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# Emission factors for open and domestic biomass burning for use in atmospheric models

**S. K. Akagi[1], R. J. Yokelson[1], C. Wiedinmyer[2], M. J. Alvarado[3], J. S. Reid[4], T. Karl[2], J. D. Crounse[5], and P. O. Wennberg[6]**

[1]University of Montana, Department of Chemistry, Missoula, MT, USA
[2]National Center for Atmospheric Research, Boulder, CO, USA
[3]Atmospheric and Environmental Research (AER), Inc., Lexington, MA, USA
[4]Naval Research Laboratory, Monterey, CA, USA
[5]Division of Chemistry and Chemical Engineering, California Institute of Technology, Pasadena, CA, USA
[6]Divisions of Engineering and Applied Science and Geological and Planetary Science, California Institute of Technology, Pasadena, CA, USA

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Correspondence to: R. J. Yokelson (bob.yokelson@umontana.edu)

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## Abstract

Biomass burning (BB) is the second largest source of trace gases and the largest source of primary fine carbonaceous particles in the global troposphere. Many recent BB studies have provided new emission factor (EF) measurements. This is especially true for non methane organic compounds (NMOC), which influence secondary organic aerosol (SOA) and ozone formation. New EF should improve regional to global BB emissions estimates and therefore, the input for atmospheric models. In this work we present an up-to-date, comprehensive tabulation of EF for known pyrogenic species based on measurements made in smoke that has cooled to ambient temperature, but not yet undergone significant photochemical processing. All the emission factors are converted to one standard form (g compound emitted per kg dry biomass burned) using the carbon mass balance method and they are categorized into 14 fuel or vegetation types. We compile a large number of measurements of biomass consumption per unit area for important fire types and summarize several recent estimates of global biomass consumption by the major types of biomass burning. Biomass burning terminology is defined to promote consistency. Post emission processes are discussed to provide a context for the emission factor concept within overall atmospheric chemistry and also highlight the potential for rapid changes relative to the scale of some models or remote sensing products. Recent work shows that individual biomass fires emit significantly more gas-phase NMOC than previously thought and that including additional NMOC can improve photochemical model performance. A detailed global estimate suggests that BB emits at least 400 $\mathrm{Tg\,yr^{-1}}$ of gas-phase NMOC, which is about 4 times larger than most previous estimates. Selected recent results (e.g. measurements of HONO and the BB tracers HCN and $CH_3CN$) are highlighted and key areas requiring future research are briefly discussed.

## 1 Introduction

Biomass burning (BB) can be broadly defined as open or quasi-open combustion of any non-fossilized vegetative or organic fuel. Examples range from open fires in forests, savannas, crop residues, semi-fossilized peatlands, etc. to biofuel burning (e.g. cooking fires, dung burning, charcoal or brick making, etc.). Savanna fires, domestic and industrial biofuel use, tropical forest fires, extratropical (mostly boreal) forest fires, and crop residue burning are thought to account for the most global biomass consumption (in the order given). Overall, BB is the largest source of primary fine carbonaceous particles and the second largest source of trace gases in the global atmosphere (Bond et al., 2004; Andreae and Merlet, 2001; Forster et al., 2007; Guenther et al., 2006).

Particles emitted and formed in BB plumes have major direct and indirect effects on climate (Hobbs et al., 1997; Rosenfeld, 1999) and contribute to dense continental-scale haze layers that occupy much of the tropical boundary layer (and sometimes large parts of the boreal boundary layer) during the dry season (Andreae et al., 1988; Reid et al., 1998; Wofsy et al., 1992; Eck et al., 2003). A multipart review by Reid et al. (2005a,b) focused on the physical and optical properties of biomass burning particles and their impacts. These topics have been the subject of much ongoing research (e.g., Andreae et al., 2004; Ramanathan and Carmichael, 2008; Grieshop et al., 2009).

The trace gases emitted by biomass burning have a significant influence on the atmosphere, which includes a major contribution to the formation of global tropospheric ozone ($O_3$); an important greenhouse gas (Sudo and Akimoto, 2007). The $O_3$ formed can also affect air quality: e.g., Pfister et al. (2007) show that BB emissions from California wildfires in 2007 increased downwind ozone concentrations in rural regions. Trace gases from BB can contribute to the secondary formation of aerosol particles (Reid et al., 1998; Alvarado and Prinn, 2009; Yokelson et al., 2009). The effect of BB trace gases on the oxidizing power of the troposphere is an important, complex issue. The hydroxyl radical (OH) is a key oxidant in the global troposphere and is mostly produced in the tropics, which is also where ~70–80% of BB is thought to

occur (Crutzen and Andreae, 1990; van der Werf et al., 2010). The carbon monoxide (CO) and NMOC produced by BB are continually removed via reaction with OH while photolysis of some of the oxygenated NMOC and the $O_3$ formed in BB plumes can be an OH source (Crutzen and Andreae, 1990; Singh et al., 1995). Coupled with this picture are large tropical biogenic emissions of isoprene, which has a complex oxidation scheme that is still under investigation, but results in some OH regeneration and significant CO production (Lelieveld et al., 2008; Paulot et al., 2009; Archibald et al., 2010; Peeters et al., 2009)

Among the earliest studies to point out the importance of biomass burning on the global scale are the seminal work of Crutzen et al. (1979) and Seiler and Crutzen (1980). Major field campaigns in the 1980's and 1990's resulted in a boom in BB related publications. These are well summarized in a number of review and compilation papers, such as Haywood and Boucher (2000), Andreae and Merlet (2001), Simoneit (2002), Lemieux et al. (2004), and Reid et al. (2005a,b). The work of Andreae and Merlet (2001), in particular, continues to have widespread use in the atmospheric modeling community. For example, the EF reported therein can be combined with databases that provide estimates of global biomass consumption such as Global Fire Emissions Database (GFED, van der Werf et al., 2006, 2010) and Fire Locating and Modeling of Burning Emissions (FLAMBE, Reid et al., 2009), to produce emission estimates for atmospheric models. Despite the continued utility of previous reviews a large number of studies have been carried out since ~2000 that benefitted from advances in instrumentation and the understanding of BB plume chemistry. The results of these studies have not been conveniently compiled in one work. Thus, to aid in the assessment of biomass burning impacts in model simulations, we present an updated compilation with the following rationale:

1. In recent years, the ability has been developed to quantify a wide range of emitted species that were previously unmeasured and thus, often ignored in modeling applications.

2. The effect of rapid plume chemistry on measured emission ratios is better understood. This has led to recognition of the need to compare or combine data from smoke samples of a similar well-defined age in a standardized way. Our compilation of "initial" EF is based on measurements made in smoke that has cooled to ambient temperature, but not yet undergone significant photochemical processing.
3. Many of the studies compiled in this work sampled smoke meeting the "freshness" criteria above *and* measured a wide range of species from a large number of fires. Studies that are more comprehensive and of fresher smoke may better represent the true regional initial emissions. These EF measurements need to be compiled for convenient use in atmospheric models to promote improved modeling results and assessments.
4. With computational capacity increasing and to promote a wide variety of applications, the link between the fire emissions and the fire type needs to be available at a high level of detail, but still allow straightforward implementation of less detailed schemes. The difference between fire types is small for the EF of some species, but can be quite large for others.
5. Methods need to be developed for dealing with the abundant, but as yet unidentified NMOC, which strongly impact plume chemistry.
6. The calculation of emission rates requires emission factors to be linked to estimates of biomass consumption. Thus we also compile a large number of measurements of biomass consumption per unit burned area for major fire types and several estimates of global biomass consumption by the main fire types.
7. The emission factor tables will be updated when warranted and available at: http://bai.acd.ucar.edu/Data/fire/.

In this paper we assess the literature on BB emission factors to address the above issues. We organized the available data into 14 different categories based on the type

of fuel burned and then analyzed each study considering the different properties of the sampled fires (e.g. amount of flaming and smoldering), the attributes of the measuring platforms, instrument sensitivity, and the number of fires sampled. We carefully selected measurements in smoke roughly 5–20 min old, after immediate condensational processes on smoke particles yet prior to most of the photochemistry that can alter the composition of a plume. The age of the smoke sample is important, since current photochemical plume models and larger-scale global atmospheric chemistry models (GACMs) use the emissions as direct inputs before chemical changes occur. Despite the difficulty of modeling rapid changes occurring after emission, initial emission measurements obtained in fresh smoke, as described above, may provide the only clearly defined point in smoke evolution for a bottom-up approach. We also briefly discuss measurements in aged smoke separately to summarize our knowledge of post-emission chemistry, which is both complex and so variable that a single EF for an advanced smoke age would be highly uncertain for most species emitted by BB. This work presents a comprehensive effort tying together recent measurements of emission factors, fuel loadings, plume chemistry, and global BB estimates for the main types of biomass fires to facilitate improved understanding of regional/global tropospheric chemistry.

## 2 Methods and results

### 2.1 Terminology and the scope of this compilation

#### 2.1.1 Emission ratios, emission factors and combustion efficiency

An excess mixing ratio (EMR) is defined as the mixing ratio of species X in smoke minus its mixing ratio in background air. The EMR of X is often denoted by "ΔX," where the measured value reflects the degree of plume dilution and the instrument response time (Andreae et al., 1988; Yokelson et al., 1999). As a standardization measure,

$\Delta X$ is often divided by an EMR of a fairly non-reactive co-emitted smoke tracer ($\Delta Y$), such as CO or $CO_2$; this molar ratio is defined as the Normalized Excess Mixing Ratio (NEMR), which can be measured anywhere within a plume. A special case of the NEMR is the "emission ratio" (ER); the molar ratio between two emitted compounds (also written as $\Delta X/\Delta Y$), which should be reserved for emission measurements taken at the source (fresh smoke). The NEMR is highly variable for reactive gases and some aerosol species downwind from fires and dependent on the details of the post-emission processing (see Sect. 3.3). Thus for a reactive compound, a NEMR measured downwind may not be equal to the emission ratio even though it is expressed in similar fashion. A simpler alternative term sometimes used to refer to downwind NEMR is the "enhancement ratio" (Lefer et al., 1994), but since it would have the same abbreviation as "emission ratio" and some species are "depleted" downwind we do not use this term in this work.

We use ER to derive emission factors (EF) in units of grams of X emitted per kilogram of dry biomass burned using the carbon mass balance method (Ward and Radke, 1993) with explicit equations shown elsewhere (e.g., Yokelson et al., 1999). The method assumes that all burned carbon is volatilized or contained in the emitted aerosol and that all major carbon-containing species have been measured. The inability to detect all carbon species can inflate emission factors by 1–2% when using the carbon mass balance method (Andreae and Merlet, 2001). The carbon content in the fuel must also be measured or estimated. In this study we assume a 50% carbon content by mass (dry weight) when a measured value is not available. Except for organic soils and dung, the carbon content of biomass normally ranges between 45 and 55% (Susott et al., 1996; Yokelson et al., 1997; McMeeking et al., 2009). EF scale linearly in proportion to the assumed fuel carbon fraction. Our calculation of EF from charcoal kilns (in units of g X per kg charcoal made) reflects the changing carbon content during the kiln lifetime, as detailed by Bertschi et al. (2003a) and briefly discussed in Sect. 2.3.9.

Combustion efficiency (CE) – the fraction of fuel carbon converted to carbon as $CO_2$ – can be estimated from measured emission ratios with the detailed equation given elsewhere (e.g., Sinha et al., 2003). The CE at any point in time during a fire, or for the fire as a whole, depends strongly on the relative contribution of flaming and smoldering combustion, with a higher CE indicating more flaming (Ward and Radke, 1993; Yokelson et al., 1996). Flaming combustion involves rapid reaction of $O_2$ with gases evolved from the solid biomass fuel and is common in foliage or dry, small diameter aboveground biomass. Flaming combustion converts the C, H, N, and S in the fuel into highly oxidized gases such as $CO_2$, $H_2O$, $NO_x$, and $SO_2$, respectively, and produces most of the black (or elemental) carbon particles. As a fire progresses smoldering combustion tends to play a more dominant role via surface oxidation (or gasification, commonly known as "glowing") and pyrolysis (mostly the thermal breakdown of solid fuel into gases), often affecting large-diameter aboveground biomass and belowground biomass. Smoldering produces most of the CO, $CH_4$, NMOC, and primary organic aerosol. Smoldering and flaming frequently occur simultaneously during a fire, and distinct combustion phases may not occur. Flaming (~1400 K) and glowing (~800–1000 K) are the two heat sources driving pyrolysis and fuel temperatures can range from unheated to that of a nearby heat source. The widely used term "fire temperature" is based on the amount of 4-micron radiation emitted by a geographic area containing a fire and may not reflect the relative amount of flaming and smoldering (Kaufman et al., 1998). We also note that smoldering is not caused by a deficiency of $O_2$; rather chemisorption of $O_2$ on char is exothermic and helps drive glowing combustion (Yokelson et al., 1996). Depletion of $O_2$ was measured at only a few percent or less within intense, open fires and $O_2$ levels may not have a large affect on the gas-phase species emitted by fires (Susott et al., 1991). Large natural variability in fuel geometry, growth stage, moisture, windspeed, etc. causes large natural variability in the relative amount of biomass consumption by flaming and smoldering combustion; even within a single fire type category. This, coupled with variation in fuel chemistry, leads to a large range in the naturally occurring EF for most species for any fire type as discussed more below.

The combustion efficiency, as stated above, can be useful in indicating the relative abundance of flaming and smoldering combustion. Since CE is hard to measure, the modified combustion efficiency (MCE), which is defined as $\Delta CO_2/(\Delta CO_2+\Delta CO)$, is commonly reported as an estimate of CE accurate within a few percent (Ferek et al., 1998). Pure flaming has an MCE near 0.99 while the MCE of smoldering varies over a larger range (~0.65–0.85), but is most often near 0.8. Thus an overall fire-integrated MCE near 0.9 suggests roughly equal amounts of biomass consumption by flaming and smoldering. Since both CE and MCE indicate the relative amount of flaming and smoldering combustion, both parameters often correlate reasonably well with EF (Fig. 4.3 in Ward and Radke, 1993; Fig. 3 in Yokelson et al., 2003). For example, in Fig. 3 of Yokelson et al. (2003) airborne measurements of EF($CH_4$) for individual fires range from ~0.5 $\text{g kg}^{-1}$ to ~3.5 $\text{g kg}^{-1}$ (a factor of 7) with decreasing MCE. Additional variation in EF and MCE would result from considering the unlofted emissions from residual smoldering combustion (RSC) (see, e.g., Bertschi et al., 2003b; Christian et al., 2007; Yokelson et al., 2008). In general, the MCE dependence of "EF(X)" for a fire type allows calculation of a specific EF(X) for any known MCE. However, we do not yet have good data on how regional average MCE may evolve with time over the course of the biomass burning season for the major types of burning. Thus, in this work we only report average EF for each fire type and (where possible) a very rough estimate of the expected naturally occurring range in the average EF appropriate for a typical group of fires. The calculation of these values is described in detail in Sect. 2.3.

### 2.1.2 NMOC, OVOC, and NMHC

Non-methane hydrocarbons (NMHC) are defined as organic compounds excluding methane ($CH_4$) that contain only C and H; examples include alkanes, alkenes, alkynes, aromatics, and terpenes. Oxygenated volatile organic compounds (OVOC) contain C, H, and O; examples include alcohols, aldehydes, ketones, and organic acids. NMHC and OVOC together account for nearly all the gas-phase non-methane organic

compounds (NMOC) emitted by fires. The distinction is important when discussing the role of NMOCs in post-emission chemistry. All of the organic compounds are important in secondary processes such as ozone and aerosol formation, but the OVOC are more abundant (60–80% of NMOC), and the OVOC and NMHC tend to have different atmospheric chemistry (Singh et al., 1995; Finlayson-Pitts and Pitts, 2000). It is also important to note that only on the order of 50% (by mass) of the observed gas-phase NMOC can be assigned to specific compounds (Christian et al., 2003; Karl et al., 2007). The remaining unidentified species are mostly high molecular weight NMOC. The unidentified species evidently play a large role in plume chemistry (Sect. 3.3, Trentmann et al., 2005; Alvarado and Prinn, 2009). Many older NMOC measurements were made with total NMHC analyzers unable to measure total OVOC carbon with great certainty (Kállai et al., 2003). We discuss NMOC in detail and estimate total global NMOC considering the large percentage of compounds that remain unidentified in Sect. 3.2.

### 2.1.3 Common terminology used in computing regional/global emission estimates

We briefly define common terms used in quantifying biomass for emission estimates. Biomass is described as primarily live (phytomass) or dead (necromass) plant material and can be discussed as total aboveground biomass (TAGB) – referring to the litter layer and everything above – or total belowground biomass (TBGB), referring to duff, peat, organic soils, and roots (Seiler and Crutzen, 1980). Both terms are normally expressed on a dry weight basis. Fuel moisture can be calculated as (wet weight-dry weight)/dry weight, and along with fuel geometry affects what biomass is likely to burn. The term "fuel" in the forestry literature refers to only that portion of the total available biomass that normally burns under specified fire conditions (Neary et al., 2005). Thus, "fuel" and "biomass" are not equivalent terms in forestry, although they are sometimes used interchangeably by atmospheric chemists. Both fuel and biomass *loading* are typically expressed as the mass of fuel or biomass per unit area on a dry weight basis. A combustion factor is the fraction of biomass exposed to a fire that was actually consumed or volatilized. The biomass loading is often multiplied by a combustion factor to derive an estimate of how much biomass was consumed, otherwise known as the biomass consumption (per unit area). An estimate of the total combusted biomass can be obtained given biomass consumption per unit area and an estimate of the area burned. Measurements of biomass consumption per unit area burned have been published and we compile these values for several main fire types (e.g. savanna, boreal and tropical forest) in Sect. 2.4.

### 2.1.4 Sampling considerations and study selection criteria for this compilation

Smoke contains numerous species with atmospheric lifetimes ranging from microseconds to years. Other than a few continuously regenerated intermediates, current technology can only measure atmospheric species that are abundant and stable enough to have lifetimes of a few minutes or longer. In practice this means that measurements show the effects of aging for some detected species unless samples are taken within 10s of meters above lab fires or within 1–2 km of fires in the field. Under these conditions, smoke typically has CO concentrations in the range 5–1500 ppmv in the lab or on the ground, and 2–30 ppmv in airborne studies. Figure 3 in Christian et al. (2003) or Figs. 2–4 in Yokelson et al. (2008) show that field samples meeting the above “freshness criteria” can often return similar emission factors for trace gases when compared to lab studies at the same MCE. Laboratory fires sometimes tend to burn with a different average MCE than fires in similar fuels burning in the natural environment, but this can be accounted for as described in Yokelson et al. (2008).

For particles and semi-volatile organic compounds (SVOC) the picture is less clear. Particulate matter (PM) is directly emitted from fires, but can also be formed through secondary processes that may involve SVOC. The lab EF(PM) vs. MCE can be quite consistent with low-level airborne measurements of EF(PM) vs. MCE (e.g., Fig. 5 of Yokelson et al., 2008). On the other hand, Babbitt et al. (1996) compared EF($PM_{2.5}$) (particles with aerodynamic diameter $<2.5\,\mu m$) measured from 30 m towers above Brazilian fires to EF($PM_{2.5}$) measured using identical gravimetric methods from an

aircraft flying over the same fires. Except for the lowest MCE fire (perhaps the coolest smoke) the ground-based EF($PM_{2.5}$) were only about one-half those measured from the air, while the agreement for volatile trace gases vs. MCE from both platforms was excellent. In order to prevent highly sensitive particle instruments from saturating, some lab BB studies employ rapid dilution with room temperature air. This could possibly differ from the dilution/cooling regime associated with injection of emissions into the atmosphere during some real fires. Thus, for now, we favored the data from low-level airborne smoke samples (taken after any rapid initial cooling, but before most of the photochemistry) to derive our “initial emissions” of particles (and SVOC if available). Our compilation does include lab-measured EF for two scenarios: 1) when the fuel burns entirely by smoldering combustion in the natural environment (e.g. peat, duff), and 2) when the fuel burns by both flaming and smoldering, but the authors took special care to realistically replicate the natural fuel complex and they report data for compounds not yet measured in the field. For example, the lab study of Goode et al. (1999) was the first to use FTIR on grass fires, but none of the results are included because they are superseded by the FTIR-based field study of savanna fires by Yokelson et al. (2003). On the other hand, we include the lab study of Christian et al. (2003), which carefully replicated savanna fires and also features the only PTR-MS measurements of the emissions from this fire type. Finally, we point out that a lot of important work has characterized particle emissions from lab fires recently (e.g., Chen et al., 2006; Engling et al., 2006; Hopkins et al., 2007; Mazzoleni et al., 2007; Chakrabarty et al., 2010), but did not report the results as EF. Thus, it was not always clear how to incorporate these results into initial particle characteristics for models that would be analogous to the trace gas and particle EF we report.

Another important consideration for field studies is that smoldering combustion can produce unlofted smoke with low MCE that is not amenable to airborne sampling. Ground-based sampling can measure these sometimes substantial emissions, but realistic estimates of the biomass consumption contributing to the two different types of smoke are needed to properly weight the ground-based and airborne measurements

(Christian et al., 2007).

## 2.2 Fire-type categories

### 2.2.1 Vegetation

We organize the available EF for landscape-scale fires into six broad types of vegetation susceptible to burning: savanna, tropical forest, boreal forest, temperate forest, peatlands, and chaparral. We split the category "extratropical forest" used by Andreae and Merlet (2001) into "boreal" (high latitude ~50–70° forested regions) and "temperate" forest. We also derive EF for "extratropical" forest fires using a weighted average of boreal and temperate emission factors (86.5% and 13.5%, respectively) based on GFED v. 3 biomass consumption estimates (van der Werf et al., 2010) to preserve the option of using this category. We present a specific category for chaparral (a type of temperate shrubland) since the emissions from chaparral fires are important in the Southwestern US and shrublands are widespread globally (Friedl et al., 2002). (Some of the temperate forest and chaparral EF we include are flagged as preliminary in the supplementary tables.) Our category "tropical forest" includes tropical evergreen forest deforestation fires, tropical dry forest deforestation fires, and tropical dry forest understory fires. Tropical dry forest is also called "seasonal" or "monsoon" forest. Tropical dry forests (TDF) differ from "woody" savanna regions in that TDF are characterized by a significant (>60%) canopy coverage or closed canopies (Mooney et al., 1995; Friedl et al., 2002). Savanna regions are qualitatively described as grassland with an "open" canopy of trees (if any). Our savanna category includes the savanna, woody savanna, and grassland categories in the detailed MODIS land cover products (Friedl et al., 2002). Our savanna and tropical forest categories contribute the most open burning emissions globally (Andreae and Merlet, 2001). While peatlands represent 3% of terrestrial cover, they hold about one third of the world's soil carbon (Rein et al., 2009; Yu et al., 2010) and can be a significant contributor to annual carbon emissions (Page et al., 2002).

### 2.2.2 Biofuel

We use "biofuel" as a specific term denoting biomass used as a domestic or industrial energy source. In assessing the impacts of biofuel it is worth recalling that, in principle, it could be regrown so is potentially "renewable" unlike fossil fuel. Rural populations in developing countries rely heavily on biomass burning as a primary source of energy (Smil, 1979; Cecelski et al., 1979; Yevich and Logan, 2003). The amount of biofuel use in urban areas of the developing world is not known, but may be significant (Christian et al., 2010). Over the 50-year period from 1950–2000, Fernandes et al. (2007) estimated a 70% growth in global annual biofuel consumption making it now the second largest type of global biomass burning after savanna fires (Andreae and Merlet, 2001), but future trends are hard to predict. In this work we present biofuel emission factors for open cooking fires, dung burning, Patsari cooking stoves, charcoal making, and charcoal burning. Open cooking fires are the single largest contributor to global biofuel emissions accounting for roughly 80% of current biofuel use worldwide (Dherani et al., 2008). Various stove designs are available, but the most complete emissions measurements have been made for Patsari stoves; therefore, we selected them to represent emissions from all types of solid-fuel stoves. Patsari stoves are used in Mexico and incorporate an insulated fire box that vents emissions outdoors via a metal chimney (Christian et al., 2010). The stoves are designed to replace traditional open three-stone fires and can reduce indoor air pollution by 70%. Stoves in general require less fuel per cooking task than open cooking fires, which reduces emissions and pressure on biofuel sources (Johnson et al., 2008; Masera et al., 2005; Zuk et al., 2007). For the above reasons there is considerable international activity to encourage switching from open cooking fires to stoves. In addition, the Patsari stove emissions were found to have different chemistry than open cooking fire emissions (Johnson et al., 2008; Christian et al., 2010), further justifying a separate category in this study. While not fully representative of all cooking stoves, the Patsari stove EF likely represent most stove emissions better than EF for open cooking fires and might be used to help assess the impact of changes in how biofuel is used.

Dung as a biofuel is mainly of note in Asia, dominated by use in India and China (Yevich and Logan, 2003). Its use in most other rural areas globally is less common than that of woodfuel (though sometimes still significant), and overall it comprises approximately 5% of the total dry matter burned as biofuel (Yevich and Logan, 2003). Charcoal is mainly produced in rural areas and often consumed in urban areas, accounting for ~10% of global biofuel use (Bertschi et al., 2003a).

### 2.2.3 Agricultural/waste burning

Crop residue and pasture maintenance fires and open burning of garbage can be common both in rural agricultural regions and peri-urban areas. For instance, sugarcane burning is the main source of PM in some Brazilian cities (Lara et al., 2005; Cançado et al., 2006). Crop residue burning has been estimated as the fourth largest type of biomass burning (Andreae and Merlet, 2001), but these emissions could be greatly underestimated given the difficulty of detecting these often short-lived, relatively small fires from space (Hawbaker et al., 2008; Smith et al., 2007; Chang and Song, 2010; van der Werf et al., 2010). Crop residue may be burned 1–3 times a year on a single site depending on the rate of annual harvest. Some crop residue is utilized as biofuel (especially in China), blurring the distinction between these categories (Yevich and Logan, 2003). A recent increase in crop residue burning is likely in large areas of the Amazon concurrent with a shift in land use from cattle ranching to crop production (Cardille and Foley, 2003; Morton et al., 2006). Pasture maintenance burning is performed every 2–3 years to prevent reconversion of pasture to forest. These fires frequently include residual smoldering combustion (RSC) of large logs that can burn for weeks after the flames have ceased (Kauffman et al., 1998). Garbage burning is normally overlooked as an emissions source. However, Christian et al. (2010) estimate that ~2000 $\text{Tg yr}^{-1}$ of garbage are generated globally and roughly half may be burned in open fires or incinerators. Partly because open garbage burning is often illegal, it is unmentioned in most inventories. We compile the few available EF for open burning of

garbage as a separate category.

### 2.3 Assessment, calculation, and application of emission factors for specific fire types

This section provides the details of how we analyzed the emission factors. We classify biomass burning into 14 categories. For each of these categories, we organize the information by study in Supplementary Tables S1–S14 for all studies meeting our selection criteria (updates at: http://bai.acd.ucar.edu/Data/fire/). For each included study we show the study-average emission factors and any additional specifics considered in calculating an overall average and estimate of the natural variation for the whole category. The rationale supporting the calculation of the category average and variation is summarized in the following sections. We present just the category average emission factors and category variability for all 14 BB categories in Tables 1 and 2. Our classification scheme allows consideration/assessment of fairly specific emission types while retaining the option of merging categories at the user's discretion. As an example, we also derive values for an "extratropical forest" category (shown in Table 1) by merging the boreal and temperate forest EF with the formula described in Sect. 2.2.1. Some users may instead desire EF in more detail than is provided by our 14 categories in Tables 1–2 and this can often be retrieved by consulting the supplementary tables. For instance, the EF for smoldering combustion of hand-piled crop residue (common in much of Asia) are very different from the EF for flaming combustion of crop residue produced by mechanized agriculture and they can be found separately in Table S13.

### 2.3.1 Savanna

The emission factors from one laboratory study and three airborne studies of savanna fires are presented and averaged in Table S1. The savanna fire average and variation is also reported in Table 1. We make several points about two of the included studies next. During the Smoke, Clouds, and Radiation-Brazil (SCAR-B) campaign, airborne EF measurements were made of fresh smoke from several different fire types. However, the EF were originally published as the overall regional average emission factors for the combination of all the different fire types observed (Ferek et al., 1998). We broke out the original fire-specific SCAR-B EF into the appropriate fire-type categories in our classification scheme based primarily on the recorded visual observations from the aircraft. However, the delineation between TDF and "wooded" or "humid" savanna fires was difficult from the aircraft and the distinction is often unclear in the literature as well. We categorized three of the SCAR-B fires as savanna fires because the Advanced Very High Resolution Radiometer (AVHRR) Continuous Fields Tree Cover product (DeFries et al., 2000) showed that the area burned had a pre-fire canopy coverage less than 40% (Matthews, 1983; Hansen et al., 2000). We used the AVHRR product because the fires burned prior to coverage by the MODIS VCF product (Hansen et al., 2003). The gas and particle emissions data from the SCAR-B fires are also converted to units of g compound/kg fuel. "Xylenes" are calculated from the sum of *p*-xylene, *m*-xylene, and *o*-xylene. Since NO and $NO_2$ are rapidly interconverted in the atmosphere, we also calculate and report an EF for "$NO_x$ as NO." The estimate of the variation in the EF is taken as the standard deviation of the EF. The volume distribution for BB particles by aerodynamic diameter shows a minimum from about 1–5 µm (Ward and Radke, 1993). Thus, in all our tables, measurements of $PM_{1.0}$–$PM_{5.0}$ are grouped together as $PM_{2.5}$ to allow averaging data from more studies. We also note that $PM_{2.5}$ is usually close to 80% of $PM_{10}$ or TPM when measured on the same BB sample (e.g., Artaxo et al., 1998). Finally, we group EF reported for elemental carbon (EC) or black carbon (BC) in a single "BC" category, although differences between the measurement techniques used for these species are the subject of ongoing research (Reid et al., 2005 a,b; Bond and Bergstrom, 2006; Schwarz et al., 2008).

We include EF from Christian et al. (2003) who measured emissions from burning grass and/or twig/leaf-litter fuels from Zambian humid savannas in 16 laboratory fires. Their reported uncertainty is ±37% factoring in 31% naturally occurring variability in NMOC (Yokelson et al., 2003), 15% prediction error (reflecting the uncertainty in using

lab data to predict field emission factors for this fire type), and 5% error in measurement. Alang-alang (*Imperata cylindrica*) is a widespread fire-maintained grass subject to frequent burning in Indonesia (Jacobs, 1988; Seavoy, 1975; Pickford et al., 1992) that was burned in five fires by Christian et al. (2003) and we categorize it as a savanna-type fuel. Most of the data in Christian et al. (2003) were collected using an open path FTIR (OP-FTIR) and a proton-transfer-reaction mass spectrometer (PTR-MS). For this study and all the other fires sampled by both FTIR and PTR-MS, we report only FTIR data when EF from both instruments were originally reported and more than one species could contribute to a mass peak. The synthesis of data from various instruments is described in more detail in the original papers. The EF(HCOOH) that originally appeared in Christian et al. (2003) and other FTIR-based studies published up to January 2010 have been rescaled to be consistent with a new reference spectrum for HCOOH (Rothman et al., 2009).

For this category and for the other categories, when sufficient data are available, we provide a rough estimate of the "naturally-occurring variation" in the average EF for a group of fires within the classification. It is common to report variability as "uncertainty," but the measurement uncertainties associated with calculating individual EF are generally quite low for the studies we include in this compilation. We adopted a relatively simple, approach to estimate the variability, which is described next in order of increasing complexity:

*The case when only one study is available:*

1. If there is only one EF value available, we do not estimate variability.
2. If there are only two EF values available, we estimate variability as the range.
3. If two or more EF values are given and both provide an estimate of variation, we average them to estimate variability.
4. If three or more EF values are given in just one study, we estimate variability as the standard deviation of the EF.

*The case when two or more studies are available*:

5. If more than one study reports EF, but only one study provides an estimate of variability, we estimate variability using the fractional variability from the one study provided.

6. If more than one study reports EF and an estimate of the variability, we took the average variability as our estimate of variability (we find that the range or standard deviation of study means can sometimes significantly underestimate natural variability).

7. When more than one study was available and there was a large difference in the amount of sampling between studies, we weighted the EF by the amount of sampling to derive a final average EF value reported in our tables, but our estimates of variation were obtained as above (without weighting).

8. Variability in total NMOC was taken as the sum of the variability of each individual NMOC (we find that equations propagating fractional uncertainties overemphasize the impact of compounds measured in low abundance on total variation).

Users preferring an alternate calculation of averages or variation can implement their scheme using the original data, which can be found in Supplementary Tables S1–S14.

### 2.3.2 Boreal forest

Boreal forest fires can consume large amounts of both above-ground and below-ground biomass (Ottmar and Sandberg, 2003; French et al., 2004). We include lab or ground-based measurements of EF for burning organic soils, peat, and woody/down/dead vegetation; the latter term including stumps, logs, and downed branches (Table 1; Table S2). Such fuels are likely to burn by RSC, which can continue long after flaming and strong convection from a fire have ceased (Bertschi et al., 2003b). We computed the average for five common components of the fuel in boreal organic soils that were

burned individually by Bertschi et al. (2003b) (identified as Lolo 1, 2, 3, NWT 1, 2 in original work) and took the standard deviation as the variability. Emissions from burning organic soil from Alaska (identified as sedge, sphagnum moss, feather moss, white spruce, and forest floor duff) were reported by Yokelson et al. (1997), for which we compute EF using the reported C content. Yokelson et al. (1997) also reported emissions measurements for boreal peat from Alaska and Minnesota. Given that the %C was not measured for Alaskan peat, we used the measured %C for MN peat (49.4%) in all of the boreal peat EF calculations. We also include Alaskan duff EF measured in a laboratory by Burling et al. (2010). Bertschi et al. (2003b) reported EF for woody/down/dead fuels (identified as Stump and Cwd 2), which are also included here. We are unaware of any measurements of the relative consumption of the different organic soil and woody fuel components for "typical boreal fires" so a straight average of the EF for the organic soil/duff and dead and down component in these lab/ground studies was used for a ground-based average (as shown in Table S2).

Four studies reported airborne measurements of boreal forest fire EF in fresh smoke for an extensive number of compounds. We include the average of the emission factors from three fires (B280, B349, and B309) sampled by Goode et al. (2000) (fire B320 was not included since the fuels were not representative of a boreal forest, see original work). Nance et al. (1993) and Radke et al. (1991) also reported boreal airborne EF measurements for one wildfire and four prescribed fires, respectively. These are included in this compilation. We include data collected from boreal wildfires measured during the Arctic Research of the Composition of the Troposphere from Aircraft and Satellites (ARCTAS) campaign during the summer of 2008 (I. Simpson, work in progress; Wisthaler and Mikoviny, unpublished results). Whole air samples (canisters) were collected onboard the DC-8 aircraft during Flights 17, 18, and 19 over Saskatchewan, Canada. Emission factors for long-lived species were based on all the canisters collected in 5 plumes. EF of "short-lived" ($k_{\mathrm{OH}} \geq 8.52 \times 10^{-12}\ \mathrm{cm}^3\ \mathrm{molecule}^{-1}\ \mathrm{s}^{-1}$) species were calculated using only samples of fresh smoke collected <7.3 km from the source of the McKay fire during Flight 18.

Due to slow decomposition large amounts of carbon are stored in boreal soils and dead/down woody fuels that burn by smoldering combustion and thus account for much of the total fuel consumption. To take this into consideration, we calculate the overall boreal forest fire EF in Table 1 (and Table S2) from a straight average of the airborne and ground-based EF averages, which is roughly equivalent to assuming ~70% of the fuel consumption is by smoldering combustion. In fact, data in Table 4 of van der Werf et al. (2010) suggest that an even larger weighting of the ground-based "smoldering" EF could be appropriate. Using Table S2, EF can be calculated based on other relative contributions of flaming and smoldering. Some smoldering compounds were measured only from the air. For these compounds we multiply the airborne EF by the average ratio of the "50–50" average to the airborne average (2.02±0.070). A similar approach is used for smoldering compounds measured only from the ground; they were multiplied 0.70±0.11 to obtain the 50–50 average. Finally, we point out that the flaming and smoldering emissions can have different injection altitudes which could be important to recognize in some applications.

### 2.3.3 Tropical forest

Our emission factors for tropical forest fires are in Table 1. We derive the EF by averaging over several types of tropical forest fires in Table S3. For tropical evergreen forest deforestation fires we include EF cited in Yokelson et al. (2008) and retrieved from the original SCAR-B data of Ferek et al. (1998). The EF in Yokelson et al. (2008) (all from the Tropical Forest and Fire Emissions Experiment (TROFFEE)) were derived using a 0.05/0.95 weighted average of the Christian et al. (2007) ground-based measurements (dominated by residual smoldering combustion) and the Yokelson et al. (2007a) airborne measurements (dominated by flaming combustion). For these types of fires, available evidence suggests that approximately 5% of biomass consumption is by residual smoldering combustion and 95% of consumption is during the convective plume forming phase of the fire (Christian et al., 2007). Taking a weighted average of the EF for residual smoldering with the EF for initially lofted emissions gives EF for

smoldering compounds that were 1.12±0.11 times higher on average than the EF from just the airborne data. Thus, since some smoldering compounds were measured only in the air, their emission factors are computed from 1.12 times the airborne average in both this work and Yokelson et al. (2008). Data from Andreae and Merlet (2001) are included for $H_2$. The EF and variation for pyrrole can be found in the discussion version of Yokelson et al. (2007a). The variation for all the other species is taken as the standard deviation in the airborne EF in Table 2 of Yokelson et al. (2007a). We also average EF from 12 airborne samples of fires from the SCAR-B campaign (Ferek et al., 1998) that represented emissions from six flaming and six smoldering fires classified as tropical evergreen deforestation fires. We do not make the small adjustment to the smoldering compounds for RSC in the SCAR-B data (Ferek et al., 1998). Average EF for the "subcategory" tropical evergreen deforestation fires are included in Table S3.

For tropical dry forest (TDF) fires we consider both deforestation and understory fire emissions. The studies we include are Yokelson et al. (2009), Ferek et al. (1998), and Sinha et al. (2004). Yokelson et al. (2009) sampled five TDF deforestation fires in the Yucatan. These were early dry season fires, which should help the average EF we derive for this category reflect the entire dry season since the other studies measured EF only during the late dry season. We calculate an EF for nitrous acid (HONO) from the mass emission ratio $\Delta\mathrm{HONO}/\Delta\mathrm{NO_x}$ measured on one tropical dry deforestation fire (Fire #2 on 23 March 2006 from Yokelson et al., 2009) times our average EF($NO_x$) for TDF. We classify three SCAR-B fires from Ferek et al. (1998) as TDF fires and estimate the variation as the standard deviation of these EF. Sinha et al. (2004) measured numerous emissions from one African tropical dry forest (Miombo) understory fire. That work includes an EF for condensation nuclei in the diameter range 0.003–3 µm expressed as number of particles $\mathrm{kg^{-1}}$ fuel burned. Finally, in theory, to derive average EF for tropical dry forest fires from the available measurements we would need to know the relative importance of understory and deforestation burns in this ecosystem (Desanker et al., 1997). Since this information is not available to our knowledge, we weight them equally here to obtain average EF for TDF. We then weight all the studies in Table S3 equally to obtain the tropical forest fire average EF that we carry over to Table 1.

### 2.3.4 Temperate forest

We include the average and standard deviation of EF from two temperate evergreen forest fires and one temperate evergreen prescribed fire measured by Radke et al. (1991), as seen in Table S4 and Table 1. We also include preliminary EF from two prescribed understory fires in coniferous forest in the Sierra National Forest measured during the SLOBB campaign (San Luis Obispo Biomass Burning study) (I. Burling, work in progress).

### 2.3.5 Peatland

Peat burns almost entirely by smoldering combustion. Christian et al. (2003) made laboratory measurements on a single Indonesian peat fire using OP-FTIR and PTR-MS. We obtained EF($CH_3OH$) by averaging data collected by both instruments since there are no interferences for methanol on the PTR-MS. For all other species we used OP-FTIR emission factors when data from both instruments was obtained. We provide no estimate of variation for EF from Christian et al. (2003) as only one fire was measured, though a general range of at least 20–40% uncertainty could be assumed. The boreal peat EF reported in Yokelson et al. (1997) and considered in calculating the boreal forest EF (Table S2) are also used in computing our global peatland EF in Table S5. The Indonesian peat sample had a 54.7% carbon content, which contributed to a significantly higher EF($CO_2$) compared with boreal peat, but we don't imply that tropical peat always has higher C content. We calculated the average peat EF in Table S5 by averaging the studies of boreal (Yokelson et al., 1997) and Indonesian (Christian et al., 2003) peat and estimate an average variability from the fractional variation in EF in Yokelson et al. (1997). Smoldering peat accounts for the bulk of the emissions from most fires in peatlands and our average peatland EF in Table S5 are based only

on the smoldering peat measurements. However, Page et al. (2002) estimated that 0.19–0.23 Gt of carbon was released into the atmosphere through peat combustion in tropical peat swamp forests, while 0.05 Gt of carbon was released from overlying vegetation during the 1997 El Niño year in Central Borneo. From these estimates we took a weighted average of the peat EF (73%) in Table S5 with the tropical evergreen forest deforestation fire EF (27%) in Table S3 to derive an average shown in Table 2 that accounts for consumption of a (tropical) forest overstory. The user can apply the average EF most suited to their application.

### 2.3.6 Chaparral

We include the average EF from three studies that measured emissions from California chaparral fires. The average EF from three fires sampled by Radke et al. (1991) was taken (Eagle, Lodi 1, and Lodi 2). We converted their EF($NO_x$), which assumes a 50/50 mix to an EF for "$NO_x$ as NO" by multiplying their original EF by a mass factor of (30/38). We include the emission factors from a laboratory study that sampled ~40 carefully replicated fires in six types of chaparral fuels at the Fire Sciences Lab in Missoula, MT (Burling et al., 2010). We also include the average EF from three chaparral fires measured during the SLOBB field campaign (I. Burling, work in progress). The emission factors from the latter campaign are flagged as preliminary, but subject to only minor changes by the time of publication.

### 2.3.7 Open cooking

Christian et al. (2010) reported the average EF of eight open cooking fires sampled in Mexico. Brocard et al. (1996, 1998) reported the average emission factors for 43 open cooking fires in Ivory Coast. We multiply their EF and variation by $MW_X/MW_C$ to convert from g C kg$^{-1}$ dry fuel to g X kg$^{-1}$ dry fuel. Some updated EF reported by Brocard et al. (1998) supersede those found in Brocard et al. (1996). Smith et al. (2000) sampled six open cooking fires in a lab using Indian fuels: variation was taken as the fractional variation in ER as seen in Bertschi et al. (2003a). We include the EF measured by Zhang et al. (2000) for wood burning in open stove types in China and EF measurements made in Honduras for traditional open "stoves" (designated as "no chimney") from Roden et al. (2006, 2009). $CO_2$ data were not included in the published work of the latter, but were graciously provided by Tami Bond and Christoph Roden. Johnson et al. (2008) included EF for 8 open cooking fires in Mexico, which we convert from g C kg$^{-1}$ fuel to g X kg$^{-1}$ fuel. Bertschi et al. (2003a) report the average EF for three open wood cooking fires in Zambia and we estimate variation from the fractional variation in their ER. We weight all 8 included studies equally to obtain the average EF shown in Table S7 and Table 2.

### 2.3.8 Patsari cooking stoves

We assume a fuel C content of 50% when converting all cook stove ER to EF. Christian et al. (2010) analyzed 26 samples collected from chimney outlets of two Patsari stoves in Mexico. Our estimate of variation is the range in the two EF measurements. We also include Patsari stove EF measurements from Johnson et al. (2008) made in 13 homes in Mexico. We report the overall Patsari stove average emission factors and variation in Table 2 and Table S8.

### 2.3.9 Charcoal making

Most of the global charcoal production is carried out in temporary kilns constructed mainly from dirt (Bertschi et al., 2003a). Charcoal making EF have been reported in the literature in at least four types of units: g compound or g C emitted, referenced to either kg of wood used or kg of charcoal made. We convert as needed and report all EF here in units of g compound kg$^{-1}$ charcoal produced. In Bertschi et al. (2003a), the kiln was charged with a tree species with a known carbon content of 48% (Susott et al., 1996). Coupling several other studies they concluded that ~45% of the wood carbon is given off as gases so that approximately 216 g C is volatilized per kg of dry wood used.

Dividing up those 216 g according to their measured ER (which included the major emissions $CO_2$, CO, and $CH_4$) then allowed straightforward calculation of the reported EF per kg wood used. Conversion to EF per kg charcoal produced was based on assuming an average charcoal yield per mass of dry wood of 28%, a factor that varies little between the many reported measurements (Bertschi et al., 2003a; Chidumayo, 1994; Pennise et al., 2001; Lacaux et al., 1994; Smith et al., 1999). Bertschi et al. (2003a) obtained their ER from averaging three 1–2 h measurements made on one kiln on three different days spread over the 4 days required to produce a batch of charcoal. They then derived EF as just described. Christian et al. (2010) made 36 spot measurements of ER (with ~1 min sampling time) during days 2–5 from three kilns that had 8 day lifetimes and then converted to EF with the procedure of Bertschi et al. (2003a). Our estimate of variation for Bertschi et al. (2003a) and Christian et al. (2010) is the fractional uncertainty in ER. Christian et al. (2007) made three spot measurements (1 min sampling time) from a single kiln in Brazil; however, measurements were made only in the last stage of the kiln lifetime and may not be representative of emissions occurring throughout the charcoal making process. The FTIR-based studies of Bertschi et al. (2003a) and Christian et al. (2007, 2010) measured a substantially different suite of NMOC than the other available studies and also differed in sampling approach so data from these 3 studies was averaged together separately using the weighting factors described next. Since Christian et al. (2007) collected only three 1 min spot measurements, we employed a weighting factor (4%) based on the minutes of actual sampling. The kiln measurements of Christian et al. (2010) and Bertschi et al. (2003a) were roughly equivalent in the extent of sampling and were weighted equally at 48%. The FTIR-based average values were then averaged with 4 other studies to obtain the overall charcoal making EF shown in Table S9 and Table 2. The four additional studies are described next. Lacaux et al. (1994) continuously monitored the emissions from a charcoal kiln in the Ivory Coast over its whole "lifetime." We convert their emission factors from g C kg$^{-1}$ wood used to g X kg$^{-1}$ charcoal produced as described above. We assume that any differing EF found in a later paper that discusses that project

(Brocard et al.,1998) supersede those found in Lacaux et al. (1994). We also include EF from Smith et al. (1999) and Pennise et al. (2001) measured in Thailand and Kenya, respectively.

### 2.3.10 Charcoal burning

We report all EF in units of g compound kg$^{-1}$ charcoal burned (Table 2 and Table S10). Unless otherwise stated, the charcoal fuel carbon content was assumed to be 72±3% (Lacaux et al., 1994; Chidumayo, 1994; Ishengoma et al., 1997; Smith et al., 1999). We recalculate the EF from the ER reported in Bertschi et al. (2003a) and a few of our EF values differ slightly from those originally reported in their work. No variation was reported for the Bertschi et al. study as emissions were measured from only one fire. Brocard et al. (1998) reported ER and fractional variation in those ER for charcoal burning, which we converted to EF. For the compounds they reported relative to $CO_2$, we estimate variation from the fractional variation in the ER. To estimate variation for the compounds they reported relative to CO, we also consider their uncertainty in $\Delta CO/\Delta CO_2$. We include Smith et al. (2000) and Kituyi et al. (2001) EF measured in India and Kenya, respectively.

### 2.3.11 Dung

Keene et al. (2006) reported the median EF of gases and total particulate carbon from two laboratory fires that burned dung collected in India. Christian et al. (2007) measured the emissions from three burning cattle dung piles encountered on a subsistence farm in Brazil. We calculate all EF assuming a 32.6% fuel carbon content on a dry weight basis, as reported by Keene et al. (2006). We compute a weighted average based on the number of samples from each study (Table S11 and Table 2).

### 2.3.12 Pasture maintenance

In Brazil many cattle ranches have been established in areas that were previously tropical forest. Pasture maintenance fires are used to prevent the re-establishment of the forest and they burn both grass and residual wood from the original forest. Within Brazil, these fires are estimated to consume as much biomass annually as primary deforestation fires (Kauffman et al., 1998). Pasture maintenance fires are thought to be much less abundant in most other tropical forest areas. Yokelson et al. (2007a) sampled one Brazilian pasture fire from an airborne platform (Table S12). We include no estimate of variation as only one fire was measured. The EF for pyrrole for that fire was reported in the discussion version of Yokelson et al. (2007a). The SCAR-B study of Ferek et al. (1998) included airborne samples of six pasture fires. We use the standard deviation in EF from their pasture fire measurements to estimate the variability in EF and we compute a weighted airborne average EF based on the number of fires sampled in these two studies. A significant fraction of the fuel consumption in pasture fires produces unlofted emissions via residual smoldering combustion (RSC) of the residual woody debris (RWD) from the former forest (Barbosa and Fearnside, 1996; Guild et al., 1998; Kauffman et al., 1998). These emissions must be sampled from the ground. We are aware of one ground-based study (Christian et al., 2007) that reported EF for RSC of RWD in pastures and we also obtained originally unpublished EF from that study for "$NO_x$ as NO," NO, and $NO_2$. For all species with both airborne and ground-based data we obtained a "EF(total)" for pasture fires from a weighted average based on the assumption that 40% of the fuel consumption was by RSC and 60% generated lofted emissions that could be sampled from the air (Christian et al., 2007).

Some compounds were measured only from the air. The EF(total) for the smoldering compounds that were measured only from an aircraft is estimated by multiplying the average EF(air) by 2.00±0.90, which was the average value of the ratio EF(total)/EF(air) for smoldering compounds not containing N that were measured from both platforms

(Yokelson et al., 2008). Two flaming compounds were measured only from the air. EF(total) for $SO_2$ is estimated by multiplying EF(air) for $SO_2$ by EF(air)/EF(total) for $NO_x$ which was measured from both platforms. Our estimate of EF(HONO) is obtained by multiplying the $\Delta HONO/\Delta NO_x$ mass ER in Yokelson et al. (2007a) times our final EF($NO_x$). Two smoldering compounds were measured only on the ground. EF(total) for acetol (1-hydroxy, 2-propanone, $C_3H_6O_2$) and phenol ($C_6H_6O$) are estimated by multiplying the EF(ground) times the average EF(total)/EF(ground) for the (non-N) smoldering compounds measured from both ground and air. We use the fractional variation in the ground-based EF to estimate the variation in species with ground or both ground and airborne data, since ground-based data appear to have greater variability than airborne data (see Figs. 2 and 4 in Yokelson et al., 2008). For species with only airborne data we estimate the uncertainty as 45% (Yokelson et al., 2008) (Table 1).

#### 2.3.13 Crop residue

Post harvest crop residue is a fine fuel that burns directly in the field and mostly by flaming in many mechanized agricultural systems. In contrast, when crops are harvested by hand the residue is often burned in large piles that may smolder for weeks. Yokelson et al. (2009) reported emission factors from airborne measurements of six crop residue fires associated with mechanized agriculture in the Yucatan, Mexico (Table S13). Christian et al. (2010) made ground-based measurements of EF from two similar burns in Central Mexico. Christian et al. (2003) measured the mostly smoldering emissions from three laboratory fires burning piled Indonesian rice straw by open path FTIR (OP-FTIR) and PTR-MS. For EF(final), the studies are weighted based on the number of fires sampled for the final reported EF in Table 1, which is equivalent to assuming a mix of manual (3/11) and mechanized (8/11) harvesting. If only one form of agriculture is known to dominate for an application, one could select the data from either the mechanized or manual agriculture studies in Table S13.

We also note that our usual approach for estimating total NMOC (summing the average values) was potentially inappropriate due to a very high acetol EF for smoldering

rice straw and a resulting EF for identified NMOC of 79.26 g $kg^{-1}$ that may be too large to represent global CR fires since acetol was not detected from the other crop residue fires. It was also not ideal to average the sum of identified NMOC from each study (33.96 g $kg^{-1}$) since two of the studies measured very few compounds due to minimal instrumentation. We used the average EF(NMOC) of the two approaches (56.61 g $kg^{-1}$), which may represent a relatively unbiased estimate of EF(NMOC) for global crop residue fires. No uncertainty is provided but it can be assumed to be large.

#### 2.3.14 Garbage burning

We consider field and laboratory measurements from Christian et al. (2010) and Lemieux et al. (2000), respectively. Christian et al. (2010) made 72 spot measurements at four Mexican landfills using a rolling, land-based FTIR and filter sampling apparatus. Emission factors were computed assuming the landfill waste was 40% C by mass. Their estimate of EF($PM_{2.5}$) is the sum of particle components measured on quartz filters with a small allowance for unmeasured species (Christian et al., 2010). Our estimate of variation is the standard deviation of the EF from the four fires measured. We report the average EF($PM_{2.5}$) and EF(HCl) from Lemieux et al. (2000) for the burning of recycled and non-recycled waste in barrels. We obtain the average from four "runs" – emissions from two avid recyclers and two non-recyclers – with $PM_{2.5}$ emissions from non-recyclers notably higher than those of avid recyclers (see Table 1 in Lemieux et al., 2000 for study details and garbage composition). We include airborne EF measurements from a garbage burning fire in Mexico (R. Yokelson, work in progress). We also include the few available USEPA AP-42 EF for open burning of municipal waste.

### 2.4 Estimates of biomass loading and biomass consumption

To project total emissions from a fire or region the EF presented above must be multiplied by the mass of biomass consumed in the fire or region. For open burning the total mass of biomass consumed is usually estimated from the product of two other estimates: 1) the mass of biomass consumed per unit area, and 2) the area burned. Airborne or ground-based measurements of the area of individual burn scars can be fairly accurate, but they are usually not available for the tropics and space-based measurements of burned area are still highly uncertain (Korontzi et al., 2004; Roy et al., 2009; Giglio et al., 2006, 2010). The biomass consumption per unit area has been measured for examples of most major types of open burning. Another approach involves calculating the fraction of the total biomass that was exposed to a fire that actually burned to determine a "combustion factor" (sometimes called "combustion completeness"). The combustion factor (CF) can then be multiplied by spatially varying estimates of biomass loading (Brown and Lugo, 1992; Brown, 1997) to estimate the biomass consumption per unit area for any burned location. The CF need not be a constant for an ecosystem. The small diameter biomass components in a "fuel complex" tend to have larger CFs than the larger diameter biomass components (Table 2 in Kauffman et al., 2003). Considering the season of CF measurements (available in the references for Table 3) reveals that CF tend to increase strongly as periods of dry weather lengthen and dry out the larger diameter fuels (van der Werf et al., 2006). Additional variation in CF results from natural variation at burn time in any of numerous factors that affect fire behavior such as relative humidity, temperature, winds, fuel geometry, etc. (Kauffman et al., 2003). For example, CF for Brazilian pasture fires ranged from 21–83% due mainly to variable consumption of the large diameter residual woody debris (Kauffman et al., 1998; Guild et al., 1998). In Southern Africa the percentage of available fuel that burned in understory fires in June (at the beginning of the dry season) in the Miombo tropical dry forest was 1% and 22% (*n*=2, Hoffa et al., 1999), while Shea et al. (1996) observed that 74% and 88% (*n*=2) of the understory fuels burned in Miombo fires in late August/early September (their Table 4). We have compiled and summarized many of the literature data for biomass loading, combustion factor, and biomass consumption sorted by vegetation/fire type in Table 3. GFED v. 3 estimates for biomass consumption are also shown in Table 3 whenever their regional estimates were likely dominated

by one vegetation type. GFED estimates 46% higher biomass consumption for North American boreal fuel types compared to the average of the other referenced measurements. However, estimates of Asian boreal biomass consumption by GFED lie within 4% of the average of the few measurements. A comparison for other fire types is difficult because the GFED biomass consumption data is presented by geographic regions that usually contain multiple fire types (van der Werf et al., 2010).

### 2.5 Global emission estimates

Operationally, most global models use temporally and spatially explicit products such as monthly GFED (van der Werf et al., 2006, 2010) or hourly FLAMBE (Reid et al., 2004, 2009) to generate open burning emissions over the course of a model run. However, estimates of the total annual biomass consumed globally by various fire types can be used, at the global scale, to assess the importance of various fire types, to develop emissions inventories for an average or model year, and to factor into budgets. We report several global estimates of combusted biomass (dry matter) for different fire types in Table 4. The individual estimates are based on data collected anywhere from 1987–2000, which explains some of the variability in comparisons. Global estimates from Andreae and Merlet (2001) and Bond et al. (2004) agree well for the main types of open burning: savanna, forest, and crop residue fires. Yevich and Logan (2003) estimated biofuel biomass consumption at 2447 $\mathrm{Tg\,yr^{-1}}$ for 1985, which suggested a dominant role of biofuels in global emissions even 25 years ago. They also estimated that biofuel use was growing at 20% per decade. Consistent with that growth, Bond et al. (2004) and Fernandes et al. (2007) independently estimated higher biofuel use for 1996 and 2000, respectively. If savanna burning remains constant on average, biofuel burning could overtake it as the primary source of BB emissions by approximately 2030; assuming the average emissions presented in Table 4 represent global emissions from the year 2003 with a 20% growth rate per decade. This projection is included to highlight the importance of biofuel use, but it is based mostly on past population/development trends and a rigorous projection of future trends is beyond the scope of this work. In general, large uncertainties in biofuel use stem from the difficulty in monitoring its usage in developing countries (Bond et al., 2004). The magnitude of industrial biofuel use remains especially uncertain given the diverse range of fuels used and the subjectivity of user surveys coupled with financial and legal issues for micro-enterprises, which form a large part of the economy of the developing world (Christian et al., 2010). A quantity with extreme uncertainty is the amount of global garbage burning with estimates ranging from 33–1000 $\mathrm{Tg\,yr^{-1}}$ (Bond et al., 2004; Christian et al., 2010 and references therein). Some of the range could be because the higher estimate includes incineration (which may be included in some inventories of anthropogenic emissions), while the low estimate is for open burning only.

## 3 Discussion

We begin this section with some comments on individual BB emissions that are important due to their reactivity (HONO) or use as BB tracers (HCN, $CH_3CN$) and for which a significant amount of new information has been recently obtained. We then briefly discuss progress in NMOC measurements as well as the large amount of NMOC emitted by BB that so far remain unidentified. We offer a new estimate for total global BB NMOC emissions. An overview of the sparse information available about atmospheric processing of BB emissions is presented. We then conclude with a brief summary of the state of the field identifying a few key gaps in our knowledge that should be targeted for future research.

### 3.1 Specific compounds

#### 3.1.1 HONO

Given the abundance of highly reactive species present in fresh smoke, OH plays a key role in the "fast chemistry" of young plumes (Hobbs et al., 2003). Photolysis is the primary daytime fate of nitrous acid (HONO), which forms OH and NO with unit quantum

yield within 10–20 min (Sander et al., 2006). Thus, HONO can be an important source of the OH radical, which then initiates attack on NMOCs (Finlayson-Pitts and Pitts, 2000). Significant, direct emissions of gas-phase HONO from BB at ~3% of $NO_x$ was first reported for a savanna fire by Trentmann et al. (2005). A range of $\Delta HONO/\Delta NO_x$ (5–30%) was then observed in a laboratory experiment burning various types of Southern African biomass (Keene et al., 2006). A comparable molar ratio of $\Delta HONO/\Delta NO_x$ (~14%) was observed by Yokelson et al. (2007a) from a pasture fire in the Amazon (Table S12). The Caltech chemical ionization mass spectrometer (CIMS) measured similar HONO/$NO_x$ ER for Yucatan BB (Yokelson et al., 2009). OP-FTIR and the NOAA Negative Ion Proton Transfer Chemical Ionization Mass Spectrometer (NIPT-CIMS) were in good agreement with each other and observed HONO/$NO_x$ ratios similar to those mentioned above when sampling laboratory fires burning North American biomass fuels (Roberts et al., 2010; Burling et al., 2010; Veres et al., 2010). A subsequent aircraft field campaign confirmed the lab HONO/$NO_x$ ratios by airborne FTIR for the same fuel types on open fires and also documented rapid post-emission HONO loss (Akagi et al., work in progress). Similar HONO/$NO_x$ ER were observed for boreal forest fires during ARCTAS (J. M. St. Clair, private communication, 2010). The collective range of ~3–30% in the ER $\Delta HONO/\Delta NO_x$ from all studies suggests that HONO emissions may be highly dependent on fuel type (Keene et al., 2006; Burling et al., 2010). High levels of OH in young BB plumes that are consistent with photolysis of significant amounts of HONO have been observed at least twice (Hobbs et al., 2003; Yokelson et al., 2009) (discussed further in Sect. 3.2). In two prognostic model applications both Trentmann et al. (2005) and Alvarado and Prinn (2009) found that adding both a "reasonable" amount of initial HONO and a continuous source from heterogeneous chemistry to detailed photochemical models of BB plumes improved the agreement between simulated and observed ozone formation.

### 3.1.2 HCN

Andreae and Merlet (2001) reported HCN emission factors only from Hurst et al. (1994a,b) and stressed the need for more measurements of this species given its potential value as a biomass burning tracer (Li et al., 2000). Within the past decade EF(HCN) has been measured frequently and the results have high variability. However, the bulk of the new data suggest average EF for most types of biomass burning that are about ten times higher than obtained in the first measurements (Tables 1 and 2). HCN has also proved useful as a tracer to deconvolute mixtures of urban and BB emissions (Yokelson et al., 2007b; Crounse et al., 2009). Currently there appears to be two main limitations in the use of HCN as a BB tracer. First, there is a high natural variability in HCN emissions even within a single or similar fire types. For example, Yokelson et al. (2009) note that the ER $\Delta$HCN/$\Delta$CO for Brazilian tropical evergreen forest deforestation fires (0.0063±0.0054) does not differ significantly from that of tropical dry forest deforestation fires in the Yucatan (0.0066±0.0041). While it is encouraging that the mean, observed ER for these two fuel types are similar, the $1\sigma$ standard deviation uncertainty in the mean is greater than 60% of the mean for both fuel types, which then contributes large uncertainty to HCN-based estimates of the BB contribution to regional pollutant levels as discussed elsewhere (Yokelson et al., 2007b; Crounse et al., 2009). Second, there is a large difference in the $\Delta$HCN/$\Delta$CO ER that can occur for some different fire types, which impacts estimates in the many areas featuring many types of biomass burning. For example, Christian et al. (2010) note that HCN levels fell below FTIR detection limits when sampling cooking fire emissions in both Mexico and Africa. Thus, more sensitive measurements of biofuel sources are needed for this source. On the other hand the EF(HCN) measured for peat fires appear to be about ten times larger than for other types of open burning (Tables 1, 2, and S5). Indonesia, as one example, has large amounts of peat fires, forest fires, and biofuel use, complicating the use of HCN as a tracer in that region.

### 3.1.3 $CH_3CN$

Biomass burning is thought to be the primary source of acetonitrile in the atmosphere (de Gouw et al., 2003). Its relatively long lifetime (5–6 months, Li et al., 2003) and near exclusive production from biomass burning suggests great potential as a biomass burning tracer. Many groups have now measured $CH_3CN$ in smoke plumes by MS (Andreae et al., 2001; Jost et al., 2003; Holzinger et al., 1999; Karl et al., 2003, 2007; Christian et al., 2003; Yokelson et al., 2007a, 2009; Crounse et al., 2009). The $\Delta CH_3CN/\Delta HCN$ ER has been shown to be quite robust in both laboratory and field measurements with molar ER of 0.39–0.56 being observed for a wide range of non-boreal fuel types burned (Crounse et al., 2009; Yokelson et al., 2008, 2009; Christian et al., 2003). The $\Delta CH_3CN/\Delta HCN$ ER for 4 boreal plumes sampled during ARCTAS was 0.30±0.11 (Wisthaler and Mikoviny, unpublished results). Analogous to HCN, $CH_3CN$ is both unmeasured for cooking fires and was produced at order of magnitude higher levels by peat fires (Table S5).

## 3.2 Gas-phase non-methane organic compounds (NMOC)

Most NMOC are reactive so it is important to speciate as many as possible and to know the total amount. Early attempts to measure total gas-phase NMOC from BB relied on instruments designed to measure total NMHC (TNMHC, e.g., Cofer et al., 1993) and returned ER for $\Delta$TNMHC/$\Delta$CO on the order of 24%. However, it is now known that ~60–80% of the NMOC emitted by BB are OVOC on a molar basis and that TNMHC instruments have a response to oxygenated compounds that is poorly characterized (Kállai et al., 2003). More recent attempts to estimate total NMOC are described next. PTR-MS is a "soft ionization" technique for measuring gas-phase NMOC with proton affinity higher than water (most NMOC) that produces essentially one peak per molecular mass and for which the proportionality between signal level and concentration falls within a narrow range for all compounds detected at each mass (Lindinger et al., 1998; Karl et al., 2007). GC and FTIR techniques can be useful for species identification when more than one compound appears at a mass, but their response factors to individual species vary over a greater range than for MS. In addition, many of the NMOC emitted by BB are too "sticky" or reactive to be measured by GC while FTIR works best for smaller molecules whose strongest absorption features are not severely overlapped by water or other co-emitted species (Goode et al., 1999). In BB studies that synthesized all three techniques the amount of MS signal due to identified and unidentified peaks was compared in a few selected "typical" PTR-MS mass spectra. In this way both Christian et al. (2003) and Karl et al. (2007) estimated that ~72% of the NMOC in fresh smoke could be identified (on a molar basis) when considering all *m/z* up to 205. Since most of the unidentified species are heavier in mass (>100 amu) this corresponds to successful identification of about 50% of the NMOC on a mass basis. In a recent laboratory-based fire study, additional advanced species identification processes were employed, but the unidentified species still ranged from 25–51% on a mass basis (C. Warneke, work in progress).

Two major concerns regarding this large amount of unidentified (and often overlooked) NMOC species are 1) the additional reactivity they contribute to plume gas-phase chemistry should be recognized in models, and 2) because of their high mass, on average, many are likely to be SVOC that could also condense after cooling or oxidation (Robinson et al., 2007). Consistent with the first effect, Trentmann et al. (2005) showed that tripling the NMOC/$NO_x$ ratio above the measured amount improved model-measurement agreement for ozone. Alvarado and Prinn (2009) added monoterpenes to the BB initial emissions in their smoke model to increase the total organic gas-phase emissions by 30% as a surrogate for unidentified NMOC and this enabled their model to better reproduce the secondary formation of particulate organic carbon observed in a plume. Both findings suggest that model simulations can be improved by including NMOC that have not been identified to date.

Because unidentified NMOC emissions are both abundant and important, we have included their estimated EF for each fire type in Tables S1–S14 and in Tables 1 and 2 as described above. In addition, we provide a new global estimate of total

NMOC emissions in Table 5, which supersedes the estimate in Table 5 of Yokelson et al. (2008). For most fuel types we calculated total identified NMOC by summing the individual average NMOC emission factors reported in Tables S1–S14. We then estimated the potential total NMOC (identified plus unidentified) in $g\,kg^{-1}$ for each fire type by multiplying identified NMOC by a factor of 2 or 3 as needed. Based on the findings of Christian et al. (2003) and Karl et al. (2007) that only about half of the NMOC mass can be identified when the emissions are measured by FTIR, GC, and PTR-MS, we employed a factor of 2 to estimate total NMOC from fire types where the emissions were measured with all these instruments. Yokelson et al. (2009) found that studies using 2 of these techniques identified only about 40% as much NMOC by mass compared to studies using all three techniques. Therefore, we use a factor of 3 to calculate potential total NMOC for those fire types that have so far only been sampled with minimal instrumentation (e.g. biofuel burning).

To scale to global production of gas-phase NMOC in $Tg\,yr^{-1}$ in Table 5, we multiply the total NMOC in $g\,kg^{-1}$ for each fire type by the estimated biomass consumption for that fire type from Table 4. Global production of NMOC from biofuels is calculated using EF(NMOC) of open cooking fires, since these types of fires are the dominant source of biofuel emissions on a global scale (Dherani et al., 2008). Summing the annual gas-phase NMOC from each fire type results in a total BB gas-phase NMOC source of 412 $Tg\,yr^{-1}$ (378 $Tg\,yr^{-1}$ without garbage burning). We then show that adding the estimated NMOC emissions from peat fires in the 1997 El Niño year brings the global total to 744 $Tg\,yr^{-1}$. The latter estimate is an upper limit for emissions during an El Niño year when a large number of fires affected the Indonesian tropical peatlands. The lower value of 378 $Tg\,yr^{-1}$ ignores the contribution from peat and garbage burning entirely. We note that this lower value is still significantly larger than ~100 $Tg\,yr^{-1}$, which would be derived from Table 2 in Andreae and Merlet (2001) based on late 1990's data. Clearly biomass burning emissions of NMOC rank well ahead of urban NMOC emissions globally and are second only to biogenic emissions (~1000 $Tg\,yr^{-1}$) as discussed in Yokelson et al. (2008). CO and black carbon (BC) are also important

BB emissions. Both as a demonstration of (and a check on) our methodology in Table 5, we also combine our biomass consumption values with the measured EF(CO) and EF(BC) for each fire type to produce global BB central estimates for CO (725 $\mathrm{Tg\,yr^{-1}}$) and BC (5.35 $\mathrm{Tg\,yr^{-1}}$) that are in good agreement with other recent estimates (Kopacz et al., 2010; Bond et al., 2004).

### 3.2.1 Post emission processing

The emission factors presented herein were calculated from fresh smoke sampled at the source that had usually cooled to ambient temperature, but undergone minimal photochemical aging. In nearly any application of this data, it is important to realize that rapid, complex photochemistry can cause large changes in smoke composition within minutes after its initial emission. There is a not a single standard processing scenario that can be applied to all smoke. Rather, there are numerous possible fates of smoke given differences in initial emissions, degree of cloud processing, dispersion altitude, temperature, humidity, time of day or night, small vs. large-scale fires (e.g. cooking fires vs. forest fires), the degree of mixing with other BB plumes or biogenic or urban emissions, etc. In theory, almost any realistic processing scenario can be modeled. In practice, very few detailed smoke evolution data have been acquired that are of value for testing or constraining the chemical mechanism in smoke photochemistry models. Next we summarize the measurements that have been made and draw a few conclusions.

The type of measurements that provide the most straightforward test of photochemical models are those that trace the evolution of a single, isolated BB plume. Measurements of this type have been obtained in Alaska, Africa, California, Mexico, and Canada. Goode at al. (2000) sampled two very large plumes up to ~56 km downwind in Alaska (their Table 6). The B280 fire plume was sampled well below the top of the plume and no ozone formation or change in a reactive hydrocarbon ($C_2H_4$) was observed over ~2.8 h of aging. However $NH_3$ decreased about 70% over that time. The B309 fire plume was sampled closer to the top and within ~2 h of aging

the $\Delta O_3/\Delta CO$ ratio had risen to ~9% (from negative values at initial injection). Both $\Delta HCOOH/\Delta CO$ and $\Delta CH_3COOH/\Delta CO$ doubled on that time scale, but the decrease in $C_2H_4$, which would have accompanied high OH, was small enough to be insignificant. The chemical evolution of five different isolated BB plumes was tracked by various aircraft during SAFARI 2000. Three of these plumes were sampled by the University of Washington Convair 580 (Fig. 5 in Yokelson et al., 2003) and $\Delta O_3/\Delta CO$ rose to ~9% in only ~40 min of aging. In addition $\Delta CH_3COOH/\Delta CO$ more than tripled from its initial value to ~9% within the same aging interval. Additional detailed analysis of the most extensively sampled plume (Hobbs et al., 2003), the Timbavati plume, revealed a rapid loss of $NO_x$ and 16 reactive hydrocarbons consistent with an average OH of $1.7\times10^7$ $\mathrm{molecules\,cm^{-3}}$. Technical issues involving the sampling system preclude us from making conclusive quantitative statements about the evolution of the particles in that plume (Alvarado and Prinn, 2009; Magi, 2009), but particle nitrate definitely increased and OC likely did also. Jost et al. (2003) sampled a Namibian BB plume at 11 different smoke ages and observed $\Delta O_3/\Delta CO$ rise to ~10% in ~2 h and also a tendency for acetone to be enhanced in the downwind plume. In another African plume, Abel et al. (2003) measured an increase in single scattering albedo from 0.84 to 0.885 over 2.4 h of aging that they attributed to condensation of non-absorbing (organic) species.

Yokelson et al. (2009) described the evolution during 1.5 h of aging of a single BB plume in the Yucatan sampled by the NCAR C-130. $\Delta O_3/\Delta CO$ rose to 10–15% in about one hour, which is almost identical to the $O_3$ formation rate observed by Hobbs et al. (2003). Rapid secondary production of peroxyacetylnitrate (PAN), hydrogen peroxide, formic acid, and peroxyacetic acid was observed. The post emission loss rates were measured for $SO_2$, $NO_x$, and HONO. No reactive NMOC were measured in the down-wind plume, but in-plume OH was directly measured for the first time in a BB plume and averaged $1.14\times10^7$ $\mathrm{molecules\,cm^{-3}}$ for the plume age interval 22–43 min. Significant post-emission formation rates for particle nitrate, ammonium, sulfate, organic aerosol (OA), and an increase in single scattering albedo were measured.

Aerosol mass spectrometry and light scattering measurements both indicated that the aerosol to CO ratio increased by a factor of ~2.4 in 1.5 h. In another isolated Yucatan plume TEM analysis indicated that slightly aged smoke had a higher abundance of tar balls (Yokelson et al., 2009), which have been linked to the concept of brown carbon (Adachi and Buseck, 2008; Andreae and Gelenesér, 2006; Chakrabarty et al., 2010). Recently the evolution of an isolated BB plume was measured in mid-coast California for ~4.5 h. A rapid increase in light scattering and formation of $O_3$, HCOOH, $CH_3COOH$, and PAN was observed along with loss of $NO_x$, HONO, $C_2H_4$, $C_3H_6$, and $NH_3$ (S. Akagi, work in progress).

The chemical evolution of one boreal forest fire plume (Flight 18, McKay Lake Fire) was extensively measured during ARCTAS (Alvarado et al., 2010). This fire plume exhibited two different types of behavior on the same day. In particular, the early evening samples were divided into two groups: those obtained below 1 km and those obtained above 2 km. The low altitude samples had low MCE (more smoldering, 0.85–0.92) and low $\Delta NO_y/\Delta CO$ (0.34–0.55%). After ~4 h of aging $\Delta PAN/\Delta CO$ reached 0.23–0.36% and $\Delta O_3/\Delta CO$ ranged from −1% to 3%. The higher altitude samples had higher MCE (more flaming, 0.96) and a higher $\Delta NO_y/\Delta CO$ (1.1–1.7%). In the high altitude samples (produced by more vigorous combustion) $\Delta PAN/\Delta CO$ reached 0.51% after only one hour, by which time $\Delta O_3/\Delta CO$ had already reached 5%. These results nicely illustrate the high natural variability in post-emission processing that can be expected for fires, which was, in this case, driven partly by variable initial emissions. In light of the potential connection between plume chemistry and plume injection altitudes it is significant that large numbers of BB plume heights can now be measured from space about once per day (Val Martin et al., 2010). However, in the boreal forest, where individual fires can burn for weeks, the fire radiative energy (and likely the biomass consumption rate and injection altitude) may not peak at mid-day in a simple diurnal cycle. In fact, the regional fire radiative energy can be higher at night (see Fig. 4b of Vermote et al., 2009), which could be driven by frontal passage or other weather or fuel conditions.

Several authors have constructed detailed photochemical models for direct comparison to the measurements in some of the plumes mentioned above (Jost et al., 2003; Trentmann et al., 2005; Mason et al., 2006; Alvarado and Prinn, 2009; Alvarado et al., 2010). Taken together, these studies show that model performance is improved by considering the initial emissions of HONO, a continuous heterogeneous source of HONO, and unidentified NMOC. In addition, Alvarado et al. (2010) discuss application-specific techniques to improve model performance at larger geographic scales.

It is also useful to compare data acquired in "intercepted," aged plumes to "probable" initial values. Yokelson et al. (2009) discuss sources of uncertainty in interpretation of data from aged smoke only in their Sect. 3.5. The comparison is least uncertain for species which have tightly constrained initial values such as $\Delta O_3/\Delta CO$, which is negative in fresh plumes (Yokelson et al., 2003). For example, Andreae et al. (1994) described encounters with 40 different tropical BB plumes about 7–10 days old for which $\Delta O_3/\Delta CO$ averaged 43±26% (ranging from 11–89%) and where $\Delta O_3/\Delta CO$ correlated positively with $\Delta NO_y/\Delta CO$ (their Table 1 and Fig. 14). At the other end of the spectrum, 1–2 day old plumes from smoldering tundra fires in the Arctic had an average $\Delta O_3/\Delta CO$ of 9.5±6% (Wofsy et al., 1992; Jacob et al., 1992). The smaller $O_3$ enhancements in the arctic plumes were attributed to younger plume age and a factor of ten lower initial $\Delta NO_x/\Delta CO$ (inferred from lower $\Delta NO_y/\Delta CO$) (Andreae et al., 1994). Nine plumes from boreal wildfires that were 6–15 days old were sampled at the PICO-NARE station in 2004 and eight of the plumes had $\Delta O_3/\Delta CO$ ranging from 9% to 89% (Table 3 of Val Martín et al., 2006; Lapina et al., 2006). However one aged plume had $\Delta O_3/\Delta CO$ of −42% (vide infra). Higher $O_3$ enhancements were correlated with higher $NO_y$. Yokelson et al. (2007a) sampled a large regional plume in Brazil containing the mixed output from many regional fires with smoke age likely ranging from 1–20 h that had enhanced $O_3$, HCOOH, and $CH_3COOH$. Reid et al. (1998) compared regional haze dominated by aged BB smoke to BB smoke <4 min old in Brazil. Smoke aging was associated with loss of gas-phase NMHC and growth of particle ammonium, organic acids, and sulfate as well as other changes including increases in particle size and single scattering albedo.

The observation at PICO-NARE that some plumes may have little or no photochemical activity is consistent with some airborne observations. One of two Alaskan plumes traced by Goode et al. (2000) and several aged plumes encountered at high altitude off the coast of the US by de Gouw et al. (2006) or in Africa by Capes et al. (2009) showed little evidence of oxidation. Most of the boreal forest fire plumes sampled during ARCTAS were photochemically active enough to generate substantial amounts of PAN, but very little $O_3$ (Alvarado et al., 2010; Singh et al., 2010). A variety of plume chemistry regimes has been observed in remote sensing studies. For example, Fishman et al. (1991) attributed widespread $O_3$ enhancements in the Southern Hemisphere to BB, while Verma et al. (2009) observed both $O_3$ formation and $O_3$ destruction in boreal forest fire plumes. In summary, the airborne, ground-based, and space-based observations consistently demonstrate that a large range in post-emission outcomes is possible. An important aspect of this variability is that despite the difficulty of using reactive initial emissions as input for regional-global models, there may not be an advanced smoke age that provides significant advantages as a starting point at which emissions could be tabulated.

A few observations have been made of cloud processing of smoke. Yokelson et al. (2003) observed rapid reduction in NO, $CH_3OH$, $NH_3$, and $CH_3COOH$ and concurrent fast $NO_2$ and HCHO formation within a small pyrocumulus cloud that capped a vertical column of smoke ~3 km above the flame front. This was modeled as the product of heterogeneous reactions of $CH_3OH$ on droplet surfaces combined with enhanced photochemistry in the cloud (Tabazadeh et al., 2004; Madronich, 1987). Unpublished data from the Yucatan plume described earlier in this section shows a very large step increase in $\Delta$HCHO/$\Delta$CO immediately after the plume RH briefly exceeded 100%, a potential sign of cloud-processing (A. Fried, T. Campos, private communication, 2010). A large fraction of the smoke on Earth resides in hazy boundary layers that are “topped” with a layer of embedded cumulus clouds. These clouds also play a role in “pumping” smoke from the boundary layer to the free troposphere. For these

Discussion Paper | Discussion Paper | Discussion Paper | Discussion Paper |

reasons, smoke-cloud interactions require much more research.

### 3.3 Knowledge gaps and sources of uncertainty

#### 3.3.1 Poorly characterized fire types and post-emission processing scenarios

Biofuel use (fuelwood, charcoal, dung, and crop residue burning for use as fuel) remains one of the most significant yet often overlooked forms of biomass burning on a global level, as these fires are small in scale and elude satellite detection. Consequently, biofuel consumption is estimated based on user questionnaires distributed in mostly rural areas (Yevich and Logan, 2003); a process that may underestimate urban biofuel use and may work better for domestic use than industrial use. The NMOC emissions from biofuel use have not yet been measured with mass spectrometry, which is an important component of modern smoke analysis as described earlier in this paper. In particular, HCN and $CH_3CN$ have yet to be observed in the emissions from burning fuelwood or dung, and there is currently no known chemical tracer to validate survey results (Christian et al., 2010). The lack of a way to check on survey results is especially problematic for industrial biofuel use, which is dominated by large numbers of small firms that utilize a dynamic variety of fuels (wood, sawdust, crop residue, used motor oil, tires, garbage, boards with lead paint, etc.). Burning these fuels is not always legal, which can affect survey results (Christian et al., 2010). Small informal firms, including those that use biofuel, have been estimated to account for over 50% of non-agricultural employment and 25–75% of gross domestic product in both Latin America and Africa (Ranis and Stewart, 1994; Schneider and Enste, 2000). More biofuel measurements are needed given the wide variety and high variability of these emissions and their dependence on geographical region, fuel availability, climate, local customs, and season (Yevich and Logan, 2003). The limited data continue to hinder our ability to predict the relative contribution of biofuel burning to total pollution in the many areas where this type of burning is common (Ramanathan et al., 2001; Christian et al., 2010).

Discussion Paper | Discussion Paper | Discussion Paper | Discussion Paper |

Garbage burning (GB) is a widespread emission source occurring in urban-rural areas in both developing and developed nations. Christian et al. (2010) made some of the first detailed measurements of open GB as part of the 2007 Mega-city Impacts Local and Global Research Observations (MILAGRO) campaign based in Central Mexico. High EF(HCl) (1.65–9.8 $\mathrm{g\,kg^{-1}}$) were observed (Table S14) traceable to the large amounts of polyvinyl chloride (PVC) found in the landfills (Christian et al., 2010). These EF(HCl) suggest GB may be the main global source of HCl and it has long been known as the main global source of dioxins (Costner, 2005, 2006). Because GB emits large amounts of PM, HCl, and $NO_x$ (R. Yokelson, work in progress) the interaction between these species could lead to reactive products that impact $O_3$ formation (Osthoff et al., 2008; Raff et al., 2009; Thornton et al., 2010). GB was found to emit high levels of several compounds such as levoglucosan sometimes used as tracers for BB.

Since ~70–80% of biomass burning occurs in the tropics (Crutzen and Andreae, 1990; van der Werf et al., 2010), relatively little research on BB emissions in temperate regions has been done. Though less significant on a global scale, these fires occur in areas where some degree of control of the amount of burning is both feasible and enacted to maintain good air quality. However, excluding fire, which is the major natural disturbance factor in most forests, can severely compromise other valid land management objectives (e.g., Mutch, 1994; Neary et al., 2005; Wiedinmyer and Hurteau, 2010). So far, regional-global models have estimated air quality impacts using emission factors for temperate-region fires derived from tropical and boreal forest data, which introduces additional uncertainty to emission estimates (C. Wiedinmyer, work in progress). Recent laboratory and field campaigns have responded to this need and sampled emissions from fires in oak savanna, chaparral, pine understory, and pocosin vegetation types from California, North Carolina, and Arizona. Some of the laboratory and preliminary field results are included in this compilation and more should be forthcoming in the near future (I. Burling, private communication, 2010).

Not only have some types of biomass burning been inadequately sampled, but there are also no measurements that quantify many important post-emission processing

scenarios such as nighttime smoke chemistry and plume mixing. This is in addition to the paucity of data on cloud-processing of smoke. Finally, the proper measurement of particle-phase light absorbing carbon (LAC, including BC, weak absorption by particulate organic carbon (OC), and strong absorption by "brown carbon") as well as the best way to represent LAC in models is unresolved and therefore an area of active research (Andreae and Gelenesér, 2006; Magi, 2009; Chakrabarty et al., 2010).

### 3.3.2 Measurement challenges

A major current instrumental-based limitation is that most of the high molecular mass NMOC (which occur in both the gas and condensed phase products) are still unidentified making it difficult to estimate/model their atmospheric impact. Moving from the current common mass resolution of ~200 to ~5000 and scanning at higher masses will allow more of these compounds to be identified. However, compounds with different structural formulas can have the exact same chemical formula and thus the exact same molecular mass even with very high resolution MS (e.g. acetic acid and glycolaldehyde) (Jordan et al., 2009). Species with identical mass can sometimes be separated by their different tendencies to form clusters, pre-separation, or MS-MS techniques, but all these approaches have limitations and many of the unknown species are semi-volatile and thus difficult to sample (Crounse et al., 2006; Karl et al., 2007). However, improved knowledge of the chemical formula of the emissions present at higher masses should still enable an improved assessment of the physical properties of these emissions (e.g. vapor pressure, reactivity, etc.).

Platform-based limitations exist for all platforms used to study BB. While each platform offers well-documented, powerful advantages, here we simply mention some key limitations and offer some ideas for overcoming them. In laboratory studies the fires are simulated and many fuel complexes are hard to replicate indoors. Additionally, the different products of flaming and smoldering combustion may mix differently in the laboratory compared to real fires. Due to wall losses and other issues, aging studies are limited to a few hours. Careful comparison/synthesis of laboratory results with field results for similar fuels (as discussed at length in Yokelson et al., 2008) should maintain a key role for laboratory studies in future BB research.

Airborne studies sample real fires, but only part of the total smoke plume. The smoke from the lofted and unlofted emissions can have different chemistry and different post-emission transport. Recently, simultaneous ground-based and airborne EF measurements of the same fire were implemented in Brazil and elsewhere (Yokelson et al., 2008). Complimentary observations such as these can help assess the overall fire impact. However, measurements of the relative biomass consumption contributing to the lofted and unlofted emissions are needed to scale to total emissions (e.g. see Sect. 2.3.12). Airborne platforms can study aging on long time scales, but real world smoke is often a complex mixture of young and old plumes and may reflect mixing with non-BB sources. The use of multiple tracers can deconvolute complex mixtures of urban and BB emissions with some success as demonstrated by Crounse et al. (2009).

Remote sensing provides numerous types of high temporal resolution global products to drive and validate models, but unfortunately, space-based estimates of the amount of biomass burned are difficult and still not as quantitative as desired. The lack of daily coverage by MODIS in the tropics and loss of information due to cloud coverage globally are serious limitations and detection efficiency of fires as hot-spots or burn scars is poor for small fires, which may comprise the majority of tropical fires (Hawbaker et al., 2008; Chang and Song, 2010; Giglio et al., 2006). Comparison of burned area or hotspot products can often reveal factor of ten or larger disagreements (e.g., Al-Saadi et al., 2008; Tables 5–7 in Chang and Song, 2010a, b). These latter authors noted that GFEDv2.1-based estimates of CO emissions from tropical Asia were 5–7 times higher than their estimates based on MODIS or L3JRC burned area products. However, based on synthesis of multiple space-based CO products, Kopacz et al. (2010) concluded that GFED2 significantly underestimates CO from biomass burning in this region and globally. Clearly more validation of remote sensing products is needed to guide their incorporation into models and hopefully to inform the development of new sensors.

The measurement limitations that impact models the most may be as follows. At the plume scale the large fraction of the reactive NMOC precursors emitted by BB that are still unidentified and the small number of measurements of aging plumes that constrain/validate plume model mechanisms are serious issues. When scaling up to regional-global models, key limitations are the uncertainty in biomass consumption, plume injection heights, and the limited amount of data that could validate model parameterizations of fast photochemical changes and physical processes that occur in smoke on spatial scales that are much smaller than the model grid (Alvarado et al., 2009, 2010; Fast et al., 2009). Expected improvements in measurements, computing power, and parameterization of faster processes could all increase model performance (Alvarado et al., 2009, 2010).

## 4 Conclusions

Major advances have been made in the past ten years in characterizing the initial emissions of trace gases and particles from biomass burning and their post-emission evolution. Instruments developed during this time span can better quantify particle species, useful BB tracers, and oxygenated NMOC, which account for the majority of the gas-phase NMOC emitted by fires. Several quality data sets have been obtained that traced the chemical evolution of smoke in the field and these observations have been modeled with some success. We have yet to identify and quantify most of the higher molecular weight NMOC, which tend to be semi-volatile and thus move between the gas-phase and condensed phase in poorly understood fashion – profoundly affecting the chemical evolution of both phases. The unidentified compounds comprise approximately one-half of the actual NMOC by mass and will likely be speciated with only limited success in the immediate future. The discrepancy between identified and actual levels of NMOC could be addressed for the time being by increasing known NMOC by a factor of 2–3 to obtain more realistic initial emissions. Incorporation of improved NMOC estimates and chemistry should improve our understanding of the impact of biomass burning on the local-global environment. Large uncertainties remain for undersampled processing scenarios (clouds, night-time) and undersampled fuel types, most notably for biofuels, which could become the largest global type of burning. Thus, on a global scale many EF, even for measureable NMOC, still have high levels of uncertainty. Our knowledge of the spatial and temporal trends in biomass burning, the transport of BB emissions on continental scales, and the injection altitudes of BB plumes has improved greatly due to advances in remote sensing. However, accurate quantification of the amount of biomass burned, which is coupled with the EF compiled in this work, remains a difficult challenge.

**Supplementary material related to this article is available online at: http://www.atmos-chem-phys-discuss.net/10/27523/2010/ acpd-10-27523-2010-supplement.zip**.

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**Table 1.** Emission factors (g kg$^{-1}$) for species emitted from different types of biomass burning.

| | Tropical Forest | Savanna | Crop Residue | Pasture Maintenance | Boreal Forest | Temperate Forest | Extratropical Forest[a] |
|---|---|---|---|---|---|---|---|
| Carbon Dioxide ($CO_2$) | 1643 (58) | 1692 (37) | 1537 (81) | 1548 (142) | 1485 (107) | 1598 (80) | 1500 (93) |
| Carbon Monoxide (CO) | 92 (26) | 59 (16) | 112 (32) | 135 (38) | 126 (45) | 104 (37) | 123 (44) |
| Methane ($CH_4$) | 5.12 (2.05) | 1.50 (0.70) | 6.01 (2.71) | 8.71 (4.97) | 5.94 (3.13) | 4.98 (2.28) | 5.81 (2.87) |
| Acetylene ($C_2H_2$) | 0.26 (0.21) | 0.24 (0.089) | 0.28 (0.12) | 0.21 (0.29) | 0.18 (0.10) | 0.22 (0.06) | 0.19 (0.08) |
| Ethylene ($C_2H_4$) | 1.07 (0.36) | 0.80 (0.34) | 1.60 (0.65) | 1.28 (0.71) | 1.42 (0.49) | 1.58 (0.43) | 1.44 (0.45) |
| Ethane ($C_2H_6$) | 0.68 (0.28) | 0.31 (0.20) | 0.68 (0.55) | 0.95 (0.43) | 1.79 (1.14) | 0.51 (0.25) | 1.62 (0.92) |
| Propadiene ($C_3H_4$) | 0.016 (0.0066) | 0.012 (0.005) | – | 0.020 (0.009) | – | – | – |
| Propylene ($C_3H_6$) | 0.65 (0.42) | 0.23 (0.15) | 0.84 (0.50) | 0.85 (0.66) | 1.13 (0.62) | 0.64 (0.14) | 1.06 (0.44) |
| Propyne ($C_3H_4$) | – | – | – | – | 0.059 | – | 0.059 |
| Propane ($C_3H_8$) | 0.126 (0.060) | 0.10 (0.067) | 0.19 (0.16) | 0.22 (0.10) | 0.43 | 0.26 (0.11) | 0.41 (0.17) |
| *n*-Butane ($C_4H_{10}$) | 0.038 (0.023) | 0.016 (0.013) | 0.042 (0.035) | 0.040 (0.018) | 0.12 | 0.083 (0.10) | 0.12 (0.14) |
| *i*-Butane ($C_4H_{10}$) | 0.011 (0.009) | 0.0043 (0.0027) | 0.014 (0.012) | 0.014 (0.0063) | 0.040 | – | 0.040 |
| 1-Butene ($C_4H_8$) | 0.079 (0.024) | 0.043 (0.022) | 0.10 (0.07) | 0.17 (0.077) | 0.15 | – | 0.15 |
| *i*-Butene ($C_4H_8$) | 0.11 (0.051) | 0.024 (0.0051) | 0.082 (0.064) | 0.11 (0.05) | 0.11 | – | 0.11 |
| 1,3-Butadiene ($C_4H_6$) | 0.039 | 0.052 (0.028) | 0.10 (0.07) | – | 0.14 | – | 0.14 |
| *trans*-2-Butene ($C_4H_8$) | 0.029 (0.013) | 0.011 (0.0055) | 0.041 (0.033) | 0.050 (0.023) | 0.038 | – | 0.038 |
| *cis*-2-Butene ($C_4H_8$) | 0.024 (0.010) | 0.0084 (0.0043) | 0.031 (0.025) | 0.040 (0.018) | 0.030 | – | 0.030 |
| *n*-Pentane ($C_5H_{12}$) | 8.03E-03 (8.03E-03) | 0.0032 (0.0032) | 0.014 (0.011) | 0.0056 (0.0025) | 0.083 | – | 0.083 |
| *i*-Pentane ($C_5H_{12}$) | 0.010 (0.010) | 0.0022 (0.0032) | 0.007 (0.006) | 0.0074 (0.0033) | 0.038 | – | 0.038 |
| *trans*-2-Pentene ($C_5H_{10}$) | 3.30E-03 | 0.0045 (0.0028) | – | – | – | – | – |
| *cis*-2-Pentene ($C_5H_{10}$) | 1.90E-03 | 0.0025 (0.0018) | – | – | – | – | – |
| 3-Methyl-1-Butene ($C_5H_{10}$) | 3.80E-03 | 0.0051 (0.0034) | – | – | – | – | – |
| 2-Methyl-2-Butene ($C_5H_{10}$) | 4.00E-03 | 0.0048 (0.0035) | – | – | – | – | – |
| 2-Methyl-1-Butene ($C_5H_{10}$) | 4.40E-03 | 0.0059 (0.0037) | – | – | – | – | – |
| Isoprene ($C_5H_8$) | 0.13 (0.056) | 0.039 (0.027) | 2.14 (0.88) | 0.12 (0.055) | – | – | – |
| Cyclopentane ($C_5H_{10}$) | – | – | 0.001 (0.001) | – | – | – | – |
| 2+3-Methylpentane ($C_6H_{14}$) | – | – | – | – | 0.034 | – | 0.034 |
| 2-Methyl-1-Pentene ($C_6H_{12}$) | 2.80E-03 | 0.0035 (0.0021) | – | – | – | – | – |
| *n*-Hexane ($C_6H_{14}$) | 0.010 | 0.013 (0.0074) | – | – | 0.055 | – | 0.055 |
| Heptane ($C_7H_{16}$) | 5.60E-03 | 0.0070 (0.0072) | – | – | 0.046 | – | 0.046 |
| Benzene ($C_6H_6$) | 0.39 (0.16) | 0.20 (0.084) | 0.87 (0.20) | 0.70 (0.32) | 1.09 | – | 1.09 |
| Toluene ($C_6H_5CH_3$) | 0.26 (0.13) | 0.080 (0.058) | 1.08 (0.35) | 0.34 (0.15) | 0.49 | – | 0.49 |
| Xylenes ($C_8H_{10}$) | 0.11 (0.082) | 0.014 (0.024) | – | 0.11 (0.050) | 0.17 | – | 0.17 |

**Table 1.** Continued.

| | Tropical Forest | Savanna | Crop Residue | Pasture Maintenance | Boreal Forest | Temperate Forest | Extratropical Forest[a] |
|---|---|---|---|---|---|---|---|
| Ethylbenzene ($C_8H_{10}$) | 0.050 (0.036) | 0.006 (0.010) | – | 0.067 (0.030) | 0.048 | – | 0.048 |
| *n*-Propylbenzene ($C_9H_{12}$) | – | – | – | – | 0.018 | – | 0.018 |
| $\alpha$-Pinene ($C_{10}H_{16}$) | – | – | – | – | 1.61 | – | 1.61 |
| $\beta$-Pinene ($C_{10}H_{16}$) | – | – | – | – | 1.43 | – | 1.43 |
| Ethanol ($CH_3CH_2OH$) | – | – | – | – | 0.055 | – | 0.055 |
| Methanol ($CH_3OH$) | 2.31 (0.88) | 0.77 (0.23) | 3.55 (1.41) | 5.84 (3.42) | 2.82 (1.61) | 2.61 (1.43) | 2.79 (1.56) |
| Phenol ($C_6H_5OH$) | 0.45 (0.088) | 0.52 (0.36) | 2.94 (0.78) | 1.68 (3.34) | 2.96 | 0.83 (0.61) | 2.67 (1.96) |
| Formaldehyde (HCHO) | 1.73 (1.22) | 0.80 (0.34) | 2.89 (0.57) | 1.90 (1.11) | 1.80 (1.22) | 2.32 (0.89) | 1.87 (1.03) |
| Glycolaldehyde ($C_2H_4O_2$) | 1.32 | 0.38 (0.18) | 5.31 (1.01) | – | 0.36 | 0.35 (0.44) | 0.36 (0.45) |
| Acetaldehyde ($CH_3CHO$) | 1.55 (0.75) | 0.57 (0.30) | 7.05 (1.59) | 2.40 (1.08) | – | – | – |
| Acrolein ($C_3H_4O$) | 0.65 (0.23) | – | – | – | – | – | – |
| Furaldehydes | 0.29 (0.0010) | – | – | – | – | – | – |
| Propanal ($C_3H_6O$) | 0.10 (0.026) | – | – | 0.16 (0.074) | – | – | – |
| Methyl Propanal ($C_4H_8O$) | 0.18 (0.075) | – | – | 0.33 (0.15) | – | – | – |
| Hexanal ($C_6H_{12}O$) | 0.01 (0.005) | – | – | 0.034 (0.015) | – | – | – |
| Acetone ($C_3H_6O$) | 0.63 (0.17) | 0.16 (0.13) | 2.53 (0.40) | 1.05 (0.47) | 0.73 | – | 0.73 |
| Methyl Vinyl Ether ($C_3H_6O$) | – | 0.16 (0.045) | 0.43 (0.07) | – | – | – | – |
| Methacrolein ($C_4H_6O$) | 0.15 (0.045) | – | – | 0.40 (0.18) | – | – | – |
| Crotonaldehyde ($C_4H_6O$) | 0.24 (0.068) | – | – | 0.60 (0.27) | – | – | – |
| 2,3-Butanedione ($C_4H_6O_2$) | 0.73 (0.22) | – | – | 1.58 (0.71) | – | – | – |
| Methyl Vinyl Ketone ($C_4H_6O$) | 0.39 (0.11) | – | – | 1.00 (0.45) | 0.19 | – | 0.19 |
| Methyl Ethyl Ketone ($C_4H_8O$) | 0.50 (0.21) | – | – | 0.94 (0.42) | 0.22 | – | 0.22 |
| 2-Pentanone ($C_5H_{10}O$) | 0.08 (0.024) | – | – | 0.17 (0.077) | – | – | – |
| 3-Pentanone ($C_5H_{10}O$) | 0.03 (0.011) | – | – | 0.08 (0.034) | – | – | – |
| Furan ($C_4H_4O$) | 0.41 (0.10) | 0.17 (0.058) | 0.62 (0.24) | 1.02 (0.43) | 0.80 (0.49) | 0.51 (0.17) | 0.76 (0.38) |
| 3-Methylfuran ($C_5H_6O$) | 0.59 (0.20) | – | – | 1.41 (0.64) | – | – | – |
| 2-Methylfuran ($C_5H_6O$) | 0.08 (0.028) | – | – | 0.20 (0.091) | – | – | – |
| Other substituted furans | 1.21 (0.016) | – | – | – | – | – | – |
| $C_6$ Carbonyls | 0.24 (0.11) | – | – | 0.61 (0.28) | – | – | – |
| Acetol ($C_3H_6O_2$) | 1.13 (0.12) | 0.94 (0.35) | 34.5 (6.74) | 6.18 (5.60) | – | – | – |
| Acetonitrile ($CH_3CN$) | 0.41 (0.10) | 0.11 (0.058) | 1.19 (0.35) | 0.55 (0.25) | 0.59 | – | 0.59 |
| Propenenitrile ($C_3H_3N$) | 0.04 (0.01) | 0.051 (0.022) | 0.19 (0.01) | – | – | – | – |
| Propanenitrile ($C_3H_5N$) | 0.090 | 0.031 (0.014) | 0.35 (0.01) | – | – | – | – |

**Table 1.** Continued.

| | Tropical Forest | Savanna | Crop Residue | Pasture Maintenance | Boreal Forest | Temperate Forest | Extratropical Forest[a] |
|---|---|---|---|---|---|---|---|
| Pyrrole ($C_4H_5N$) | 0.12 (0.038) | – | – | – | – | – | – |
| Formic Acid (HCOOH) | 0.27 (0.23) | 0.18 (0.071) | 1.06 (0.55) | 0.20 (0.64) | 0.57 (0.46) | 0.22 (0.08) | 0.52 (0.33) |
| Acetic Acid ($CH_3COOH$) | 3.11 (0.96) | 1.58 (0.64) | 6.91 (2.28) | 10.4 (6.8) | 4.41 (2.66) | 3.13 (1.52) | 4.24 (2.32) |
| Hydrogen Cyanide (HCN) | 0.42 (0.26) | 0.35 (0.092) | 0.47 (0.10) | 0.46 (0.45) | 1.52 (0.81) | 0.79 (0.14) | 1.42 (0.56) |
| Dimethyl Sulfide ($C_2H_6S$) | 1.35E-03 (1.71E-03) | 0.0013 (0.0011) | – | – | 4.59E-03 | – | 4.59E-03 |
| Carbonyl Sulfide (OCS) | 0.025 | – | – | – | 0.46 (0.47) | – | 0.46 (0.47) |
| Chloromethane ($CH_3Cl$) | 0.053 (0.038) | 0.055 (0.036) | – | 0.29 (0.13) | 0.058 | – | 0.058 |
| Methyl Bromide ($CH_3Br$) | 2.83E-03 (2.38E-03) | 8.53E-04 (8.62E-04) | – | 5.71E-03 (2.57E-03) | 3.58E-03 | – | 3.58E-03 |
| Methyl Iodide ($CH_3I$) | 2.50E-03 (3.45E-03) | 5.06E-04 (3.88E-04) | – | 3.48E-03 (1.56E-03) | 7.78E-04 | – | 7.78E-04 |
| Dichloromethane ($CH_2Cl_2$) | – | – | – | – | 2.42E-03 | – | 2.42E-03 |
| Trichloromethane ($CHCl_3$) | 2.94E-04 (6.75E-03) | 0.012 (0.020) | – | 6.32E-04 (2.84E-04) | – | – | – |
| Dichlorodifluoromethane ($CCl_2F_2$) | 2.80E-03 | – | – | – | – | – | – |
| Ethylchloride ($C_2H_5Cl$) | – | – | – | – | 7.37E-04 | – | 7.37E-04 |
| Trichloroethylene ($C_2HCl_3$) | – | – | – | – | 1.78E-04 | – | 1.78E-04 |
| Ammonia ($NH_3$) | 0.76 (0.56) | 0.49 (0.38) | 2.27 (1.65) | 1.47 (1.29) | 2.72 (2.32) | 1.45 (0.80) | 2.55 (1.83) |
| Methyl Nitrate ($CH_3ONO_2$) | 8.29E-03 (1.60E-02) | 5.1E-04 (3.7E-04) | – | – | 2.85E-03 | – | 2.85E-03 |
| Ethyl Nitrate ($C_2H_5NO_3$) | 5.70E-03 | – | – | – | 1.73E-03 | – | 1.73E-03 |
| *n*-Propyl Nitrate ($C_3H_7NO_3$) | 0.0003 | – | – | – | 3.09E-04 | – | 3.09E-04 |
| *i*-Propyl Nitrate ($C_3H_7NO_3$) | 0.001 | – | – | – | 3.17E-03 | – | 3.17E-03 |
| 2-Butyl Nitrate ($C_4H_9NO_3$) | 0.0006 | – | – | – | 3.78E-03 | – | 3.78E-03 |
| 3-Pentyl Nitrate ($C_5H_{11}NO_3$) | – | – | – | – | 7.19E-04 | – | 7.19E-04 |
| 2-Pentyl Nitrate ($C_5H_{11}NO_3$) | – | – | – | – | 9.41E-04 | – | 9.41E-04 |
| 3-Methyl-2-Butyl Nitrate ($C_5H_{11}NO_3$) | – | – | – | – | 1.13E-03 | – | 1.13E-03 |
| Hydrogen ($H_2$) | 3.21 (1.39) | – | 2.70 (1.79) | – | – | – | – |
| Sulfur Dioxide ($SO_2$) | 0.40 (0.19) | 0.48 (0.27) | – | 0.32 (0.14) | – | – | – |
| Nitrous Acid (HONO) | 2.18 | – | – | 0.16 (0.07) | – | 0.41 (0.30) | 0.41 (0.30) |
| Nitrogen Oxides ($NO_x$ as NO) | 2.56 (1.42) | 2.8 (0.65) | 3.52 (1.34) | 0.75 (0.59) | 0.90 (0.69) | 1.72 (0.78) | 1.01 (0.64) |
| Nitrous Oxide ($N_2O$) | – | – | – | – | 0.41 | 0.16 (0.21) | 0.38 (0.49) |
| NMHC | 1.7 | 3.4 (2.3) | – | – | – | – | – |
| NMOC (identified) | 23.7 (8.3) | 9.1 (4.1) | 55.4 | 44.8 (30.1) | 28.3 (10.0) | 14.0 (6.4) | 26.4 (10.8) |
| NMOC (identified+unidentified) | 47.4 | 18.2 | 166.3 | 89.6 | 56.5 | 28.1 | 52.7 (28.4) |
| Total Particulate Carbon | 5.24 (2.91) | 3.00 (1.43) | – | 10.6 (4.8) | – | – | – |
| Total Suspended Particulate (TSP) | 13 | – | – | – | 18.1 | 28.4 (13.3) | 19.5 (9.1) |

**Table 1.** Continued.

| | Tropical Forest | Savanna | Crop Residue | Pasture Maintenance | Boreal Forest | Temperate Forest | Extratropical Forest[a] |
|---|---|---|---|---|---|---|---|
| CN (particles 0.003–3 µm diameter)[b] | 5.90E+16 | – | – | – | – | – | – |
| $PM_{2.5}$[c] | 9.0 (3.6) | 6.93 (3.70) | 5.81 (1.39) | 14.8 (6.7) | 15.3 (7.0) | 18.8 (12.1) | 15.8 (8.8) |
| $PM_{10}$ | 18.5 (4.1) | – | – | 28.9 (13.0) | – | – | – |
| Black Carbon (BC) | 0.52 (0.28) | 0.37 (0.20) | 0.055 | 0.91 (0.41) | – | – | 0.56 (0.19)[d] |
| Organic Carbon (OC) | 4.71 (2.73) | 2.62 (1.24) | 5.92 | 9.64 (4.34) | – | – | 8.6–9.7[d] |
| Oxylate ($C_2O_4$) | 0.04 (0.034) | 0.0055 (0.0055) | – | 0.040 (0.018) | – | – | – |
| Nitrate ($NO_3$) | 0.11 (0.050) | 0.016 (0.013) | – | 0.14 (0.063) | – | – | – |
| Phosphate ($PO_4$) | 5.56E-03 (8.99E-03) | 0.0045 (0.0060) | – | 1.07E-03 (4.80E-04) | – | – | – |
| Sulfate ($SO_4$) | 0.13 (0.088) | 0.018 (0.009) | – | 0.19 (0.086) | – | – | – |
| Ammonium ($NH_4$) | 5.64E-03 (1.72E-02) | 0.0035 (0.0035) | – | 3.97E-03 (1.79E-03) | – | – | – |
| Cl | 0.15 (0.16) | 0.23 (0.055) | – | 0.24 (0.11) | – | – | – |
| Ca | 0.085 (0.089) | 0.021 (0.018) | – | 0.020 (0.009) | – | – | – |
| Mg | 0.040 (0.034) | 0.016 (0.007) | – | 0.030 (0.014) | – | – | – |
| Na | 6.37E-03 (5.46E-03) | 0.0055 (0.0045) | – | 0.030 (0.014) | – | – | – |
| K | 0.29 (0.28) | 0.23 (0.053) | – | 0.34 (0.15) | – | – | – |

[a] EF calculated from a weighted average of boreal and temperate forest EF based on GFED v.3 biomass consumption estimates.

[b] Number of particles $kg^{-1}$ fuel burned

[c] $PM_1$–$PM_5$ categorized as $PM_{2.5}$

[d] Source is Andreae and Merlet (2001)

**Table 2.** Emission factors ($g\,kg^{-1}$) for species emitted from different types of biomass burning.

| Compound | Peatland | Chaparral | Open Cooking | Patsari Stoves | Charcoal Making[a] | Charcoal Burning[b] | Dung Burning | Garbage Burning |
|---|---|---|---|---|---|---|---|---|
| Carbon Dioxide ($CO_2$) | 1563 (65) | 1703 (38) | 1548 (125) | 1610 (114) | 1626 (244) | 2385 | 859 (15) | 1453 (69) |
| Carbon Monoxide (CO) | 182 (60) | 71 (13) | 77 (26) | 42 (19) | 255 (52) | 189 (36) | 105 (10) | 38 (19) |
| Methane ($CH_4$) | 11.8 (7.8) | 2.74 (0.74) | 4.86 (2.73) | 2.32 (1.38) | 39.6 (11.4) | 5.29 (2.42) | 11.0 (3.3) | 3.66 (4.39) |
| Acetylene ($C_2H_2$) | 0.14 (0.093) | 0.20 (0.08) | 0.97 (0.50) | 0.28 (0.01) | 0.21 (0.02) | 0.42 | – | 0.40 (0.28) |
| Ethylene ($C_2H_4$) | 1.79 (0.72) | 0.81 (0.18) | 1.53 (0.66) | 0.46 (0.12) | 3.80 (1.15) | 0.44 (0.23) | 1.12 (0.23) | 1.26 (1.04) |
| Ethane ($C_2H_6$) | – | 0.36 (0.11) | 1.50 (0.50) | – | 12.2 (9.3) | 0.41 (0.13) | – | – |
| Propylene ($C_3H_6$) | 2.3 (0.74) | 0.41 (0.13) | 0.57 (0.34) | 0.03 | 4.12 (1.89) | – | 1.89 (0.42) | 1.26 (1.42) |
| Propane ($C_3H_8$) | – | 0.19 (0.09) | – | – | – | – | – | – |
| Butane ($C_4H_{10}$) | – | 0.14 (0.07) | – | – | – | – | – | – |
| Isoprene ($C_5H_8$) | 1.07 (0.44) | – | – | – | – | – | – | – |
| Toluene ($C_6H_5CH_3$) | 1.21 (0.69) | – | – | – | – | – | – | – |
| Benzene ($C_6H_6$) | 2.46 (1.21) | – | – | – | – | – | – | – |
| Methanol ($CH_3OH$) | 5.36 (3.27) | 0.91 (0.29) | 2.26 (1.27) | 0.39 (0.39) | 54.9 (27.9) | 1.01 | 4.14 (0.88) | 0.94 (1.25) |
| Acetol ($C_3H_6O_2$) | 3.43 (0.36) | – | – | – | 21.6 (35.3) | – | 9.60 (2.38) | – |
| Phenol ($C_6H_5OH$) | 4.36 (5.06) | 0.44 (0.08) | 3.32 | – | 10.4 (6.6) | – | 2.16 (0.36) | – |
| Furan ($C_4H_4O$) | 1.51 (0.37) | 0.22 (0.09) | 0.40 | – | 3.94 (2.30) | – | 0.95 (0.22) | – |
| Formaldehyde (HCHO) | 1.69 (1.62) | 0.88 (0.27) | 2.08 (0.86) | 0.37 (0.40) | 3.62 (2.42) | 0.60 | – | 0.62 (0.13) |
| Glycolaldehyde ($C_2H_4O_2$) | 1.22 (1.95) | 0.06 (0.19) | 0.66 | – | – | – | – | – |
| Acetaldehyde ($CH_3CHO$) | 2.81 (1.36) | – | – | – | – | – | – | – |
| Carbonyl Sulfide (OCS) | 1.20 (2.21) | – | – | – | – | – | – | – |
| Acetic Acid ($CH_3COOH$) | 7.08 (3.40) | 1.32 (0.45) | 4.97 (3.32) | 0.34 | 44.8 (27.3) | 2.62 | 11.7 (5.08) | 2.42 (3.32) |
| Formic Acid (HCOOH) | 0.54 (0.71) | 0.06 (0.03) | 0.22 (0.17) | 0.0048 | 0.68 (0.20) | 0.063 | 0.46 (0.31) | 0.18 (0.12) |
| Acetone ($C_3H_6O$) | 1.08 (0.29) | – | – | – | – | – | – | – |
| Hydrogen Cyanide (HCN) | 5.00 (4.93) | 0.46 (0.11) | – | – | 0.21 (0.17) | – | 0.53 (0.30) | 0.47 |
| Methyl Ethyl Ketone ($C_4H_8O$) | – | – | – | – | – | – | – | – |
| Hydrogen Chloride (HCl) | – | 0.17 (0.14) | – | – | – | – | – | 3.61 (3.27) |
| Methyl Vinyl Ether ($C_3H_6O$) | 0.85 | – | – | – | – | – | – | – |
| Acetonitrile ($CH_3CN$) | 3.70 (0.90) | – | – | – | – | – | – | – |
| Sulfur Dioxide ($SO_2$) | – | 0.68 (0.13) | – | – | – | – | 0.06 | 0.5 |
| Hydrogen ($H_2$) | – | – | – | – | – | – | – | 0.091 |
| Ammonia ($NH_3$) | 10.8 (12.4) | 1.26 (0.62) | 0.87 (0.40) | 0.03 | 1.24 (1.44) | 0.79 | 4.75 (1.00) | 0.94 (1.02) |
| Nitrogen Oxides ($NO_x$ as NO) | – | 3.29 (1.02) | 1.42 (0.72) | – | 0.22 (0.22) | 1.41 | 0.5 | 3.74 (1.48) |
| Nitrous Oxide ($N_2O$) | – | 0.25 (0.18) | – | – | – | 0.24 | – | – |

**Table 2.** Continued.

| Compound | Peatland | Chaparral | Open Cooking | Patsari Stoves | Charcoal Making[a] | Charcoal Burning[b] | Dung Burning | Garbage Burning |
|---|---|---|---|---|---|---|---|---|
| Nitrous Acid (HONO) | – | 0.39 (0.15) | – | – | – | – | – | – |
| TNMHC as $CH_4$ | – | – | 2.89 (1.21) | 3.76 (4.53) | – | – | – | – |
| TNMHC as g C | – | – | 2.27 (2.07) | – | – | – | – | – |
| NMOC (identified) | 48.8 (30.3) | 6.6 (2.3) | 18.5 (7.6) | 1.87 (0.92) | 161 (115) | 5.56 | 32.6 (10.2) | 11.2 (10.8) |
| NMOC (identified+unidentified) | 97.6 | 13.3 | 55.4 | 5.62 | 321 | 11.1 | 97.7 | 33.5 |
| Total Suspended Particulate (TSP) | – | 15.4 (7.2) | 4.55 (1.53) | 3.34 (1.68) | 2.5 (3.5) | 2.38 | – | – |
| Total Particulate C | – | – | – | – | – | – | 22.9 | – |
| $PM_{2.5}$[c] | – | 12.2 (5.7) | 6.64 (1.66) | – | – | – | – | 9.8 (5.7) |
| Black Carbon (BC) | 0.17 (0.092) | 1.3 | 0.83 (0.45) | 0.74 (0.37) | 0.02 (0.02) | 1.0[d] | 0.53[d] | 0.65 (0.27) |
| Organic Carbon (OC) | – | 3.7 | 2.89 (1.23) | 1.92 (0.90) | 0.74 (0.72) | 1.3[d] | 1.8[d] | 5.27 (4.89) |
| Organic Aerosol (OA) | – | 5.93 | – | – | – | – | – | – |
| Sulfate ($SO_4$) | – | 5.98E-03 | – | – | – | – | – | – |
| Nitrate ($NO_3$) | – | 0.21 | – | – | – | – | – | – |
| Ammonium ($NH_4$) | – | 0.10 | – | – | – | – | – | – |
| Cl | – | 0.08 | – | – | – | – | – | – |

[a] EF reported in units of g of compound emitted per kg of charcoal produced

[b] EF reported in units of g of compound emitted per kg of charcoal burned

[c] $PM_1$–$PM_5$ categorized as $PM_{2.5}$

[d] Source is Bond et al. (2004)

**Table 3.** Biomass loading, combustion factor, and biomass consumption estimates for various fuel types.

| Fuel Type | Reference | Location | Vegetation specifics | Biomass Loading ($Mg\,ha^{-1}$) | Combustion Factor (%) | Biomass Consumption ($Mg\,ha^{-1}$) |
|---|---|---|---|---|---|---|
| Tropical Dry Forest | Kauffman et al. (2003) | Mexico | Deforestation | 118.2–134.9 | 62.4–80.2 | 73.7–108.1 |
| | Kauffman et al. (1993) | Brazil | Deforestation | 73.8 | 87 | 64 |
| | Jaramillo et al. (2003) | Mexico | Deforestation | 112.2 | – | – |
| | Shea et al. (1996) | Zambia | Understory | 5.10–5.77 | 88–74 | 4.5–4.3 |
| | Hoffa et al. (1999) | Zambia | Understory | 10.435 | 22.3 | 2.30 |
| | Ward et al. (1992) | Brazil | | 9.3 | 78 | 7.3 |
| | van der Werf et al. (2010), GFED3 | Central America | Central America (CEAM) | – | – | 29.8 |
| Evergreen Tropical Forest | Ward et al. (1992) | Brazil | | 292.4 | 53 | 155 |
| | Fearnside et al. (1993) | Brazil | | 265 | 27.5 | 73 |
| | Carvalho et al. (1998) | Brazil | | 401.5 | 20.47 | 82 |
| | Carvalho et al. (2001) | Brazil | | 496 | 50 | 248 |
| | Hughes et al. (2000) | Mexico | | 403 | 95 | 380 |
| | Kauffman et al. (1995) | Brazil | | 355.4 | 51.6 | 185 |
| | Guild et al. (1998) | Brazil | | 354.8 | 47 | 167 |
| | van der Werf et al. (2010), GFED3 | Equatorial Asia | Equatorial Asia (EQAS) | – | – | 190 |
| Crop Residue | Zarate et al. (2005) | Spain | Cereal crops | – | 80 | 1.14 |
| | Hughes et al. (2000) | Mexico | Cornfield | 23 | – | – |
| | Lara et al. (2005) | Brazil | Sugarcane | – | – | 20 |
| Peatland | Page et al. (2002) | Indonesia | | – | – | 510 |
| Pasture | Hughes et al. (2000) | Mexico | | 24 | – | – |
| | Guild et al. (1998) | Brazil | | 66.3 | 31 | 21 |
| | Kauffman et al. (1998) | Brazil | | 81.7 | 50 | 34 |
| | Kauffman et al. (2003) | Mexico | | 29.0–40.3 | 75–63 | 21.8–25.4 |
| | Jaramillo et al. (2003) | Mexico | | 26.9 | – | – |
| Savanna | Ward et al. (1992) | Brazil | Tropical savanna | 7.2 | 99 | 7.1 |
| | Savadogo et al. (2007) | West Africa | Woodland savanna | – | – | 4.1 |
| | Shea et al. (1996) | South Africa | | 3.807 | 76 | 2.9 |
| Boreal Forest | Goode at al. (2000) | Alaska, USA | | – | – | 36 |
| | S. Drury (unpublished work, 1998) | Alaska, USA | Wildfire B309, 28 Jun 1997 | – | – | 37 |
| | van der Werf et al. (2010), GFED3 | North America | Boreal North America (BONA) | – | – | 53.2 |
| | FIRESCAN Science Team (1996) | Bor Forest Island, Siberia | Prescribed crown fire | – | – | 38 |

**Table 3.** Continued.

| Fuel Type | Reference | Location | Vegetation specifics | Biomass Loading ($Mg\,ha^{-1}$) | Combustion Factor (%) | Biomass Consumption ($Mg\,ha^{-1}$) |
|---|---|---|---|---|---|---|
| | Cofer et al. (1998) | Northwest Territories, Canada | Prescribed crown fire | – | – | 42.7 |
| | van der Werf et al. (2010), GFED3 | Asia | Boreal Asia (BOAS) | – | – | 39.6 |
| | Kasischke et al. (1999) | Global estimate | | – | – | 10–60 |
| | Stocks (1991) | Global estimate | | – | – | 25 |
| | Cahoon et al. (1994, 1996) | Global estimate | | – | – | 25 |
| | de Groot et al. (2009) | Canada | | – | – | 22 |
| Temperate Forest | Sah et al. (2006) | Florida, USA | Florida Keys pine forests | 60.6 | – | – |
| | Snyder (1986) | Florida, USA | Everglades NP | 75–90 | – | – |
| | van der Werf et al. (2010), GFED3 | North America | Temperate North America (TENA) | – | – | 12.5 |
| Chaparral/Shrub | Cofer et al. (1988) | S. California, USA | Chaparral | – | – | 20–70 |
| | Clinton et al. (2006) | S. California, USA | Chaparral | 28.25 | – | – |
| | Ottmar et al. (2000) | S. California, USA | Chaparral | – | – | 15.0 |
| | Hardy et al. (1996) | S. California, USA | Chaparral | – | – | 24.5 |

**Table 4.** Global estimates of biomass consumption in units of mass of dry matter burned (Tg) per year.

| Year measured | 1990's | mid 1990's | 2000 | 1993/1995 | 1985 | | |
|---|---|---|---|---|---|---|---|
| | Andreae and Merlet (2001)[a] | Bond et al. (2004)[b] | Fernandes et al. (2007)[c] | Ludwig et al. (2003)[d] | Yevich and Logan (2003)[e] | Other[f] | Average |
| Savanna | 3160 | 3572 | – | – | – | – | 3366 |
| Forest | 1970 | 1939 | – | – | – | – | 1955 |
| Tropical forest | 1330 | – | – | – | – | – | 1330 |
| Extratropical forest | 640 | – | – | – | – | – | 640 |
| Biofuel | 2897 | – | 2458 | – | 2447 | – | 2601 |
| Cooking Stoves | – | – | 1351 | – | – | – | 1351 |
| Open Cooking (fuelwood) | – | – | – | 1062 | 1714 | – | |
| Charcoal Burning | 38 | – | 39 | 24 | – | – | 39 |
| Charcoal Making | 43 | – | – | – | – | – | 43 |
| Crop Residue (for biofuel) | – | – | 495 | – | 597 | – | 546 |
| Dung | – | – | 75 | – | 136 | – | 106 |
| Industrial | – | – | 498 | – | – | – | 498 |
| Peat | – | – | – | – | – | 3400 | 3400 |
| Pasture Maintenance | – | – | – | – | – | 240 | 240 |
| Crop Residue (field burning) | 540 | 475 | – | – | 451 | – | 489 |
| Garbage Burning | – | – | – | – | – | 1000 | 1000 |

[a] Source is Andreae and Merlet (2001). Value of 640 Tg yr$^{-1}$ is cited in original work as "extratropical forest", which encompasses both boreal and temperate forest types. "Biofuel" global estimate derived from the sum of biofuel burning, charcoal making, and charcoal burning estimates.
Charcoal making estimate of 43 Tg yr$^{-1}$ was calculated assuming a 27% charcoal yield (Bertschi et al., 2003a). The biomass consumption estimates were derived using methods described in Lobert et al. (1999).

[b] Source is Bond et al. (2004). Estimates from Table 4 in original work.

[c] Source is Fernandes et al. (2007). Original work defines "biofuel" as fuelwood (open cooking), charcoal burning, crop residues and dung.

[d] Source is Ludwig et al. (2003).

[e] Source is Yevich and Logan (2003). "Biofuel" defined as woodfuel, charcoal burning, crop residues and dung.

[f] Other. Garbage burning estimate of 1000 Tg yr$^{-1}$ from Christian et al. (2010), peat estimate of 3400 Tg yr$^{-1}$ from Page et al. (2002), and pasture maintenance estimate of 240 Tg yr$^{-1}$ from Yokelson et al. (2008).

**Table 5.** Measured and predicted estimates of NMOC emitted from biomass burning on an annual basis.

| Fuel Type | Total Combusted Biomass (Tg yr$^{-1}$)[a] | EF(CO) (g kg$^{-1}$ dry biomass)[b] | CO global production (Tg yr$^{-1}$) | EF (NMOC, identified) (g kg$^{-1}$ dry biomass)[b] | EF (NMOC, estimation of total) (g kg$^{-1}$ dry biomass)[c] | NMOC global production (Tg yr$^{-1}$) | EF (BC) (g kg$^{-1}$ dry biomass) | BC global production (Tg yr$^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| Savanna | 3366 | 59 | 199 | 9.1 | 18.2 | 61 | 0.37 | 1.25 |
| Extratropical | 640 | 123 | 79 | 26.4 | 52.8 | 34 | 0.56 | 0.36 |
| Tropical Forest | 1330 | 92 | 122 | 23.7 | 47.4 | 63 | 0.52 | 0.69 |
| Biofuel | 2601 | 77 | 200 | 18.5 | 55.5 | 144 | 0.83 | 2.16 |
| Open Cooking/Cooking Stoves | 1351 | 59.5 | 80 | 10.2 | 30.6 | 41.3 | 0.79 | 1.06 |
| Dung Burning | 106 | 105 | 11.1 | 32.6 | 97.8 | 10.4 | 0.53 | 0.056 |
| Charcoal Making | 43 | 255 | 11.0 | 161 | 322 | 13.8 | 0.02 | 8.6E-04 |
| Charcoal Burning | 39 | 189 | 7.4 | 5.56 | 11.1 | 0.43 | 1.0 | 0.039 |
| Pasture Maintenance | 240 | 135 | 32 | 44.8 | 89.6 | 21.5 | 0.91 | 0.22 |
| Crop Residue | 489 | 112 | 55 | 55.4 | 110.8 | 54.2 | 0.055 | 0.027 |
| Garbage Burning | 1000 | 38 | 38 | 11.2 | 33.6 | 33.6 | 0.65 | 0.65 |
| Peat[d] | 3400 | 182 | 619 | 48.8 | 97.6 | 332 | 0.14 | 0.48 |
| Avg. model year – global estimate | – | – | 725 | – | – | 412 | – | 5.35 |
| El Niño year – global estimate | – | – | 1344 | – | – | 744 | – | 5.83 |

[a] Total combusted biomass estimates are from Table 4 averages. Charcoal making estimate is in units of Tg charcoal made per year. Charcoal burning estimate is in units of Tg charcoal burned per year.

[b] Data are from Tables 1 and 2 of this work. EF for open cooking fires was used to represent EF for all biofuel since cooking fires are the dominant source of biofuel emissions globally. EF for open cooking/cooking stoves was taken as the averages of open cooking and cooking stove EF. Charcoal making EF in units of g kg$^{-1}$ charcoal made. Charcoal burning EF in units of g kg$^{-1}$ charcoal burned.

[c] Multiplication factors to estimate total EF(NMOC) (as identified+unidentified NMOC) is described in Sect. 3.2.

[d] Emissions from peat are added to global totals to estimate emissions during the 1997 El Niño year.