Significant Revisions

Title change to: Combustion efficiency and emission factors for wildfire season fires in mixed conifer forests of the northern Rocky Mountains, US

SR.1 (P34, L1 – P35, L27)

In the US wildfires and prescribed burning present significant challenges to air regulatory agencies attempting to achieve and maintain compliance with air quality regulations. Wildland fire emission inventories (EI) provide critical inputs for atmospheric chemical transport models used by air regulatory agencies to understand and to predict the impact of fires on air quality. Fire emission factors (EF) are essential input for the emission models used to develop wildland fire EI. Most previous studies quantifying wildland fire EF of temperate ecosystems have focused on emissions from prescribed burning conducted outside of the wildfire season. Little information is available on EF for wildfires in temperate forests of the conterminous US. The goal of this work is to provide information on emissions from wildfire season forest fires in the Rocky Mountains, US.

Over 8 days in August of 2011 we deployed airborne chemistry instruments and sampled emissions from 3 wildfires and a prescribed fire that occurred in mixed conifer forests of the northern Rocky Mountains. We measured the combustion efficiency, quantified as the modified combustion efficiency (MCE), and EF for CO_2, CO, and CH_4. Our study average values for MCE, $EFCO_2$, EFCO, and $EFCH_4$ were 0.883, 1596 g kg^{-1}, 135 g kg^{-1}, 7.30 g kg^{-1}, respectively. Compared with previous field studies of prescribed fires in temperate forests, the fires sampled in our study had significantly lower MCE and $EFCO_2$ and significantly higher EFCO and $EFCH_4$. An examination of our work and 54 temperate forest prescribed fires (PF) from previously published studies shows a clear trend in MCE across US region/fire type: southeast PF (MCE=0.935) > southwest PF (MCE=0.922) > northwest PF (MCE=0.900) > this study (MCE=0.883).

The fires sampled in this work burned in areas reported to have moderate to heavy components of standing dead trees and down dead wood due to insect activity and previous fire, but fuel consumption data was not available. However, fuel consumption data was available for 18 prescribed fires reported in the literature. For these 18 fires we found a significant negative correlation (r = -0.83, p-value = 1.7e-5) between MCE and the ratio of coarse fuel (large diameter dead wood and duff) consumption to total fuel consumption. This analysis suggests the relatively low MCE measured for the fires in our study resulted from the availability of coasre fuels and conditions that facilitated combustion of these fuels. More generally, our measurements and the comparison with previous studies indicate that fuel composition is an important driver of variability in MCE and EF.

This study only measured EF for CO_2, CO, and CH_4; however, we used our study average MCE to provide rough estimates of wildfire season EF for $PM_{2.5}$ and 4 non-methane organic compounds (NMOC) using MCE and EF data reported in the literature. The wildfire season $EFPM_{2.5}$ estimated in this analysis was nearly twice that reported for temperate forests in a two widely used reviews of biomass burning emission studies. Likewise, western US wildfire season forest fire $PM_{2.5}$ emissions reported in a recent national emission inventory are based on an effective $EFPM_{2.5}$ that is only 60% of that estimated in our study. If the MCE of the fires sampled in this work are representative of the combustion characteristics of wildfire season fires in these forest types across western US then the use of EF based on prescribed fires may result in a significant underestimate of wildfire $PM_{2.5}$ and NMOC emissions. Given the magnitude of biomass consumed by western US wildfires, this may have important implications for the forecasting and management of regional air quality.

SR2 (P36, L23 – P37, L28)

Wildland fires in the US may be divided into two classes: prescribed fires and wildfires. A prescribed fire is any fire ignited by management actions to meet specific objectives. Two common land management objectives pursued with prescribed fire are wildfire hazard reduction and ecosystem restoration (Agee and Skinner, 2005; Finney et al., 2005; Varner et al., 2005). While prescribed burning dominates fire activity in the southeastern US (~75% of area burned between 2002-2010; (NIFC, 2012)), wildfires are dominant in the western US (defined here as: Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, Wyoming)(~85% of area burned between 2002-2010; (NIFC, 2012)). The majority of prescribed burning in the western US occurs outside of the wildfire season which typically runs June – September (October in California) (Urbanski et al., 2011). From 2008 to 2011 77% of the area treated with prescribed fire on federal lands burned between January – May or October – December (Lahm, 2013). The seasonal wildfire activity originates in the southwest (Arizona, New Mexico, Nevada, and Colorado) in June. During July, the fire activity expands northward along the Rocky Mountains and through the Great basin with the epicenter of activity migrating into northern Nevada and southern Idaho. Fire activity occurs throughout the interior west and Pacific Northwest over July. By August, the center of fire activity moves into the northern Rocky Mountains and Pacific Northwest. Fire activity decreases in September and outside of California is minimal in October. Wildfires in California exhibit more seasonal variability and significant fire activity may occur throughout June – October. In California fire activity is usually greatest October and in some years wildfire conditions persist into well into November. The extent to which wildfires impact different western US ecosystems varies intra-seasonally and inter-annually. In recent years forests have comprised 44% of wildfire burned area (based on a geospatial overlay of 2001-2010 Monitoring Trends in Burn Severity fire boundaries (MTBS, 2013) and a Remote Sensing Application Center / Forest Inventory Analysis forest map (Ruenfenacht et al., 2008)). However, owing to the greater fuel loadings (dead and live biomass available for combustion per unit area (Albini 1976; Sandberg et al., 2001)) of forests compared to grasslands and shrublands fuel consumption and emissions are dominated by forest fires (Urbanski et al., 2011).

Recent emission estimates published prior to this study suggest that wildland fires account for a sizeable fraction of the annual total PM2.5 and CO emissions in the western US (as much as 39% and 20%, respectively)(Urbanski et al., 2011). Because wildfire emissions are episodic and highly concentrated both temporally and spatially (Urbanski et al., 2011), such annualized comparisons may greatly understate the potential impact of the wildfires on the day time scale that is pertinent to air quality forecasting and management.

Wildland fire emission inventories (EI) provide critical input for atmospheric chemical transport models used by air regulatory agencies to understand and to predict the impact of fires on air quality. Fire emission factors (EF), which quantify the amount of pollutants released per mass of biomass burned, are essential input for the emission models used to develop EI. Over the past decade substantial progress has been realized characterizing the composition of fresh BB smoke and in quantifying BB EF (see Akagi et al., 2011). Yet significant gaps in the current knowledge of EF remain in many areas. Little information is available on EF for forest fires that occur in the western US during the wildfire season. Emission estimates for US wildfire season forest fires rely largely on EF measurements from prescribed fires. However, because prescribed fires are usually conducted outside of the wildfire season they may not be a suitable proxy for wildfires. The combustion characteristics of a fire, in particular the relative amounts of flaming and smoldering combustion, have a significant influence on the chemical composition of the smoke. Smoldering combustion is less efficient than flaming combustion and per unit of fuel consumed produces more CO, CH_4, non-methane organic compounds (NMOC), and particulate matter and less CO_2 (Bertschi et al., 2003; Burling et al., 2010; Lobert, 1991; Yokelson et al., 1996, 2008). Smoldering combustion is prevalent in coarse woody debris (CWD, large diameter (> 7.62 cm) dead wood) and duff while fine fuels (grasses, shrubs, foliage, litter, and fine woody debris (FWD, small diameter (< 7.62 cm) dead wood)) tend to burn by mostly flaming combustion (Ottmar, 2001; Sandberg et al., 2002). Therefore, the characteristics of the fuels consumed in a wildland fire, which is determined by the fuels present and environmental conditions, should have an important influence on the composition of emissions.

SR.3 (P38, L 1-21)

Conditions during the western U.S. wildfire season, low fuel moistures, and high-intensity fire fronts, are favorable for the consumption of CWD and duff and these fuels may comprise a significant portion of total fuel consumed in a forest fire (Campbell et al., 2007; Reinhardt et al., 1991). Conversely, prescribed burning is generally characterized by low intensity fire and is conducted outside the wildfire season when the moisture of CWD and duff are moderate to high (Finney et al., 2005; Hardy, 2002), conditions which minimize consumption of these fuels relative to fine fuels. Thus, forest fires occurring during the wildfire season might be expected to burn with more smoldering combustion than typical prescribed fires and have higher EF for species associated with smoldering combustion (and lower EF for species related to flaming combustion). This reasoning suggests EF based on prescribed fires may not be appropriate for modeling emissions from wildfire season (WFS) forest fires.

We present smoke emissions data from airborne field measurements of fires that occurred in mixed conifer forests of the Middle and Northern Rocky Mountains during the 2011 wildfire season. We report measurements of modified combustion efficiency (MCE) and EF for CO_2, CO, and CH_4 and compare these with previous field studies of temperate forest fires. The MCE measured in our field study are used to estimate WFS forest fire EF for $PM_{2.5}$ (particulate matter with an aerodynamic diameter < 2.5 μm), ethane (C2H6), propylene (C3H6), formaldehyde (HCHO), and methanol (CH3OH) using previously published EF – MCE relationships. This new WFS EF dataset for Rocky Mountain forest fires is compared with a recent review article and a national emissions inventory. We also examine MCE and fuel consumption data from previous studies of 18 prescribed fires to identify possible drivers of fire combustion characteristics.

SR.4 (P39, L6 – P41, L2)

The locations and vegetation involved for the fires studied are provided in Table 1. All 4 fires occurred in montane mixed conifer forests of Lodgepole Pine, Douglas-Fir, Engelmann Spruce, and Subalpine Fir in the Northern or Middle Rocky Mountains. The methods used to determine the vegetation involved and fire elevation are described in Supplement 1. The size, growth, and activity of the fire and fuel moisture conditions on each day of sampling are provided in Table S1. Fuel moisture conditions were similar across all sites on the days the fires were sampled and were typical of wildfire season conditions in the Northern and Middle Rocky Mountains. The afternoon 10-hour fuel moisture (moisture of fine dead wood (diameter < 2.5 cm)) was ~ 5% and the 1000-hour fuel moisture (the moisture typical of large diameter (> 7.5 cm) dead wood) was between 9% and 14%. The estimated daily fire growth, derived from incident fire perimeters and ICS-209 reports (Supplement 1), ranged from ~ 0 to 774 ha (Table S1).

The daily fire activity was not limited to areas of perimeter growth. Fires occurring in complex, forested terrain often burn with mixed-severity creating a mosaic that includes areas that were unburned or where fuel consumption was limited. The burning of previously unburned areas and the smoldering of large diameter woody fuels and duff can continue for days after the passage of the fire front. On most days of sampling we observed wide spread smoldering and regions of active burning (flaming combustion) scattered within the fire perimeter. An example of post-frontal combustion is provided by Figure 1 which shows fire perimeters, areas of active burning, and the region of smoke sampling from the Saddle Complex on August 24. The perimeters, as observed via airborne IR sensor on the evenings (August 23 and 24), indicate that on August 24 the fire growth occurred mostly on the west and east ends, with some minor growth along the northern and southern edges. In addition to the active fire fronts on the perimeter we also observed many pockets of burning scattered within the perimeter while sampling on the afternoon of the August 24. The MODIS burn scar data (RSAC, 2012b) and active fire detections (RSAC, 2012a) for August 24 captured some of this activity (Fig. 1).

2.1.1 Hammer Creek Fire

The Hammer Creek Fire was ignited by lighting on July 19, 2011 in the Bob Marshall Wilderness in northwestern Montana and burned an estimated 2555 ha (elevation of 1360 m to 2250 m a.m.s.l. (above mean sea level)) before being declared under control on October 7, 2011 (Carbonari, S., 2011c). The fire burned in the South Fork Flathead drainage where Douglas-fir / Lodgepole Pine forests are maintained by mixed-severity fire regime (Arno et al., 2000). In addition to the dominant forest types listed in Table 1, the area burned by the Hammer Creek Fire included emergent Larch and Ponderosa Pine as important ecosystem components (Arno et al., 2000; Larson, 2013). The incident management team reported the fire was burning in "mature timber with moderate to heavy dead standing and dead down" trees and also in the area of previous burns with "moderate to heavy component of dead/down fuel" (Carbonari, S., 2011b). The previous burns which occurred in 2003 were the Little Salmon Creek Fire and the Bartlett Mountain Fire (MTBS, 2012). The Hammer Creek Fire was sampled only on August 22, a day when the fire activity involved mostly creeping and smoldering and growth of the fire was minimal (Table S1).

2.1.2 Big Salmon Lake Fire

The Big Salmon Lake Fire started from an unknown cause on August 16, 2011 in the Bob Marshall Wilderness in northwestern Montana, about 10 km northwest of the Hammer Creek Fire. The fire burned in steep terrain at elevation of 1320 m to 2400 m a.m.s.l. and its perimeter encompassed ~2200 ha when declared controlled on October 7, 2011 (Carbonari, S., 2011a). With the exception of a ~70 ha pocket, the area of the Big Salmon Lake Fire had not been significantly impacted by fire in over 25 years (MTBS, 2012). An aerial forest health survey conducted in 2010 found ~10% of the area burned by the Big Salmon Lake fire area was impacted by mortality due to beetles (USDA, 2012b).

The Big Salmon Lake Fire was sampled on August 17, 22, and 28. On August 17 we sampled the fire in the late afternoon when it spread rapidly with significant spotting and sustained crown runs with wind-driven and terrain induced spread. (ref-ICS). While the Big Salmon Lake Fire was reported at 14:00 local time on August 16 it was not detected by MODIS until the following afternoon. The size of the Big Salmon Lake Fire was estimated as 295 ha on the late afternoon of August 17 and the fire perimeter measured on that night's IR aerial survey placed the fire size at 900 ha. On August 22, the Salmon Lake fire spread was low to moderate with creeping, smoldering, and some single tree torching (ref-ICS). The fire grew ~ 405 ha on August 28 and the fire activity included spreading ground fire and group torching of tree crowns (ref-ICS).

2.1.3 Saddle Complex

The Saddle Complex was a fire complex in the Middle Rocky Mountains along the Idaho – Montana border that formed when the Saddle Creek Fire and Stud Fire merged on August 18, 2011. The Saddle Creek Fire was ignited by lightning on August 10 in the Bitterroot National Forest in Montana. The Stud Fire was also caused by lightning and began on August 14 in the Salmon-Challis National Forest in Idaho. The fire complex was managed as two separate fire incidents (Salmon-Challis Branch and the Bitterroot Branch). The fire burned in complex terrain at an elevation of 1040 m to 2650 m a.m.s.l. with a final perimeter area of 13,770 ha. A substantial portion of the trees in the impacted forest were dead from insect kill (> 40% Bitterroot Branch and 20-45% Salmon-Challis Branch; (Central Idaho Dispatch, 2011; McKee, K., 2011)). The forest burned by the Saddle Complex had not been impacted by wildfire in over 25 years (MTBS, 2012).

On the four days the Saddle Complex was sampled its perimeter grew 200 to 800 ha day-1 and fire activity included group torching of tree crowns as well as running crown fire (Table S1). MODIS active fire detections and daily burn scars (RSAC, 2012a, 2012b) were consistent with our airborne observation that significant fire activity and smoke emissions occurred within the perimeter, especially on Aug 24 and 25 (Figure 1).

2.1.4 North Fork Prescribed Fire

The North Fork Prescribed Fire was actually two burns ignited on August 12, 2011 in the North Fork District of the Clearwater National Forest in northern Idaho. The prescribed burn targeted diseased and insect-infested areas (Lubke, M., 2011). Fire history data for the Clearwater National Forest indicate the burn unit had not been impacted by significant wildfire activity since 1910 (USDA, 2013). The fires were ignited using a combination of aerial and hand ignition and allowed to burn with the goal of 400 ha eventually burning. The fire was sampled on August 13. MODIS active fire detections indicate the fire persisted until August 25 (RSAC, 2012a).

SR.5 (P 42, L 11 – P43, L4)

Emissions were measured by sampling the smoke above the flame front and up to 40 km downwind at elevations between 300 m above ground level (agl) and plume top. A sample run was level altitude flight segment that began outside of the smoke plume in clean background air, entered the smoke plume, continued through the smoke plume and eventually exited the smoke plume into the background air. The portion of each sample run prior to plume entry provided a background measurement of at least 20 data points (2400 m at typical flight speed of 60 m s^{-1}) and the average CO, CO_2, and CH_4 mixing ratios of these data points were used as the background for calculating excess mixing ratios (see Sect. 2.4) for that sample run. The pre-plume background CO was used as a baseline to identify samples inside the smoke plume (the plume penetration) which were used as the smoke sample data points (see Sect. 2.4). Two flight profiles, parallel and perpendicular, were used for smoke sampling. The parallel profile began a few km upwind of the fire in smoke free air on a trajectory that was roughly parallel to the direction of the plume transport. In the parallel profile the aircraft penetrated the smoke plume immediately above or near the fire front. After passing over a segment of the fire front the sample run would continue to sample smoke in the plume for a several km downwind.

The parallel profile may be illustrated by considering smoke sampling from the Saddle Complex on August 24. On this day fresh smoke samples were obtained along the northern edge of the fire perimeter (Fig. 1). Winds were from the WSW and the initial portion of our sampling runs captured emissions emanating from the within the western area of the perimeter just downwind as they reached neutral buoyancy. The runs proceeded to the ENE sampling smoke above the fire front on the northern perimeter and then continued downwind with the plume that entrained smoke from across the fire complex. The perpendicular flight profile involved transecting the smoke plume on a level altitude trajectory that was roughly perpendicular to the direction of smoke transport at distances of 2 – 40 km downwind of the fire front. Sampling in the perpendicular mode typically crossed the entire width of the plume and provided measurements of background air on one or both ends of the sample run. The extensive downwind perpendicular transects of smoke obtained in this study may be used for the validation of smoke dispersion models. However, the focus of this paper is limited to EF.

Sample runs often encountered multiple smoke plumes as interior regions of the perimeter with active fire were transected. It is common for wildfires in complex forested terrain to spread unevenly across the landscape due to changing weather conditions and variability in fuels and terrain resulting in a burn with mixed severity (Arno, 1980; Hudec and Peterson, 2012; Schwind, 2008). The wildfires sampled in this study had active fire occurring, often discontinuously along a large portion of the fires' perimeters. We frequently observed group torching of tree crowns and short runs of crown fire along portions of the fire front. Active fire was also typically scattered throughout the perimeter as areas unburned or lightly burned during progression of the initial fire front burned/re-burned. Pockets of vigorous fire activity within the perimeter appeared to entrain and loft smoke from the surrounding smoldering fuels.

SR6 (P 45, L13 – P 47, L9)

CRDS measurements for a typical fresh smoke sample run, in this case the Saddle Complex on August 24 (Table 2, sample SC2402) are shown in Fig. 2. The horizontal dashed line in each plot marks the background mixing ratios measured upwind of the fire. The background mixing ratios for this sample (CO_2 = 382.56 ppm, CH_4 = 1.856 ppm, and CO = 0.110 ppm) were typical of the background for fire-days. The smoke sampling was conducted from an unpressurized aircraft in conditions that were often very turbulent. Under these challenging circumstances the study average in-flight measurement precision, defined as the 14 s standard deviation while measuring a calibration standard, was 0.024 ppmv for CO, 0.281 ppmv for CO_2, and 0.005 ppmv for CH_4. The uncertainties for our EMR, ΔX, were estimated as $\sqrt{\sigma_{bkgd}^2 + \sigma_{prec}^2}$, where σ_{bkgd} is the standard deviation of X_{bkgd} and σ_{prec} is the 14 s standard deviation of X during the CRDS calibration applied to that sample. The study average (n = 63) uncertainty in sample average ΔX (Table 2) was ±5% for CO, ±7% for CO_2, and ±9% for CH_4.

EF, MCE, and average ΔX for all 63 fresh smoke samples are given in Table 2. The fire-day average EF (Eq. 1) agreed within 10% with the EF that were calculated from zero-forced linear regression of the emission ratios of the 2 s data points. The number of data points for each smoke sample depended on the flight profile, aircraft speed, and dispersion conditions. At a typical aircraft ground speed of 60 m s^{-1} each 2 s data point represents a 120 m segment. Some plume samples were taken significant distances downwind of the source. In particular, on August 17, samples were taken 40 km downwind of the Big Salmon Lake Fire. The afternoon atmospheric sounding at Great Falls, Montana (NOAA, 2012) on this day indicated the transport winds were ~11 m s^{-1} implying a smoke age of ~ 60 minutes for these samples. However, since CO_2, CO, and CH_4 are fairly non-reactive in the atmosphere (CO, the most chemically reactive of the 3 gases, has a lifetime > 30 days with respect to chemical reaction (Seinfeld and Pandis, 2006)) the age will not impact the measured EF for these gases.

Figure 3(a-c) shows the average, range, and ±1σ of MCE, EFCO, and $EFCH_4$ for each fire-day. The Big Salmon Lake Fire and the Saddle Complex were sampled on multiple days and as mentioned previously, we have treated these sampling days as separate fires, identifying each as a ‘fire-day’ (Sect. 2.1). The extreme fire-days of the study were the North Fork Fire and Hammer Creek Fire. The North Fork Fire had the lowest MCE (0.867) and the highest EFCO (153.6) and EFCH4 (7.89) while the Hammer Creek Fire had the highest MCE (0.897) and the lowest EFCO (119.3) and EFCH4 (6.43). The low MCE and high EFCO and EFCH4 of the North Fork Fire may reflect the lack of canopy fire activity. The North Fork Fire was the only fire for which we did not observe canopy fire activity. Because ICS-209 reports were not filed for the North Fork Fire (Table S1) we do not have independent verification of our airborne observations. The North Fork Fire and the Hammer Creek Fire were sampled only one day each and we cannot speculate if the emissions we report were characteristic of these fire events over time. We did observe a fair amount of inter-day emissions variability for the Big Salmon Lake Fire and the Saddle Complex. The EFCO and EFCH4 for the Big Salmon Lake Fire on August 17 and 22 fell on opposite ends of the fire-day range and the samples for these days were significantly different (Mann-Whitney test, $p < 0.01$).

The Hammer Creek Fire and Big Salmon Lake Fire, which were located about 10 km apart, were both sampled on August 22. Interestingly, the Hammer Creek Fire had highest MCE of the study while the MCE for the Big Salmon Lake Fire measured on this day was the lowest measured for that particular fire (and third lowest of all fire-days). This may reflect the differential fire behavior at the two sites, the HC fire had group torching along the perimeter while the BSL fire which had only isolated torching of single trees (Table S1). Since the burning of conifer crowns, which consumes needles and fine branch wood, occurs with high MCE (ref), emissions from group torching or a crown fire run may have a significant influence on the EF measured.

We did observe some large variations in the sample EF within fire-days. In particular, the range in sample EFCO was 83.7, 56.8, and 55.9 g kg-1 for the Big Salmon Lake Fire on 8/28, the Hammer Creek Fire on 8/22, and the Saddle Complex on 8/27, respectively. These fire-days also had the largest inter-sample range in EFCH4 and MCE. In general, we believe the sample variability within fire-days is partially attributable to the sporadic nature of the crown fire activity of the fires. Since the consumption of conifer needles and fine branch wood occurs with high MCE (Chen et al., 2007; Yokelson et al., 1996), emissions from group torching or a crown fire run could have a significant influence on the EF measured during a sample run. The canopy fire activity we observed was patchy and intermittent, observations corroborated by the fire management team reports (Table S1) and the USFS Rapid Assessment of Vegetation Condition after Wildfire (RAVG) analysis of these wildfires (RAVG, 2013). The fact the three fire-days with the largest sample range also had the three lowest EFCO (and highest MCE) of the study supports the notion that their large range in EFCO resulted in part from the relatively high contribution of canopy fire emissions during some periods while the fires were being sampled.

Our study average values (average of the 9 fire-day values) for MCE, $EFCO_2$, EFCO, and $EFCH_4$ are 0.883±0.010, 1596±23 g kg^{-1}, 135±11 g kg^{-1}, 7.30±0.58 g kg^{-1}, respectively (uncertainties are 1 standard deviation). The fire-day average values of EFCO and $EFCH_4$ are confined to a fairly narrow span of 26% and 21% of the study average, respectively, and the standard deviations are only ~10% of the study average (Table 2). This limited inter-fire-day variability supports the idea that the dataset average values are more broadly representative of wildfire season forest fires in the western US. We note that despite the limited span of MCE and EFCH4 observed in our study, our measurements are sufficiently precise to reveal an MCE – EFCH4 relationship. CH4 is produced by smoldering combustion processes, and as expected, EFCH4 has a strong inverse correlation with MCE (Fig. 3d; r = -0.87, p-value = 0.002).

SR.7 (Paragraph added at P50, L3)

Hornbrook et al. (2011) report on 7 biomass burning plumes from California wildfires measured from the NASA DC-8 aircraft during the ARCTAS experiments in June and July of 2008. They do not report EF for CO2, CO or CH4, but they do report MCE. The average of MCE of the 7 biomass burning plumes was 0.911. Analysis of fire data, (MTBS, 2012; RSAC, 2012; Urbanski et. al (2011)) and vegetation maps (Ruenfenacht et al., 2008) indicates the area burned in California during ARCTAS (Jun 15 – July 15, 2008) was 30% non-forest and 70% forest (40% western oak, 23% California mixed conifer, and 10% ponderosa pine). Using the DC-8 back trajectories from the ARCTAS data archive (http://www-air.larc.nasa.gov/cgi-bin/arcstat-c) and fire data (MTBS, 2012; RSAC, 2012; Urbanski et. al (2011)) we attempted to identify the source fires or source regions of the 7 California wildfire plumes measured in Hornbrook et al (2011). We could only confidently associate 2 of the 7 California biomass burning plumes (plumes #12 and #18 in Table 1 of Hornbrook et al.) with coherent sources. Plume 18 (MCE=0.88, sampled on June 26) emissions clearly originated from the wide spread wildfires occurring in the mountains (northern Sierra Nevada, Klamath, southern Cascade, and Coastal mountains) on the northern end of the Central Valley. The fire data were combined with vegetation maps (Ruenfenacht et al., 2008) to estimate the ecosystems involved (by area) as 83% forest (52% California Mixed Conifer, 22% Western Oak, 9% other forest types) and 17% non-forest. The Back trajectories indicate the Basin Complex Fire was the main contributor to the biomass burning sampled in plume 12 (MCE = 0.91, sampled on June 18). Fire data and vegetation maps indicate the fuels involved for this plume were (by area) 40% forest (Western Oak) and 60% non-forest (mostly chaparral).

SR.8 (P50, L4 – P50, L 24)
This work measured only EF for CO_2, CO, CH_4. However, our study average MCE can be used to provide rough estimates of EF for additional species using EF and MCE data from previous studies. The NW prescribed and wildfire studies of R91, H96, and B11 have an average MCE of 0.888, very close to that measured in our study (.883). Likewise, their average $EFCH_4$ (8.2) is in good agreement with our average $EFCH_4$ (7.3) differing by only 11%. This agreement suggests that the average EF of other species reported in these studies may serve as reasonable estimates for wildfire season fires in western US mixed conifer forests. All three studies report EF for smoke particles. R91 and H96 reported $EFPM_{3.5}$; however, since coarse mode particles (2.5 – 10 μm diameter) typically account for only ~10% of the mass fraction of fresh smoke particles (Reid et al., 2005), $EFPM_{3.5}$ will not be significantly different from EFPM2.5. The nine fires from the studies have an average $EFPM_{2.5}$ of 23.2 ± 10.4 (uncertainty 1 standard deviation) and we adopt this as our best estimate of $EFPM_{2.5}$ for wildfire season fires in mixed conifer forest of the northwestern US. Since EF for many species are correlated with MCE we may use the larger body of published field measurements to estimate $EFPM_{2.5}$ at our wildfire season MCE. The $EFPM_{2.5}$ based on MCE relationships may be used to gauge the uncertainty of our best estimate $EFPM_{2.5}$. In addition to the nine NW fires of R91, H96, and B11, we also considered B11's SE airborne measurements and the tower based measurements of U09 (NW, SW, SE). We conducted linear regressions of $EFPM_{2.5}$ against MCE for four combinations of data: NW measurements (B11, H96, R91, U09), airborne measurements (B11, H96, R91), and all measurements (B11, H96, R91, U09). Plots of $EFPM_{2.5}$ vs. MCE and statistics for the linear fits are provided in Supplement S1. While the coefficients of determination for the fires were only moderate (R^2 ~ 0.60), the results were consistent with our initial rough estimate predicting an $EFPM_{2.5}$ of 23 to 25 g kg^{-1}. Due to a lack of data in R91 and H96 we relied on EF – MCE regressions to provide rough estimates of EF for CH_3OH, HCHO, C_3H_6, and C_2H_6 at our wildfire season MCE using data from R91, B11 (NW and SE), U09, (NW, SW, and SE) and Akagi et al. (2013). While the source studies provide EF for a wider range of compounds, we report estimates for only those moderately correlated with MCE (R^2 > 0.60). Fig. 4d shows EFCH4 plotted vs. MCE for data from R91, B11 (NW and SE), U09, (NW, SW, and SE), Akagi et al. (2013), and this study and a linear regression line based on the data from the previous studies. Our individual fire-day average $EFCH_4$ fall closely along the regression line and our study average $EFCH_4$ differs from the regression prediction by only 5% (7.30 g kg^{-1} vs. 7.71 g kg^{-1}).

Plots of EF vs. MCE for CH3OH, HCHO, C3H6, and C2H6 and statistics for the linear fits are provided in Supplement 1.

Rough estimates of EF for $PM_{2.5}$, CH_3OH, HCHO, C_3H_6, and C_2H_6 at our average wildfire season MCE are given in a Table 3 along with the EF measured in this work. For comparison we have included the temperate forest and boreal forest EF from the recent review of Akagi et al. (2011) (hereafter, A11) and effective EF from the U.S. EPA 2008 National Emission Inventory version 2 (USEPA, 2012a) (hereafter, NEI).

SR.9 (P51, L11 – P52, L9)

The EFCO and EFCH4 measured in our study are significantly larger than the A11 temperate forest (TF) values, but in good agreement with their boreal forest (BF) values (Table 3). However, our estimated wildfire season $EFPM_{2.5}$ is substantially larger than both the TF and BF recommendations of A11. The 50% difference between our $EFPM_{2.5}$ estimate and the A11 BF value is a bit surprising considering the similar MCE of the datasets (0.883 vs. 0.882). The comparison of our estimated EF with A11 for ethane, propylene, formaldehyde, and methanol gives mixed results. Notable differences are our estimated EFC2H6 (EFHCHO) which are roughly 50% lower (higher) than the A11 BF values.

NEI effective EF for wildfires in forests (hereafter referred to as simply NEI EF) could only be estimated for five of the species in Table 3. The NEI $EFPM_{2.5}$ and EFHCHO are 40% and 100% lower, respectively than our estimates. However, the NEI $EFCH_4$ is in close agreement with the $EFCH_4$ measured in our work. The NEI MCE of 0.847 is significantly lower than our wildfire season MCE and the NEI EF for CH_4, HCHO, and $PM_{2.5}$ are substantially lower than the EF – MCE regression equations used in this work predict (Supplement 1). The ratios of NEI EF to predicted EF are 0.70, 0.38, and 0.45 for CH_4, HCHO, and $PM_{2.5}$, respectively (Table S2).

SR.10 (P52, L10 – P53, L2)

Our field measurements show that some wildfire season forest fires in the western U.S. burn with an MCE that is significantly lower than most of the temperate forest prescribed fires reported in the literature (U09, B11, R91) and used in the development of EF recommendations for atmospheric modeling (A11; Andreae and Merlet, 2001). The lower MCE of the wildfire season forest fires we have studied indicates these fires have larger EF for species associated with smoldering combustion processes ($PM_{2.5}$ and NMOC) than are reported for temperate forests in previous studies and reviews (U09; B11; A11; Andreae and Merlet, 2001). In the western US wildfires are account for the vast majority of wildfire burned area and emissions (Urbanski et al., 2011). Because the average fuel mass consumed per unit area burned by forests is ~ 3 times that of non-forest fuels (Urbanski et al., 2011), annual wildfire emissions are dominated by forest fires even though they account for only ~44% of the area burned (see Sect. 1). The fires sampled in our study burned in Lodgepole Pine, Douglas-fir, and Engelmann Spruce / Subalpine Fir forests. While the fires were located in the Rocky Mountains of Idaho and Montana, the forest types involved are found throughout the Rocky Mountains, the Cascade Mountains, and portions of the Sierra Nevada Mountains and North Coast Ranges in California. From 2001 – 2010 these forest types comprised 19% of the total area and 43% of the forested area that was burned by wildfires in the western US (based on a geospatial overlay of 2001-2010 fire boundaries (MTBS, 2013) and a forest type map (Ruenfenacht et al., 2008)). If the fires sampled in our study are representative of wildfires in these forest types across the western US, the use of EF based on temperate forest prescribed fires may significantly underestimate $PM_{2.5}$ and NMOC emissions. Likewise, the effective $EFPM_{2.5}$ used in the NEI for wildfires in western US forests was only 60% of that estimated in this work, suggesting the inventory may underestimate $PM_{2.5}$ emissions from wildfires in the forest types addressed by our study. In the western US wildfires are an important source of $PM_{2.5}$ and other air pollutants (Urbanski et al., 2011; Wiedinmyer et al., 2011) and the forest types addressed in our study constitute a sizeable share of this wildfire activity. Our work suggests the contribution of wildfires in these ecosystems to NAAQS $PM_{2.5}$ and Regional Haze may be underestimated by air regulatory agencies due to the use of EF that are not representative of wildfire season emissions.

Emission factors are not the only source of uncertainty in emission inventories. Biomass burning emission models typically estimate emissions as the product of area burned, fuel load, combustion completeness, and EF (Urbanski et al., 2011; Wiedinmyer et al., 2011; van der Werf et. al. 2010; Larkin et al., 2009). The contribution of these components to uncertainty in emission estimates is not equal and varies with spatial and temporal scale (Urbanski et al., 2011). In general, fuel loading is considered to be the greatest uncertainties emission estimates (Urbanski et al., 2011; French et al., 2011). The impact of biomass burning emissions on air quality depends not only on emissions but also on plume rise, transport, and chemistry, all of which introduce additional uncertainty (Goodrick et al., 2013; Achtemeier et al., 2011).

SR.11 (P53, L4 – P 56, L27; 3.4 MCE, EF, and fire characteristics)

The MCE we measured for wildfires are significantly lower than those reported in the literature for prescribed fires in temperate conifer forests. There are also distinct regional differences in the published prescribed fire MCE (Fig. 4a). Factors that affect the combustion process, in particular environmental conditions (e.g. wind speed, topography) and fuel characteristics (e.g. moisture, chemistry, the state of decay of dead wood, geometry and arrangement of fuel particles) (Ottmar, 2001; Sandberg et al., 2002) will also influence MCE. Fine fuels, those with high surface to volume ratios, such as grasses, conifer needles, and fine woody debris (diameter < 7.6 cm) have a tendency to burn by flaming combustion with a high MCE (Chen et al., 2007; Ottmar, 2001; Sandberg et al., 2002; Yokelson et al., 1996). Smoldering combustion, which has a lower MCE, is more prevalent in CWD, duff, and organic soils (Bertschi et al., 2003; Burling et al., 2011; Ottmar, 2001; Sandberg et al., 2002; Yokelson et al., 1997). Reviews of field studies show that fires in ecosystems dominated by fine fuels such as grasslands and savannas burn with a higher MCE than forest fires (Akagi et al., 2011; Andreae and Merlet, 2001; Urbanski et al., 2009). In addition to fuel geometry and arrangement, recent laboratory studies suggest a linkage between fuel moisture and MCE, with MCE tending to increase with decreasing fuel moisture for homogeneous fine fuels (Chen et al., 2010b; McMeeking et al., 2009). An analysis of emission field measurements for multiple biomes found evidence that the spatio-temporal variability in MCE could be partially attributed to fraction of tree cover and monthly precipitation (Van Leeuwen and Van der Werf, 2011), the later which is presumably a surrogate for fuel moisture.

Considering the influence of fuel moisture and the tendency of certain fuel types to favor flaming or smoldering combustion, one might expect higher fuel moisture and/or the involvement of coarse fuels (CWD and duff) to result in fires with lower MCE. However, the combustion completeness of CWD and duff increases with decreasing fuel moisture (Albini and Reinhardt, 1997; Brown et al., 1991; Ottmar et al., 2006; Ottmar, 2001), while that of fine woody debris, grasses, and litter is relatively insensitive to moisture once ignition is achieved (Ottmar et al., 2006; Ottmar, 2001). Because the moisture contents of different types of fuel particles respond to environmental conditions with different time-lags, there can be a large difference in the moisture content of fuel bed components. The moisture content of fine fuels like cured grasses, litter, and small twigs (< 0.64 cm diameter) adjusts to environmental conditions with a time-lag on the order of 1 h (these are often referred to as 1-h fuels; (Bradshaw et al., 1984)). In contrast, CWD and duff respond with a time-lag of around 1000 h (1000-h fuels; (Bradshaw et al., 1984; Brown et al., 1985; Harrington, 1982)). Therefore, at a given forest stand, under conditions typical of a springtime prescribed burn, consumption of coarse fuels may be minimal due to the high fuel moisture content of these components. However, at the same site under wildfire conditions, when the moisture content of coarse fuels is low, these components may comprise the majority of fuel consumed. Thus, despite the lower fuel moisture during the wildfire season, one might expect a fire with lower MCE compared with a springtime prescribed fire in the same forest stand due to the greater involvement of coarse fuels which favor smoldering combustion processes. The prescribed fires studies of B11 and A13 showed evidence of such an effect. The B11 North Carolina prescribed fires burned in the spring under conditions of high fuel moisture and MCE were high, averaging 0.948. While occurring in nominally similar forests, the prescribed fires studied in A13 were burned during the fall prescribed fire season before the region had fully recovered from a prolonged drought. The average MCE of the A13 fires was 0.931.

We believe the relatively low MCE of the fires we sampled and the general trend in MCE across regions is partially attributable to the differential consumption of coarse fuels. The Big Salmon Lake, Hammer Creek, Saddle Complex wildfires and the North Fork Prescribed fire involved significant areas of dead standing and dead down trees (Sect. 2.1). The 6 SE understory conifer fires reported in B11 occurred under conditions of high duff moisture and the fuels burned were predominantly shrubs, litter, grass, and fine woody debris (B11; Reardon, 2012). Pre and post fuel loading measurements taken at two of the B11 NC sites (the two Camp Lejeune burns) indicate CWD and duff were < 15% of the fuel mass consumed (Reardon, 2012). While the SE burns of B11 involved predominantly fine fuels, their Sierra Nevada burns (Turtle burn and Shaver burn) involved moderate to heavy loadings of dead wood. At the Turtle, burn site litter and 1-h dead wood comprised only ~ 1/3 of the surface dead fuel loading (Gonzalez, 2009). The site of the Shaver burn had dead woody fuel loadings of up to 28 kg m^{-2} due to mountain pine beetle activity and the lack of previous fire (B11). Perhaps coincidently, the MCE measured for the Shaver burn (0.885) was roughly equal to the average MCE (.883) of the wildfires studied in this work which also burned in forests with areas of standing dead trees and heavy loadings down dead wood.

In contrast to the B11 SE burns, the Shaver and Turtle burns occurred when coarse fuels had fairly low moisture content (1000-h = 18%, (WFAS, 2012)) and these fuels likely comprised a significant portion of the fuel mass consumed. This comparison of the B11 prescribed fires and the wildfires suggests the presence of coarse fuels (CWD and duff) and conditions favorable for their burning results in fires with a greater fraction of smoldering combustion, a lower MCE, and higher emissions of species associated with smoldering.

Given the lack of fuel consumption data for the wildfires and all but 2 of the B11 prescribed fires our argument is highly speculative. However, fuel consumption data is available for 13 prescribed fires from U09 and for the 3 prescribed fires of H96. To test our argument that the consumption of coarse fuels favors lower MCE we compared the ratio of coarse fuel consumption to total fuel consumption (CFF) versus MCE for the 18 prescribed fires with fuel consumption data (see Appendix A for details). The results, plotted in Fig. 5, show a strong negative correlation between CFF and MCE ($r = -0.83$, p-value = 1.7e-5), as CWD and duff comprise a larger fraction of the total fuel consumed the fire average MCE decreases.

The analysis presented in Fig. 5 indicates the consumption of coarse fuels favors smoldering combustion, a finding consistent with previous ground based studies of prescribed burns in logging slash and guidelines for smoke management (Ottmar, 2001; Sandberg et al., 2002). However, we emphasize that our conclusion is based on a small sample size and involves significant uncertainty regarding the representativeness of emission sampling. The fuel consumption measurements quantify the fuel consumed over the entire life of the burn. Since smoldering combustion may continue for many hours after the active flame front has passed (Ottmar, 2001; Sandberg et al., 2002) it is unlikely the emissions sampling is properly weighted for smoldering emissions. Due to this temporal mismatch between emissions and fuel sampling it is possible the contribution of smoldering emissions may be underrepresented in the MCE and EF measurements. Further, given the variability in fuel loading and fire characteristics (spread rate, ignition method) the degree of sampling bias with respect to smoldering emissions may vary among burns. For these reasons we stress that the data and the analysis are not intended to be applied for predicting MCE. Nonetheless, the analysis identifies relative CWD and duff consumption as a driver of fire average MCE and a likely factor behind the differences in MCE measured for temperate forest fires.

van Leeuwen and van der Werf (2011) developed a global, biome-independent MCE model. This continuous MCE model, a multivariate regression of field measured MCE versus coarse-scale (monthly, 0.5° × 0.5°) environmental parameters, was driven primarily by monthly precipitation and fraction tree cover (FTC), and explained about 34% of the variability in the field measured MCE. They were unable to account for fuel composition due to lack of consistent data, but suggested it may be a crucial factor driving the MCE variability not captured by their analysis. The authors also explored biome stratified emissions data and highlighted a strong negative correlation between MCE and FTC for fires in Australian Savannas and deforestation fires in Brazil. If the loading of CWD is proportional to FTC, then the coarse fuel combustion – MCE dependence we have identified may help explain their observed FTC-MCE relationship. Since their model is biome-independent and aggregates across grasslands, savannas, and forests it is possible the correlation they have observed between FTC – MCE reflects fuel composition, with FTC serving as a proxy for CWD loading.

SR.12 (P57, L1 – P 59, L7; 4. Conclusions)

Over 8 days in August of 2011 we sampled emissions from 3 wildfires and a prescribed fire that occurred in mixed conifer forests of the northern Rocky Mountains. We measured MCE and EF for CO_2, CO, and CH_4 using a CRDS gas analyzer deployed on an airborne platform. We believe this study may be the first to apply in-flight CRDS technology to characterize the emissions from open biomass burning in the natural environment. The combustion efficiency, quantified by MCE, of the fires sampled in this work was substantially lower than the average MCE measured in previous field studies of prescribed fires in similar forest types (conifer dominated temperate forests) and that reported in recent review articles of biomass burning emissions. In comparison to previous field studies of prescribed fires and review articles, the fires studied in this work measured lower MCE and $EFCO_2$ and higher EFCO and $EFCH_4$. An examination of results from our study and 54 temperate forest fires from previously published studies show a clear trend in MCE across region/fire type: southeast prescribed fires (MCE=0.935) > southwest prescribed fires (MCE=0.922) > northwest prescribed fires (MCE=0.900) > this study (MCE=0.883). The fires sampled in this work burned in areas reported to have moderate to heavy loadings of standing dead trees and down dead wood due to insect activity and previous burns. Of previously published field measurements of prescribed fires the few with MCE similar to that measured in our study also burned in forest with heavy loadings of large diameter dead wood and/or duff.

Fuel consumption data was not available for any of the fires sampled in this study; however, it was available for 18 prescribed fires reported in the literature. For these 18 fires we found a significant negative correlation between MCE and the ratio of coarse fuel (CWD and duff) consumption to total fuel consumption. This observation suggests the comparatively low MCE measured for the fires in our study results from the availability of coarse fuels and conditions that facilitate combustion of these fuels (e.g. low moisture content). More generally, our measurements and the comparison with previous studies indicate that fuel composition is an important driver of EF variability. Considering the accumulation of fuels in western US forests due to factors such as fire exclusion and insect induced mortality (see for example Klutsch et al., 2009), the MCE and EF measured in this study and those we have estimated based on EF – MCE relationships, may be representative of wildfires in mixed conifer forests across the western US.

The temperate forest EF reported in the literature are based on fires which burned with higher combustion efficiency (i.e. a lower relative fraction of smoldering combustion) than the fires sampled in our study. Because the EF of many smoldering combustion species have a strong negative correlation with MCE, the EF found in the literature may significantly underestimate the true EF for smoldering species for fires with combustion characteristics similar to the wildfires measured in this work. EF – MCE relationships from the literature and our study average MCE were used to derive rough estimates of wildfire season EF for 5 species. If the MCE of the fires sampled in this work are representative of wildfire season fires in similar western US forest types, this analysis indicates that the use of literature EF may result in a significant underestimate of wildfire $PM_{2.5}$ and NMOC emissions. The most recent national emission inventory reports western forest wildfire emissions of $PM_{2.5}$ based on an effective $EFPM_{2.5}$ that is only 60% of that estimated in this study. Given the magnitude of biomass consumed by western US wildfires, the use of an $EFPM_{2.5}$ that is low for wildfires could have important implications for the forecasting and management of regional air quality. The contribution of wildfires to NAAQS $PM_{2.5}$ and Regional Haze may be underestimated by air regulatory agencies.

Our study sampled 4 fires over 8 days for a total of 9 fire-day observations. The fires burned in similar environments: montane, mixed conifer forest of Lodgepole Pine, Douglas-Fir, and Engelmann Spruce/Subalpine Fir with significant insect induced tree mortality and moderate to heavy loadings of standing dead and down dead wood. Our measured MCE and EF and the EF estimated from EF – MCE relationships may not be applicable to all wildfires in western US forests. High loadings of down dead wood may have been the main factor driving the MCE and EF of these fires. Our measurements did not include fires in Ponderosa Pine dominated forests which are characterized by lower loadings of dead wood, especially CWD (Graham et al., 1994). Other forests types or forests with a different disturbance history may not have similar loadings of coarse fuels and the therefore the MCE and EF (measured and estimated) reported here may not be applicable. Future emission studies focusing on other regions (e.g. southern Rocky Mountains), forest types (e.g. ponderosa pine dominated), and forests with different disturbance histories are needed to better quantify $PM_{2.5}$ and NMOC emissions from wildfires in the western US.