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degree in chemistry

Line W. H. Alnes

**Indoor Air Pollution in
China: Analysis of Global
Warming Contributions
and Exposure to Particles**

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ABSTRACT

About 60% of the Chinese population lives in rural areas, where biomass and coal are the main sources of energy for cooking and heating. Indoor air pollution from household fuel burning is a major health concern, responsible for at least 420,000 premature deaths annually in China alone. However, less is known about the emissions of greenhouse compounds from the household sector. Therefore, this work aims to quantify the global warming contribution (GWC) from main household fuels in rural China. Global warming potentials (GWPs) are used to compare the impact of different greenhouse compounds (GHCs) relative to CO₂. This is combined with emission data from Zhang et al. (2000), to estimate the total GWC in CO₂ equivalents from household fuel combustion stratified by main fuel categories. The key factors affecting GWC are the GHCs included in the calculations, whether the biomass fuels are harvested renewably or not and choice of time horizon. There are large differences in GWC between different fuel categories, with estimates for a 20 year time horizon ranging from 130 g CO₂ equivalents per MJ delivered energy for gas to 1560 g CO₂ equivalents / MJ for coal, when all GHCs are taken into account. However, the uncertainty for GWC is large, with standard deviations in the range of 5-30% for different fuel categories due to both high variation in the emission data and uncertainty associated with the GWP values. On average, CO₂ is the compound that contributes most to total global warming, followed by black carbon and carbon monoxide. Organic carbon has the largest cooling effect, while the contribution from sulfate is negligible.

To be able to consider potential co-benefits in terms of health improvements by switching to cleaner fuels, the GWC results were compared to population exposure to respirable particles, stratified by fuel groups. The results show that gas is the most preferable fuel, both when it comes to global warming contribution and public health. The picture is more complicated when it comes to choosing between coal and biomass, where a fuel switch to biomass increased the exposure while most likely cut the GWC. However, only 3 fuel categories were included in the exposure estimate. In reality many more options exist, including improved stoves. Further work should investigate this as well as aim to reduce the uncertainty by obtaining less variable data.

Table of contents

ACRONYMS	5
1.0 INTRODUCTION	6
1.1 GREENHOUSE GAS EMISSIONS.....	6
1.2 HEALTH EFFECTS.....	7
1.3 THE PRESENT STUDY	7
2.0 EMISSION METRIC FOR COMPARING GREENHOUSE COMPOUNDS	8
2.1 GLOBAL WARMING POTENTIAL (GWP)	9
2.1.1 Carbon dioxide (CO ₂).....	11
2.1.2 Methane (CH ₄).....	11
2.1.3 Carbon monoxide (CO)	11
2.1.4 Non-methane hydrocarbons (NMHC)	12
2.1.5 Nitrogen oxides (N ₂ O, NO _x).....	13
2.1.6 Particles (BC, OC, Sulfate)	15
2.1.7 GWP summary.....	17
2.2 GLOBAL WARMING COMMITMENT (GWC)	17
3.0 GLOBAL WARMING COMMITMENT OF MAIN HOUSEHOLD FUELS	18
3.1 DATABASE OF EMISSION FACTORS	18
3.2 GLOBAL WARMING COMMITMENT RESULTS.....	23
3.2.1 GWC: dependence on the GHCs considered.....	23
3.2.2 Stoves, individual GHC contributions and time horizon.....	26
3.2.3 Global warming impact of the household sector.....	31
3.3 UNCERTAINTIES	32
3.3.1 Uncertainty estimates	32
3.3.2 Measurement variability.....	34
3.3.3 GWP uncertainty	35
3.3.4 Caveats and other sources of uncertainty	36
4.0 POPULATION EXPOSURE STRATIFIED BY MAIN FUELS	38
4.1 POPULATION EXPOSURE BASED ON DATA FROM MESTL ET AL. (2007A)	38
4.2 POPULATION EXPOSURE BASED ON DATA FROM EDWARDS ET AL. (2007) ...	42

5.0 EVALUATION OF GLOBAL WARMING COMMITMENTS VS. POPULATION EXPOSURE	45
5.1 GWC COMPARED TO PARTICLE EMISSIONS.....	45
5.2 GWC COMPARED WITH POPULATION EXPOSURE	47
6.0 CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER WORK	50
ACKNOWLEDGEMENTS	52
REFERENCES	53
APPENDIX	58
APPENDIX A: LIFETIMES WHERE $AGWP_{20} \approx AGWP_{100}$	58
APPENDIX B: ANALYSIS OF IAP CONCENTRATION CHANGE OVER TIME.	59
<i>Rural Biomass (South)</i>	60
<i>Rural Coal (South)</i>	60
<i>Rural Biomass, Indoors (bedroom and living room) (North)</i>	61
<i>Rural Biomass, Kitchen (North)</i>	61
<i>Rural Coal, Living room (North)</i>	62
<i>Rural Coal, Kitchen (North)</i>	62
<i>Rural Coal, Bedroom (North)</i>	63

ACRONYMS

AGWP	Absolute global warming potential
BC	Black carbon
GHG	Greenhouse gas
GHC	Greenhouse compound
GWC	Global warming commitment
GWP	Global warming potential
HDP	Health damaging pollutants
IAP	Indoor air pollution
IPCC	Intergovernmental Panel on Climate Change
NCE	Nominal combustion efficiency
NISP	National improved stove program
OC	Organic carbon
OE	Overall efficiency
OM	Organic matter
PIC	Products of incomplete combustion
PM	Particulate matter
PWE	Population weighed exposure
RF	Radiative forcing
SGWP	Sustained global warming potential
TSP	Total suspended particles
WBT	Water boiling test

1.0 INTRODUCTION

About 60% of China's 1.3 billion people lives in rural areas (NBS 2006). Nearly all of the rural population still uses coal or biomass as the main source of energy for cooking or space heating (Zhang and Smith, 2005). The household stoves generally have low energy efficiency, and incomplete combustion generates high levels of air pollutants such as carbon monoxide and particulate matter (Zhang et al., 2000). This has two main consequences: it leads to emissions of greenhouse compounds that in turn cause global warming, and it poses a major health risk through deteriorating indoor air quality.

1.1 Greenhouse gas emissions

“It has been said that wood is the fuel that heats you twice – once when you chop it and once again when you burn it. Like fossil fuels, however, biofuels also have the potential to heat you a third time as a result of enhanced greenhouse warming due to the gases released by combustion.” (Smith, 1994)

If household stoves had a 100% nominal combustion efficiency, all the carbon in the fuel would be converted to CO₂. However, this is not the case. Stoves generally have poor combustion efficiencies, and fuels are of varying quality. This leads to emissions of other substances like carbon monoxide, methane, non-methane hydrocarbons, nitrogen oxides and particles (Zhang et al., 2000). These are all greenhouse compounds (GHCs) that either directly or indirectly affect our climate. In fact, they generally have an even greater climate impact than CO₂ (on a mass basis). Here, the term GHC will be used when referring to both greenhouse gases (GHGs) and particles.

Even though the emissions from a household stove would seem too small to have any impact, the several hundred millions of stoves in use in China every day have the potential to contribute significantly to GHC emission inventories. In Asia, the household sector accounts for more than half of the total anthropogenic BC emissions (Streets et al., 2003). This is noteworthy, considering that BC is claimed to be the second or third most important individual anthropogenic warming agent after CO₂ and possibly methane (Hansen and Nazarenko, 2004; Jacobsen, 2000). Still, not much attention has been given to possible climate impacts of the household sector.

1.2 Health effects

Greenhouse compound emissions will ultimately affect public health through global warming in the long term. However, the immediate health concern is that the fuel consumption in rural homes leads to high levels of indoor air pollution (IAP) that frequently exceed the Chinese standard of $150 \mu\text{g PM}_{10}/\text{m}^3$ (Edwards et al., 2007). Chronic exposure to IAP can cause severe health effects. There is strong evidence for an association between IAP from household combustion and acute lower respiratory infections, chronic obstructive pulmonary disease and lung cancer (Smith et al., 2004). Estimates from the World Health Organization (WHO) indicate that indoor air pollution from solid fuel use is responsible for approximately 420,000 premature deaths annually in China alone (Zhang and Smith, 2005). This is based on binary calculations that classify the population as either exposed or not exposed according to fuel use. Calculations by using the actual IAP concentrations to which the population is exposed have resulted in even higher estimates of between 800,000 and 3.5 million premature deaths per year (Mestl et al., 2007b).

1.3 The present study

It is clear that IAP is an important problem when it comes to health impacts, but less is known about the GHC emissions from the household sector in China. The aim of the present study is to add to the knowledge of global warming contribution from rural fuel use by first establishing an emission metric to be able to compare the climate impact of different greenhouse compounds (Chapter 2), and then applying the method to evaluate the global warming commitments (GWCs) from some common household fuels in China (Chapter 3). The most important factors that influence the GWC results are identified. If the fuels with the largest climate impact are the same that cause highest population exposure to harmful pollutants, co-benefits may arise from switching to cleaner fuels. This is investigated further in the remaining chapters. An overview of population exposure to health damaging particles stratified by main fuels are presented in Chapter 4, and finally both the GWC and the population exposure are compared to determine the overall effects of fuel choice (Chapter 5).

2.0 EMISSION METRIC FOR COMPARING GREENHOUSE COMPOUNDS

The substances that are emitted from household burning of solid fuels have quite variable characteristics and lifetimes in the atmosphere. Here, lifetime refers to the time required to turn over the global atmospheric burden (defined as the burden (Tg) divided by the mean global sink (Tg/yr) for a gas in steady state (IPCC, 2001)). To be able to compare the contribution from different greenhouse compounds, it is necessary to use a metric to place the emissions on a common scale in terms of their climate impact. However, creating such a metric is not a simple task; it requires simplifications of complex chemical and physical atmospheric and climatic processes, as well as value-driven decisions, for example what timescales should be considered. Some economists argue that emission metrics also need to account for the economic dimensions of the problem they are intended to address (IPCC 2007).

The Intergovernmental Panel on Climate Change (IPCC) has used two simple and purely physical metrics; radiative forcing (RF) and global warming potential (GWP). RF represents the change in radiative budget of the surface-troposphere system, for example due to a change in greenhouse gas concentration. A more precise description is given by the IPCC (2001, 2007) where RF is defined as “the change in net (down minus up) irradiance (solar plus long wave, in W/m^2) at the tropopause after allowing for stratospheric temperatures to re-adjust to radiative equilibrium, but with surface and tropospheric temperatures and state held fixed at the unperturbed value.” RF does not take into account the different lifetimes of the forcing agents, and thus cannot be used to assess the potential climate change associated with emissions. For this, global warming potential is used. GWP relies heavily on the RF concept and compare the integrated RF over a specified period (usually 100 or 20 years) from a unit mass pulse emission relative to CO_2 (IPCC, 2007). (See Section 2.1 for a more thorough definition of GWP)

GWP was developed by the IPCC and adapted for use in the Kyoto protocol. The Kyoto protocol only included six groups of substances; CO_2 , CH_4 , N_2O , sulfur hexafluoride (SF_6) hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs). As will be seen in Chapter 3, this may pose a problem when it comes to portraying a complete picture of the climate impact from sectors that emit many different compounds. The GWP metric is criticized for not taking the climate sensitivity of different compounds into account, and being problematic for

short lived gases or aerosols that are not well mixed in the atmosphere. Alternatives to the GWP index include the global temperature potential, that compares the global mean temperature change by different compounds at the end of a given time horizon, and revised GWP formulations that also include the efficacy of a forcing agent (Fuglestedt et al., 2003). However, since GWP is the dominant metric employed in the international climate regime, and the one in use in the Kyoto protocol it is this approach that will be used in the present study.

2.1 Global warming potential (GWP)

The GWP is defined as the time integrated global mean radiative forcing (RF) of a pulse emission of 1 kg of compound i relative to that of 1 kg of the reference gas CO_2 (IPCC, 2007):

$$\text{GWP}_i = \frac{\int_0^{TH} RF_i(t) dt}{\int_0^{TH} RF_r(t) dt} = \frac{\int_0^{TH} a_i [C_i(t)] dt}{\int_0^{TH} a_r [C_r(t)] dt}$$

TH is the time horizon, RF_i is the global mean radiative forcing of component i , a_i is the RF per unit mass increase in atmospheric abundance of component i (radiative efficiency), $[C_i(t)]$ is the time dependent abundance of i , and RF_r , a_r and $C_r(t)$ are the corresponding values for the reference gas CO_2 . GWP_i is thus strongly influenced by the time horizon chosen if the compounds have different lifetimes in the atmosphere.

The numerator and denominator are called the absolute global warming potential (AGWP) of i and the reference gas r , respectively. The AGWP values for CO_2 for 20 and 100 year time horizons are $2.47 * 10^{-14}$ and $8.69 * 10^{-14} \text{ W m}^{-2} \text{ yr (kg CO}_2\text{)}^{-1}$ (IPCC 2007). For short lived compounds, the AGWP will not change significantly between 20 and 100 year time horizon, see Figure 1 for illustration. This fact can be used to convert GWP_i between 100 and 20 year time horizons for short lived compounds by multiplying GWP_i with the corresponding AGWP ratio for CO_2 , i.e. $\text{GWP}_{i, 20} = \text{GWP}_{i, 100} * 3.52$ (where $3.52 = 8.69 * 10^{-14} \text{ W m}^{-2} \text{ yr (kg CO}_2\text{)}^{-1} / 2.47 * 10^{-14} \text{ W m}^{-2} \text{ yr (kg CO}_2\text{)}^{-1}$). If the lifetime is longer than approximately 4 years, the AGWP value will start to differ between 20 and 100 year time horizon. A lifetime

of 5 years corresponds to about 2% lower AGWP value for 20 vs. 100 year time horizon, while a lifetime of 10 years corresponds to a difference of around 14% (see Appendix A for calculations).

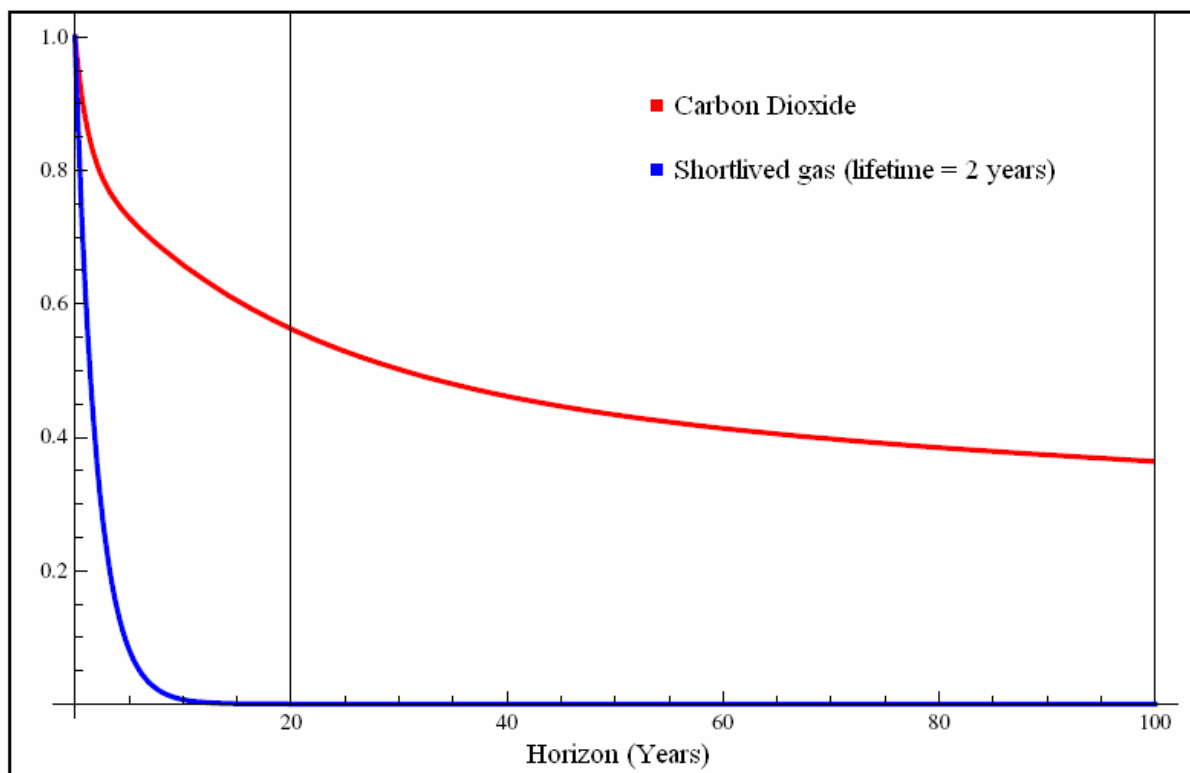


Figure 1: This graph illustrates the decay of a short lived gas followed by a pulse emission, compared to the decay of CO₂. Recall that AGWP is the time integrated value of concentration times radiative efficiency. For a short lived compound, it is evident that the AGWP value will not differ between a time horizon of 20 and 100 years (because the area under the blue curve does not change). CO₂ on the other hand, has a long lifetime and will naturally have a larger AGWP value for a longer time horizon.

In the following, the GWPs for all relevant substances will be presented, for both 20 and 100 year time horizon. Data are as up to date as possible and location dependent values are Asia or China specific where feasible. For the well mixed gases (e.g. CH₄ and N₂O) the GWPs have been taken directly from the IPCC (2007), but for the short lived species there is less consensus and the GWP values have been calculated using relevant papers. Overall, there is a large degree of uncertainty associated with estimation of GWPs. The GWP of a GHG depends on parameters that are difficult to measure, such as atmospheric lifetime, as well as assumptions about future atmospheric conditions such as the concentration of other gases and free radicals. This may again affect the lifetime or the RF of the GHG, or the reference gas CO₂. As a result, a guesstimate of the uncertainty range has been given based on literature

review and an overall evaluation. To simplify, a normal distribution was assumed even though it can be argued that this may not always be the case. For instance, a lognormal distribution or another skewed distribution seems more likely where the uncertainty is large. IPCC (2007) reported uncertainties for RF as 90% confidence intervals, and the distributions were normal for all the effects listed here, except for tropospheric ozone.

2.1.1 Carbon dioxide (CO₂)

The GWP for CO₂ is 1 by definition.

2.1.2 Methane (CH₄)

Methane emissions have both direct and indirect radiative impacts on climate. Four indirect effects of methane emissions have been identified: The enhancement of its own lifetime through changes in the OH concentration, enhancement of tropospheric ozone, enhancement of stratospheric water vapor levels and production of CO₂. The largest forcing, about two thirds, is due to methane itself, while ozone, stratospheric H₂O and production of CO₂ contributes 23%, 8% and 2%, respectively (IPCC 2007).

IPCC's fourth assessment report recommends $GWP_{100} = 25$ and $GWP_{20} = 72$, and these estimates are not subject to wide uncertainty ranges compared to most of the compounds mentioned below. CH₄ has a lifetime of about 12 years (IPCC 2007), and is thus well mixed in the atmosphere. The uncertainty in RF for CH₄ emissions was estimated to be $\pm 20\%$ by Shindell et al. (2005). However this does not include the uncertainty in RF for CO₂ (as GWP is a ratio). "Direct GWPs are usually believed to be known reasonably accurately ($\pm 35\%$)" according to IPCC (2001). In the further assessment a standard deviation of 15% has been used for methane, which corresponds to a 95% confidence interval that the true value lays within a range of $\pm 30\%$.

2.1.3 Carbon monoxide (CO)

CO has an insignificant direct GWP, but indirect radiative effects arise through enhancement of ozone and reduced OH levels, that in turn lead to enhanced concentrations of CH₄. IPCC

2007 reports an averaged value of 1.9 for GWP_{100} . Berntsen et al. (2005) have calculated GWPs that are Asia specific, and thus these estimates have been used in this study. The GWP_{100} was 2.4, and because of the short lifetime of about 1-3 months (IPCC 2001), GWP_{20} was calculated by multiplying with 3.52 as described in the previous section. This gives a GWP_{20} of 8.4. By doing the calculations analogous to NO_x , see below, the result is 7.4. These values are in good agreement with GWP estimates from the literature; Fuglestvedt et al. (1996) reports $GWP_{20} = 10$ and $GWP_{100} = 3.0$, while Johnson and Derwent (1996) have $GWP_{100} = 2.1$. In the following, $GWP_{20} = 8$ has been used with a standard deviation of 30%, the same as for NMHC (see below).

2.1.4 Non-methane hydrocarbons (NMHC)

Non-methane hydrocarbons contribute indirectly to global warming by enhancement of tropospheric ozone, reduction of hydroxyl and peroxy radicals which leads to build up of methane, and production of CO_2 . Collins et al. (2002) have used a global three-dimensional Lagrangian chemistry transport model to calculate GWPs for 10 different NMVOC. IPCC (2007) uses a weighted average of these resulting in a GWP of 3.4 for the 100-year estimate, without specifying further how it was weighted. Six of these compounds are more relevant for biomass and coal burning (namely ethane, propane, butane, ethylene, propylene and toluene (IPCC, 2001)), and so the average of these six compounds has been used here, which gives $GWP_{100} = 3.5$. The emissions of NMHC are frequently given on a per gram carbon basis. The GWP (total mass) can be adjusted to give GWP (mass carbon) by dividing by the carbon fraction. This gives $GWP_{100}(C) = 4.2$.

NMHCs have lifetimes in the range of fractions of a day to months (IPCC 2001), so GWP_{20} was calculated by multiplying GWP_{100} with 3.52, which gives $GWP_{NMHC\ 20} = 12$ and $GWP_{NMHC-C\ 20} = 15$.

It was hard to find other estimates in the literature with which to compare. Due to the short lifetimes of NMHCs and the nonlinearity involved in ozone and OH chemistry, Collins et al. (2002) estimated an uncertainty range of -50% to +100%. They did not specify further how it was calculated, but it seems they assumed a lognormal distribution. Here, a standard deviation of 30% has been used, which corresponds to 95% confidence that the true value lies within $\pm 60\%$ from the reported GWP.

2.1.5 Nitrogen oxides (N₂O, NO_x)

Nitrous oxide (N₂O) has a long atmospheric lifetime (114 years) and is well mixed in the atmosphere. It has a direct radiative impact on climate, and a GWP₁₀₀ value of 298 and GWP₂₀ value of 289 was reported by IPCC (2007). N₂O will not be included in calculations in the present study, so no uncertainty estimate is given. However, the effects are reasonably well known.

For NO_x (NO and NO₂) however, the situation is more complex. NO_x has a short lifetime in the atmosphere and a complex nonlinear chemistry. There are two opposing indirect effects through ozone enhancement and methane reduction, which make calculations of GWP for NO_x emissions highly uncertain. Furthermore, the effect of nitrate aerosol formation can be significant, but this has not yet been included in model calculations. Due to the nonlinear chemistry, the net RF of NO_x emissions depends strongly on both the location and timing (daily, seasonal) of the emissions (IPCC, 2007). As a result, a central estimate for GWP_{NO_x} was not presented by IPCC (2007).

Berntsen et al. (2005) have estimated GWP for surface emissions of NO_x in Asia, and by using four different models, they calculated GWP₁₀₀ values of 9.1, 5.5, 7.4 and 3.0. The average is 6.3, and the rounded off value of 6 has been used in further calculations for a 100 year time horizon.

For the 20 year time horizon, it was necessary to convert the GWP based on sustained (step) emissions published in Berntsen et al. (2005) to pulse based emissions. Four different estimates of sustained GWP (SGWP) were presented in the article for the Asia region. A conversion of SGWP to pulse based GWP was done by considering the effect on ozone and methane from NO_x emissions separately. As illustrated in Figure 2, the GWP/SGWP ratio depends on the lifetime of the gas in question as well as the time horizon. Methane has a lifetime of approximately 10 years, while ozone has a lifetime of less than one year. GWP/SGWP ratios of 0.58 and 0.85 were therefore used for ozone and methane respectively.

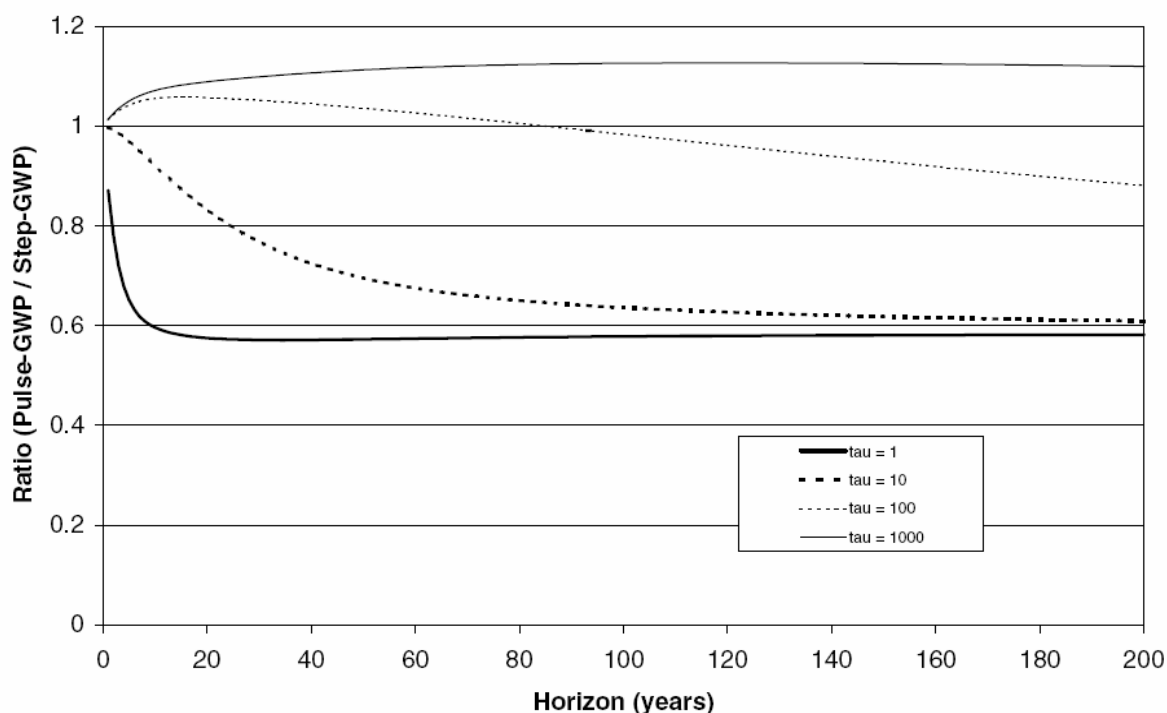


Figure 2: Ratio between pulse-based and step-based GWPs as function of time horizon for four selected lifetimes, namely 1, 10, 100 and 1000 (τ). [Source: Berntsen et al., 2005].

The SGWP values for ozone and methane was obtained by personal communication from Jan Fuglestvedt (Cicero), as only the total values were published in Berntsen et al. (2005). The calculations are reproduced in the Table 1, and a rounded-off GWP₂₀ value of 80 has been used for NO_x in the further assessment.

Table 1: Conversion of sustained GWP to pulse based GWP for NO_x with 20 years time horizon.

Model	SGWP			GWP		
	CH ₄	O ₃	TOTAL	CH ₄ (*0.85)	O ₃ (*0.58)	TOTAL
UiO / ECHAM4	-111	286	175	-94	166	72
UiO / UREAD	-111	275	164	-94	160	65
LMDzINCA / ECHAM4	-126	356	230	-107	206	99
LMDzINCA / UREAD	-126	338	213	-107	196	89

As mentioned, GWP estimates of NO_x are highly uncertain because there are two opposing climate effects. The literature estimates differ a great deal, also because the effects of NO_x emissions are highly dependent on location, both geographically and by elevation. A standard deviation of 50% has been used in further calculations because of the large uncertainty.

2.1.6 Particles (BC, OC, Sulfate)

Black carbon influences the climate directly by absorbing radiation, which exerts a warming effect and increases the amount of energy retained by the earth-atmosphere system. An opposing effect that is cooling the atmosphere is provided by light-scattering particles like organic carbon and sulfates (Bond et al., 2004). In addition, particles have an indirect impact on climate through enhancement of clouds, which cause a negative forcing. In Aunan et al. (2008), the climate impact of constituents emitted from household fuels in Asia are quantified. The radiative forcings attributable to the household sector are reported for different compounds, including black carbon (BC), organic carbon (OC) and sulfates. It is extremely difficult to simulate the effects on clouds in climate models, so this was not accounted for in the estimates. In the present study, the GWPs for particles have been calculated by using the RF from Aunan et al. (2008). The RF of BC, OC and sulfate were divided by the biomass and fossil fuel household emissions of BC, OC and SO₂ (emission inventory based on Streets et al. (2003), exact values were obtained by personal communication with K. Aunan). The resulting AGWP was divided by the AGWP for CO₂ for 20 and 100 year time horizons in order to obtain the GWP values reported in Table 2. The model that was used to estimate RF takes geographic location of emissions into account, and because biomass and fossil fuel use vary spatially, calculation based on the two different fuel groups gives different GWPs as seen in Table 2:

Table 2: This Table shows the calculation of the GWP values for particles, based on the RF values from Aunan et al. (2008). The emissions for sulfate are given as SO₂.

Calculation of GWP for particles:	Biomass			Fossil fuels		
	BC	OC	Sulfate	BC	OC	Sulfate
RF (mWm ⁻²)	26	-39	-2,4	5,5	-1,3	-5,9
Emission (Gg/yr)	1283	6413	1107 (SO ₂)	365	296	2690 (SO ₂)
Calculated AGWP (Wm ⁻² yr / kg)	2,03E-11	-6,08E-12	-2,17E-12	1,51E-11	-4,40E-12	-2,19E-12
GWP 20	821	-246	-88	610	-178	-89
GWP 100	233	-70	-25	173	-51	-25

The following averaged values have been used in this report for 20 and 100 year time horizons, respectively: GWP_{BC} 700 and 200, GWP_{OC} -200 and -60, GWP_{sulfate-SO₂} -90 and -25.

Other papers have reported substantially higher GWP values for BC, with global GWP_{20} estimates of 2200, and $GWP_{100} = 680$ (Bond and Sun, 2005) or 480 (Reddy and Boucher, 2007). Estimated GWPs by using results from Koch et al. (2007) give the following values for particles for the south east region (roughly China): BC: $GWP_{20} = 1740$, $GWP_{100} = 500$, OC: $GWP_{20} = -82$, $GWP_{100} = -23$, sulfate: $GWP_{20} = -57$ and $GWP_{100} = -16$. By using data for China from Berntsen et al. (2006) instead, the values are: BC: $GWP_{20} = 1230$, $GWP_{100} = 350$, OC: $GWP_{20} = -135$, $GWP_{100} = -38$, sulfate: $GWP_{20} = -69$ and $GWP_{100} = -20$. The GWP values for BC used in the current study are roughly 3 times lower than the global estimates, and approximately 2 times lower than the Chinese estimates. The OC and sulfate estimates used in this study are in better agreement with the literature. However, they seem a bit high compared to the literature values presented here.

The calculated GWP values in Table 2 are based specifically on household emissions. The GWPs for BC seem to be much lower than other estimates, and suggests that it may be necessary to use different GWPs for different sectors for such short lived compounds. The radiative impacts of particles are highly dependent on temporal and spatial distribution of the emissions. BC over a snow covered area will for instance have much larger RF than if it is dispersed over a dark surface like the ocean (Aunan et al., 2008). Household emissions are seasonally variable, with higher emissions in the wintertime, especially in northern China. Emissions from households also depend on location, as use of biomass and coal differ between provinces. This was taken into account in the models that estimate RF.

The estimates of GWP values for particles are highly variable, as was just illustrated. For uncertainty estimation in Fuglestvedt et al. (2008), a normal distribution was assumed for RF modeling uncertainty for particles. A standard deviation of 33, 47 and 44% was used for BC, OC and sulfate, respectively. For the further assessment in the present study, standard deviations of 50% for all the particles have been used.

2.1.7 GWP summary

A summary of all the global warming potentials discussed in this chapter, as well as the standard deviation that will be used in further uncertainty assessment, are given in Table 3:

Table 3: Summary of global warming potentials. For NMHC, NO_x and Sulfate the GWP is given relative to mass of carbon, nitrogen and sulfur dioxide, respectively (denoted in parenthesis).

Compound	GWP 20 years	GWP 100 years	Standard deviation (%)
CO ₂	1	1	0
CH ₄	72	25	15%
CO	8	2.4	30%
TNMHC (C)	15	4.2	30%
NO _x (N)	80	6	50%
N ₂ O	298	289	-
Sulfate (SO ₂)	-90	-25	50%
BC	700	200	50%
OC	-200	-60	50%

2.2 Global warming commitment (GWC)

Use of GWP makes it possible to place the impact of different GHCs on a common scale. Since multiple GHCs are emitted by household fuel burning, an aggregate index called global warming commitment (GWC) is employed to estimate the total impact. The GWC is the sum of the emissions multiplied by the GWP associated with each GHC:

$$GWC = \sum_j GHC_j * GWP_j$$

where GHC_j is the emitted mass of compound j and GWP_j is the global warming potential of that particular GHC (Smith et al., 2000). The GWC is expressed in relation to some denominator of interest, for example, per unit fuel mass, per unit fuel energy or per unit energy delivered to the pot. Because the GWPs depend on the time horizon, the GWC will differ depending on what time horizon is used, and so it is necessary to specify whether GWC is based on GWP_{20} or GWP_{100} .

3.0 GLOBAL WARMING COMMITMENT OF MAIN HOUSEHOLD FUELS

In this chapter, the method described in Chapter 2 will be applied to quantify the GWC from main household fuels by using emission data from Zhang et al. (2001). This is a comprehensive database of emission factors from common fuel and stove combinations in China, and Section 3.1 of this chapter presents the most relevant information from this database. The emission data was combined with the GWPs to calculate global warming commitments, and the results are presented in Section 3.2. The most important factors affecting the GWC were also identified. Finally, a discussion of the uncertainties and caveats is included in Section 3.3 to illustrate the level of confidence about these estimates.

3.1 Database of emission factors

The database from Zhang et al. (2000) provides emission factors for greenhouse gases and other airborne pollutants from common fuels and stoves in China. 28 fuel and stove combinations were tested, of which 26 were included in the current study. One of the coal types tested in two stoves was omitted because the coal came from a special coal mine in Shanxi province and was included in the original study because of its high emissions relative to other coal types (Edwards et al. (2003, 2004) also excluded these data). The remaining fuels were selected to represent what is commonly used in China, and all the solid fuels were obtained from a local market. The stove types were those most typical for burning each type of fuel and were the most popular models found in rural households or the market. The improved stoves were similar to the traditional ones in shape and structure, but had a better design to improve stove thermal efficiencies. All the fuel and stove combinations that have been included in the present study are summarized in Table 4:

Table 4: Description of the fuel and stove combinations, sorted by fuel category. The abbreviated name will be used in the following illustrations. v = vented, i.e. with flue. Based on Zhang et al. (2000).

Nr	Fuel category	Name	Fuel Description	Stove Description
1	Fuel wood	Wood-Brick -v	Fuel wood (Harvested from mature trees or their big branches)	Brick stove with a flue
2		Wood-India	— “ —	Metal stove from India without flue
3		Wood-Imp -v	— “ —	Improved brick stove with a flue
4	Brush wood	Brush-Brick -v	Brush wood (Thin branches of brushes which normally grow faster than trees)	Same as #1
5		Brush-India	— “ —	Same as #2
6		Brush-Imp -v	— “ —	Same as #3
7	Crop Residue	Wheat-Brick -v	Wheat residue (Dried wheat stalks)	Same as #1
8		Wheat-Imp -v	— “ —	Same as #3
9		Maize-Brick -v	Maize residue (Dried corn stalks)	Same as #1
10		Maize-Imp -v	— “ —	same as #3
11	Coal Briquette	CoalBriq-Metal	Coal briquette (Made of coal powder and clay)	Metal coal stove without flue
12		CoalBriq-Metal -v	— “ —	Metal coal stove with a flue
13		Honey-Metal	Honeycomb coal briquette (Made of coal powder and clay, and possess a honeycomb-like shape)	Same as #11
14		Honey-Metal -v	— “ —	Same as #12
15		Honey-Imp	— “ —	Improved metal coal stove without flue
16	Coal	Coal-Brick -v	Coal (Unprocessed coal powder)	Same as #1
17		Coal-Metal	— “ —	Same as #11
18		Coal-Metal -v	— “ —	Same as #12
19		WashCoal-Metal -v	Washed coal (Coal powder soaked in water for 24 hours and subsequently sun dried)	Same as #12
20	Kerosene	Kero-Press	Kerosene (A middle distillate from petroleum refining. It is in liquid form under normal atmospheric conditions)	Kerosene pressure stove without flue
21		Kero-Wick	— “ —	Kerosene wick stove without flue
22	Gas	LPG-Trad	Liquefied petroleum gas (From petroleum refining. LPG is available in bottles, and is in gaseous form under normal atmospheric conditions.)	Traditional LPG stove without flue
23		LPG-IR	— “ —	LPG stove with an infrared head without flue
24		NatGas-Trad	Natural gas (From a natural gas field about 1000 km northeast of Beijing.)	Traditional gas stove without flue
25		NatGas-IR	— “ —	Gas stove with an infrared head without flue
26		CoalGas-Trad	Coal gas (Synthesized from coal gasification and distributed locally to households through pipelines.)	Same as #24

All fuel/stove combinations were tested in a simulated village kitchen house, except those stoves using piped gas fuels, which were tested in actual homes. A water boiling test (WBT) was used to estimate emissions relative to energy use. The WBT is a standard method for comparing the efficiencies of different stoves, by imitating common cooking procedure in which a specified quantity of water is used to simulate food. Three successful tests with complete burn cycles were conducted for each fuel/stove combination, and airborne combustion products were collected using a stainless steel sampling probe connected to a filter holder, a pump and then a gas-collection bag. For stoves with flues, the sampling probe was inserted into the flue. Stoves without flues were placed under a hood built for the test purpose, and the sampling probe was inserted into an exhaust vent for the hood. A full description of methods and quality controls can be found in Zhang et al. (2000).

The database reports emission factors for CO_2 , CH_4 , CO , TNMHC_C (total non-methane hydrocarbons given as gram of carbon), NO_x , SO_2 and total suspended particles (TSP). Measurements of N_2O were not included, but even though N_2O is a powerful GHG with climate impact almost 300 times that of CO_2 on a mass basis, the emissions are likely so low that the total GWC impact from N_2O is negligible. A similar analysis for India showed that N_2O on average contributed less than 1% of the total global warming commitment (Edwards et al., 2004; Smith et al., 2000). Particles on the other hand, are likely to play an important role. Results from Aunan et al. (2008) demonstrate that RF from particles from household combustion can be several times larger than the other greenhouse gases. On the other hand there are both absorbing and scattering particles, hence the net effect on climate may not be very large (Aunan et al., 2008). Unfortunately, only total suspended particles were reported in Zhang et al. (2000), and not the individual BC and OC emissions. Unlike industrial burning, particle emissions from residential burning emit mostly carbonaceous material (Bond et al., 2004; Streets et al., 2001). Other constituents could include ionic compounds and trace metals, but at least for wood smoke these are generally less than 1%. The carbonaceous aerosols consist of organic matter (OM) and BC, while OM consist of OC and associated elements (Roden et al., 2006). BC and OC thus make up the total carbon fraction of the particles. In Zhang et al. (2000), the molar emission ratio of TSP_C to CO_2 was given, and so it was possible to calculate the total carbon fraction of the particles by multiplying the molar emission ratio with the emission factor for CO_2 and correcting for the mass difference ($\text{C} = 12 \text{ g/mol}$, $\text{CO}_2 = 44\text{g/mol}$). (See Section 3.3.4 for a further discussion of the accuracy of this

estimate). In the further calculations, it was assumed that the TSP_C had a OC:BC ratio of 5:1 for biomass fuels, and 1:1 for the fossil fuels. This is based on emission factors for China in Streets et al., (2003). For comparison, global emission factors had a OC:BC ratio of 0.9 for residential coal, 4 for residential wood and 3.9 for crop residues (Bond et al, 2004), while estimates for residential biofuels in Asia gave a somewhat higher value of 4.7 (Venkataraman et al., 2005). In view of this, the OC:BC ratio employed here seems reasonable.

The emission factors in the database are reported either on a fuel mass basis, or on a delivered energy basis. Because different fuels have different energy content, it may be misleading to compare fuels on a mass basis, as high energy fuels such as gases will have high relative emissions per kg fuel burned. Thus, for this work, the latter approach was used because the emission per unit energy (MJ) delivered to the pot is more relevant in a comparison of different alternatives. Table 5 summarizes the emission factors for all relevant compounds on a delivered energy basis:

Table 5: Emission factors as gram of compound per MJ energy delivered to the pot. See Table 4 for further description of the fuel/stove combinations. The color coding corresponds to the 7 different fuel categories in Table 4; light green = fuel wood, green = brush wood, light brown = crop residue, light blue = coal briquette, blue = coal, orange = kerosene and yellow = gas. Key: c.v. = coefficient of variation (standard deviation / mean), nd = not detected, na = not applicable, (i) = instant emissions, (u) = ultimate emissions. Based on Zhang et al. (2000).

Fuel/ Stove	CO ₂		CH ₄		CO		TNMHC _C		NO _x		SO ₂		TSP		TSP (C)	
	Mean	c.v.	Mean	c.v.	Mean	c.v.	Mean	c.v.	Mean	c.v.	Mean	c.v.	Mean	c.v.	Mean	c.v.
Wood (i)- Brick -v	750	0.20	0.294	0.43	11.100	0.1	0.075	1.01	0.219	0.91	nd	na	0.759	0.36	0.595	0.25
Wood (i)- India	458	0.59	0.663	0.38	14.600	0.6	0.531	0.5	0.266	0.64	0.0005	1.01	0.971	0.84	0.923	0.77
Wood (i)- Imp -v	388	0.08	0.831	0.73	17.200	0.58	1.330	0.94	0.136	0.71	0.0075	0.92	1.080	0.47	1.036	0.44
Wood (u)- Brick -v	779	0.22	1.250	0.72	14.900	0.29	0.903	0.88	0.227	0.92	nd	na	1.090	0.49	0.618	0.27
Wood (u)- India	502	0.47	2.190	1.08	21.500	0.54	1.880	1.2	0.280	0.54	0.0007	1.15	1.460	0.67	1.012	0.69
Wood (u)- Imp -v	393	0.07	1.000	0.7	18.000	0.57	1.490	0.87	0.137	0.7	0.0076	0.93	1.140	0.45	1.049	0.44
Brush (i)- Brick -v	714	0.08	1.010	0.59	32.200	0.34	0.399	0.98	0.925	0.21	0.0024	1.39	1.280	0.24	1.011	0.20
Brush (i)- India	545	0.01	1.950	0.19	36.600	0.16	1.700	0.17	0.606	0.07	0.0009	1.73	1.600	0.13	1.445	0.13
Brush (i)- Imp -v	636	na	3.350	na	44.400	na	1.070	na	0.655	na	nd	na	2.080	na	1.490	na
Brush (u)- Brick -v	719	0.08	1.190	0.6	33.000	0.35	0.544	0.91	0.932	0.21	0.0024	1.39	1.340	0.26	1.018	0.20
Brush (u)- India	552	0.01	2.180	0.21	37.800	0.15	1.900	0.22	0.613	0.06	0.0009	1.73	1.670	0.13	1.463	0.13
Brush (u)- Imp -v	647	na	3.770	na	46.400	na	1.400	na	0.666	na	nd	na	2.250	na	1.516	na

Wheat-Brick -v	976	0.07	2.010	0.52	46.300	0.4	2.700	0.66	0.810	0.7	0.0223	0.48	3.650	0.66	1.661	0.59
Wheat-Imp -v	447	0.22	4.190	0.94	83.900	0.29	4.560	1.08	0.110	0.64	0.0388	1.73	8.480	0.62	5.193	0.72
Maize-Brick -v	666	0.16	0.983	0.71	24.600	0.67	1.860	0.99	0.703	0.21	0.0080	0.57	1.020	0.66	0.245	0.52
Maize-Imp -v	348	0.09	2.020	0.65	28.900	0.39	0.846	0.77	0.202	0.5	0.0649	0.93	1.340	0.43	0.636	0.41
CoalBriq-Metal	312	0.15	0.004	0.91	3.890	0.74	0.018	1.69	0.191	0.56	0.0233	0.67	0.006	0.26	0.001	0.23
CoalBriq-Metal -v	415	0.48	0.003	0.84	5.400	0.42	0.002	1.41	0.075	0.61	0.5120	0.78	0.049	0.35	0.008	0.50
Honey-Metal	573	0.18	0.004	1.73	14.600	0.32	0.006	1.46	0.097	0.11	0.0259	0.99	0.062	0.35	0.005	0.25
Honey-Metal -v	824	0.11	0.003	0.23	19.000	0.21	0.000	1.73	0.142	0.31	0.0602	0.55	0.070	0.48	0.006	0.49
Honey-Imp	303	0.06	nd	na	6.520	0.58	0.001	0.94	0.049	0.55	0.0092	1.69	0.053	1.16	0.005	1.19
Coal-Brick -v	542	0.55	0.267	1.14	44.100	0.63	0.145	0.83	0.503	0.35	0.0559	1.23	0.430	0.78	0.210	0.60
Coal-Metal	643	na	2.650	na	18.100	na	0.619	na	0.039	na	0.0381	na	2.240	na	1.964	na
Coal-Metal -v	742	0.49	1.080	1.5	25.600	0.66	0.378	1.44	0.119	0.97	0.1160	1.14	1.250	1.47	1.054	1.27
WashCoal-Metal -v	855	0.09	5.190	0.37	32.200	0.2	0.686	0.74	0.063	1.19	0.3730	1.49	5.160	0.33	4.104	0.29
Kero-Press	158	0.07	0.001	1.27	0.378	0.49	0.021	0.82	0.078	0.03	0.0006	0.87	0.009	0.48	0.002	0.50
Kero-Wick	162	0.08	0.002	0.58	0.446	0.17	0.015	0.33	0.032	0.59	0.0017	0.87	0.004	0.39	0.001	0.34
LPG-Trad	140	0.05	0.023	1.73	0.100	1.41	0.151	0.99	0.148	0.12	nd	na	0.025	1.72	0.004	1.72
LPG-IR	153	0.23	0.016	0.96	1.030	0.84	0.240	1.18	0.004	0.35	0.0000	1.73	0.001	0.45	0.000	0.46
NatGas-Trad	125	0.02	nd	na	0.010	1.73	0.003	1.01	0.105	0.29	0.0001	1.17	0.004	0.76	0.001	0.76
NatGas-IR	110	0.01	0.001	1.73	nd	na	0.005	1.19	0.019	0.45	nd	na	0.006	0.27	0.001	0.26
CoalGas-Trad	93	0.1	nd	na	nd	na	0.000	1.73	0.089	0.03	0.0834	0.33	0.010	0.22	0.002	0.33

The mean and standard deviation were derived from 3 measurements for each of the tested fuel/stove combinations, so when considering the standard deviations, the low number of measurements must be kept in mind. For two of the fuel/stove combinations, namely brush wood in the improved stove and unprocessed coal in the metal stove, only one measurement was done and thus no standard deviation was estimated. Some of the compounds were not detectable, meaning that the concentration was too low to be measured or that the flue gas concentration was lower than the background concentration. In the further calculations it was assumed that “not detectable” equals zero.

Two set of results were reported for wood; instant emissions and ultimate emissions. Instant emissions reflect emissions during a particular test. When burning wood, part of the carbon is often left at the end of the cooking period as charcoal. The char is often saved for use later and consequently produce additional emissions when burned. The ultimate emissions are an

estimate of the total emissions, and even though it was not measured directly by Zhang et al. (2000), it was calculated using other data and included in the database. If char and wood had the same combustion it would be no difference between instant and ultimate emissions, since the emission factors are given per unit delivered energy. However, wood seems to provide more energy with less PIC production per MJ, thus the ultimate emissions are a bit higher as seen in Table 5. To simplify only the instant emission factors have been used in the further calculations (the same approach was used in Edwards et al. (2004)). It should be noted that this may underestimate the GWC for wood when saving char for later use is a common practice. Also, the ultimate emissions are higher for fuel wood in proportion to brush wood. Since fuel wood generally have lower emissions, the difference in GWC between fuel wood and brush wood may have been overestimated by using only instant emissions.

3.2 Global warming commitment results

3.2.1 GWC: dependence on the GHCs considered

The total GWC estimates depend strongly on what GHCs are included in the calculations. The following graph (Figure 3) illustrates this point and demonstrates how much the results change depending on which GHCs are considered. For instance, if one is evaluating a clean development mechanism (CDM) project, one would perhaps only be interested in the GHCs included in the Kyoto agreement. In this case, only the GWC contribution from CO₂ and CH₄ will be counted. This is illustrated as the “Kyoto” scenario in Figure 3. Many different compounds are emitted from household fuel burning, so to present a more complete picture, it is necessary to include a more comprehensive list of GHCs in the calculations. Scenario two in Figure 3 is called “GHG” and represents the GWC when the contribution from all GHGs are included (CO₂, CH₄, CO, NMHCs and NO_x). Scenario three, called “GHC”, also includes the particles (sulfate, black carbon and organic carbon). One might argue for the use of either scenario. On one hand the uncertainties associated with GWPs for particles are high, which makes the standard deviation for the total GWC very large. On the other hand it is important to include all the relevant compounds to give the most comprehensive presentation of total global warming impact.

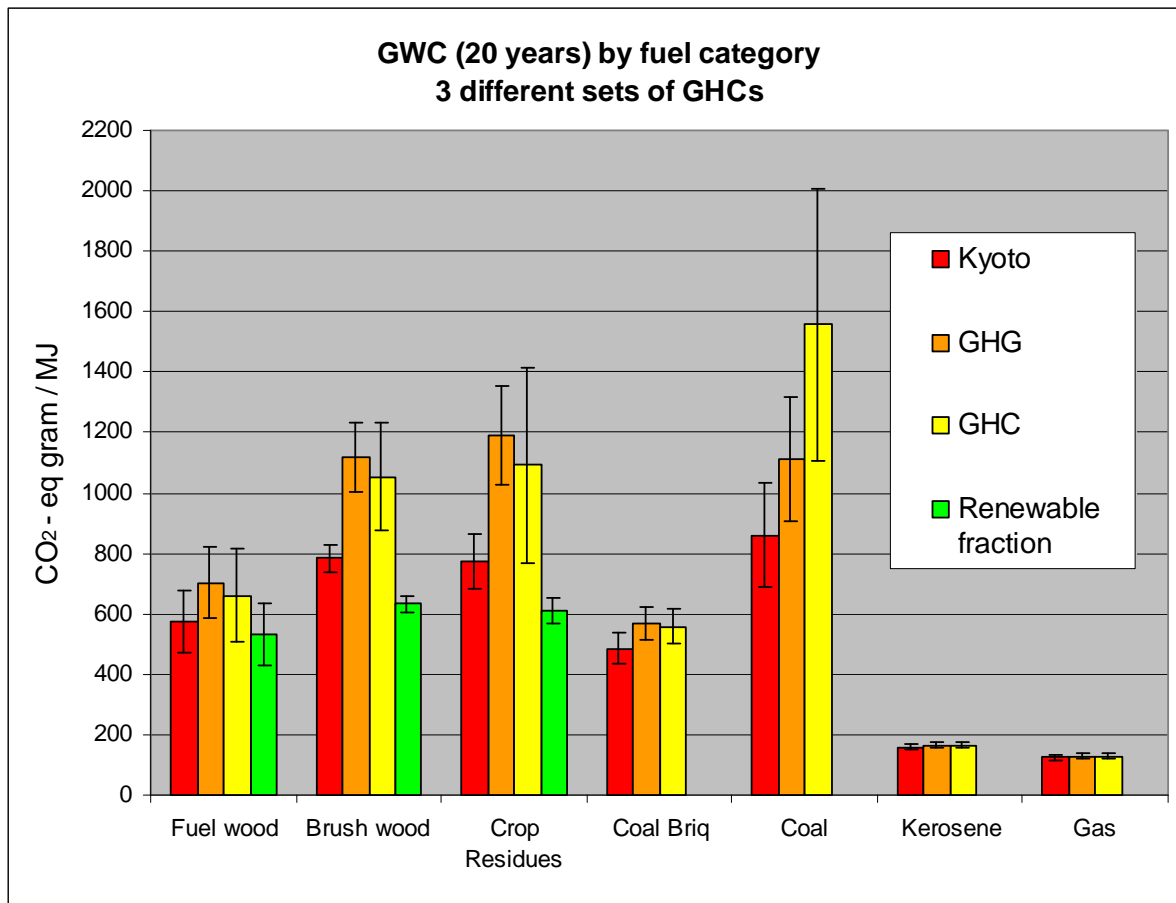


Figure 3: This graph illustrates the average GWC₂₀ (as gram CO₂ equivalents per MJ) from each fuel type, when different sets of GHCs are included. The red column represents the GWC contribution from the Kyoto gases; in this case CO₂ and CH₄. The orange column represents the contribution from the GHGs, while the yellow column also includes the particles. The green column is the CO₂ contribution and equals what should be subtracted if the biomass fuels are harvested renewably. The error bars correspond to one standard deviation, including both measurement variability and uncertainty associated with GWPs.

As evident from Figure 3, it is not irrelevant which set of GHCs that are included in GWC calculations. Not only do the total values differ, but also the ranking of fuels, because combustion of different fuels does not lead to proportional emissions of the different greenhouse compounds. The “Kyoto” scenario gives the lowest GWC, not surprisingly as it only includes two GHGs. Including all the GHGs presents a more complete picture and the GWCs increase. With the exception of fuel wood, the GWC increases more for biomass than for coals for “GHG” compared to the “Kyoto” scenario. This is because brush wood and crop residues on average have higher emissions of other GHGs like CO and NO_x than the coals. The “GHG” scenario present crop residues as the “worst” fuel with respect to climate effects instead of coal, although the differences are not statistically significant. A certain fraction of renewable harvesting will, however, imply a lower net GWC value for the biomass fuels (cf. the green column in Figure 3, see description below). When the particles are also included,

the picture changes. The total GWCs of the biomass fuels are reduced for the “GHC” compared to the “GHG” scenario, because the cooling contribution from OC exceeds the positive effect from BC. The opposite is the case for raw coal, where GWC increases due to high particle emissions. The cooling effect of sulfate is negligible, and because the BC:OC fraction was assumed to be 1:1, the warming contribution dominates. For coal briquettes the particle emissions are minimal, and the net change in GWC for the “GHC” scenario compared to “GHG” is actually slightly negative due to a small cooling contribution from sulfate. This may seem counterintuitive since the sulfur content is lower in coal briquettes than in coal, but because the carbonaceous particle emissions are highly reduced for coal briquettes, the sulfate becomes relatively more important. However, as can be seen from the error bars, the uncertainty is large and no definite conclusions should be drawn on the GWC trend for small differences. The most important finding illustrated by Figure 3 is that the GWC may change significantly depending on what set of GHCs are included in the calculations. One exception is the combustion of kerosene and gas, which emits almost solely CO₂. Thus, in this case, it does not matter much which set is considered.

CO₂ emissions from biomass fuels are portrayed in the fourth column in Figure 3. If the biomass is harvested renewably, the CO₂ emissions will be completely recycled and there will be no net increase in GWC from CO₂. It is very likely that the crop residue is harvested renewably, while it’s hard to tell whether the wood is harvested renewably or not. As a consequence, two values are applicable for biomass; one for renewable and another for non-renewable harvesting. The degree of renewable harvesting will determine the actual GWC between these two extremes. It is clear from Figure 3 that whether the fuel is harvested renewably or not makes a very large difference. Interestingly, the biomass fuels are not carbon neutral even when harvested renewably, because of products of incomplete combustion (PICs) and to a small extent NO_x. Even more than before, the results depend strongly on what set of compounds are considered. Renewable biomass fuels seem particularly favorable if we only look at the “Kyoto” scenario, and are then comparable or even better than kerosene and gas. This gives a rather misleading picture, as in practice this means that only the contribution from methane is counted. In the further calculations, the full set of GHCs will be used.

3.2.2 Stoves, individual GHC contributions and time horizon

Figure 3 presented the total GWC, averaged for each fuel category. There are large variations between different stove types within the same fuel category, which were not apparent in Figure 3. Figure 4 provides the GWC for all fuel and stove combinations, as well as a more detailed breakdown of how large the global warming contributions are for individual GHCs.

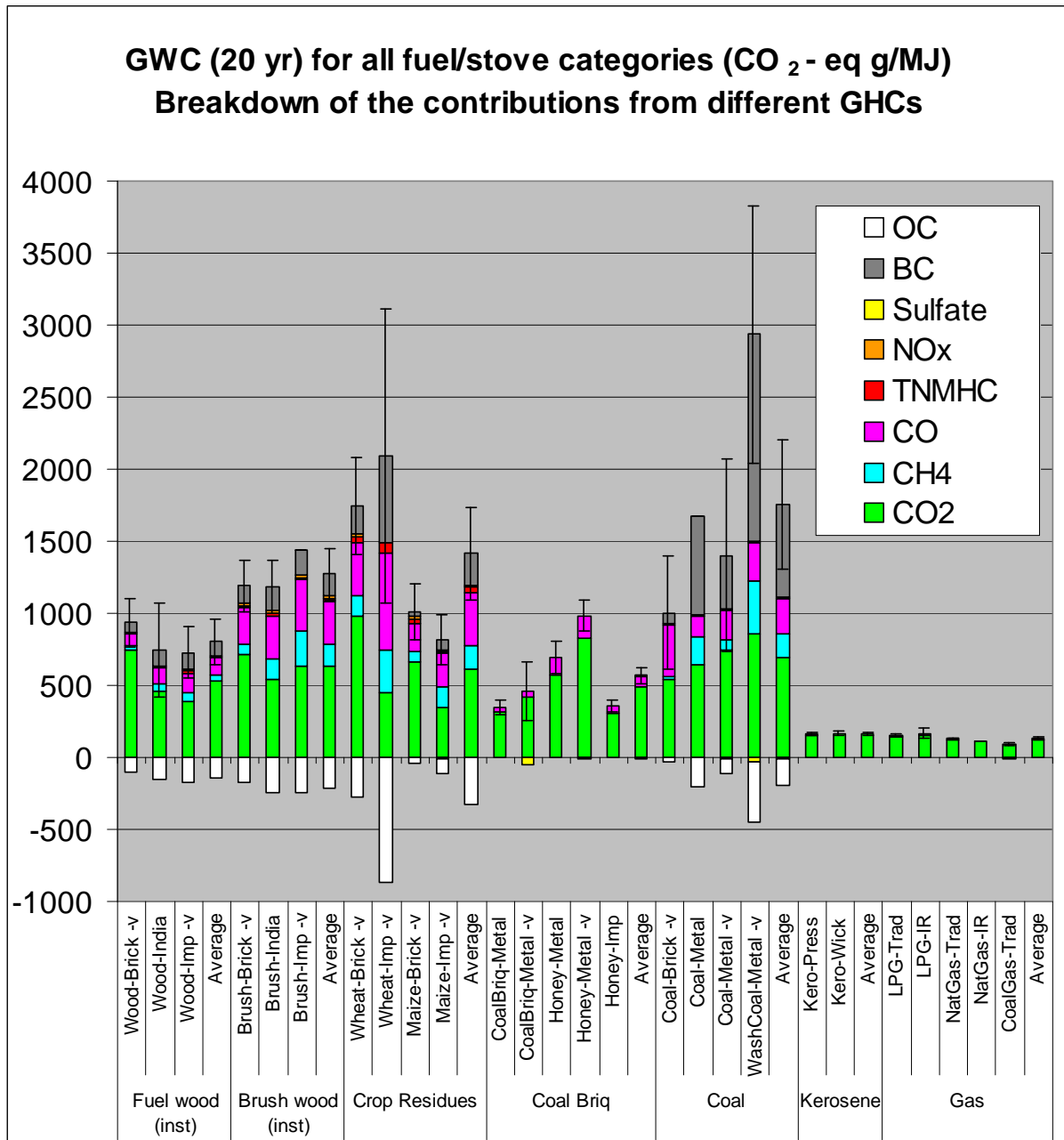


Figure 4: GWC₂₀ for all the 26 fuel and stove combinations, plus the average for each fuel category. See Table 4 for explanation of fuel/stove names. The contributions from the different GHCs in CO₂ equivalents (g/MJ) are shown in different colors; CO₂ is green and represent the renewable fraction for the biomass fuels. The contributions from methane, carbon monoxide, non-methane hydrocarbons, nitrogen oxide and black carbon are shown in blue, pink, red, orange and black, respectively. Sulfate and organic carbon have cooling effects, and are represented in yellow and white. The error bars correspond to one standard deviation for the total GWC.

Figure 4 illustrates the variation between the different stoves, and the relative contribution from the different GHCs. It is not clear which stove is best; the ranking changes for different fuels. Moreover, it also depends on what GHCs are taken into account. For instance, the improved stove has the lowest GHG emissions in the fuel wood category, while the opposite is the case for brush wood. In the crop residue category the improved stove has the lowest total GWC, while the particle emissions (which are also relevant for health effects) are significantly higher than for the brick stove. It appears like the improved biomass stove has the least complete combustion, and hence emits a larger fraction of PICs and less CO₂. It may seem counterintuitive that the improved stove should have poorer combustion, but an explanation might be that even though the improved stoves here have higher overall efficiency (OE), the nominal combustion efficiency (NCE) is lower. Overall stove efficiency is a combination of two internal efficiencies; NCE which indicates the percentage of the fuel carbon converted to CO₂, and heat transfer efficiency (HTE) which refers to the percentage of heat released by combustion that makes its way into the pot (Edwards et al., 2004). If OE is improved by increasing HTE at the expense of NCE, the result is that even though fuel use is reduced, the PIC emissions are greater. China's national improved stove program (NISP) was focused mostly on improving fuel efficiency, and the disseminated stoves were not evaluated for emissions of health damaging pollutants (HDPs) or GHCs (Edwards et al., 2007). In this light it is not surprising that the improved biomass stoves show the highest PIC emissions (see also Section 3.3.4). The improved coal briquette stove, on the other hand, has lower emissions compared to the other stoves. Because of these inconsistent results between stoves, and because of high uncertainty associated with lab measurements of emission factors for different stoves (see next section), it is the average emissions for each fuel category that will be used in Chapter 5.

When it comes to the relative contribution from each GHC, it is evident from Figure 4 that kerosene and gas emit almost solely CO₂. The coal briquettes also have very clean combustion, with some contribution to the total GWC from carbon monoxide. Table 6 sums up the relative contributions from each compound. Washed coal has the highest overall emissions, and poor combustion. This is a bit unexpected, as coal washing is meant to reduce sulfur content as well as remove other impurities from the coal, and theoretically should give more efficient operation and reduced particle emissions (Aunan et al., 2004). A simple household level washing process was used (i.e. coal powder soaked in water and sun dried),

so no inference about industrial washing and emissions should be drawn from these results. The washing process reduced the sulfur content of the coal from 0.85% to 0.35% (Zhang et al., 2000), but still the washed coal had the highest SO₂ emissions per MJ of all the coals. This implies that more washed coal was needed to produce the same amount of energy. One explanation might be that the emissions were higher compared to unprocessed coal due to increased water content (Edwards et al., 2004), as the coal contained more than twice as much water after washing (Zhang et al., 2000). However, there might also be some kind of measurement error, as the SO₂ emissions per fuel mass also seem to be highest for the washed coal compared to unprocessed coal. The overall contribution from sulfate to the total GWC is negligible for all fuel categories (see Table 6). This is somewhat surprising, as sulfate had a central role in the radiative budget for household fossil fuels in Aunan et al. (2008), which was based on the emission inventory by Streets et al. (2003). In Aunan et al. (2008), the magnitude of the cooling effect from sulfate exceeded the positive forcing from BC for fossil fuels, making the net effect from particles negative. The discrepancy between the importance of sulfate in this work and Aunan et al. (2008) can possibly be explained, at least partly, by the low sulfur content in the coal tested in Zhang et al. (2000) compared to other coals (0.26%, 0.17%, 0.85% and 0.35% sulfur in coal briquettes, honeycomb briquettes, unprocessed coal and washed coal, respectively.) For comparison, the national average sulfur content in coal is 1.16% (Vennemo et al., 2006).

Table 6: The relative GWC (20 years) contribution from the individual GHCs for each fuel type. The numbers are given as percentage of the positive GWC contribution, so the positive numbers add up to 100% except for rounding off errors.

Fuel type	CO ₂	CH ₄	CO	NMHC	NO _x	Sulfate	BC	OC
Fuel wood	66	5	14	1	1	0	12	-18
Brush wood	50	12	24	1	1	0	12	-17
Crop Residues	43	12	26	3	1	0	16	-23
Coal Briq	85	0	14	0	0	-2	0	0
Coal	40	9	14	0	0	-1	37	-10
Kerosene	97	0	2	0	1	0	0	0
Gas	95	0	1	1	1	-1	0	0
Average	68	6	14	1	1	-1	11	-10

Table 6 present the percentage contribution from each GHC relative to the total positive GWC, for each fuel category. CO₂ is the largest contributor, with an average of 68%, followed by CO (14%), BC (11%) and CH₄ (6%). Organic carbon has an average cooling contribution of one tenth of the positive GWC value. There are large differences between the

fuel categories, however. NMHC, NO_x and sulfate do not have any significant effects on the total GWC. The relative contributions from the different GHCs changes if the time horizon is 100 years instead of 20, as evident in Figure 5 and Table 7:

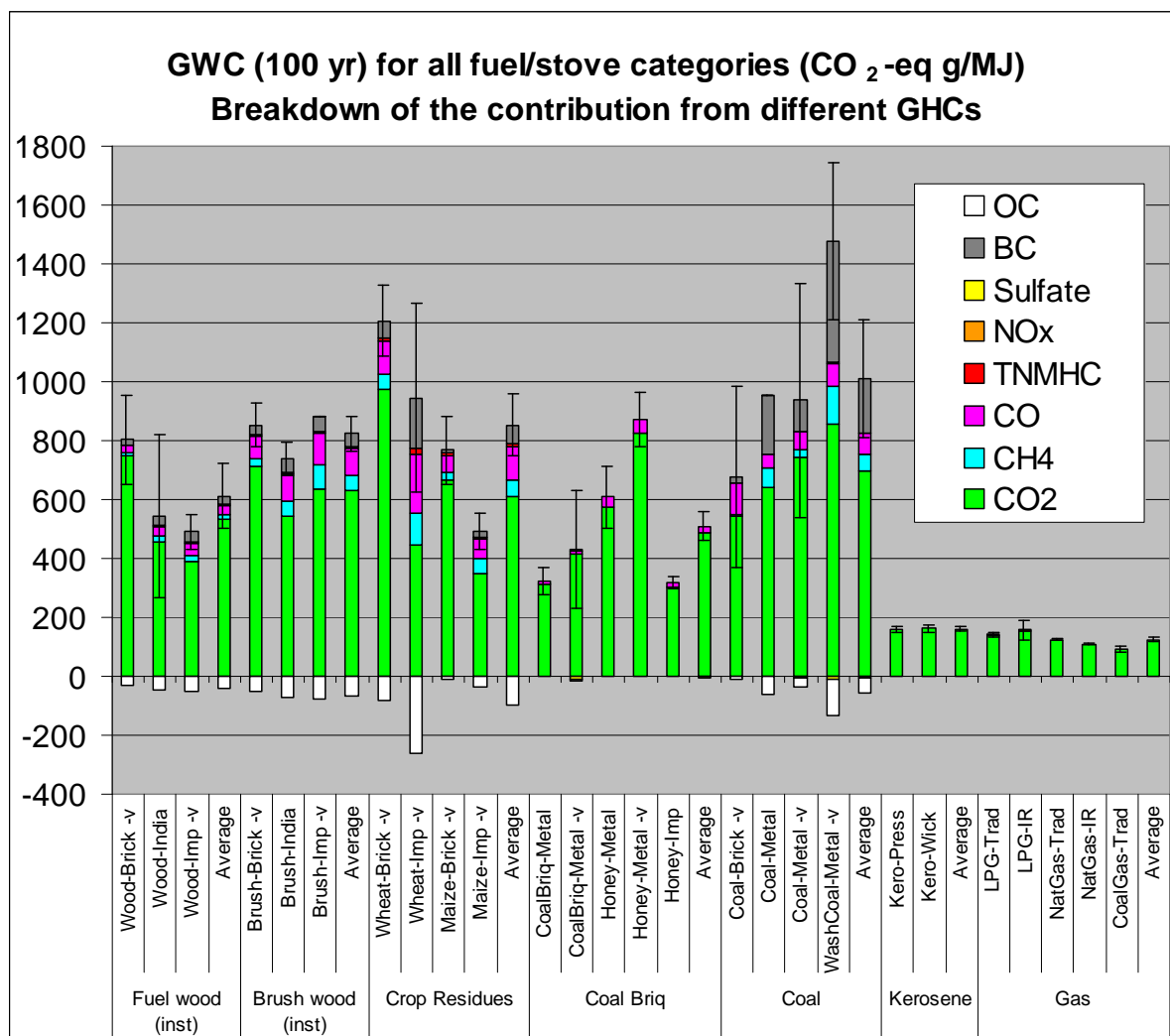


Figure 5: GWC₁₀₀ for all 26 fuel and stove combinations, plus the average for each fuel category. See Table 4 for explanation of fuel/stove names. The contributions from the different GHCs in CO₂ equivalents (g/MJ) are shown in different colors, corresponding to Figure 4. The error bars equals one standard deviation for the total GWC.

Table 7: The relative GWC contribution from the individual GHCs for time horizon of 100 years, for each fuel type. The numbers are given as percentage of the positive GWC contribution, so the positive numbers add up to 100% except for rounding off errors.

Fuel type	CO ₂	CH ₄	CO	NMHC	NO _x	Sulfate	BC	OC
Fuel wood	87	2	6	0	0	0	5	-7
Brush wood	77	6	11	1	0	0	5	-8
Crop Residues	71	7	13	1	0	0	8	-11
Coal Briq	95	0	5	0	0	-1	0	0
Coal	69	6	7	0	0	0	18	-5
Kerosene	99	0	1	0	0	0	0	0
Gas	99	0	0	0	0	0	0	0
Average	85	3	6	0	0	0	5	-5

It is evident that all the other GHCs now play a less important role compared to CO₂. This is as expected, since CO₂ has a much longer lifetime than the other compounds. For a time horizon of hundred years, CO₂ contributes 85% of the GWC on average. CO only contributes an average of 6%, while BC, OC and CH₄ follow with averages of 5%, -5% and 3%, respectively. The total GWC is lower for 100 year time horizon than for 20, and the uncertainty also decreases (see Table 8). This is because the GHCs with high standard deviations for GWP become less important relative to CO₂. The differences are smaller for fuels that have cleaner combustion. For instance, when time horizon changes from 100 to 20 years a minor increase in GWC for kerosene and gas (2-3%) is observed, while coal briquettes and fuel wood increase by approximately 10-15%, the remaining biomass fuels increase by about 40-45% and coal increase by 64%.

Clearly, the choice of time horizon will affect the conclusions, but the choice is not obvious. On one hand the Kyoto protocol has adopted GWPs for a time horizon of 100 years (Fuglestvedt et al., 2003), but on the other hand a long time horizon will diminish the effect of the short lived GHCs, one of the arguments against the GWP concept. The choice of time horizon will ultimately have to reflect the purpose of the study. According to Smith et al. (2000), they “use 20-year GWPs for the analysis because they wish to compare investment options for reducing emissions. A 20-year GWP is roughly equivalent to a discount rate of 4-5%, which is about the lowest that can be justified for social investments.” For comparisons of improvement options for IAP, such as fuel switching, it seems reasonable to use a 20 year time horizon to avoid penalizing near generations on behalf of later ones.

Table 8: Summary of total GWC (gram CO₂ equivalents per MJ) for the full set of GHCs, for both 20 and 100 year time horizon, sorted by fuel type. “Percent increase” indicates how much GWC changes when time horizon is adjusted from 100 to 20 years. The numbers in parenthesis specify the GWC for renewably harvested biomass fuels.

Fuel type	GWC (100 year)	Std dev (%)	GWC (20 year)	Std dev (%)	Percent increase
Fuel wood	570 (38)	19 (91)	661 (129)	24 (91)	16
Brush wood	759 (127)	8 (41)	1053 (421)	17 (42)	39
Crop Residues	755 (146)	14 (66)	1091 (482)	30 (67)	44
Coal Briq	507	10	557	10	10
Coal	952	21	1557	29	64
Kerosene	161	5	165	5	2
Gas	125	6	128	6	3

3.2.3 Global warming impact of the household sector

To get an impression of the magnitude of the global warming impact from household fuel use, the GWC can be used to calculate the total emissions from different fuel categories in CO₂ equivalents. Data for Chinese rural household energy use for 2000 was obtained for wood, crop residues, coal and petroleum products (Jingjing et al., 2001). Only data for total coal was presented, so this was weighed to get the individual coal and coal briquette fractions. In 2000, coal briquettes made up about 5% of rural coal use (NBS, 2002). For fuel wood vs. brush wood, no data were available, so a 1:1 relationship was assumed. GWC for both gas and kerosene were multiplied by the “petroleum product” energy use, so the total emissions (in CO₂ equivalents) from gas and kerosene overall are somewhere in between the two reported values in Table 9 depending on the fuel use ratio.

Table 9: Summary of the total emissions (in CO₂ equivalents) by fuel category. Two estimates are given based on the two time horizons. Also, the total emissions of CO₂ are reported. The numbers in parenthesis represent renewably harvested biomass fuels. The two reported values for petroleum products represent 100% gas use vs. 100% kerosene use. The standard deviations are based on GWC only, as no uncertainty estimate was given for the energy use in Jingjing et al. (2001). Tg = 10¹²g.

Fuel category	Energy use (PJ)	All GHC (100 years)		All GHC (20 years)		Only CO ₂	
		Tg CO ₂ eq	Std dev (%)	Tg CO ₂ eq	Std dev (%)	Tg CO ₂	Std dev (%)
Fuel wood		800 (53)	19 (91)	928 (181)	24 (91)	746	19
Brush wood	2806	1064 (178)	8 (41)	1477 (591)	17 (42)	886	5
Crop Residues	2277	1720 (332)	14 (66)	2484 (1097)	30 (67)	1387	7
Coal Briq		132	10	145	10	127	10
Coal	5218	4718	21	7720	29	3448	23
Gas / Kerosene	228	29 / 37	6 / 5	29 / 38	6 / 5	28 / 36	6 / 5

According to Jingjing et al. (2001), almost half of the energy in rural households comes from coal. As coal had the highest GWC, the total emissions are very high with 4718 million ton CO₂ equivalents per year based on 100 year time horizon, or 7720 million ton based on GWC₂₀. The biomass emissions are lower, and are cut drastically if the harvesting is renewable. Kerosene and gas together emit between about 30 and 40 million ton CO₂ equivalents per year. Most emission data only report CO₂ emissions, hence this is presented in the last columns in Table 9. Crop residues are likely harvested renewably, and if 50% of the wood is also renewable, the total CO₂ emissions from rural households would be about 4420 Tg. For comparison, the total CO₂ emissions for China was about 3820 Tg in 2000 (Streets et al., 2003). Emissions from the rural household sector that exceeds the total

Chinese emissions is obviously an irrationally high estimate. For Asia, the household sector accounted for roughly 25% of the total CO₂ emissions (Streets et al., 2003).

The results in Table 9 must be considered as rough estimates. They are highly uncertain, due to several factors. First of all, the energy use data from Jingjing et al. (2001) may not be accurate. Data from Streets et al. (2003) give quite different estimates for domestic energy use in China for 2000, with 2420 PJ from coal use and 7178 PJ for biofuels. This is more in accordance with the common belief that most of the energy consumption in rural households are covered by biomass. According to Zhang and Smith (2005) about 80% of the energy in rural household in 2003 was from biomass and nearly 10% from coal. If these data are used instead with the same assumptions as above, and crop residues account for 64% of the biomass (NBS, 2002), the CO₂ emissions would be 2440 Tg from the residential sector. This seems more reasonable compared to China's total CO₂ emissions, but it is still a pretty high estimate in terms of share of total emissions.

Secondly, if the use of coal briquettes is more common than the 5% share that was assumed for the calculations, it will reduce the estimates considerably, as coal have approximately twice or three times as high GWC as coal briquettes for 100 and 20 year time horizon. Lastly, the GWCs estimated on the basis of cooking are not necessarily representative for other energy tasks like space heating (for further discussion of this, see Section 3.3.4).

3.3 Uncertainties

3.3.1 Uncertainty estimates

For the estimation of standard deviations, the following formulas have been used:

The standard deviation for sums ($z = x + y + q + \dots$) have been added in quadrature (that is, squared, added and then square rooted):

$$sd_z = \sqrt{(sd_x)^2 + (sd_y)^2 + (sd_q)^2 + \dots}$$

The standard deviation for weighted sums ($z = ax + by + cq + \dots$, where a, b, c etc. are constants) have been added in quadrature (that is, squared, added and then square rooted):

$$sd_z = \sqrt{(a \cdot sd_x)^2 + (b \cdot sd_y)^2 + (c \cdot sd_q)^2 + \dots}$$

The standard deviation for products ($z = x$ times or divided by y times or divided by q etc.) have been found by first estimating the percent standard deviation of z (sd_z/z), by adding the percent standard deviation in all of its components in quadrature:

$$\frac{sd_z}{z} = \sqrt{\left(\frac{sd_x}{x}\right)^2 + \left(\frac{sd_y}{y}\right)^2 + \left(\frac{sd_q}{q}\right)^2 + \dots}$$

These are all special cases of the more general Gaussian error propagation rule:

$$sd_z = \sqrt{\left(\frac{\partial z}{\partial x} sd_x\right)^2 + \left(\frac{\partial z}{\partial y} sd_y\right)^2 + \left(\frac{\partial z}{\partial q} sd_q\right)^2 + \dots}$$

where dz/dx , etc are the partial derivatives of z with respect to its component variables (Kirchner, 2006).

In these formulas x , y , q etc. are uncorrelated with each other. No correlation between the variables has been assumed for all the estimates in this study. However, this is not valid for GWPs, as GWP is expressed as a ratio to the AGWP for CO_2 . As a result, the total variation might have been underestimated for the GWC values. On the other hand, the main interest here is the difference between the fuel groups, and not the total GWC value. Even if the AGWP value for CO_2 changed, the relative difference between the fuels would still be preserved.

The average fuel emissions for each fuel group were weighted by number of stoves only. The standard deviations of each compound for the fuel average were calculated by weighting by stove numbers and adding the standard deviations in quadrature, as described above. For two of the fuel groups, brush wood and coal, one of the stoves had no standard deviations. Here it was assumed that the variation in the measurements for the other stoves in the same fuel group were representative for the variability, and the total standard deviation for the fuel group was calculated on the basis of the remaining stoves. Another option would have been to use the average standard deviation of the other stoves as estimate of the variability for the last stove, and then calculated the variation of the whole group. However, this would have

resulted in a lower estimate of the standard deviation for the whole fuel group, and thus this approach was not used.

3.3.2 Measurement variability

Only 3 measurements were done for each of the fuel/stove combinations, and the variability was large between measurements. Figure 6 below illustrates the measurement variability in each compound for each fuel category. Some of the compounds, especially NMHC and sulfate, have very large relative standard deviations for the measurement variability. However, since they only have minor contributions to the GWC, it does not influence the total uncertainty estimation very much.

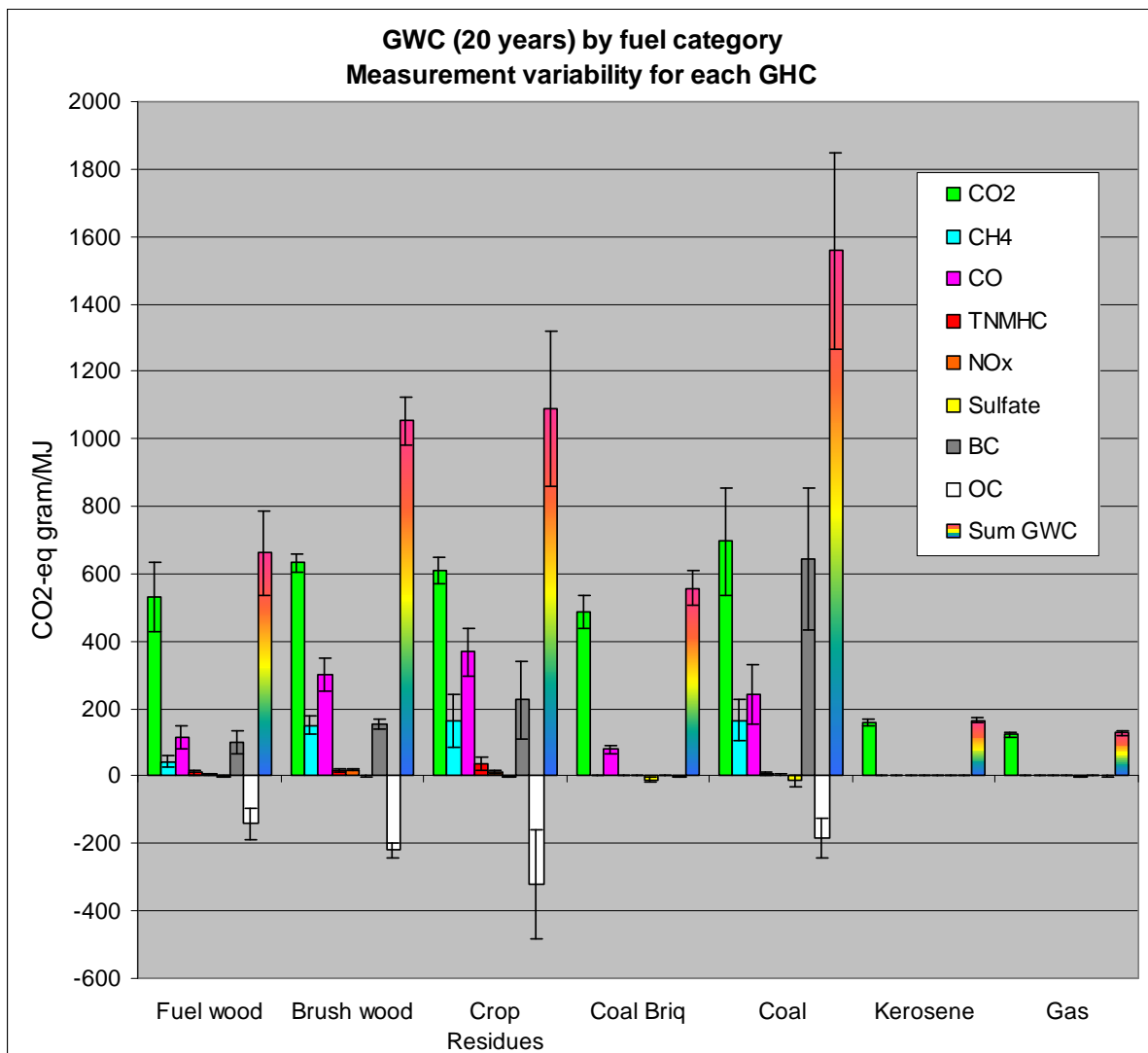


Figure 6: This graph shows the relative contribution of each GHC to the total GWC (gram CO₂ equivalents per MJ) for each fuel category. The error bars represent one standard deviation based on the measurement variability (not including uncertainty associated with GWP). The color coding for the GHCs is analogous to Figure 4 and 5. The total GWC (sum of the positive contribution minus the cooling portion) for each fuel category is shown in rainbow colors.

3.3.3 GWP uncertainty

The large uncertainty associated with the GWP concept contributes to additional uncertainty. In the following graph (Figure 7) the total GWC₂₀ is presented for each fuel category, with only the measurement variability represented for the first column. In the second column, the standard deviation for GWP is also included, while in the third column the sensitivity to changes in GWP is tested by increasing the standard deviation in GWP by 50%.

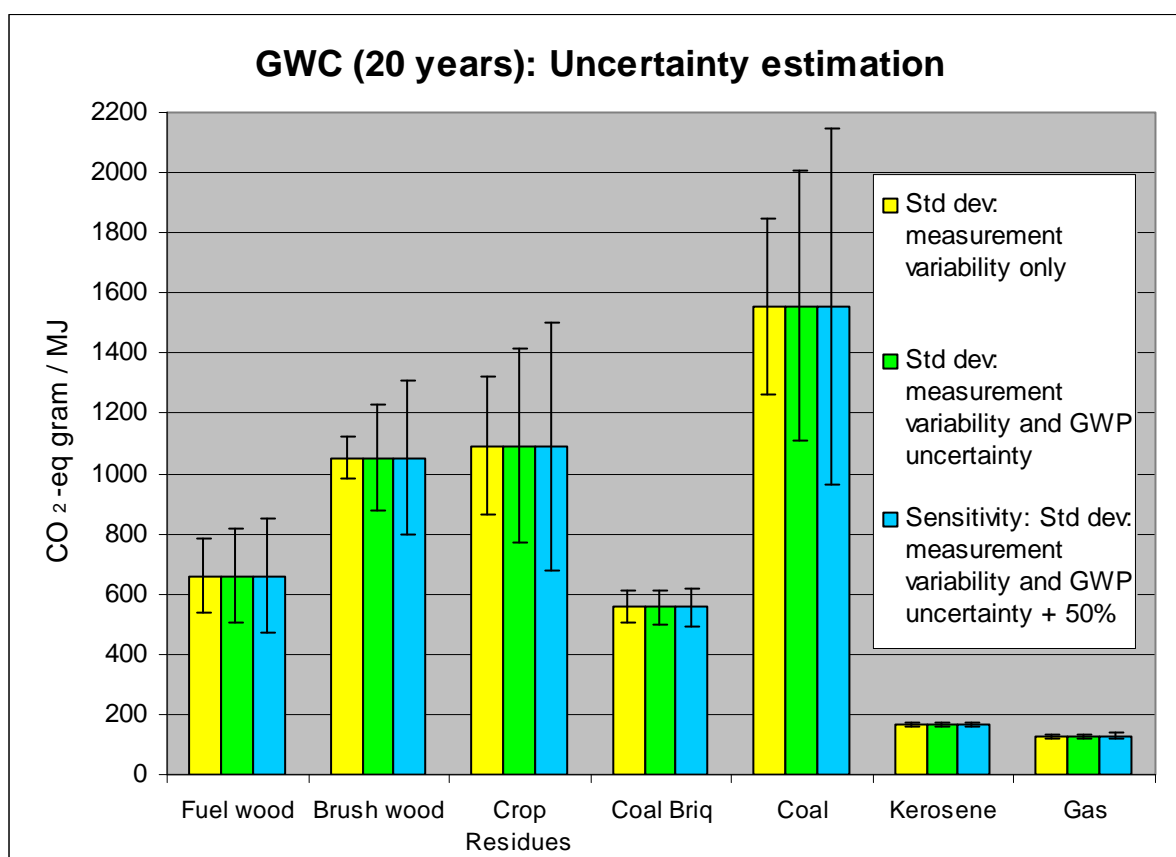


Figure 7: GWC (gram CO₂ equivalents per MJ) for the sum of all GHCs by fuel category. The error bars represent one standard deviation for 3 different scenarios: The yellow bars only include measurement variability, the green bars also include the uncertainty from GWP, while the blue bars illustrate what happens if the standard deviation for GWP increase by 50%.

Figure 7 illustrates that even though most of the GWPs have large uncertainties, the overall results are not that sensitive. An increase in standard deviation for GWP by 50% increased the standard deviation by almost 20% on average. This is probably because CO₂ and CH₄ represent almost three quarters (68 and 6% respectively) of the total positive GWC contribution, and these two gases are associated with low uncertainty. This means that even if

some of the GWP values change, especially for NMHC, NO_x or sulfate, it will not change the results.

3.3.4 Caveats and other sources of uncertainty

Individual emission factors for BC and OC was not reported in Zhang et al. (2000), and therefore the emissions were estimated by calculating the carbon fraction of the particles, and assuming a ratio between BC and OC based on literature. This contributes to additional uncertainty for the GWC calculations, which has not been included. The carbon content of the particles was calculated as 90, 80, 45, 12, 75, 19 and 19% on average for fuel wood, brush wood, crop residues, coal briquettes, coal, kerosene and gas, respectively. The values seem pretty low for coal briquettes, kerosene and gas, which may underestimate the effects of BC and OC for these fuel groupings. However, by assuming that 100% of the particle mass is pure carbon, the contributions from BC and OC would be of only 1-3%, and would not have changed any of the results.

Additional uncertainty arises because there may be large discrepancies between measurements done in a simulated kitchen as in Zhang et al (2001) and what is emitted in real homes during authentic cooking. This is not quantified here, but according to Johnson et al. (2007) the difference is probably large. Further, Johnson et al. (2007) showed that there is a discrepancy between emissions from traditional and improved stoves between lab and in-field measurements. In their study, the mean CO/CO₂ ratio during daily cooking activities was double that found during controlled WBTs, while the CH₄/CO₂ ratio increased threefold for open fires. The improved stoves showed the opposite trend however, with mean CO/CO₂ and CH₄/CO₂ ratios 1.6 and 1.8 times greater for WBTs in simulated kitchens than in homes during daily stove use. The reason is believed to be that the WBT does not reflect the combustion conditions during normal cooking activities. However, these measurements were done in Mexico where 90% of cooking tasks do not involve cooking of water. The cooking is done mostly in a low power phase where, for instance, tortillas are prepared. It is possible that the water boiling test is more comparable to the Chinese cooking style, as the diet in all probability includes more rice, noodles, steamed bread and wok, which is high power cooking tasks. In any case, it is clear that the results from Mexico do not compare directly to China, and the difference between lab measurements and field emissions may be less for Chinese households.

Another fact that needs to be considered is that only the direct emissions from burning are taken into account in this study, not the whole life cycle of the fuels. To get a complete picture and identify the full global warming impact, the whole fuel cycle has to be analyzed where emissions from production, refinery, transport etc are also taken into account.

Furthermore, this work builds on a few studies that are not necessarily representative for China as a whole, as China is a large and diverse country. Based on the information available today, it is not possible to account for the myriad of different stove designs and varied fuel qualities in use in China.

Finally, these measurements were done for cooking, and the GWC estimates do not necessarily apply to other energy uses. For space heating it is possible that the GWC would be even higher, because the fuel probably has a longer smoldering phase that is associated with high emissions. Then again cooking tasks probably require more tending of the fire and more frequent events of lighting and putting out the fire. During a combustion cycle, it is these events that have the highest emissions (Roden and Bond, 2006). In that case the GWC would be lower for heating purposes than cooking. Without obtaining emission factors directly related to heating fuel use, these will remain speculations.

Even though the uncertainty is large as just discussed, the objective is not to estimate the exact global warming commitment from the household sector, but to give an indication of what the situation is like.

4.0 POPULATION EXPOSURE STRATIFIED BY MAIN FUELS

The health effects due to a given air pollutant are determined by the intake fraction that is actually absorbed in the lungs of a person – the dose. The dose depends on many factors, so it is not sufficient to use emission data like what was presented in Section 3.1. For example, the dose attributable to IAP from fuel combustion depends on indoor concentration (determined by factors such as emissions, proximity to the source and ventilation), breathing rate, exposure duration (how much time the person spends in the polluted area), and absorption fraction (how much of the inhaled particles that are actually absorbed in the body). It can be quite difficult to measure all the parameters necessary for a full dose assessment. As a result the exposure, i.e. the level of pollution a person is being exposed to over a given period of time, is frequently used as a proxy for the dose.

It is common to use particulate matter (PM) as an indicator of indoor air pollution, even though there are many different compounds that cause harmful health effects. Furthermore, only the respirable fraction is of interest, as larger particles are not inhaled into the lungs and thus cause less harmful effects. The respirable fraction is defined as PM₁₀ (particles with an aerodynamic diameter of less than 10 µm). The best way to determine exposure is by using personal measurement devices. However, this is both expensive and difficult to accomplish in practice. A good option is to combine the pollution concentrations with the proportion of time spent in the different microenvironments. This can be done for different demographic groups according to fuel use category, by using the equation:

$$EXP_{j,f} = \sum_k t_{j,k,f} \cdot q_{k,f}$$

where $EXP_{j,f}$ is the annual exposure to PM₁₀ for population group j belonging to fuel use category f , t is the fraction of time spent in a given micro-environment k , and q is the PM₁₀ concentration in µg/m³ (Mestl et al., 2007a). This is the approach that has been used to estimate the exposure in the following two cases (Section 4.1 and 4.2).

4.1 Population exposure based on data from Mestl et al. (2007a)

For the population exposure classified by cooking fuel use, the present study draws on work done by Mestl et al. (2007a). They used published indoor air pollution (IAP) data and time

activity patterns together with comprehensive demographic data to estimate the exposure experienced by different demographic groups based on age, sex, household fuel, and geographic location based on climate zone and urban or rural classification. For the indoor air pollution data, Mestl et al. (2007a) used a database from Sinton et al. (1995) that cites more than 110 papers on IAP published between 1980 and 1994. 39 papers fulfilled certain selection criteria and were used in the study, of which 18 reported IAP measurements for rural areas. Some more recent studies were also included, but only one of them characterized rural areas. Pollution levels and standard deviation were estimated for different microenvironments (kitchen, bedroom, living room, indoors at home, indoors away from home and outdoors) for urban and rural population living in the north and south, by using a two-dimensional Monte Carlo simulation (2D-MC). This is a two loop method to account for both variability and uncertainty in the data. For more detailed information of the method see Mestl et al. (2006, 2007a). The resulting IAP values for the rural population living in north and south are illustrated in Figure 8 and 9:

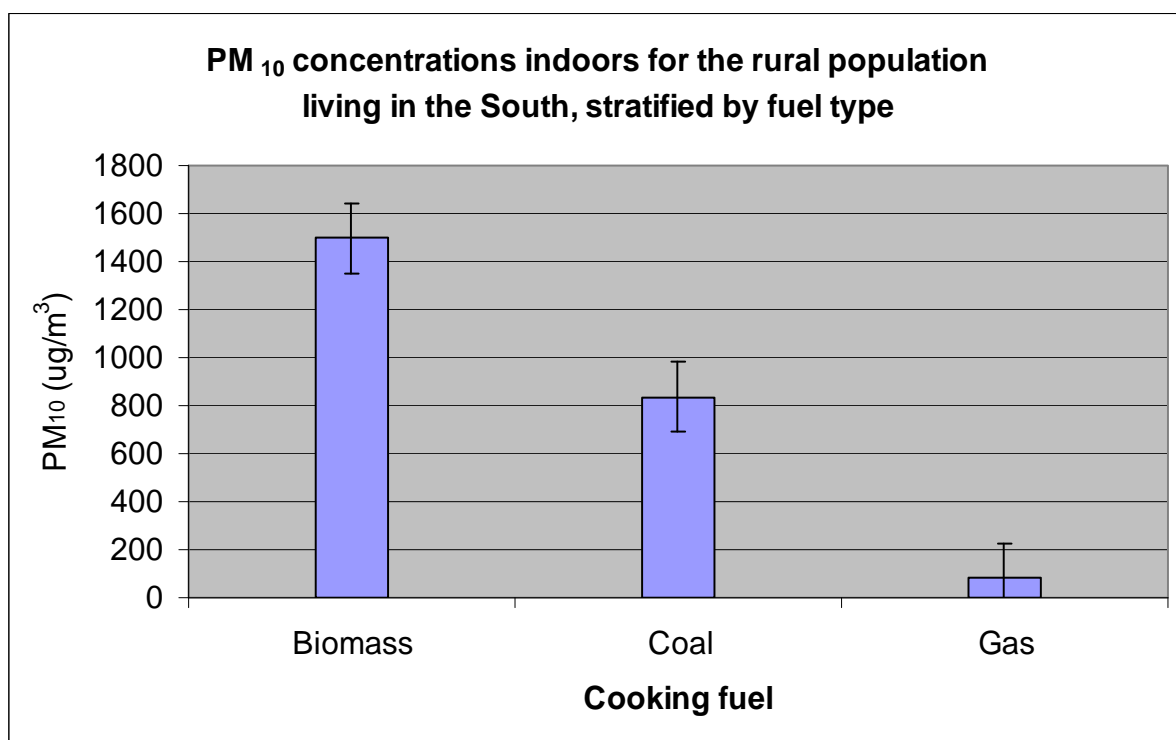


Figure 8: Estimated PM₁₀ concentrations (ug/m³) for the biomass, coal and gas users in the south of China. There were insufficient data to estimate values for different microenvironments, so this represents the pooled “indoors at home” values. No seasonal variations in IAP values were estimated for the south. The error bars represent one standard deviation. Data from Mestl et al. (2007a).

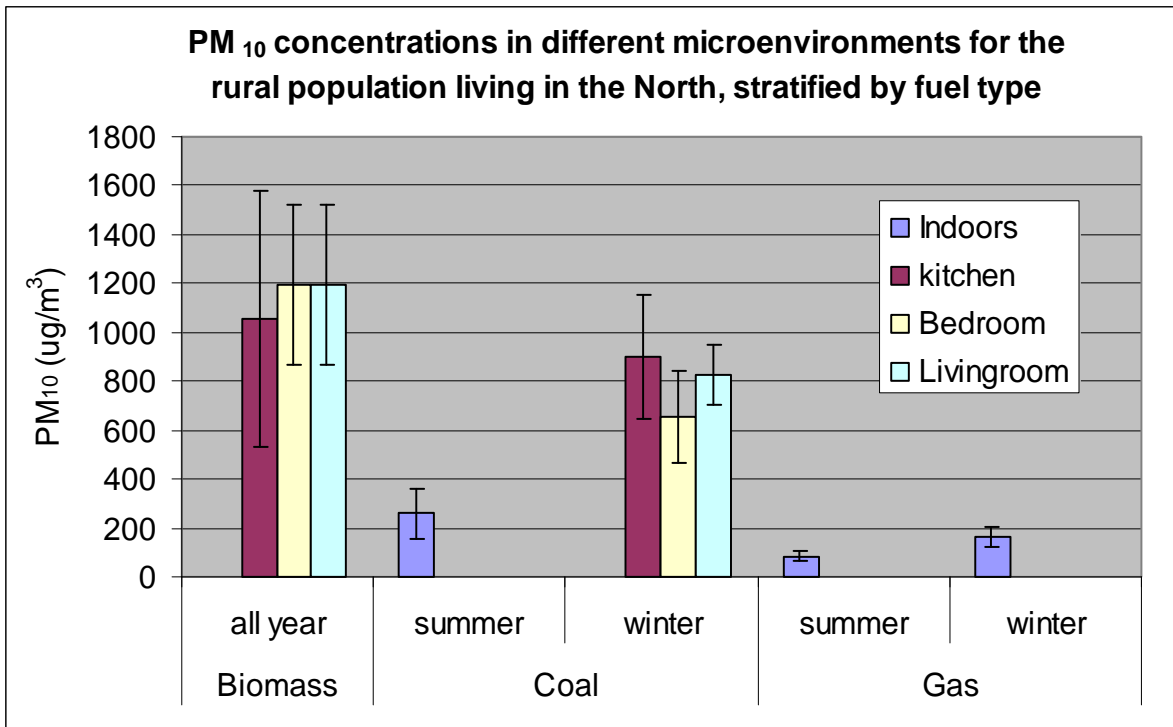


Figure 9: Estimated PM₁₀ concentrations (ug/m³) in indoor environments in the north of China, stratified by fuel type. For gas and summer coal users, there were insufficient data to estimate values for different microenvironments, so this represents the pooled “indoors at home” values. There was not enough data to estimate seasonal variation for the biomass users. The error bars represent one standard deviation. Data from Mestl et al. (2007a).

The annual exposure was estimated by combining the air pollution concentration with the time activity patterns as explained in the beginning of this chapter. The exposure for different demographic groups was then weighted by population, to get the exposure for the urban and rural population living in the north and south belonging to different fuel groups. Here, we are only interested in the exposure fraction that is attributable to the indoor air pollution in rural homes, so the results were modified to give the population weighted exposure (PWE) due to IAP for the rural population, by multiplying with the IAP fraction and weighting the exposure by how many lived in the north and south. Personal communication with the author revealed that most of the uncertainty was due to IAP, but it was not possible to tell exactly how big that fraction was. To be conservative it was assumed that all the uncertainty stemmed from IAP. Outdoor exposure was excluded from the PWE estimate, but because the study also included exposure from “indoors away from home”, some additional exposure that is independent of fuel user classification may overestimate the PWE. The results are illustrated in Figure 10:

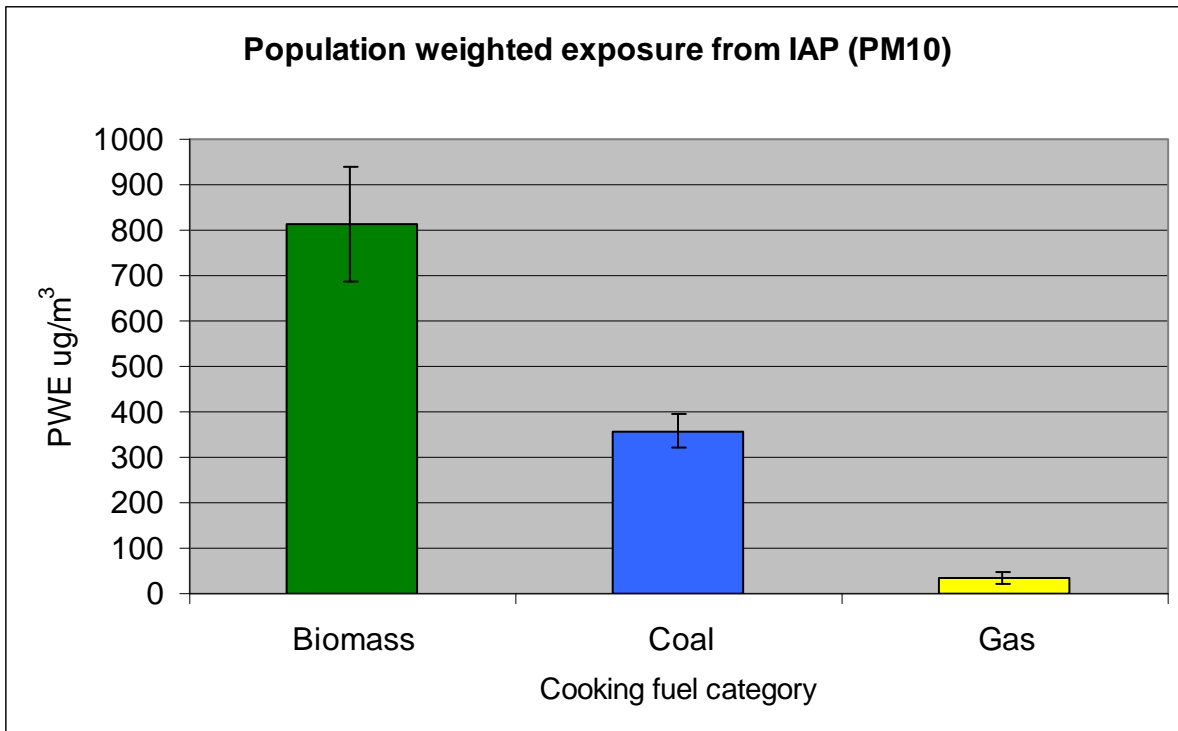


Figure 10: PWE to PM₁₀ for the rural population grouped by cooking fuel category. The data are modified from Mestl et al. (2007a), to only include exposure due to IAP. The error bars represent one standard deviation.

Because most of the studies included in Mestl et al. (2007) were from the late eighties and early nineties, it can be questioned whether the results are representative for China today. To get an idea if this is the case, the IAP concentrations for the available microenvironments in each fuel category in the rural areas were plotted against the publication year of the paper (data from personal correspondence with H. Mestl. PM₁₀ conversions were done by multiplying TSP and PM₄ fractions with 0.7 and 1.4 respectively, in accordance with the method in Mestl et al. (2007a)), and a linear regression analysis were performed (see Appendix B). All trend lines were negative, implying that the IAP concentrations on average have decreased over time. However, most of the fits were poor, with r^2 ranging from 0.03 to 0.36. Better correlations were obtained for “coal north” kitchen and bedroom with $r^2 = 0.69$ and 0.49 respectively (6 measurements in both cases). “Biomass north kitchen” had a clear negative trend with $r^2 = 0.97$, but it only included 3 measurements, and so the result can not be seen as robust. The overall trend was about minus 30-70 ug/m³ PM₁₀ per year. The results indicate that IAP levels probably have declined over time, but there is not enough data to draw definite conclusions. The decline can also be due to other factors, for example change in measurement practices over time.

4.2 Population exposure based on data from Edwards et al. (2007)

More recent data on indoor air pollution levels measured in 2001-2003 were presented in Edwards et al. (2007). As part of a large-scale household survey including 3500 households, a sub-sample of almost 400 households were monitored for PM₄ in kitchens and living rooms. The research was performed in order to evaluate the effects of China's national improved stove program (NISP) on indoor air quality. Three provinces were included in the study; Zhejiang, Hubei and Shaanxi, with different levels of income and climate. However, given the range of conditions found in China, the survey does not provide a statistically representative picture of the national IAP. The study found highly diverse fuel usage patterns, with 34 different fuel combinations used in kitchens in winter, and 28 in the summer. On average, 2.6 types of fuels were used per household in the summer, and 1.9 in the winter (Edwards et al., 2007). IAP concentrations for summer and winter are presented in Figure 11, stratified by main cooking fuel. The households included here reported no heating fuel use in the winter.

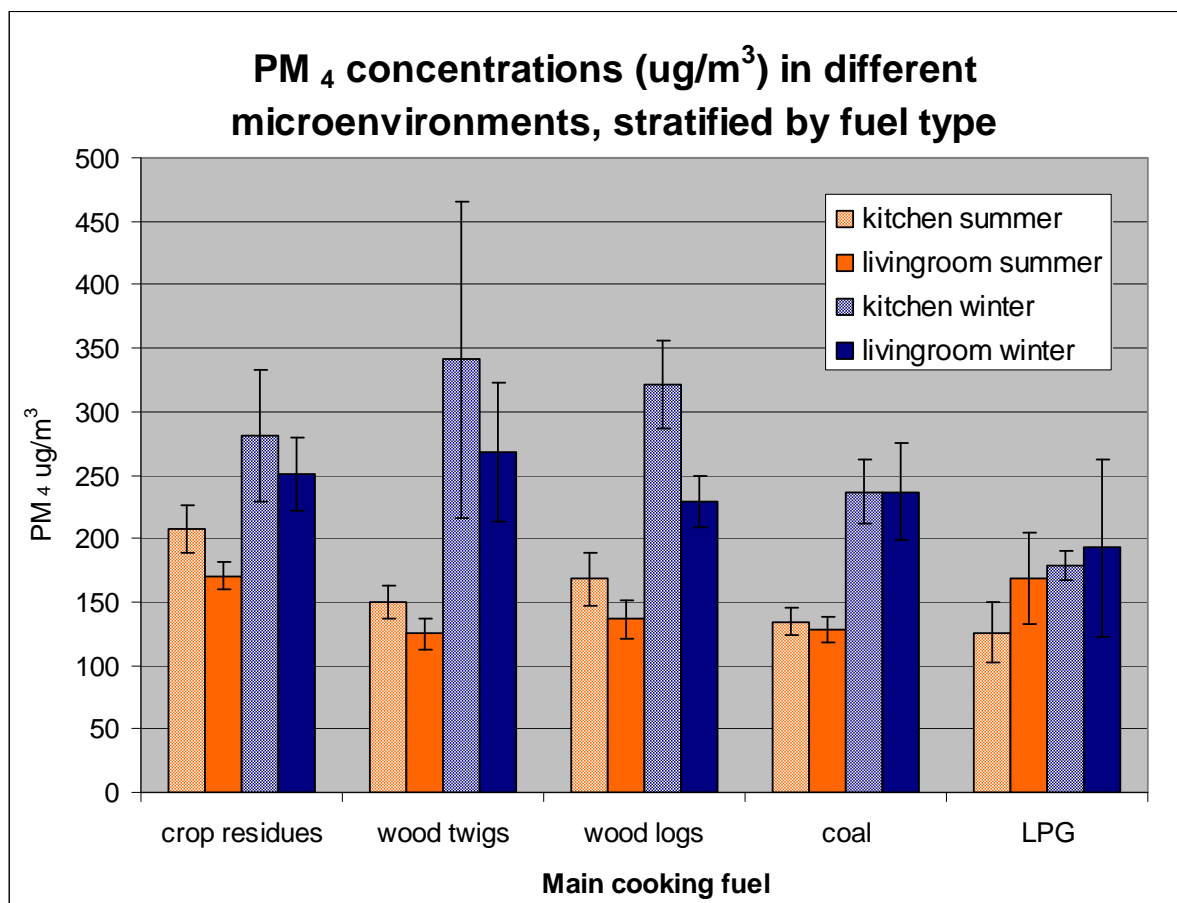


Figure 11: This graph illustrates the indoor summer (orange) and winter (blue) PM₄ concentrations for kitchen

(“tile”) and living room (solid). The error bars represent one standard error (sd/\sqrt{n}). Based on Edwards et al. (2007), exact numbers obtained from personal correspondence with R. Edwards.

Even though there are no fuels used for heating, the concentrations are higher in the winter, probably because of less ventilation. For LPG, it is evident that some other source of pollution is present, as the living room concentrations are higher than the kitchen.

For the exposure estimate the average time-activity data for men and women age 15-64 years living in the north was used (Mestl et al., 2007a). Because there was no data for the bedroom it was assumed to have the same concentration level as the living room. This should be a reasonable assumption, as fuel combustion only occurred in the kitchen. Winter time was estimated as 5 months, as in Mestl et al (2007a). PM_4 were measured in the original study, but was converted to PM_{10} by multiplying by a factor of 1.4 to be comparable to the estimates in Section 4.1. The results from the exposure estimate are presented in Figure 12:

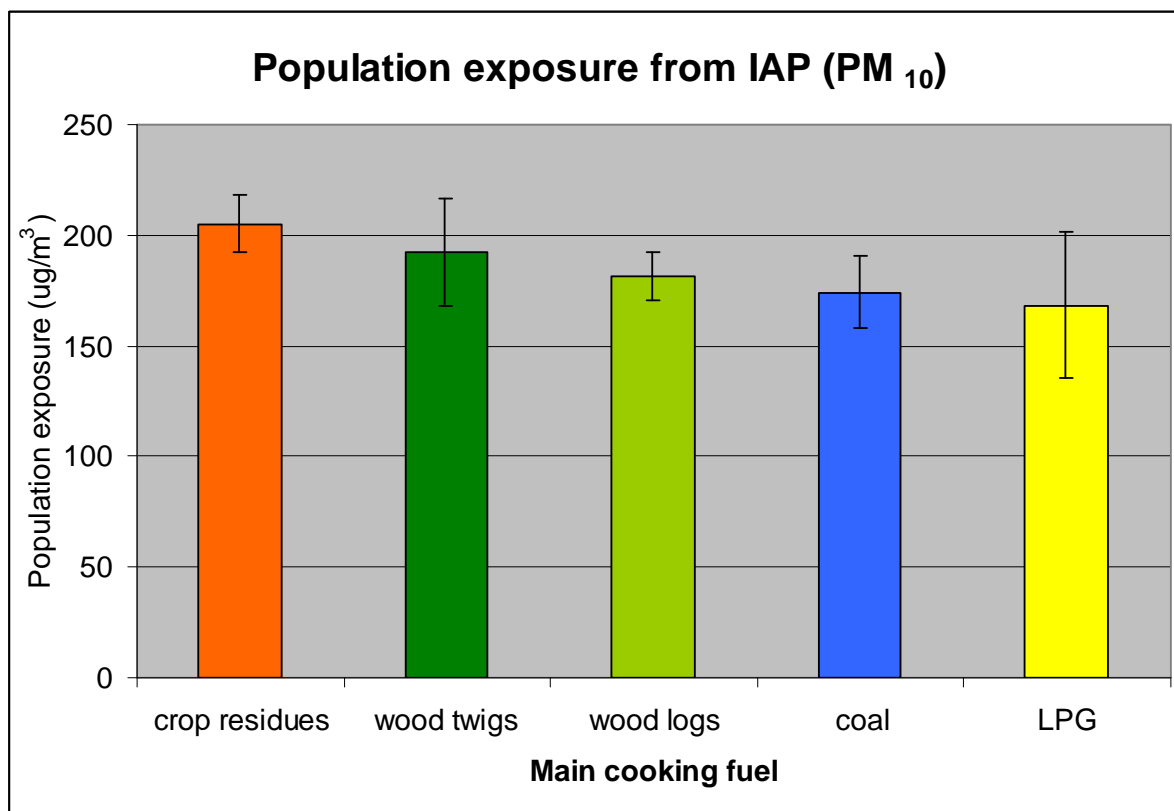


Figure 12: The average exposure to PM_4 for the rural population grouped by cooking fuel category. No exposure occurring outside the home was included. The error bars represent one standard deviation.

In contrast to the estimate based on data from Mestl et al. (2007a), the exposure estimates here hardly differ between fuel categories. The ranking is in the same order as in figure 10, with biomass > coal > gas, but there are no statistically significant differences between the fuel categories. Recall that the LPG values almost certainly are too high. In addition, these

data were classified by main cooking fuel. In the study, a lot of different fuel combinations were found. If multiple fuels were in use, it could help explain the small difference between fuel categories. The overall exposure is much lower than in Section 4.1, with the exception of gas users. The difference is roughly a factor of 4 for biomass users, and 2 for coal users. The lower values based on Edwards et al (2007) is probably attributable to at least three factors. First of all, the data are of a more recent date, so this might reflect general improvements over the last 20 odd years. Secondly, it is part of the NISP study, so improved stoves are also included that had a positive effect on the indoor air quality, at least for biomass users (Edwards et al., 2007). Thirdly, the data from Mestl et al. (2007a) also included households with heating use in the wintertime, which lead to elevated IAP concentrations. It is also worth mentioning that all exposure outside the home was excluded in the estimate presented in this section, while in the 4.1 there might have been some additional exposure independent of cooking fuel classification, due to exposure “indoors away from home”.

In the following chapter, it is the population weighed exposure data based on Mestl et al. (2007a) that will be applied to investigate the relationship between population exposure and emissions of greenhouse compounds.

5.0 EVALUATION OF GLOBAL WARMING COMMITMENTS vs. POPULATION EXPOSURE

To be able to recommend a particular fuel from an environmental perspective, one should ideally take both the emissions of greenhouse compounds and the population exposure to health damaging pollutants into account. Other factors are of course also important for fuel choice, for example the price of fuels, but that will not be considered further here. For co-benefits to arise from fuel upgrades, both the GWC and the population exposure have to decrease when switching fuel. If only one factor is reduced, it becomes difficult to decide what fuel is “better”. Some kind of value assessment is then needed. One way to do this is to convert the GWC and exposure to monetary values to be able to place it on a common scale. The global market price for CO₂ quotas could be used to estimate the value of GWC, even though this quantity has been variable and might not reflect actual costs of CO₂ reductions. Exposure/response functions would be needed to convert the exposure estimates to health effects, and then the value of the avoided health outcomes could be established. This is hard because it involves ethical considerations, but possible options include using hospital expenses, cost to society or the value of a statistical life determined by for example willingness to pay. The results will differ to a large extent depending on how climate and health effects are valued, and such estimations will not be done here.

5.1 GWC compared to particle emissions

The emission factors for total suspended particles (TSP) are reported in the database from Zhang et al. (2000). Even though emissions do not equal exposure to the population, it can give some preliminary suggestions as to what fuels probably are better in a health perspective. Figure 14 illustrates the combined effects from emissions of health damaging particles and GWCs:

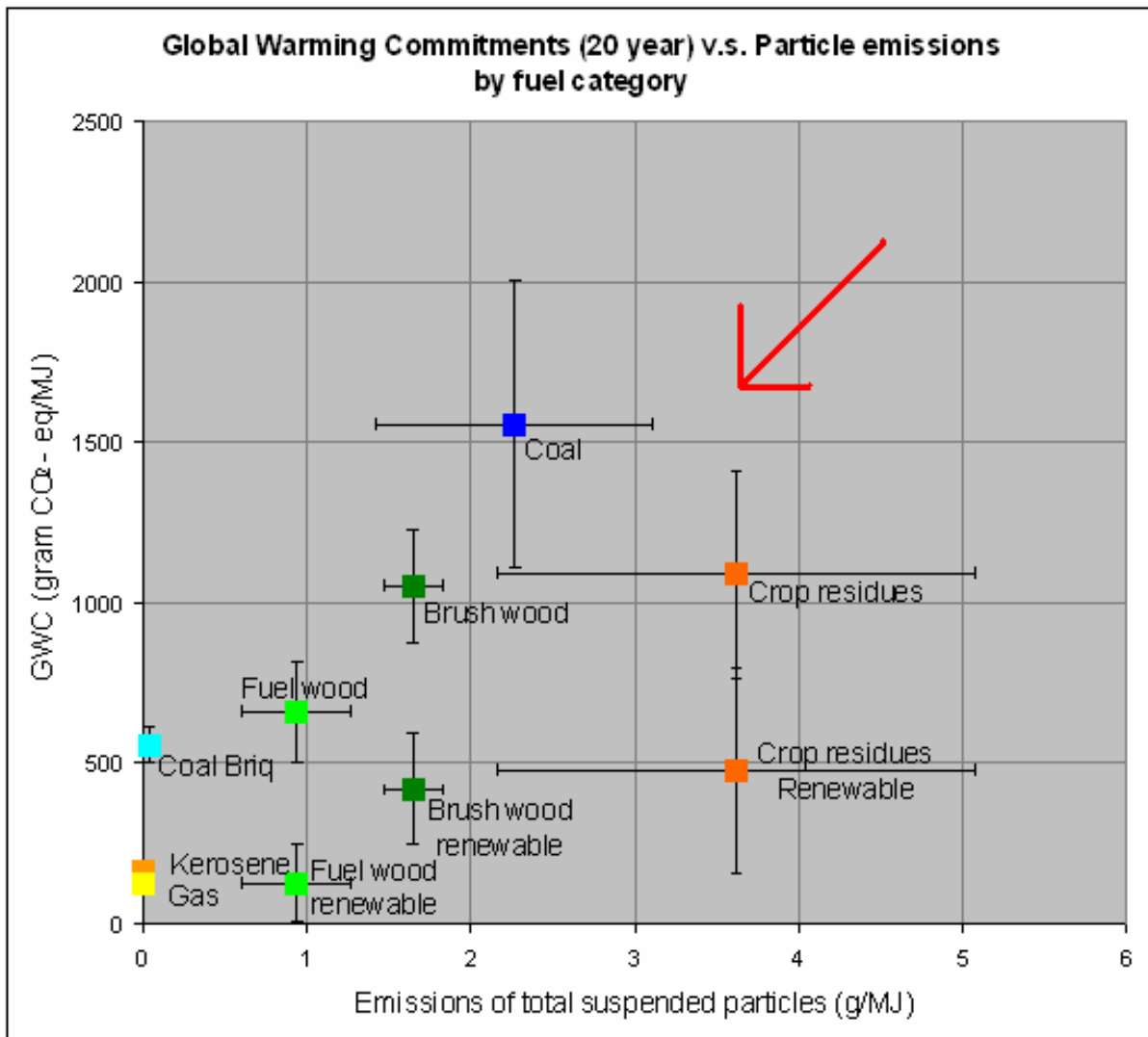


Figure 13: TSP (gram per MJ) plotted against the global warming contribution (gram CO₂ equivalents /MJ) for each fuel category. The color coding correspond to Table 5, while the red arrow indicates the general direction one needs to change fuels for co benefits to arise. The error bars represent one standard deviation.

Switching from non-renewable biomass fuels to renewable harvesting represents a vertical drop on the graph (Figure 13). It is preferable to use renewable biomass fuels from a climate point of view, but it makes no difference for the population’s exposure to harmful compounds. For co-benefits to arise from fuel change, one needs to shift in the direction of the red arrow in Figure 13. From the graph it becomes clear that the “worst” fuel when it comes to GWC emissions is not necessarily the same as the one that emits most health damaging pollutants. Coal has the highest GWC, while the emission of particles is 2.3 (± 0.8 s.d.) g/MJ. Crop residues on the other hand, has lower GWC (especially when harvested renewably, which is probably the case), while emissions of particles are 3.6 (± 1.5 s.d.) g/MJ. The variation is large however, so it can not be concluded whether crop residues have larger TSP emissions than coal on a statistically significant level. Gases and kerosene are clearly the

best fuels overall, while renewable harvested fuel wood can compete on the climate side. Because of the scale, it seems like coal briquettes have about the same TSP emissions as gas and kerosene (0.009 (± 0.009 s.d) g/MJ and 0.006 (± 0.002 s.d) g/MJ, respectively). Briquette emissions are actually notably higher (0.05 (± 0.02 s.d) g/MJ). While gas and kerosene are best from an environmental point of view, they are also the most expensive fuels. Therefore, an alternative to raw coal could be to make coal briquettes instead, as both the particle emissions and GWC will be significantly reduced. Renewably harvested biomass fuels have low GWC, while the particle emissions are somewhat high.

5.2 GWC compared with population exposure

The population exposure does not necessarily relate to the emissions, so in Figure 14 the exposure data from Chapter 4 is combined with the GWC results from Chapter 3. Unfortunately, this reduces the fuel groupings down to just gas, coal and biomass, and therefore the GWC data have been weighted by fuel use in China to provide estimates for each broad type of fuel. Domestic energy fuel use data was available for 1995, 2000 and 2005, but because the exposure estimate mainly builds on older data, the values for 1995 were used for coal. In 1995 coal briquettes made up less than 0.5% of the total coal use (NBS 1997), so only the GWC value for coal was used. Crop residue use has been stable on about 64% of the total biomass (NBS 1997, 2002, 2007). It was not specified what the firewood breakdown was, so it was assumed that fuel wood and brush wood had a relationship of 1:1.

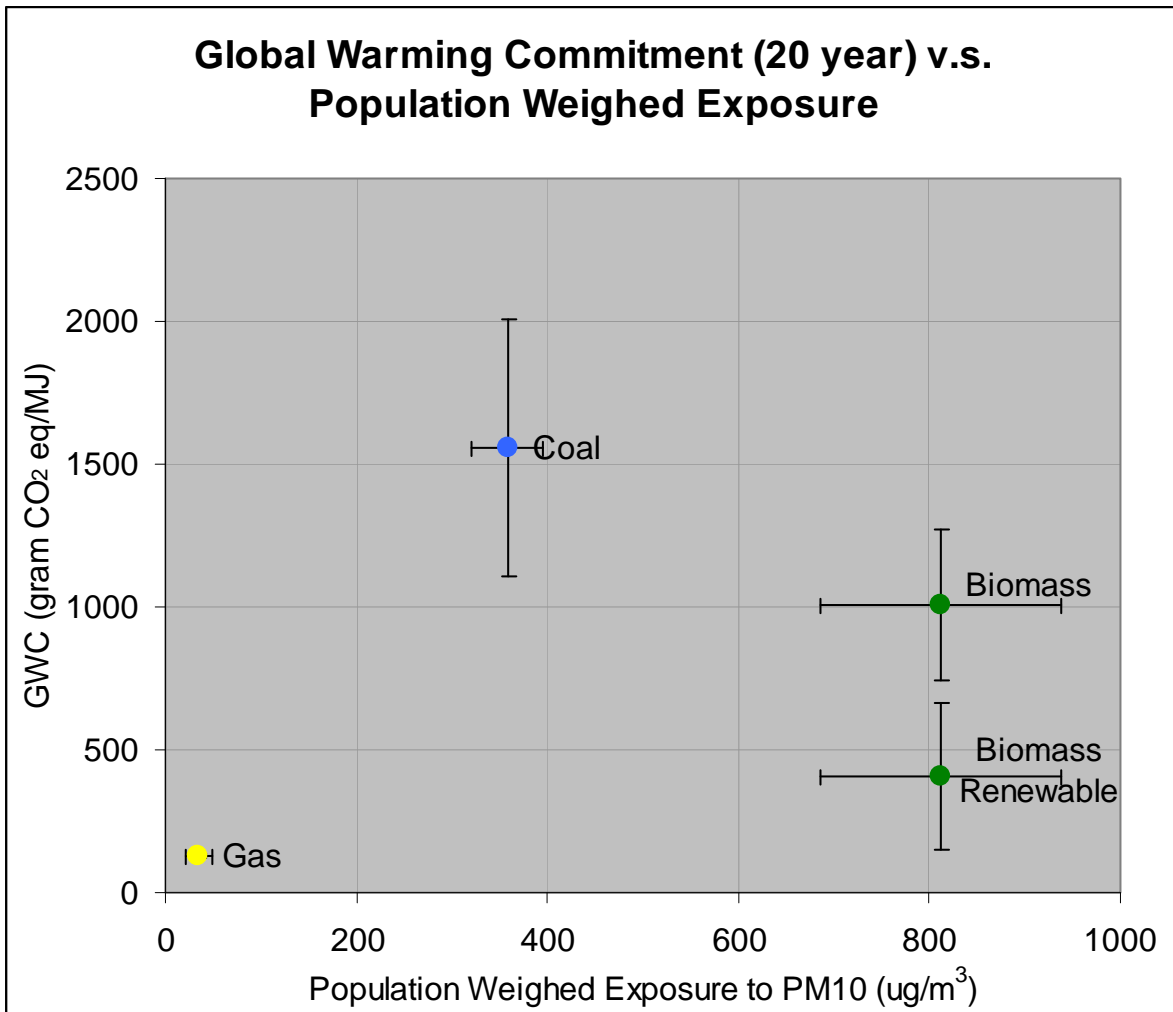


Figure 14: PWE (PM₁₀ ug/m³) is plotted against GWC (gram CO₂ equivalents /MJ) for the fuel categories coal, biomass and gas. The error bars represent one standard deviation.

Figure 14 shows similar tendencies as Figure 13. The exposure from biomass is higher than would be expected from the biomass emissions in Figure 13, but if about two thirds of the biomass users use crop residues, it is likely that this is reflected in the studies that were included in the exposure estimation in Mestl et al. (2007a).

Changing fuel from coal or biomass to gas is certainly advisable. It would reduce both global warming commitment and the population exposure to PM₁₀ significantly. A fuel switch from coal to gas cuts GWC by 1430 (±500) CO₂ equivalents in gram /MJ, while PWE is reduced by 320 (±40) µg/m³. Replacing biomass by gas cuts GWC by 280 (±260) or 880 (±270) CO₂ equivalents in gram /MJ, depending on whether biomass is harvested renewably or not. PWE is reduced by 780 (±130) µg/m³. This gives quite a large potential for improvement, considering how many rural families rely on biomass or coal for energy.

For choosing between coal and biomass, it is hard to give advice on which fuel is better. PWE is increased by $450 (\pm 130) \mu\text{g}/\text{m}^3$ by a fuel change to biomass. On the other hand, GWC is reduced by between $1150 (\pm 520)$ and $550 (\pm 520)$ CO₂ equivalents in gram /MJ, for renewable and non-renewable harvesting, respectively. The standard deviation is very large, so it is not possible to conclude whether there in fact will be an improvement. In situations like this, a value assessment that places GWC and PWE on a common scale is needed to determine which fuel option to recommend.

Figure 14 gives the impression that there is a clear trade off between coal and biomass. In reality, other options are also available. For example, Figure 13 illustrated that coal can be a good alternative as long as it is processed before combustion. Using improved stoves is another option to reduce the emissions. The improved stoves in this study did not demonstrate very good results, in fact the improved biomass stoves had the highest particle emissions. One reason might be that the measurements in simulated kitchens do not reflect the emissions in real homes under authentic cooking conditions, as discussed in Chapter 3. Controlled “before and after” evaluations of improved biomass stove dissemination programs in other parts of the world have shown significantly lower IAP concentrations. For instance, PM_{2.5} concentrations in India were reduced by 24 and 49% for two types of improved stoves compared to the traditional model (Dutta et al., 2007), while another study in India showed improvements of 44% for PM_{2.5} (Chengappa et al., 2007). A similar assessment from Mexico demonstrated PM_{2.5} concentration reductions of 67% (Masera et al., 2007). In China, evaluation of the NISP showed that IAP was reduced with improved biomass stoves, with average kitchen PM₄ concentrations of $152 \mu\text{g}/\text{m}^3$ (133-172, 95% confidence interval) compared to $268 \mu\text{g}/\text{m}^3$ (208-328, 95% confidence interval) for traditional stoves (Edwards et al., 2007). Modern biomass burning devices have much more efficient combustion than the current ‘improved stoves’ available in China and elsewhere. These show great potential, with considerably lower particle emissions than first generation improved biomass stoves. This would make biomass a much better alternative than what is presented here.

When it comes to gas, an interesting option that was not included in this study is biogas. Since it is a gas it will premix easily with air to give clean combustion, and the particle emissions are likely on the same scale as the other gases measured by Zhang et al. (2000). The combustion products consist of almost entirely CO₂ and water, so the GWC will be virtually zero as long as the biogas is produced renewably (Smith et al., 2000).

6.0 CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER WORK

The global warming contribution (GWC) from main household fuels has been quantified, and the results show that kerosene and gas by far had the lowest GWC per MJ, with the exception of 100% renewably harvested fuel wood. There are big differences between fuel categories; coal as fuel group had the largest global warming commitment, about a factor of ten higher than the best fuel. There are however also large differences within fuel categories. Estimates of GWC when all GHCs are taken into account can be found in Table 8, for both 100 and 20 year time horizon. The GWC results from the current study differ a bit from similar studies (Edwards et al., 2004, Smith et al., 2000) because this work has included more GWC, and to a minor extent because of use of different GWPs. It is found that estimates of total global warming commitment will change substantially depending on what set of GHCs are included in the calculations, whether the biomass fuels are harvested renewably or not, and the choice of time horizon. Even when the biomass was harvested renewably it still had a significant climate impact, at least for brush wood and crop residues.

On average, CO₂ was the compound that contributed most to total global warming commitment, followed by carbon monoxide and black carbon. Organic carbon had the largest cooling effect, while the contribution from sulfate was negligible, possibly due to low sulfur content in the coal. The uncertainty for GWC was large, with standard deviations ranging from 5-30% for different fuel categories, due to both high variation for the emission data and uncertainty associated with the GWP values. This makes it hard to talk about statistically significant differences, at least between the fuel groups with the largest GWCs.

Estimates of total emissions by using energy use data are hampered by large uncertainties, but seem to indicate that the household emissions contribute significantly to China's greenhouse compound inventory.

For the investigation of potential co benefits it became clear that the "worst" fuel in terms of population exposure is not necessarily the same as the one that has the largest climate impact. However, more options should be included in the assessment to give a better depiction of the real world. Even though coal briquettes had much lower particle emissions than biomass fuels, the GWC was higher if the biomass was harvested renewably. Coal briquettes had a

clean combustion with emissions of mostly CO₂ and some CO, and it will probably be hard to reduce the GWC much further. It seems like it is easier to cut the particle emissions of biomass, which showed large room for improvements. Design changes of the stove to reduce population exposure, in combination with renewably harvested biomass seems like a very promising option. It would be interesting to evaluate second generation improved biomass stoves, to investigate their global warming impact and potential for reducing IAP levels.

Further work should focus on reducing the uncertainty associated with the GWC estimates. To do this, it is essential to obtain less variable emission factors for China. Once more accurate measurements become available, it will be possible to estimate the GWC from Chinese households more accurately by using the method described in this paper. Future studies need to measure at least CO₂, CH₄, and CO, and also BC and OC instead of TSP. Sulfates contributed minimally to the total GWC, but it may be important to measure SO₂ if the coal has a higher sulfur content than the coal used in the Zhang et al. (2000) study. It is highly recommended to do measurements in the field, instead of simulated kitchen measurements. This would also have great value in itself; if the results are compared with emission factor databases such as the one used here (Zhang et al., 2000), it would be possible to examine if the discrepancy is as big between in-field measurements and simulated kitchens in China as in Mexico, where the cooking style is different. More than 3 measurements for each fuel/stove are probably necessary to obtain less variable emission factors. The uncertainties here can be used to estimate how many measurements are needed. Measurements of emission factors should also include indoor air quality measurements for the same fuel/stove combinations to be able to investigate potential co-benefits.

In addition to the variability associated with the emission factors, other factors influenced the confidence of the GWC estimates and should be addressed in further assessments. Monte Carlo simulation using the published range for GWP values is suggested to quantify the uncertainty associated with GWP. The difference between renewable and non-renewable biomass was large, so data for the proportion of the biomass that is harvested renewably should be obtained.

Regarding the population exposure to respirable particles, it would also be interesting to study further if the concentrations in Chinese homes really have declined over the years and to what extent.

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APPENDIX

Appendix A: Lifetimes where $AGWP_{20} \approx AGWP_{100}$.

If $C(t)$ is the time dependent abundance of a GHC, and its removal from the atmosphere can be expressed by simple exponential decay, the abundance will be determined by:

$$C(t) = C_0 \exp(-t/\tau)$$

where C_0 is the initial abundance and τ is the lifetime.

Remember that $AGWP = \int_0^{TH} a[C(t)]dt$, which then integrates to:

$$AGWP = a \tau C_0 \exp(0) - a \tau C_0 \exp(-TH/\tau) = a \tau C_0 (1 - \exp(-TH/\tau)).$$

We are interested in the case where $AGWP_{20} \approx AGWP_{100}$, meaning

$$f(\tau) = \frac{\int_0^{20} a[C(t)]dt}{\int_0^{100} a[C(t)]dt} = \frac{1 - \exp(-20/\tau)}{1 - \exp(-100/\tau)} \approx 1$$

From the Figure below it is evident that GHCs with lifetimes less than approximately 5 years will have $AGWP_{20}$ values pretty much equal to $AGWP_{100}$ (For $\tau = 5$ years, $f(5) = 0.982$, meaning $AGWP_{20}$ is 1.8% lower than the $AGWP_{100}$ value. For lifetimes much longer than 5 years, the discrepancy becomes larger and larger, and for $\tau = 10$ the $AGWP_{20}$ is 13.5% lower than the $AGWP_{100}$ value ($f(10) = 0.865$).

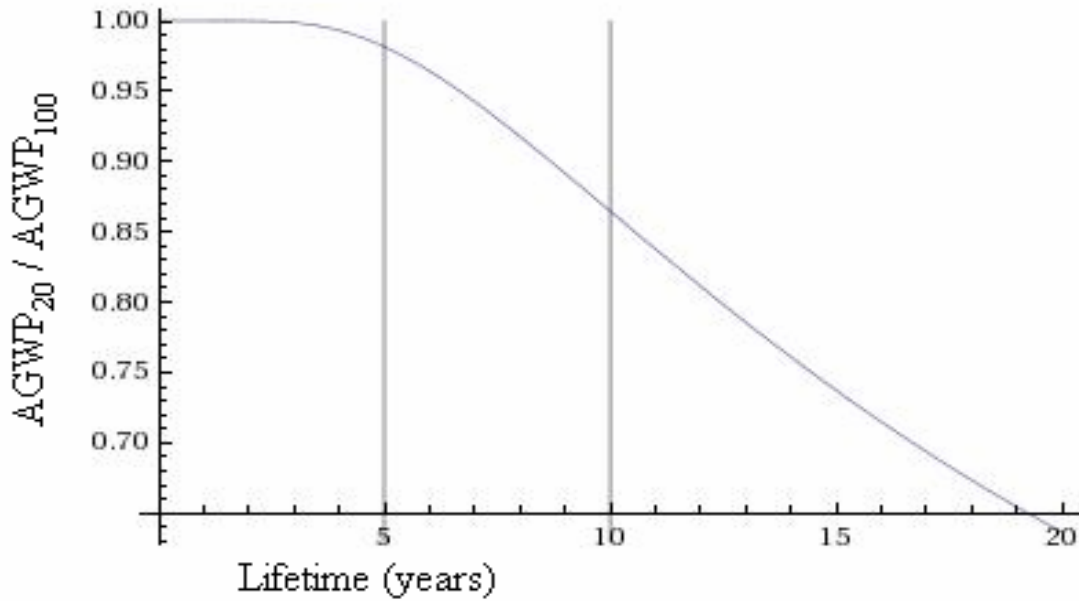


Figure A: Illustration of what lifetimes the AGWP₂₀ value will start to differ significantly from the AGWP₁₀₀ value.

Figure 1 is based on the following equations:

The decay of a pulse emission of the short lived gas: $\exp(-t/\tau)$ where $\tau = 2$ years

The decay of a pulse emission of CO₂: $a_0 + \sum_{i=1}^3 a_i \cdot \exp(-t/\tau_i)$ where $a_0 = 0.217$, $a_1 = 0.259$, $a_2 = 0.338$, $a_3 = 0.186$, $\tau_1 = 172.9$ years, $\tau_2 = 18.51$ years and $\tau_3 = 1.186$ years. (IPCC, 2007)

Appendix B: Analysis of IAP concentration change over time.

The data used in Mestl et al. (2007a) was converted to PM₁₀ and plotted against publication year for each of the microenvironment groupings. A linear regression was performed, and the results show that the trend was negative for all categories. The linear fit is in the form $f(x) = a + bx$, where a denote the interception point with the y axis and b gives the annual trend. r^2 represents how much of the variation that can be explained by the model. The following figures show the results for each microenvironment:

Rural Biomass (South)

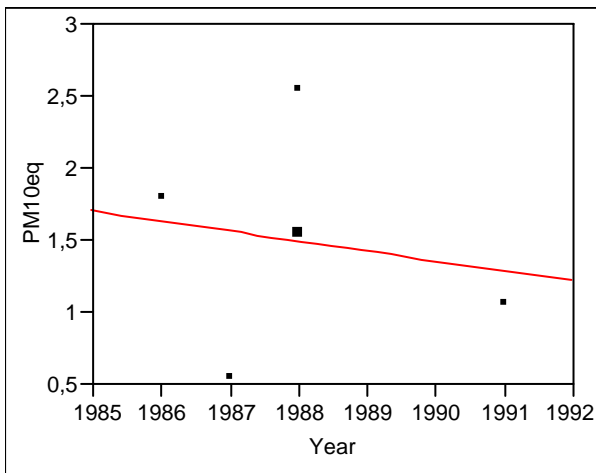


Figure B1: Linear fit of PM₁₀ equivalents (mg/m³) by publication year.

$$PM_{10eq} = 139 - 0.07 \text{ Year}, r^2 = 0.0296$$

Rural Coal (South)

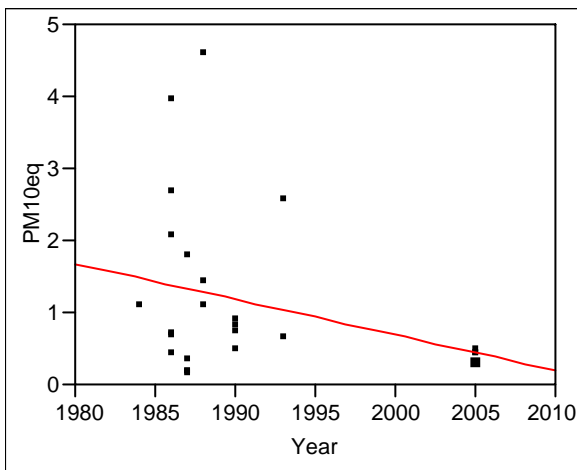


Figure B2: Linear fit of PM₁₀ equivalents (mg/m³) by publication year.

$$PM_{10eq} = 99 - 0.05 \text{ Year}, r^2 = 0.081$$

Rural Biomass, Indoors (bedroom and living room) (North)

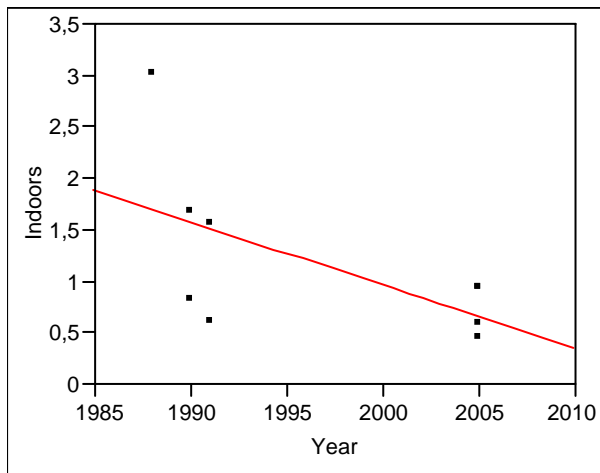


Figure B3: Linear fit of PM₁₀ equivalents (mg/m³) by publication year.

$$PM_{10eq} = 123 - 0.06 \text{ Year}, r^2 = 0.363$$

Rural Biomass, Kitchen (North)

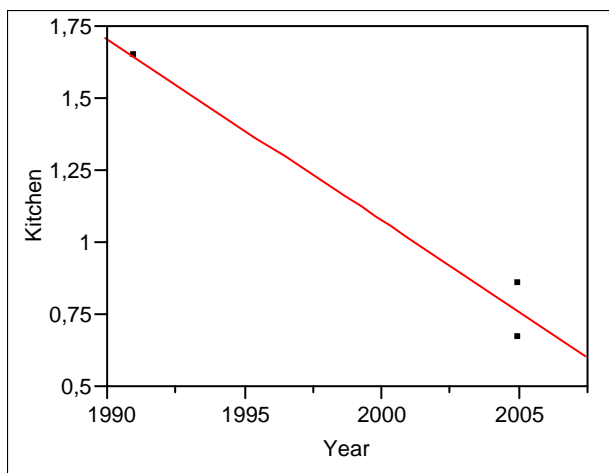


Figure B4: Linear fit of PM₁₀ equivalents (mg/m³) by publication year.

$$PM_{10eq} = 127 - 0.06 \text{ Year}, r^2 = 0.968$$

Rural Coal, Living room (North)

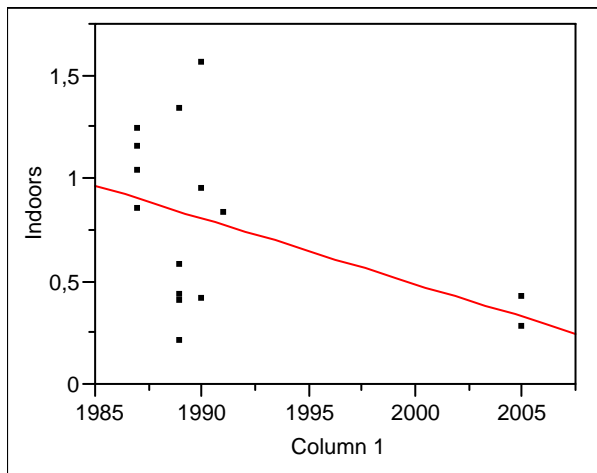


Figure B5: Linear fit of PM₁₀ equivalents (mg/m³) by publication year.

$$PM_{10eq} = 64 - 0.03 \text{ Year}, r^2 = 0.194$$

Rural Coal, Kitchen (North)

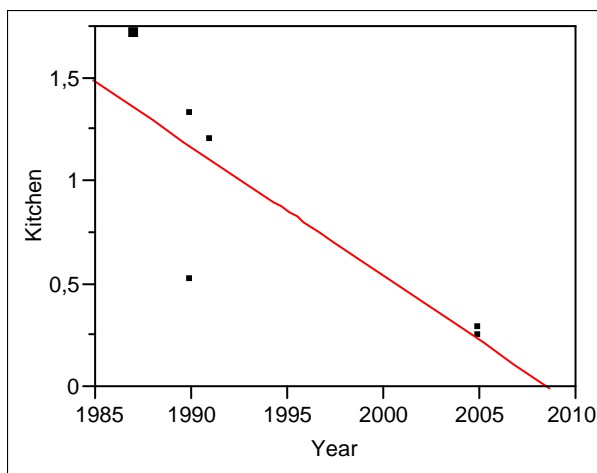


Figure B6: Linear fit of PM₁₀ equivalents (mg/m³) by publication year.

$$PM_{10eq} = 126 - 0.07 \text{ Year}, r^2 = 0.689$$

Rural Coal, Bedroom (North)

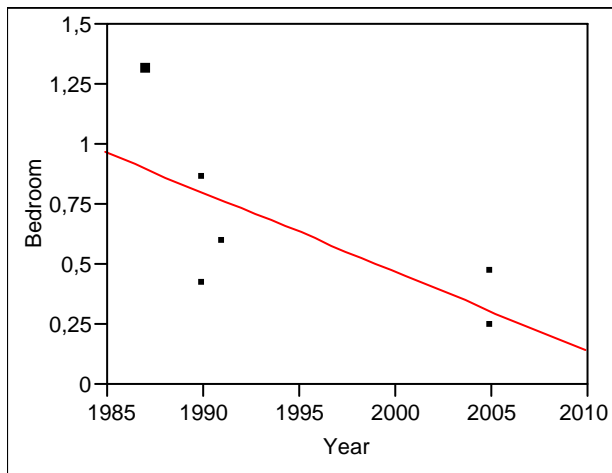


Figure B7: Linear fit of PM₁₀ equivalents (mg/m³) by publication year.

$$\text{PM}_{10\text{eq}} = 66 - 0.03 \text{ Year}, r^2 = 0.491$$