in this analysis to relative to CO.

Analogies were drawn between Brocard et al.'s (*15*) stages of combustion and those reported in this study as in Table 4. In the calculations for wood-burning stoves, emissions ratios for "dying fire" were assumed to be the average of "burning" and "hot coals". This provides a more complete gradation of the burn regime than grouping this state with one of the two adjacent ones.

The results of the conversion to ERs relative to CO, shown in Table 5 as boldface entries, were added to the measured ratio of TSP to CO for each phase of combustion in each day's measurements. The sum across each row in Table 5 is defined as K' (used to estimate the mass of C emitted in each species of pollutant following eqs 3 and 4).

Results

Total Emissions. Figure 2 shows the estimated mass of carbon emitted, disaggregated by pollutant (Figure 2a) and by phase

TABLE 3. Emissions Ratios for Firewood and Charcoal Combustion Reported by Brocard et al. (*15*)[a]

	firewood combustion (%)[b]					charcoal (%)	
	weighted average[c]	ignition	cooking	end-cooking	end-fire	making	burning
CO/CO_2	7.9 ± 1.5	26.1 ± 4.8	5.7 ± 1.1	15.0 ± 2.8	21.0 ± 2.7	24.0 ± 3.0	15.5 ± 3.0
CH_4/CO_2	0.38 ± 0.11					6.8 ± 0.6	0.25 ± 0.20
$NMHC/CO_2$	0.57 ± 0.24					1.3 ± 0.3	0.06 ± 0.007
TSP/CO_2	1.17 ± 0.63					3.3 ± 0.7	0.314[d]

[a] The blank cells indicate data omitted from Brocard et al. (*15*). We estimated these by assuming that the ratio for every pollutant can be scaled in proportion to the ratios that were reported for CO/CO_2. This gave ERs relative to CO_2 for every pollutant during each burn regime. All values are percentages. [b] The authors provide emissions ratios for all stages of combustion only for CO/CO_2 from firewood. Their report did not provide different ratios for other gases from firewood combustion, nor did it provide differentiated ratios for charcoal making and burning. [c] The authors calculated a weighted average for firewood by assuming that 80% of the mass of wood is consumed in the flaming stage, 15% is consumed in the glowing stage, and 5% is consumed in the smoldering stage. [d] Brocard et al. (*15*) did not report any emissions ratio for TSP from charcoal combustion; however, Smith et al. (*13*) report a value of 0.314% for an insulated charcoal stove from India similar in design to the KCJ.

TABLE 4. Matching the Stages of Combustion from Brocard et al. (*15*) with Observations from This Study

Brocard et al.	this study: wood	this study: charcoal
ignition	starting	starting
cooking	burning dying fire	burning coals
end-cooking	hot coals	hot coals
end-fire	dying coals	dying coals

TABLE 5. Emissions Ratios for Firewood and Charcoal Combustion Used To Estimate GHG Emissions in This Study

obsd phase of fire	CO_2/CO	CH_4/CO	NMHC/CO	TSP[a]/CO	K′ [b]
Three-Stone Fire and Ceramic Wood Stoves					
starting	**3.8**	**0.26**	**0.050**	0.215	4.33
burning	**17.5**	**0.048**	**0.072**	0.016	17.64
dying fire	**9.7**	**0.025**	**0.023**	0.028	9.78
hot coals	**6.7**	**0.017**	**0.004**	0.018	6.74
dying coals	**4.8**	**0.32**	**0.062**	0.024	5.21
Charcoal Stoves (KCJ and Loketto)					
starting	**4.2**	**0.28**	**0.054**	0.00064	4.53
burning coals	**6.5**	**0.016**	**0.004**	0.00038	6.52
hot coals	**5.1**	**0.18**	**0.034**	0.00076	5.31
dying coals	**4.2**	**0.28**	**0.054**	0.0019	4.54

[a] TSP/CO are averaged empirical observations. [b] *K′* is the sum of each row of ERs.

of combustion (Figure 2b) for all measurement days. The figure illustrates that the estimated emissions varied considerably across households using different stove–fuel combinations and between households using the same fuels. For example, the total emissions of non-CO_2 compounds in charcoal-burning households ranged from 550 to over 1400 g of C/d. Households burning wood in three-stone fires showed less variability, with a range of emissions between 350 and 780 g of C/d. Households using ceramic stoves had the lowest variability, with a range of emissions between 700 and just over 1240 g of C/d. Such variation was evident even among the same households on different measurement days, as indicated by household number codes along the horizontal axis. This variation arose largely due to differing levels fuel consumption and different patterns of fire maintenance (see below).

The averaged daily emissions of each pollutant by stove type are shown in Table 6. The table presents emissions in terms of carbon released (not weighted by GWP) and in terms of carbon in CO_2 equivalent units (weighted by 20-yr GWP) in the left- and right-hand sides of the table, respectively.

Table 7 shows the average daily breakdown of times in each combustion phase as well as the daily fuel consumption in each stove–fuel category. Charcoal-using households consumed less fuel because charcoal has a higher energy content than wood and because charcoal stoves are generally more efficient than woodstoves. Thus, charcoal use resulted in lower emissions when emissions were measured on the basis of carbon mass. However, charcoal tends to burn less efficiently than wood. Therefore, charcoal has higher emissions of non-CO_2 GHG, which leads to a higher GWC from charcoal-burning households with or without the assumption of sustainable harvesting.

Emission Factors. The total GHG emissions estimated above depend on fire maintenance practices and the amount of fuel burnt on the day of observation, which varied from 2 to 10 kg for charcoal and from 8 to 22 kg for wood-burning households in our sample.

Considering emissions factors rather than absolute emissions normalizes the variability fuel consumption and stresses the impact of variability in fire maintenance, which is largely beyond experimental control when the measurements are performed in field conditions. However, emission factors defined in terms of mass are not directly comparable across different fuels because firewood and charcoal (and other household fuels) have substantially different carbon contents per unit mass and their emissions vary accordingly. Defining emission factor with respect to energy rather than mass accounts for this. Smith et al. (*13*, *14*) defined an alternative emissions factor in terms of useful energy delivered to the pot to account for differences in the heat transfer efficiency of each stove. However, our day-long data show that many people allow fuel to burn throughout the day, even when they are not cooking, which complicates a definition of useful energy and reduces the applicability of the heat transfer efficiency of the stove in our estimation.

Emission factors were estimated from ERs (as in eqs 3–5) obtained from previous work (*14*, *15*) and applied to each phase of combustion for each stove–fuel combination. Because *K′* was the same within stove types and combustion phases, estimates of EFs varied little for a given phase of combustion within households using the same type of stove and fuel. However, daily averages were estimated by weighting combustion phase EFs by the fraction of time the fire was in each combustion phase.

For example, for CO emissions from three-stone wood fires in the starting phase of combustion, we estimated average CO emissions of approximately 182 g of CO/kg of fuel consumed in that phase for the sampled households. Estimates from other phases of combustion for this stove–fuel combination were 52 g of CO/kg of fuel in the burning phase, 91 g in the dying fire phase, 127 g in the hot-coal phase, and 158 g in the dying-coal phase with little variation across households. However, because the fraction of day that each household allowed a fire to burn or smolder varied considerably, there was interhousehold variation in the total daily emissions. Therefore, the average CO EF for each

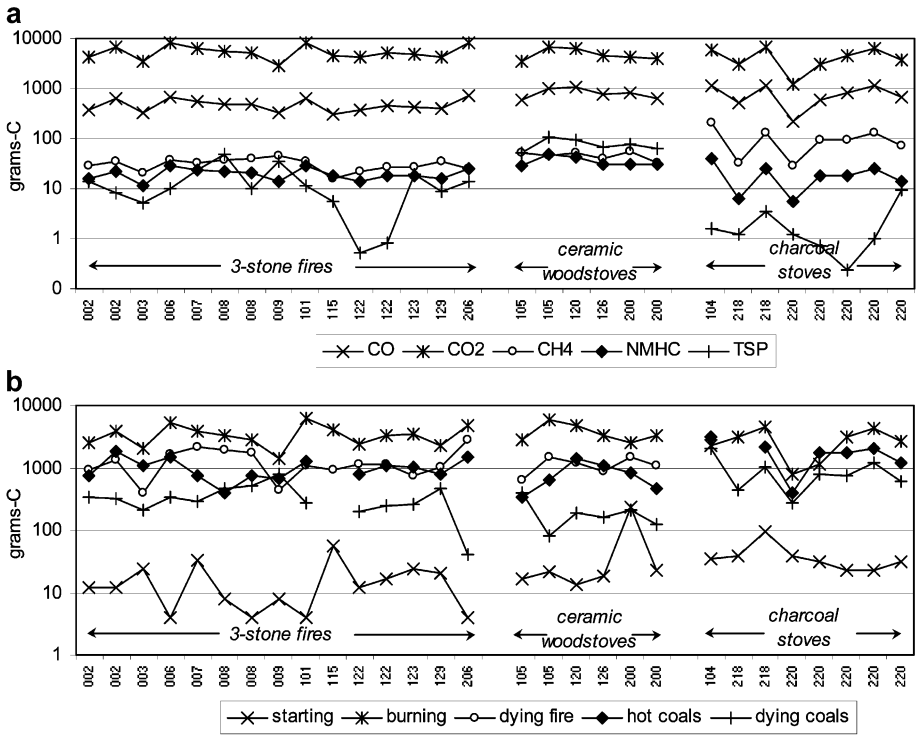

FIGURE 2. Daily carbon emissions by pollutant and phase of combustion (all households). Both panels use logarithmic vertical scales, and emissions are not weighted by GWP. Numbers on the horizontal axis indicate household identification numbers. Panel a shows emissions from each household disaggregated by pollutant. Panel b shows emissions from each household disaggregated by phase of combustion.

TABLE 6. Mean, Median and Standard Deviation[a] of Estimated Daily Emissions in g of C (Left) and GWC Weighted by 20-yr GWP (Right)

	avg daily emissions (g of C)									avg daily GWC (g of C in CO_2 equiv: 20-yr GWP)								
	three-stone fire			ceramic woodstoves			charcoal stoves			three-stone fire			ceramic woodstoves			charcoal stoves		
	mean	median	SD	mean	median	SD	mean	median	SD	mean	median	SD	mean	median	SD	mean	median	SD
CO_2	5450	5273	1700	4937	4446	1342	4300	4163	1900	5450	5273	1700	4937	4446	1342	4300	4163	1857
CO	480	469	140	810	795	191	780	740	100	1920	1875	540	3240	3181	765	3120	2960	410
CH_4	30	33	8	46	48	9	98	95	56	701	754	188	1042	1081	195	2201	2140	1270
NMHCs	20	19	5	36	31	9	19	18	11	240	224	60	430	376	104	230	222	130
TSP	14	10	13	76	71	21	2	1	3									
non-CO_2 GHGs	545	546	144	968	945	221	899	845	406	2860	2722	680	4711	4631	919	5550	5174	2700
total GHGs	5998	5833	1843	5905	5392	1553	5202	5008	2257	8310	8039	2400	9649	9077	2207	9850	9197	4600

[a] The standard deviations in this table reflect variability across different measurement days in the estimates of household emissions but are not representative of the uncertainty in GWPs or in the assumed ERs. Discussed in detail in Table 7.

TABLE 7. Average Times of Each Combustion Phase and Average Daily Fuel Consumption

	three-stone fire ($n = 15$)			ceramic woodstoves ($n = 6$)			charcoal ($n = 8$)		
avg time (min)	mean	median	SD	mean	median	SD	mean	median	SD
starting	17	15	11	20	22	10	22	20	8
burning	255	250	96	258	275	118	223	245	103
dying fire	139	130	56	97	110	47			
hot coals	166	180	81	133	125	83	164	185	75
dying coal	205	200	121	140	110	118	247	247	55
avg daily fuel consumption (kg)	14.3	14.0	4.4	11.9	12.0	5.5	6.9	6.9	2.8

household using the three-stone fire ranged between 60 and 95 g of CO/kg of fuel (79 ± 7 g of CO/kg of fuel). Similar estimates were made for each GHG and stove–fuel combination with the results shown in Table 8, including comparisons to findings from other studies.

Most of the results in Table 8 are consistent with the results of previous studies (*14*, *15*) as well as the default factors used by the IPCC (*20*) to estimate emission baselines. There are, however, some disparities such as CH_4 and TSP for charcoal stoves. In addition, there is a lack of agreement for the emission factors of NMHCs among the other studies, with the results of this analysis falling somewhere in the middle. The largest disparity was the emissions factor for CH_4 from charcoal. This is particularly important because CH_4 has a large GWP and because the net GWC is quite sensitive to CH_4 emissions.

TABLE 8. Average Emission Factors per Unit Mass of Fuel Consumed for Each Stove–Fuel Grouping[a]

	estimations from this study (mean ± SD)			findings from other studies						
				Brocard et al. (*15*)		Smith et al. (*14*)			IPCC default factors (*20*)	
	three-stone fire	ceramic wood	charcoal	three-stone fire	charcoal	three-stone fire	ceramic wood	charcoal	wood	charcoal
CO_2	1390 ± 19	1400 ± 10	2280 ± 34	1470	2260	1370	1350	2410	1370	2400
CO	79 ± 7	74 ± 6	260 ± 10	70	211	64.7	79.0	275	80	200
CH_4	3.2 ± 1.5	2.5 ± 0.9	18 ± 6	2.0	2.4	9.40	3.42	7.91	5	6
NMHC	1.6 ± 0.2	1.6 ± 0.1	3.2 ± 0.9	2.9	0.42	9.65	12.6	10.5	9	3
TSP[b]	1.1 ± 1.2	5.9 ± 0.4	0.4 ± 0.5	5		2.05	3.32	2.38	2.1	2.4

[a] All factors are reported in g of pollutant/kg of fuel except where otherwise stated. [b] TSP is reported in g of carbon only.

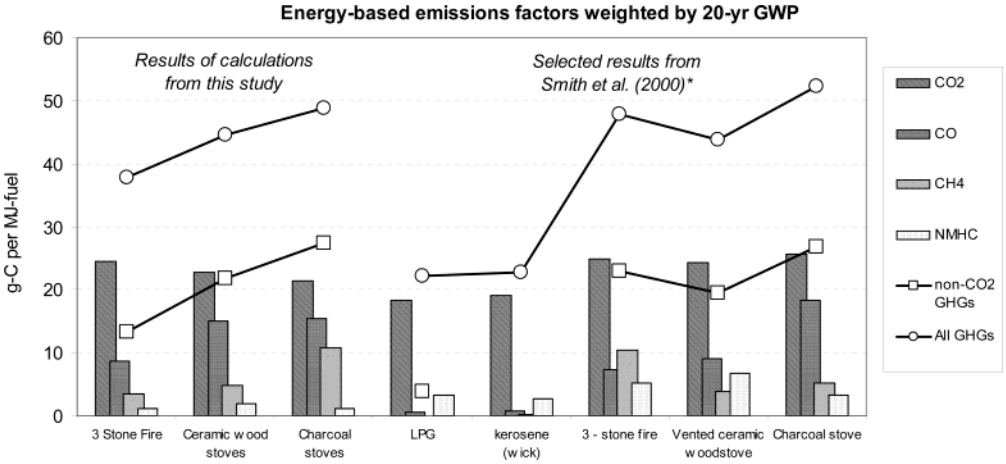

FIGURE 3. Comparison of energy-based emission factors by stove–fuel category from this study and selected results reported by Smith et al. (*14*). The height of each bar shows the average emission factor of each pollutant, while the lines show the sum of the GWP for each GHG with CO_2 (circles) and without CO_2 (squares). For biomass fuels, the latter represents fuels that are sustainably harvested so that biomass stocks are not depleted over time, while the former is applicable if stocks of biomass are fully depleted. Because fossil fuels do not allow for CO_2 replacement, the accounting of GHGs always includes CO_2, and the non-CO_2 line is omitted for these fuels. (*) Woodstoves from Smith et al. (*14*) were tested with Acacia species.

The energy density of charcoal is approximately double that of wood, and households tend to use less charcoal than wood. Replacing mass-based EFs with energy-based EFs reduces the estimated emissions from charcoal stoves by about half relative to woodstove EFs. Despite the favorable decrease of energy-based emissions for charcoal stoves relative to woodstoves, Figure 3 shows that, even on an energy basis, charcoal stoves still had higher GHG EFs than woodstoves. The results of Smith et al. (*13, 14*), included in the figure, show a similar pattern for wood and charcoal.

Figure 3 also shows that both LPG and kerosene have energy-based emission factors that are comparable to, if not lower than, the emissions from renewable biofuels and are far lower than the emissions from biofuels when they are not used renewably. This contrast becomes more pronounced in the analysis of Smith et al. (*13, 14*) because, as discussed above, they base their analysis on useful energy. Fossil fuel stoves are more efficient than biofuel stoves in both combustion and heat transfer, and an analysis of emissions per unit energy delivered to the cooking pot privileges kerosene and LPG over solid biofuels. Cooks do not allow fossil fuels to burn throughout the day as they do with wood or charcoal. Hence, accounting for stove efficiency is more appropriate when fossil fuels are used, but it is not appropriate in this analysis.

Analysis of Variance (ANOVA). Several factors contributing to the variability in our results were analyzed through ANOVA. Non-CO_2 GHG emissions weighted by GWP showed that the fraction of variation in absolute GHG emissions explained by sampling in different households is 23 times the fraction explained by stove–fuel combination, emphasizing the importance of interhousehold variability. This is most likely a result of differences in the amount of fuel consumed. Using EFs, which minimize the influence of absolute fuel consumption, reduced the ratio of the fraction of variance explained by interhousehold variation to that explained by stove–fuel combination to 0.7. Although this reduction indicates that much of the interhousehold variability in emissions is due to differences the amount of fuel, the ratio of 0.7 illustrates that "behavioral" aspects remain important; the users' handling of the stove and time allotted to different stages of combustion (captured by interhousehold variability) were responsible for nearly as much variability in EFs as the choice of stove and fuel.

Sensitivity Analysis. To test the sensitivity of the net GWC estimates to the assumed ERs, the analysis was conducted with the ERs in Table 3 ranging from 0.10 to 2.0 times their original (baseline) values. Changing the ER for each gas individually showed that the estimated emissions of woodstoves were most sensitive to changes in CO ERs, while estimated emissions of charcoal stoves were slightly more sensitive to changes in CH_4 ERs than those of CO. For example, considering the total GWC of all GHGs (the bottom row in the right-hand side of Table 6), a 25% increase in CO emissions relative to CO_2 resulted in a net increase of the estimated total GWC of roughly 15% for both types of woodstoves and 6% for charcoal stoves. Alternatively, a similar increase in CH_4 relative to CO_2 resulted in a 6% increase in estimated total GWC for three-stone fires, 4% increase for ceramic woodstoves, and 9% increase for charcoal stoves. Results for each stove–fuel category, weighted by 20-yr GWP, are shown in Figure 4. In each graph, the lines represent the

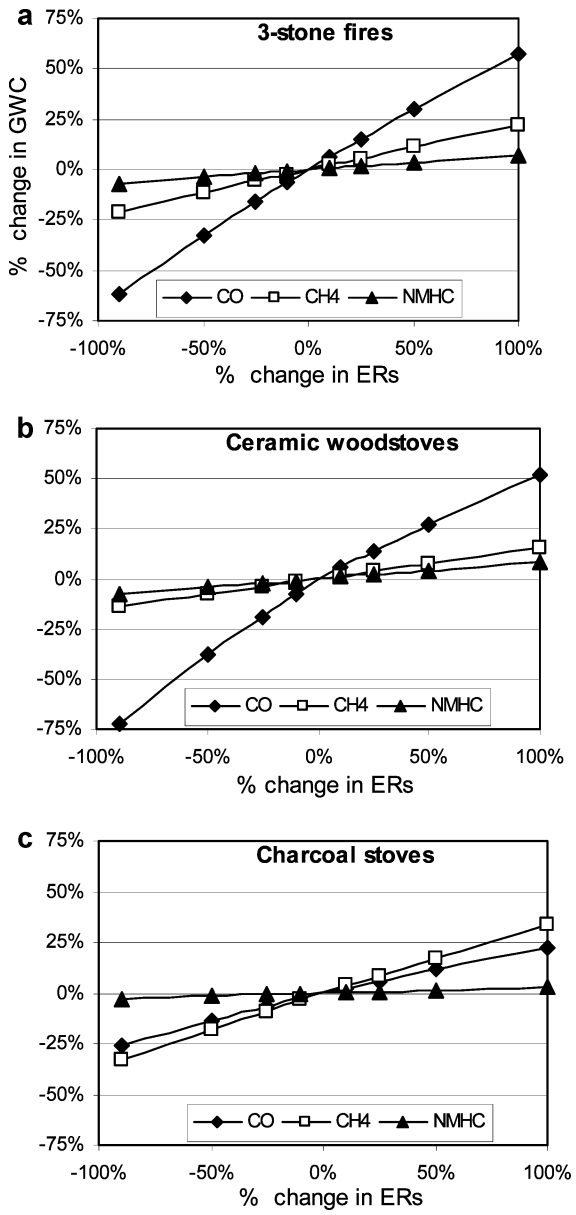

FIGURE 4. Sensitivity of estimated GWCs (including CO_2) to changes in ERs with respect to CO_2 for each stove−fuel category.

percent change in net GWC, including CO_2, occurring when the ER for CO, CH_4, and NMHCs are varied from 0.10 to 2.0 times the values from Brocard et al. (*15*) used in our baseline calculations (Table 3). TSP was not included in sensitivity analysis because it was measured directly and because it does not factor directly into the GWC calculations.

Discussion

Our estimates of GHG emission factors and average daily emissions for three different types of common biomass fuels and cookstoves used in rural Kenya under conditions of actual use showed that charcoal stoves tend to have lower absolute emissions of GHGs in terms of carbon mass emitted. However, the mix of compounds emitted by stoves burning charcoal usually has a higher fraction of CO and CH_4 than the products of wood combustion, which leads to a larger GWC because of the high GWP of these compounds. The potential climate change impacts of charcoal become more acute when one considers the entire life cycle of the fuel. Unlike woodfuel, which involves few, if any, GHG emissions prior to its use in the stove, charcoal combustion only represents a fraction of the net GHG emissions from the charcoal life cycle. Pennise et al. (*12*) measured the emission of GHGs from Kenyan earth mound kilns, the country's most common production method, and found that producing 1 kg of charcoal emits more than 1800 g of CO_2, 220 g of CO, 44 g of CH_4, 92 g of NMHC, and 30 g of TSP.

Assuming that the charcoal is produced sustainably so that the CO_2 is recycled and summing the other pollutants weighted by 20-yr GWP, over 1800 g of C of non-CO_2 GHGs (in CO_2 equivalent units) are emitted per kilogram of charcoal produced. We estimated that burning 1 kg of charcoal releases another 800 g of C (measured in the same units); therefore, charcoal production and use emits over 2600 g of C/kg or roughly 90 g of C/MJ, even when stocks of biomass are not depleted and emissions resulting from transport of the fuel are not considered.

In comparison, emissions of non-CO_2 GHGs from firewood were in the range of 200−400 g of C (CO_2 equivalent units, 20 yr-GWP)/kg of fuel consumed across a range of stove types, consistent with estimates of Smith et al. (*14*). In energy terms, woodstoves released between 13 and 24 g of C/MJ (CO_2 equivalent units and 20-yr GWP). While including stove efficiencies in the analysis would reduce the relative global warming contribution of charcoal, this fuel remains a greater emitter of GHGs than woodstoves regardless of the analytic methodology and assumptions about how ideal efficiency translates to daily emissions.

Charcoal production and use have other environmental impacts in sub-Saharan Africa, particularly with respect to deforestation (*21*−*24*). Previous work has shown that while charcoal production does not always lead to permanent loss of tree cover, it may be associated with land degradation as a result of a combination of ecological and socioeconomic factors (*21*−*24*). In Kenya, the consensus among the environmental community is that current charcoal production practices are having a negative effect on many of the country's forests and woodlands. The evidence for these effects, however, is anecdotal, and to our knowledge there no recent systematic studies of charcoal industry's ecological impact on specific woodlands or on a national scale (*25*).

Public Health. While emissions from charcoal production and end-use are associated with higher GWC as compared to firewood in Kenya, charcoal use offers public health benefits over fuelwood, especially if clean-burning cooking fuels such as kerosene and natural gas are unavailable or unaffordable (see below). Ezzati and Kammen (*26*) found that transition from three-stone fire to charcoal reduced PM_{10} exposure of household members by 75−95% on average for different demographic groups of the study population, resulting in an estimated 45% decrease in childhood acute lower respiratory infections (ALRI), the leading cause of morbidity and mortality globally (*27*), in addition to adult health benefits. Poor nations such as Kenya that contribute very little to the total global release of GHGs would likely gain more from the immediate health benefits associated with fuel substitution from wood to charcoal than they would from discouraging its use because it carries a heavy GHG burden, especially given our increasing awareness of the impact of household energy on the health of the world's poor (*28*). At the same time, if the decision is made to promote charcoal consumption because of its public health benefits, steps must also be taken to ensure more efficient production methods and a sustainable supply of wood or an alternative biomass feedstock.

Fuel Switching and Charcoal Markets. Household survey data show that, in urban areas of Kenya where kerosene and, to a lesser degree, LPG are available, their use increases with increasing household expenditure (*29*). This indicates that ability to pay is likely to be one factor limiting the adoption of cleaner fuels in poor urban households. In rural areas,

however, LPG and kerosene are rarely used, even in households with incomes comparable to the 3rd and 4th expenditure quintiles of urban areas indicating that, in addition to affordability, availability is likely to be a limiting factor in the adoption of LPG and kerosene in rural areas. In urban Kenya, as in many other sub-Saharan African countries, charcoal is readily available, can be purchased in small quantities, and requires no expensive equipment to use. For these reasons and because it is relatively clean, safe, and stores well, charcoal is the preferred fuel for many urban households as well as some well-off rural families. Therefore, despite the environmental effects described above, attempts to curtail charcoal consumption are likely to be met with public resistance unless policies specifically designed to increase access to alternative stoves and fuels such as kerosene and LPG.

Household energy policy is further complicated because charcoal markets in many sub-Saharan African countries operate within a complex political economy that can be hard to characterize and still more difficult to regulate. Even where regulations have been put forth, as in some West African countries, they are often poorly enforced and/or circumvented by powerful interest groups who control one or more parts of the commodity chain (*30*). In Kenya, charcoal production is periodically prohibited, yet thousands of people make their living by participating in one or more steps of the charcoal supply chain, and half of the urban population, some 1 million households, continue to use charcoal as their primary cooking fuel (*25*). To take advantage of the potential benefits of charcoal consumption while minimizing the negative impacts associated with its production and use, a much more coherent policy framework is required. Such a framework would legalize and regulate charcoal production and ensure that sustainable levels of production are maintained while consumer needs are met with prices that reflect the true cost of production including harvesting and regeneration, conversion, transportation, and sales (*31*).

Carbon Credits and Mitigating GHG Emissions. While charcoal consumption carries a larger burden of GHG emissions than firewood use, it also has more potential to attract investment in GHG mitigation activities. Emissions from charcoal can be reduced at both production and consumption components of its life cycle. Emission reductions in charcoal end-use can be achieved by disseminating improved (high-efficiency and low-emissions) charcoal stoves, which reduce emissions by improving both combustion and heat transfer efficiency. Furthermore, users should see substantial fuel savings. Such charcoal stoves have been widely disseminated and adopted in urban Kenya, although they are still short of saturation levels and the potential remains for wider dissemination, particularly into rural areas (*32*). In addition, little research has been done to assess field performance of stoves currently on the market for household use or to document the dissemination of substandard stoves since donors and nongovernmental groups have stopped participating in stove design and dissemination projects (*33*).

Moreover, rather than focusing on stove efficiencies as the sole project deliverable, intervention programs should take multiple aspects of household energy use into account. Alternatively, behavior-based intervention programs that optimize fuel consumption by increasing the fraction of fuel energy delivered to the cooking pot should be considered together with housing design factors such as the levels of ambient lighting or lighting alternatives as well as levels of household insulation and ventilation. All of these factors affect the level of biofuel consumption and the extent to which stoves are left burning throughout the day, which as seen earlier is an important determinant of emissions.

While some work has addressed charcoal consumption, researchers are only beginning to consider charcoal production in Kenya (*12*). Arguably larger GHG emission reductions and energy conversion efficiency improvements can be achieved by addressing charcoal production because roughly 70% of non-CO_2 GHG emissions from charcoal production and use occur during the production process (*12*).

Assessing GHG emissions from biofuels should draw attention to an aspect of domestic biofuel use that has been overshadowed by more immediate deforestation as well as health concerns relating to pollution emissions and exposures. Two critical categories of combustion emissions, health-damaging pollutants and greenhouse gases, result from similar processes of incomplete combustion. Expanding the field of indoor air quality in developing countries to include GHG emissions should direct more attention and financial resources to understanding and mitigating one of the world's leading risk factors of morbidity and mortality while reducing long-term damage in the form of global climate change.

Acknowledgments

We thank Bernard Mbinda, Mark Egelian, Peter Ekuam, Mary Lokeny, and Jackson Ngisirkale for assistance in data collection and the residents of Mpala Ranch for their hospitality. K. R. Smith and D. Pennise provided helpful comments in assessing GHG emissions and GWPs. In addition, we appreciate the insights offered by two anonymous reviewers in preparation of this paper. Our research was supported by grants from the Summit and Compton Foundations, the Social Science Research Council (SSRC), the Energy Foundation, the Link Foundation, and Princeton University's Council on Regional Studies and Center of International Studies (through a grant from the MacArthur Foundation).

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Received for review August 14, 2002. Revised manuscript received January 28, 2003. Accepted February 7, 2003.

ES026058Q