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

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Abstract. In the US, wildfires and prescribed burning present significant challenges to air regulatory agencies attempting to achieve and maintain compliance with air quality regulations. Fire emission factors (EF) are essential input for the emission models used to develop wildland fire emission inventories. 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 northern Rocky Mountains, US.

In August 2011, we deployed airborne chemistry instruments and sampled emissions over eight days from three 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 $EFCH_4$.

The fires sampled in this study 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, an analysis of MCE and fuel consumption data from 18 prescribed fires reported in the literature indicates that the availability of coarse fuels and conditions favorable for the combustion of these fuels favors low MCE fires. This analysis suggests that fuel composition was an important factor contributing to the low MCE of the fires measured in this study.

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 four non-methane organic compounds (NMOC) using MCE and EF data reported in the literature. This analysis suggests the $EFPM_{2.5}$ for wildfires that occur in forests of the northern Rocky Mountains may be significantly larger than those reported for temperate forests in the literature and that used in a recent national emission inventory. If the MCE of the fires sampled in this work are representative of the combustion characteristics of wildfire-season fires in similar forest types across the western US then the use of EF based on prescribed fires may result in an 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.

1 Introduction

Biomass burning (BB, defined here as the open burning of biomass which includes wildfires and prescribed fires in forests, savannas, grasslands, and shrublands, and agricultural fire, such as the burning of crop residue) is a major source of global trace gases and particles (van der Werf et al., 2010; Wiedinmyer et al., 2011). In terms of total global source, BB emissions are estimated to account for 40 % of carbon monoxide (CO), 35 % of carbonaceous particles, and 20 % of nitrogen oxides (NO_x) (Langmann et al., 2009). The contribution of BB in the conterminous US to global BB

emissions are minor (van der Werf et al., 2010; Wiedinmyer et al., 2011). However, emission estimates published prior to this study suggest that wildland fires (defined here as wildfires and prescribed fires in forests, savannas, grasslands, and shrublands) account for a sizeable fraction of the annual total $PM_{2.5}$ 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 daytime scale that is pertinent to air quality forecasting and management.

In the US, emissions from wildland fires have a significant impact on air quality and present major challenges to air regulatory agencies responsible for achieving and maintaining compliance with federal National Ambient Air Quality Standards (NAAQS; USEPA, 2012c) for ozone (O_3) and $PM_{2.5}$ (particulate matter with an aerodynamic diameter < 2.5 µm) and the Regional Haze Rule (USEPA, 1999). The purpose of the Regional Haze Rule is to reduce pollution which causes visibility impairment in national parks and wilderness areas. Particulate matter is the primary contributor to visibility impairment in areas covered by the Regional Haze Rule. Because O_3 is a secondary pollutant resulting from complex chemistry, quantifying the individual contribution of wildfires or prescribed fires to O_3 formation is difficult. A thorough review of regulatory issues associated with wildfire O_3 production is provided by Jaffe and Wigder (2012). Acute impacts of wildfires and prescribed fires on $PM_{2.5}$ levels in urban areas have been reported in numerous studies (DeBell et al., 2004; Liu et al., 2009; Sapkota et al., 2005) and documented by air regulatory agencies (USEPA, 2012b). Wildland fires have also been identified as important contributors to visibility reduction in areas protected by the Regional Haze Rule (Brewer and Moore, 2009).

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 (June–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, 2012). The seasonal wildfire activity begins 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 in October and, in some years, wildfire conditions persist well into November. The extent to which wildfires impact different western US ecosystems varies intra-seasonally and inter-annually. In recent years, forests (of all types) have comprised 44 % of wildfire burned area (based on a geospatial overlay of 2001–2010 Monitoring Trends in Burn Severity fire boundaries (MTBS, 2012) and a Remote Sensing Application Center/Forest Inventory Analysis forest map, Ruefenacht et al. (2008)). However, due to the greater fuel loadings of forests compared to grasslands and shrublands, fuel consumption and emissions are dominated by forest fires (Urbanski et al., 2011) (fuel loading is the dead and live biomass available for combustion per unit area, Sandberg et al., 2001).

Wildland fire emission inventories (EI) provide essential 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 made characterizing the composition of fresh BB smoke and in quantifying BB EF (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.

Conditions during the western US wildfire season are often favorable (low fuels moisture and high intensity fire fronts) 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 frequently characterized by low intensity fire and is usually conducted outside the wildfire season when the moisture of CWD and duff tend to be moderate to high (Finney et al., 2005; Hardy, 2002), conditions which minimize consumption of these fuels relative to fine fuels. Therefore, it is possible that forest fires occurring during the wildfire season might 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 forest fires.

We present smoke emissions data from airborne field measurements of fires that occurred in mixed conifer forests of the 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 wildfire-season forest fire EF for $PM_{2.5}$, ethane (C_2H_6), propylene (C_3H_6), formaldehyde (HCHO), and methanol (CH_3OH) using previously published EF–MCE relationships. This new wildfire-season 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.

2 Methodology

This study measured smoke emissions from three large wildfires (Saddle Complex, Big Salmon Lake Fire, and Hammer Creek Fire) and a large prescribed fire (North Fork prescribed fire) in the northern Rocky Mountains in August 2011. The Big Salmon Lake Fire and the Saddle Complex were sampled on multiple days and we have treated these sampling days as separate fires, identifying each as a "fire-day", resulting in a total of nine fire-day emission datasets. We believe this treatment is justified given the complex terrain, heterogeneous fuels, and the inter-day variability in meteorology conditions and observed fire behavior (see Table 1 and Table S1). One day is an appropriate temporal scale for atmospheric chemical modeling applications, since most BB EI provide estimates on a daily basis from which models then create an hourly profile based on assumptions about diurnal fire behavior cycles. The fire activity and meteorological conditions associated with the fires on each day of sampling is provided in Supplement: Table S1. All fires in this study were sampled using a US Forest Service Cessna 206 aircraft equipped with atmospheric chemistry instrumentation as described in Sect. 2.2.

2.1 Site descriptions

The locations and vegetation involved for the fires studied are provided in Table 1. All four fires occurred in montane mixed conifer forests of Lodgepole pine, Douglas fir, Engelmann spruce, and Subalpine fir in the northern Rocky Mountains. The methods used to determine the vegetation involved and elevation are described in Supplement S1. 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 Rocky Mountains. The afternoon 10 h fuel moisture (moisture of dead wood (diameter < 2.5 cm)) was ~ 5 % and the 1000 h fuel moisture (the moisture typical of CWD) was between 9 % and 14 %. The estimated daily fire growth, derived from incident fire perimeters and ICS-209 reports (Supplement S1), 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 (Schwind, 2008). 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 Fig. 1, which shows fire perimeters, areas of active burning, and the region of smoke sampling from the Saddle Complex on 24 August. The perimeters, as observed via airborne IR sensor on the evenings (23 and 24 August), indicate that on 24 August 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 24 August. 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 19 July 2011 in the Bob Marshall Wilderness in northwestern Montana. It burned an estimated 2555 ha (elevation of 1360–2250 m a.m.s.l. (above mean sea level)) before be-

Table 1. Name, location, and vegetation involved of the fires sampled in this study.

Fire	Location	Latitude ° N	Longitude ° W	Vegetation involved as percent of burned area	
North Fork Prescribed	Clearwater National Forest, Idaho, US	46.768	−114.978	Engelmann spruce/Subalpine fir	39
				Lodgepole pine	27
				Douglas fir	24
				Grand fir	10
Big Salmon Lake	Bob Marshall Wilderness, Montana, US	47.609	−113.414	Lodgepole pine	38
				Douglas fir	33
				Engelmann spruce/Subalpine fir	24
				Whitebark pine	5
Hammer Creek	Bob Marshall Wilderness, Montana, US	47.530	−113.268	Lodgepole pine	51
				Douglas fir	40
				Engelmann spruce/Subalpine fir	9
Saddle Complex	Salmon-Challis National Forest, Idaho, US and Bitterroot National Forest, Montana, US	45.520	−114.419	Lodgepole pine	47
				Douglas fir	38
				Engelmann spruce/Subalpine fir	15

ing declared under control on 7 October 2011 (Carbonari, 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, 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 22 August, 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 16 August 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 an elevation of 1320–2400 m a.m.s.l. and its perimeter encompassed ∼2200 ha when declared controlled on 7 October 2011 (Carbonari, 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 yr (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, 2012).

The Big Salmon Lake Fire was sampled on 17, 22, and 28 August. On 17 August, 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 (Owen, 2011). While the Big Salmon Lake Fire was reported at 14:00 LT on 16 August, it was not detected by MODIS until the following afternoon. The size of the Big Salmon Lake Fire was estimated at 295 ha on the late afternoon of 17 August and the fire perimeter measured on that night's IR aerial survey placed the fire size at 900 ha. On 22 August, the fire behavior of the Big Salmon Lake Fire involved low to moderate rates of spread with creeping, smoldering, and some single tree torching (Carr, 2011). The fire grew ∼405 ha on 28 August and the fire activity included spreading ground fire and group torching of tree crowns (Gallarado, 2011).

2.1.3 Saddle Complex

The Saddle Complex was a fire complex in the northern Rocky Mountains along the Idaho–Montana border that formed when the Saddle Creek Fire and Stud Fire merged on 18 August 2011. The Saddle Creek Fire was ignited by lightning on 10 August in the Bitterroot National Forest in Montana. The Stud Fire was also caused by lightning and began on 14 August in the Salmon-Challis National Forest in Idaho. The fire complex was managed as two separate fire incidents (Salmon-Challis Branch and the Bitter-

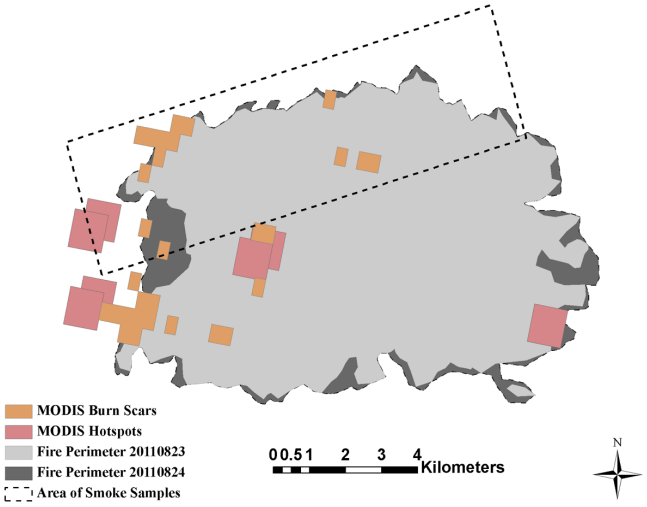

Fig. 1. Region of smoke sampling, fire perimeters, and area of active burning for Saddle Complex on 24 August 2011. The MODIS active fire detections and MODIS burn scars are from 24 August 2011.

root Branch). The fire burned in complex terrain at an elevation of 1040–2650 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, 2011)). The forest burned by the Saddle Complex had not been impacted by wildfire in over 25 yr (MTBS, 2012).

On the four days the Saddle Complex was sampled, its perimeter grew 200–800 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, b) were consistent with our airborne observation that significant fire activity and smoke emissions occurred within the perimeter, especially on 24 and 25 August (Fig. 1).

2.1.4 North Fork prescribed fire

The North Fork prescribed fire was actually two burns ignited on 12 August 2011 in the North Fork District of the Clearwater National Forest in northern Idaho. The prescribed burn targeted diseased and insect-infested areas (Chaney, 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 13 August and MODIS active fire detections indicate the fire persisted until 25 August (RSAC, 2012a).

2.2 Cavity ring-down spectroscopy (CRDS) trace gas analyzer

Continuous measurements of CO_2, CO, and CH_4 were obtained using a flight ready CRDS trace gas analyzer (Picarro Inc., CA, USA, model G2401-m)[1]. In the CRDS technique, the gas sample flows through an optical cavity with partially reflecting mirrors. Light of a specific wavelength from a continuous wave laser is injected into the optical cavity through one of the partially reflecting mirrors. While the laser is on, light builds up in the optical cavity. The laser is abruptly turned off and the decay of light intensity is monitored with a photodetector after the light exits the cavity through a second partially reflecting mirror. The measured light decay is used to determine the optical absorbance of the gas sample and provide a mixing ratio measurement of a particular gas species. A specific gas is measured by scanning a continuous wave laser over an individual spectral line of the targeted gas. The G2401-m analyzer used in this study scans lasers over the individual spectral lines of CO_2, CO, CH_4, and H_2O at wavelengths between 1560 nm and 1650 nm. The precise wavelengths used for monitoring are considered proprietary information and would not be released by the instrument's manufacturer. The analyzer tightly controls the gas sample pressure and temperature at ± 0.0002 atm and ± 0.005 °C and to provide stable, well-resolved spectral features and ensure high precision measurements. The data acquisition rate was 2 s.

Frequent in-flight three-point calibrations were used to maintain accuracy of the CRDS measurements and quantify the measurement precision. The in-flight standards were gas mixtures of CO_2, CO, and CH_4 in Ultrapure air and included or were cross calibrated against two NIST-traceable gas mixtures (concentration in ppm $\pm$ reported analytical uncertainty: $CO_2 = 351 \pm 4$ and 510 ± 5; $CO = 0.092 \pm 0.0092$ and 3.03 ± 0.06; $CH_4 = 1.493 \pm 0.015$ and 3.03 ± 0.03) (Scott-Marrin, Inc., Riverside, CA, USA). In the laboratory, a five-point calibration using an additional high span CO standard and Ultrapure air were used to ensure linearity of the CO calibration between the instrument limit of detection ($\sim$ 030 ppm CO, defined as the 14 s standard deviation while measuring a calibration standard) and 10 ppm.

In recent years, the CRDS technique has been successfully used for high accuracy/high precision measurements of CO_2, CH_4, and CO from airborne platforms (Beck et al., 2012; Chen et al., 2010a). To our knowledge, our study is the first to employ this technique for the in situ measurement of biomass burning emission factors.

[1] Tradenames are presented for informational purposes only and do not constitute endorsement by the US Department of Agriculture.

2.3 Airborne sampling

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 a 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 a typical flight speed of $60\,\mathrm{m\,s^{-1}}$ and 2 s data acquisition rate) 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 kilometers 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 several kilometers downwind.

The parallel profile may be illustrated by considering smoke sampling from the Saddle Complex on 24 August. On this day, 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 their 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 that were 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.

2.4 Emission factor calculations

Multiple smoke samples were collected on each day of fire sampling. For each smoke sample the excess volume mixing ratio (EMR) of compound X (ΔX) was calculated for each 2 s data point by subtracting the average background ($X_{\mathrm{background}}$) for that sample run ($\Delta X = X_{\mathrm{smoke}} - X_{\mathrm{background}}$). Sample emission factors for each compound X, EFX (in unit g of X per kg of dry fuel burned), were calculated from the 2 s ΔX using the carbon mass balance method (Yokelson et al., 1999) following two approaches. Approach 1 (Eq. 1) used the integrated ΔX for each plume sample while, the second approach used emission ratios determined from linear regression fits, with the intercept forced to 0, of ΔX vs. ΔCO or ΔCO_2 using the 2 s data points. The emission ratio of $CH_4 : CO_2$ ($\Delta CH_4/\Delta CO_2$) was calculated as the product of $\Delta CH_4/\Delta CO \times \Delta CO/\Delta CO_2$. The average EF calculated using the two methods agreed within 10 %. In Eq. (1), ΔC_i are the excess mass mixing ratios of carbon in each species X, MM_X is the molar mass of X ($\mathrm{g\,mole^{-1}}$), 12 is the molar mass of carbon ($\mathrm{g\,mole^{-1}}$), and F_c is the mass fraction of carbon in the dry biomass, assumed to be 0.50. We assumed $F_C = 0.50$ based on studies which found that Fc ranged between 0.45 and 0.55 for the vegetation types involved in this study (Burling et al., 2010). The majority of carbon mass (> 95 %) in biomass smoke is contained in CO_2, CO, and CH_4, therefore our neglect of other carbon-containing species in the carbon mass balance method over estimates the EF by ∼ 5 % (Yokelson et al., 2007b).

$$\mathrm{EFX} = F_c \times 1000\ \left(\mathrm{g\,kg^{-1}}\right) \times \frac{\mathrm{MM}_X}{12} \times \frac{\Delta X}{\Delta C_{\mathrm{CO_2}} + \Delta \mathrm{C_{CO}} + \Delta \mathrm{C_{CH_4}}} \quad (1)$$

The chemical composition of emissions from biomass fires are related to the combustion characteristics of the fire, in particular the relative amounts of flaming and smoldering combustion. Some species are emitted almost exclusively by flaming or smoldering, while the emissions of others are significant from both processes. Flaming combustion produces the gases CO_2, NO, NO_2, HCl, SO_2, HONO and N_2O (Burling et al., 2010; Lobert, 1991) and black carbon particles (Chen et al., 2007; McMeeking et al., 2009). The species CO, CH_4, NH_3, many NMOC, and primary organic aerosol (OA) are associated with smoldering combustion (Burling et al., 2010; McMeeking et al., 2009). The species C_2H_2, C_2H_4, CH_3COOH, and HCHO have been linked with both flaming and smoldering combustion (Burling et al., 2010; Lobert, 1991; Yokelson et al., 1996).

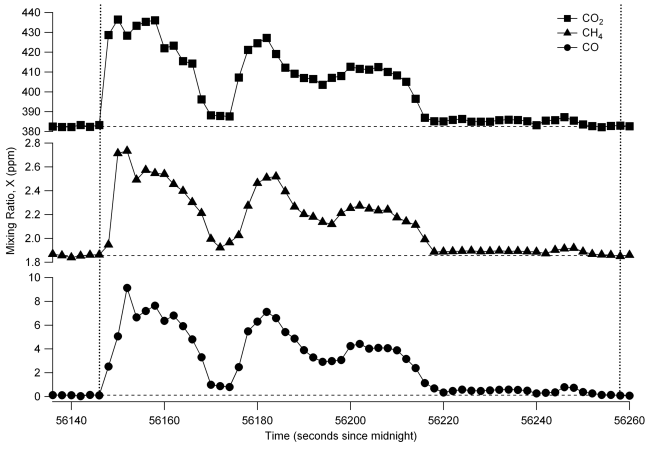

Fig. 2. CRDS measurements of CO_2, CH_4, and CO for a smoke sample run on the Saddle Complex on 24 August 2011. The horizontal dashed line in each panel shows the background mixing ratios measured upwind of the fire on approach for the smoke sample. The vertical dotted lines mark the start and end of the plume.

Modified combustion efficiency (MCE; Eq. 2) is used to characterize the relative amount of flaming and smoldering combustion (Akagi et al., 2011; Ward and Radke, 1993). Laboratory studies have shown that MCE is ~ 0.99 for pure flaming combustion (e.g., fine fuels completely engulfed in flame, Chen et al., 2007; Yokelson et al., 1996), while the MCE for smoldering combustion varies over ~ 0.65–0.85, with 0.80 being a typical value (Akagi et al., 2011). Since many species are predominantly emitted during either flaming or smoldering combustion, the EF of many compounds correlates with MCE. Laboratory studies of the combustion of fine fuels (Burling et al., 2010; Christian et al., 2003; McMeeking et al., 2009; Yokelson et al., 1997) and recent field measurements of emissions from prescribed fires (Burling et al., 2011; Akagi et al., 2013) have found a strong correlation between EF and MCE for many species. Given the utility of MCE for characterizing combustion characteristics and its potential for estimation of EF for a range of compounds, we have calculated MCE for all smoke samples. It should be noted that two laboratory studies of pure smoldering combustion of duff, organic soils, and CWD found poor correlation between MCE and EF (Bertschi et al., 2003; Yokelson et al., 2007a). However, laboratory studies may not be a good proxy for these fuels since the combustion of CWD and duff in the natural environment is dependent on fuel bed characteristics, such as the loading and arrangement of CWD and the presence of fine dead wood and litter (Albini et al., 1995; Ottmar et al., 1989). It is possible though that wildland fires involving a large component of CWD and duff may not show a strong MCE–EF relationship. Burling et al. (2011) and Akagi et al. (2013) used ground based sampling to measure EF for isolated pockets of residual smoldering combustion from prescribed burns in the southeast and found poor correlation between MCE and EF for most species that were measured.

$$\mathrm{MCE} = \frac{\Delta CO_2}{\Delta \mathrm{CO_2} + \Delta \mathrm{CO}} = \frac{1}{1 + \frac{\Delta \mathrm{CO}}{\Delta \mathrm{CO_2}}} \quad (2)$$

3 Results and discussion

3.1 Emission measurements

CRDS measurements for a typical smoke sample run, in this case the Saddle Complex on 24 August (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 all 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 ppm for CO, 0.281 ppm for CO_2, and 0.005 ppm for CH_4. The uncertainties for our EMR, ΔX, were estimated as $\sqrt{\sigma_{\mathrm{bkgd}}^2 + \sigma_{\mathrm{prec}}^2}$, 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 smoke samples are given in Table 2. The fire-day average EF (Eq. 1) agreed within 10 % with the EF that was 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 at significant distances downwind of the source. In particular, on 17 August, samples were taken 40 km downwind of the Big Salmon Lake Fire. The afternoon atmospheric sounding at Great Falls, Montana (Table S1), on this day indicated the transport winds were ~ 13 m s^{-1}, implying a smoke age of ~ 50 min 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 three 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 3a–c show the average, range, and $\pm 1\sigma$ 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

Table 2. Smoke sample EMR, MCE, and EF and fire–day average (±1 standard deviation) MCE, EF, and EMR. EMR are sample average or fire-day average of the individual 2 s data points. EMR are in units of ppmv. EF are in units of g kg^{-1}.

Sample	Time	Location	n^1	ΔCO_2	ΔCO	ΔCH_4	MCE	$EFCO_2$	EFCO	$EFCH_4$
North Fork prescribed fire – 13 August 2011										
NF1301	17:03	at source	13	8.19	1.12	0.11	0.879	1594	139.1	7.48
NF1302	17:04	at source	14	18.15	2.75	0.28	0.868	1571	151.7	8.66
NF1303	17:07	at source	27	8.06	1.17	0.11	0.873	1582	146.5	7.74
NF1304	17:18	4–8 km	85	5.09	0.90	0.08	0.850	1537	172.2	9.18
NF1305	17:47	at source	36	8.46	1.27	0.11	0.869	1575	150.8	7.71
NF1306	18:06	at source	25	16.07	2.50	0.20	0.865	1569	155.5	7.26
NF1307	18:14	at source	49	6.26	0.95	0.08	0.868	1574	152.4	7.32
NF1308	18:18	at source	11	14.00	2.27	0.19	0.861	1559	160.7	7.81
		Average					0.867 ± 0.009	1570 ± 17	153.6 ± 9.8	7.89 ± 0.68
Big Salmon Lake Fire – 17 August 2011										
BSL1701	15:55	17–20 km	61	9.96	1.07	0.09	0.903	1641	112.6	5.51
BSL1702	16:04	30 km	62	7.92	0.88	0.08	0.900	1637	115.3	5.65
BSL1703	16:51	40 km	79	5.52	0.78	0.07	0.876	1588	143.2	7.34
BSL1704	17:03	23–29 km	78	14.29	1.58	0.14	0.901	1637	114.9	5.87
BSL1705	17:18	at source	47	13.60	1.54	0.15	0.898	1631	117.4	6.55
BSL1706	17:24	at source	38	8.03	0.99	0.09	0.890	1615	126.9	6.74
BSL1707	17:30	at source	55	13.84	1.71	0.16	0.890	1615	126.8	6.80
BSL1708	17:42	16–35 km	132	7.35	0.89	0.08	0.892	1619	124.9	6.62
BSL1709	17:49	40 km	81	6.42	0.80	0.07	0.889	1613	128.4	6.77
		Average					0.893 ± 0.008	1622 ± 17	123.4 ± 9.6	6.43 ± 0.61
Big Salmon Lake Fire – 22 August 2011										
BSL2201	15:25	source to 10 km	80	6.25	0.82	0.07	0.884	1605	133.5	6.65
BSL2202	15:52	at source	32	5.56	0.72	0.06	0.886	1608	131.8	6.80
BSL2203	16:01	at source	30	8.96	1.18	0.12	0.884	1601	134.0	7.83
BSL2204	16:03	at source	16	9.74	1.45	0.15	0.871	1576	149.0	8.53
BSL2205	16:05	at source	31	3.02	0.56	0.05	0.845	1527	178.7	9.46
		Average					0.874 ± 0.017	1583 ± 34	145.4 ± 19.9	7.85 ± 1.18
Hammer Creek Fire – 22 August 2011										
HC2201	16:10	at source	53	13.30	1.44	0.12	0.903	1642	112.8	5.27
HC2202	16:14	source to 10 km	38	11.68	1.27	0.10	0.902	1641	113.6	5.15
HC2203	16:21	at source	20	13.91	2.15	0.21	0.866	1567	154.2	8.57
HC2204	16:36	at source	27	11.00	1.29	0.12	0.895	1624	121.7	6.53
HC2205	16:39	at source	15	21.37	2.35	0.23	0.901	1636	114.5	6.31
HC2206	16:41	at source	14	10.47	1.18	0.11	0.899	1632	117.2	6.28
HC2207	16:50	at source	40	16.05	1.99	0.21	0.890	1612	127.4	7.57
HC2208	16:53	4–8 km	79	8.76	1.04	0.10	0.894	1621	122.9	7.05
HC2209	17:08	at source	41	9.53	1.01	0.10	0.904	1642	110.9	6.32
HC2210	17:09	at source	14	47.39	4.35	0.35	0.916	1668	97.4	4.50
		Average					0.897 ± 0.013	1628 ± 26	119.3 ± 14.7	6.36 ± 1.20
Saddle Complex Fire – 24 August 2011										
SC2401	15:32	at source	52	9.22	1.47	0.13	0.863	1562	158.4	8.17
SC2402	15:37	at source	55	20.00	2.86	0.28	0.875	1585	144.0	8.09
SC2403	15:42	at source	41	5.02	0.78	0.07	0.866	1568	154.6	8.13
SC2404	15:46	at source	65	3.07	0.45	0.04	0.873	1582	147.1	7.46
SC2405	15:51	at source	49	4.95	0.57	0.05	0.897	1629	119.2	6.20
SC2406	16:05	8 km	125	4.74	0.66	0.06	0.878	1592	140.7	7.36
SC2407	17:02	5 km	126	1.57	0.22	0.02	0.876	1590	142.5	7.20
SC2408	17:19	at source	62	4.09	0.56	0.05	0.879	1596	139.5	6.41
SC2409	17:23	at source	62	3.88	0.62	0.06	0.863	1562	158.3	8.14
SC2410	17:29	at source	69	6.71	1.06	0.10	0.864	1563	157.0	8.52
		Average					0.873 ± 0.011	1583 ± 21	146.1 ± 12.0	7.57 ± 0.79

Table 2. Continued.

Sample	Time	Location	n[1]	ΔCO_2	ΔCO	ΔCH_4	MCE	$EFCO_2$	EFCO	$EFCH_4$
Saddle Complex – 25 August 2011										
SC2501	14:55	at source	83	1.58	0.21	0.02	0.885	1603	132.9	7.74
SC2502	14:60	at source	23	11.30	1.40	0.15	0.889	1612	127.5	7.76
SC2503	15:06	at source	68	1.78	0.28	0.03	0.865	1566	155.1	8.65
SC2504	15:09	at source	57	5.97	0.65	0.06	0.902	1639	113.7	5.68
		Average					0.885 ± 0.015	1605 ± 30	132.3 ± 17.2	7.46 ± 1.26
Saddle Complex Fire – 26 August 2011										
SC2601	15:06	6 km	44	3.14	0.37	0.03	0.895	1625	121.4	6.39
SC2602	15:58	at source	33	4.65	0.66	0.08	0.876	1582	143.1	9.62
SC2603	16:04	at source	85	3.56	0.53	0.05	0.871	1578	149.0	7.70
SC2604	16:15	source to 8 km	115	5.11	0.66	0.06	0.886	1607	131.3	7.15
		Average					0.882 ± 0.011	1598 ± 22	136.2 ± 12.3	7.71 ± 1.38
Saddle Complex Fire – 27 August 2011										
SC2702	15:08	28–34 km	156	4.58	0.41	0.04	0.918	1671	94.9	4.95
SC2703	15:39	at source	95	3.86	0.55	0.05	0.876	1588	142.7	7.69
SC2704	15:50	at source	174	3.03	0.43	0.04	0.876	1587	143.1	7.78
SC2705	16:01	at source	109	3.74	0.56	0.06	0.869	1572	150.8	8.71
		Average					0.889 ± 0.021	1612 ± 42	128.5 ± 24.4	6.96 ± 1.58
Big Salmon Lake Fire – 28 August 2011										
BSL2801	14:57	9–13 km	43	14.65	1.22	0.11	0.923	1681	89.3	4.40
BSL2802	15:14	at source	20	21.65	2.63	0.23	0.892	1620	125.1	6.23
BSL2803	15:18	at source	48	2.59	0.37	0.03	0.875	1586	144.8	7.07
BSL2804	15:22	at source	22	10.63	1.60	0.17	0.869	1571	150.9	9.00
BSL2805	15:26	at source	27	10.50	1.50	0.17	0.875	1581	143.8	9.50
BSL2806	15:27	at source	33	6.43	0.71	0.07	0.901	1637	114.4	6.03
BSL2807	15:32	at source	14	10.91	1.94	0.24	0.849	1528	173.0	12.09
BSL2808	15:54	12–16 km	15	8.35	1.08	0.10	0.885	1605	132.5	7.18
		Average					0.884 ± 0.022	1601 ± 46	134.2 ± 25.3	7.69 ± 2.41
Study Average							0.883 ± 0.010	1596 ± 23	135 ± 11	7.30 ± 0.58

[1] n = number of 2 s data points.

the lowest MCE (0.867) and the highest EFCO (153.6) and $EFCH_4$ (7.89), while the Hammer Creek Fire had the highest MCE (0.897) and the lowest EFCO (119.3) and $EFCH_4$ (6.36). The low MCE and high EFCO and $EFCH_4$ 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 $EFCH_4$ for the Big Salmon Lake Fire on 17 and 22 August 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 22 August. Interestingly, the Hammer Creek Fire had the 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). Since conifer crowns are expected to burn with a high MCE (see below), this may result from the differential fire behavior at the two sites, the Hammer Creek fire had group torching along the perimeter while the Big Salmon Lake Fire only had isolated torching of single trees (Table S1).

We did observe some large variations in the sample EF within fire-days. In particular, the range in sample EFCO, $EFCO_{maximum}$–$EFCO_{minimum}$, was 83.7, 56.8, and 55.9 g kg^{-1} for the Big Salmon Lake Fire on 28 August, the Hammer Creek Fire on 22 August, and the Saddle Complex on 27 August, respectively. These fire-days also had the largest inter-sample range in $EFCH_4$ 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) analyses of these wildfires (RAVG, 2013). The fact that 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 nine fire-day values) for MCE, $EFCO_2$, EFCO, and $EFCH_4$ are 0.883 ± 0.010, $1596 \pm 23\,g\,kg^{-1}$, $135 \pm 11\,g\,kg^{-1}$, $7.30 \pm 0.58\,g\,kg^{-1}$, respectively (uncertainties are one 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 $EFCH_4$ observed in our study, our measurements are sufficiently precise to reveal a MCE–$EFCH_4$ relationship. CH_4 is produced by smoldering combustion processes, and as expected, $EFCH_4$ has a strong inverse correlation with MCE (Fig. 3d; $r = -0.87$, $p = 0.002$).

3.2 Comparison with other studies

We compared our results with previous field studies of emissions from fires in temperate conifer dominated forests in the US and southern British Columbia, Canada: the airborne measurements of Akagi et al., 2013 (hereafter A13), Burling et al., 2011 (hereafter B11), Hobbs et al., 1996 (hereafter H96) and Radke et al., 1991 (here after R91), and the tower based study of Urbanski et al., 2009 (hereafter U09). A13 studied understory prescribed fires in conifer dominated forests of South Carolina. Their study included ground based measurements of residual smoldering combustion, but here we consider only the airborne measurements. B11 studied understory prescribed fires in conifer dominated forests of North Carolina and the Sierra Nevada of California. H96 studied three prescribed burns of clear cut logging slash on the Washington and Oregon coasts and the Corral-Blackwell Complex wildfire, which occurred in northern Idaho. From R91, we consider their results for two wildfires in western Oregon and a prescribed burn of "Hemlock, deciduous, Douglas fir logging debris" in British Columbia. U09 reported on understory prescribed fires in mixed conifer forests of the interior mountain west and in conifer and conifer/hardwood forests of the southeast US.

For comparison, we grouped the prescribed fires of the five previous studies into three regions: southeast (SE), southwest (SW), and northwest (NW). The U09 Arizona fires were assigned to SW and the Montana, Oregon, and British Columbia fires were assigned to NW. The southeastern conifer and conifer/hardwood forest fires in U09 were assigned to SE. We included 6 fires listed in the grasslands and shrublands section of Table A1 in U09 (EB1, EB2, FL5, SC9, FS1, ICI3) in SE since these fires were, in fact conifer/hardwood understory burns. They were listed as grassland/shrub in U09 since the fuel consumed was overwhelmingly grass and shrubs in the understory. In this sense, these fires were very similar to the southeast burns studied in B11. We assigned the six North Carolina fires of B11 to SE and the two Sierra Nevada fires to NW. The H96 and R91 prescribed fires were included in the NW set and the A13 fires were assigned to the SE.

First, we compare our results with the prescribed fire data. Fire average MCE, EFCO, and $EFCH_4$ from this study, A13, B11, U09, H96, and R91 are shown in Fig. 4a–c. In the following discussion, the wildfire measurements of H96 and R91 are not included in the prescribed fire averages. On average, the fires sampled in our study burned with a lower MCE compared to the prescribed fires. The data show a clear trend in average MCE across categories: SE (0.935) > SW (0.924) > NW (0.900) > WF (0.883) ("WF" includes only the fires measured in this study). There is no overlap of the WF MCE with those of the SE fires. The MCE of B11's Shaver fire (0.885) and the average MCE of H96 (0.877) are both close to the WF average MCE. These four prescribed fires involved heavy loads of down dead wood due to logging in the case of H96 and pine beetle activity in the Shaver fire (see Sect. 3.3). There is also a pronounced trend across categories for EFCO (WF (135) > NW (111) > SW (88) > SE (75)) and $EFCH_4$ (WF (7.30) > NW (6.29) > SW (3.32) > SE (2.15)). This work does not report $EFPM_{2.5}$; however, we note that the $EFPM_{2.5}$ of B11 and U09 exhibit a similar trend (NW (18.0) > SW (14.5) > SE (12.6)).

There is limited temperate forest wildfire data with which we can compare our measurements. Figure 4a shows the R91 wildfires had MCE (0.921, 0.907) above the range measured in our study, while the Corral-Blackwell Complex MCE (0.810) was significantly lower. The Corral-Blackwell Complex burned in forest and terrain very similar to that of the fires studied in this work. The fire, which occurred 120 km west of the Saddle Complex, burned in mixed stands of Lodgepole pine, Engelmann spruce, and Subalpine fir at an elevation of 600–2735 m (Farris et al., 2008). H96 provide little information on the Corral-Blackwell Complex, but do note that it was sampled "during smoldering combustion". The MCE of smoldering combustion has been found to range from $\sim$ 0.65–0.85, but typically being near 0.80 (Akagi et al.,

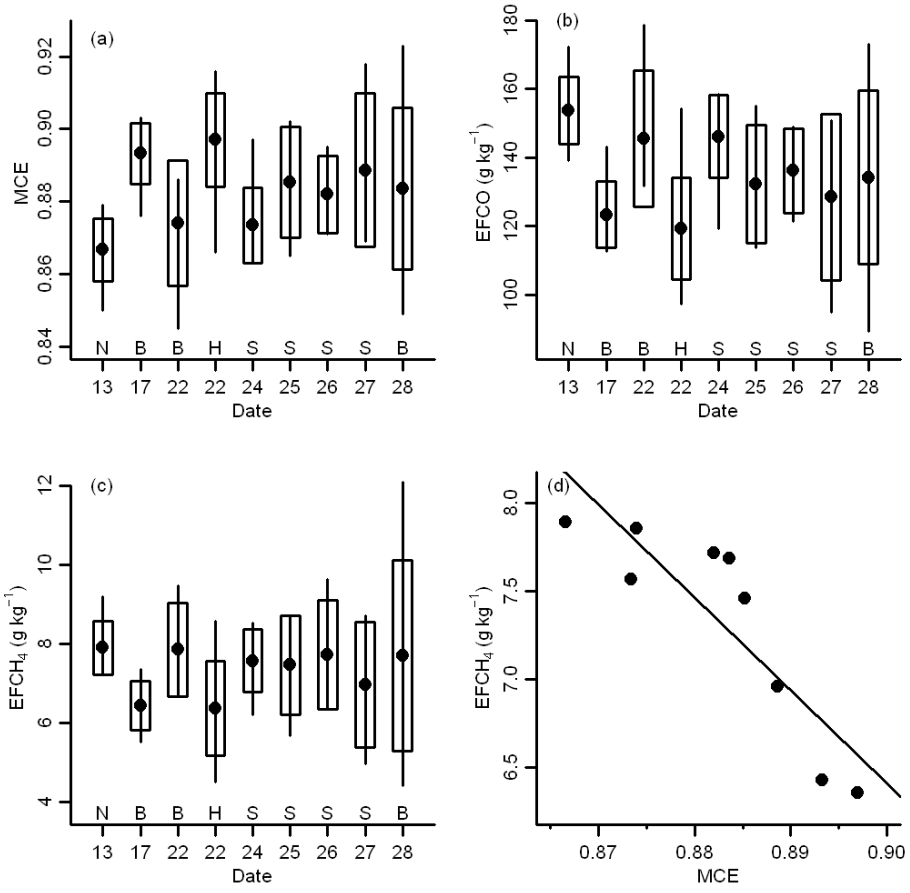

Fig. 3. (a) MCE, **(b)** EFCO, and **(c)** $EFCH_4$ by fire-day. The date is August 2011, N = North Fork prescribed fire, B = Big Salmon Lake Fire, H = Hammer Creek Fire, S = Saddle Complex. Solid circles are the fire-day average, boxes are $\pm 1\sigma$, vertical bars mark the minimum and maximum. **(d)** Fire-day average $EFCH_4$ vs. MCE. Solid line is linear least square fit: $y = -52.8\times + 53.9$, $r^2 = 0.76$.

2011). Using a ground based FTIR, B11 measured post-flame front emissions from nearly pure smoldering combustion of dead tree stumps for a prescribed burn at Camp Lejeune, NC (the ground-based measurements of pure smoldering are not included in the B11 results discussed thus far and reproduced in Fig. 4). Since CWD can smolder for an extended period of time and can comprise a large share of fuel consumed in western forest fires (Brown et al., 1991; Reinhardt et al., 1991), the smoldering stump measurements of B11 serve as a reasonable analog for assessing the Corral-Blackwell Complex data reported by H96. The four dead stumps measured by B11 yielded averages of MCE = 0.795 and $EFCH_4 = 17.4\,g\,kg^{-1}$, similar to the Corral-Blackwell Complex values reported by H96 (MCE = 0.81, $EFCH_4 = 18.0\,g\,kg^{-1}$). If H96 sampled almost exclusively smoldering combustion this may explain the very low value of their MCE compared to that measured in this work for wildfires in similar terrain and forest. During our sampling, we observed active flaming combustion on all days for all fires, usually including torching of tree crowns, and the emissions we measured originated from both active flaming and post-flame front smoldering combustion (see Fig. 1 and related discussion).

The Silver and Myrtle/Fall Creek wildfires sampled by R91 occurred in southwestern Oregon, which has a Mediterranean climate. They burned in different vegetation and elevation than the wildfires sampled in our work. The Silver Fire burned in Douglas fir/Tanoak/Pacific Madrone forest between elevations of 75 m and 1500 m with an average of 800 m. R91 described the vegetation burned in the Myrtle/Fall Creek wildfires as "standing pine, brush, and Douglas fir" and they burned at elevation of 300 m and 900 m (average = 615 m). The different vegetation involved in the R91 fires may explain the relatively high MCE they measured compared to the wildfires sampled in this work. The vegetation involved in the Silver Fire was determined from a combination of geospatial overlays of the fire perimeters (MTBS, 2012) and an existing vegetation map (LANDFIRE, 2012) and literature (Thompson and Spies, 2010). Fire elevation for both R91 wildfires was obtained from geospatial overlays of the fire perimeters and a digital elevation map (LANDFIRE, 2012).

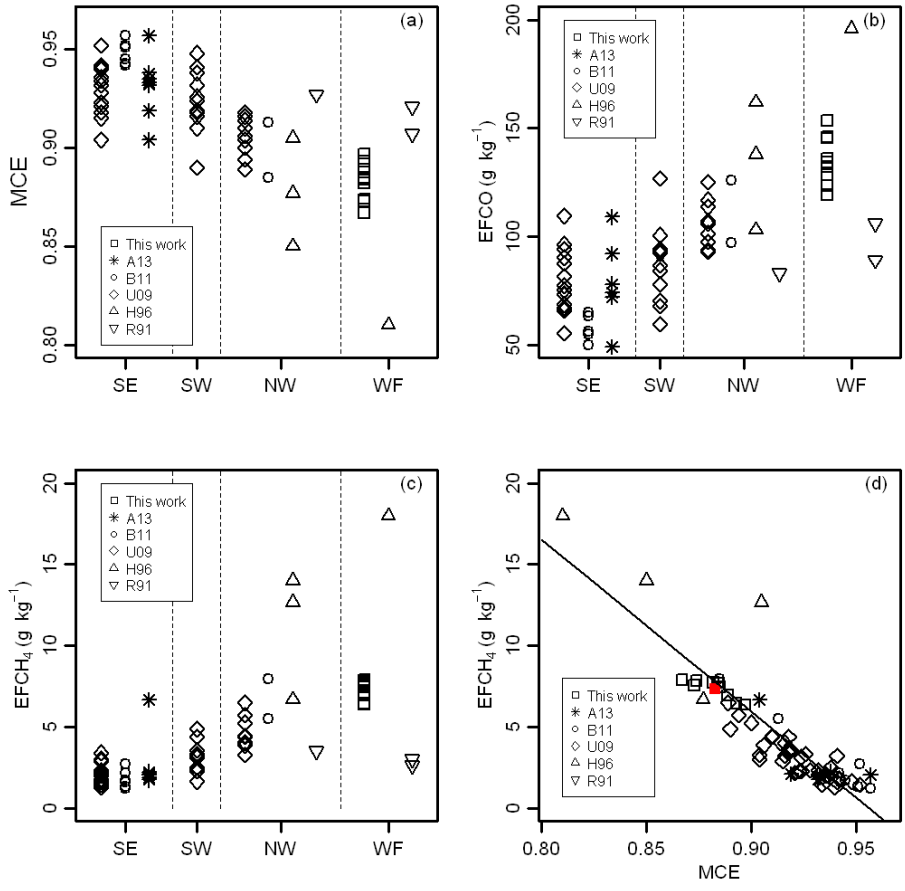

Fig. 4. (a)MCE, **(b)** EFCO, and **(c)** $EFCH_4$ for this work and from several other studies plotted by region for prescribed fires (SE = southeast, SW = southwest, NW = northwest). WF = wildfires. **(d)** EFCH4 vs. MCE for this work and several other studies. The filled red square is the average for our study (Table 2). The solid line is the linear regression best fit to data from previous studies only (A13, B11, U09, H96, and R91). Best fit line is $y = -106.10\times + 101.4$, $r^2 = 0.77$.

Hornbrook et al. (2011) report on seven 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 CO_2, CO or CH_4, but they do report MCE. The average MCE of the seven biomass burning plumes was 0.911. Analysis of fire data (MTBS, 2012; RSAC, 2012; Urbanski et al., 2011) and vegetation maps (Ruefenacht et al., 2008) indicates the area burned in California during ARCTAS (15 Jun–15 July, 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, 2012a, b; Urbanski et al., 2011), we attempted to identify the source fires or source regions of the seven California wildfire plumes measured in Hornbrook et al. (2011). We could only confidently associate two of the seven California biomass burning plumes, plumes #12 and #18 in Table 1 of Hornbrook et al. (2001) with coherent sources. Plume 18 (MCE = 0.88, sampled on 26 June) 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 was combined with vegetation maps (Ruefenacht 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 18 June). 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).

Wigder et al. (2013) reported normalized enhancement ratios (NER) of PM_1 (particulate matter < 1 µm diameter) to CO ($\Delta PM_1/\Delta CO$) for 32 wildfire smoke plumes of different ages measured at Mount Bachelor Observatory in central Oregon, US. They did not measure CO_2 and do not report EF for any species. However, comparison of their NER with NER calculated from published EFPM and EFCO may be

Table 3. MCE and EF for this work, A11 (temperate forest (TF) and boreal forest (BF)), and NEI and the ratio of EF from this work to EF from A11 and NEI.

Species	This work	A11-TF	A11-BF	NEI	This work / A11-TF	This work / A11-BF	This work / NEI
MCE	0.883 ± 0.010	0.921	0.882	0.847		ratio	
Carbon Dioxide (CO_2)[1]	1596 ± 23	1637	1489	1466	0.98	1.07	1.09
Carbon Monoxide (CO)[1]	135.4 ± 11	89	127	169	1.52	1.06	0.80
Methane (CH_4)[1]	7.30 ± 0.58	3.92	5.96	8.06	1.87	1.23	0.91
	Estimated EF						
$PM_{2.5}$[2]	23.2 ± 10.4	12.7	15.3	13.8	1.83	1.52	1.68
Ethane (C_2H_6)[3]	0.93 ± 0.26	1.12	1.79		0.83	0.52	
Propylene (C_3H_6)[3]	0.94 ± 0.20	0.95	1.13		0.99	0.83	
Formaldehyde (HCHO)[3]	2.63 ± 0.74	2.27	1.86	1.29	1.16	1.41	2.04
Methanol (CH_3OH)[3]	2.91 ± 0.77	1.93	2.82		1.51	1.03	

[1] EF value for this study is the fire-day average and standard deviation from Table 2. [2] $EFPM_{2.5}$ value estimated based on previous studies (see text). [3] EF value estimated from previous studies using EF vs. MCE regressions (see text and Supplement S1). Uncertainty is 95 % confidence interval of linear regression fits at MCE = 0.833.

used to infer possible $EFPM_1$ for the wildfires they sampled. They found their average $\Delta PM_1/\Delta CO$ for 10 relatively fresh smoke plumes (< 1 day old) of 0.17 $\mu g\,m^{-3}\,ppbv^{-1}$was similar to the $0.16 \pm 0.11\,\mu g\,m^{-3}\,ppbv^{-1}$ value they calculated from the $EFPM_{2.5}$ and EFCO reported by Akagi et al. (2011) for temperate forest fires. The agreement in PM to CO NER suggests the $EFPM_1$ for the 10 wildfire smoke plumes may have been similar to the $EFPM_{2.5}$ of $12.7 \pm 7.5\,g\,kg^{-1}$ reported in Akagi et al. (2013). This $EFPM_1$ inferred from the Wigder et al. (2013) measurements is about half the $EFPM_{2.5}$ we have estimated for our wildfires. The significant difference in estimated EFPM suggests the wildfires sampled in Wigder et al. (2013) and our study had very different combustion characteristics. These apparent differences in wildfire combustion characteristics may be related to the vegetation involved. Widger et al. (2013) do not provide details on vegetation burned by the fires they sampled. However, the source locations of the 10 wildfire plumes considered here were in eastern Oregon and it is quite possible the fuels and intensity of these fires may have been quite different from the northern Rockies fires sampled in this study. The difference in estimated EFPM between our study and Widger et al. (2013) suggests the wildfires sampled in this work may not be applicable to wildfires in forest outside the northern Rocky Mountains.

3.3 Estimated wildfire emission factors and comparison with reference data

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 (0.883). 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 northwestern 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 $EFPM_{2.5}$. 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 airborne only measurements (B11, H96, R91), NW measurements (B11, H96, R91, U09), airborne measurements all regions (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. The results were consistent with our initial rough estimate predicting an $EFPM_{2.5}$ of 23 to 25 $g\,kg^{-1}$ at our study average MCE of 0.883 (Table S2). 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 A13. While the source studies provide EF for a wider range of compounds, we report estimates for only those moderately correlated with MCE ($R^2 > 0.60$). Figure 4d shows $EFCH_4$ plotted vs. MCE for data from R91, H96, B11, U09, A13, and this study. The figure also shows a linear regression line based on the data from the previous studies (regression statistics are provided in Table S2). 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. $7.65\,g\,kg^{-1}$). Plots of EF vs. MCE for CH_3OH, HCHO, C_3H_6, and C_2H_6 and statistics for the linear fits are provided in Supplement S1.

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 US EPA 2008 National Emission Inventory version 2 (USEPA, 2012a) (hereafter, NEI). The NEI documentation reports that wildland fire EF were estimated "using the Fire Emissions Prediction Simulator which relies on EFs from the literature apportioned by flaming and smoldering combustion", but does not provide the complete set of EF used (USEPA, 2012a, documentation, Chapter 5, page 125). For comparison with EF from this study, we derived effective forest fire EF from the NEI Supporting Data and Summaries, Section 5 ("Fires") supplemental data (USEPA, 2012a). The effective EF were derived as follows: from the wildland fire location file (WF_locations_ALL.xlsx), we extracted all wildfires in forest ecosystems (fire type = "WF" and "canopy" > 0) in the western US and then from the extracted fires calculated effective EF for species X as the sum of emissions of X for all fires divided by the sum of estimated fuel consumed for all fires. The NEI effective MCE was based on $\Delta CO/\Delta CO_2$ calculated from the effective EF and molecular masses of CO and CO_2 ($\Delta CO/\Delta CO_2 = MM_{CO_2}/MM_{CO} \times EFCO/EFCO_2$).

The EFCO and $EFCH_4$ 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 CH_3OH, HCHO, C_3H_6, and C_2H_6 gives mixed results. Notable differences are our estimated EFC_2H_6 (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 S1).

Our field measurements show that some wildfire-season forest fires in the western US 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 account for the majority of wildland fire 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. These forest types comprised 19 % of the total area and 43 % of the forested area that was burned by wildfires in the western US from 2001–2010 (based on a geospatial overlay of 2001–2010 fire boundaries (MTBS, 2012) and a forest type map (Ruefenacht 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. If the northern Rockies fires sampled in our work are representative of wildfires in similar forest types across the western US, then 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 uncertainty in 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 (A13; Goodrick et al., 2013; Achtemeier et al., 2011; Alvarado et al., 2009).

3.4 MCE, EF, and fire characteristics

The MCE we measured for wildfires are significantly lower than the average MCE reported in the literature for ~ 60 prescribed fires in temperate conifer forests. Only a handful of these 60 prescribed fires had an MCE that overlapped the wildfire MCE measured in our study. 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; B11; 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 (A11; Andreae and Merlet, 2001; U09). 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 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 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, such as 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 consumption of coarse fuels, which favor smoldering combustion processes. The prescribed fire 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 the MCE were high, averaging 0.948. While occurring in nominally similar forests, the prescribed fires studied in A13 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 six 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 $\sim 1/3$ of the surface dead fuel loading (Gonzalez, 2009). The site of the Shaver burn had dead woody fuel loadings of up to $28\,\mathrm{kg\,m^{-2}}$ 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 (0.883) of the wildfires studied in this work, which also burned in forests with areas of standing dead trees and with heavy loadings of downed dead wood.

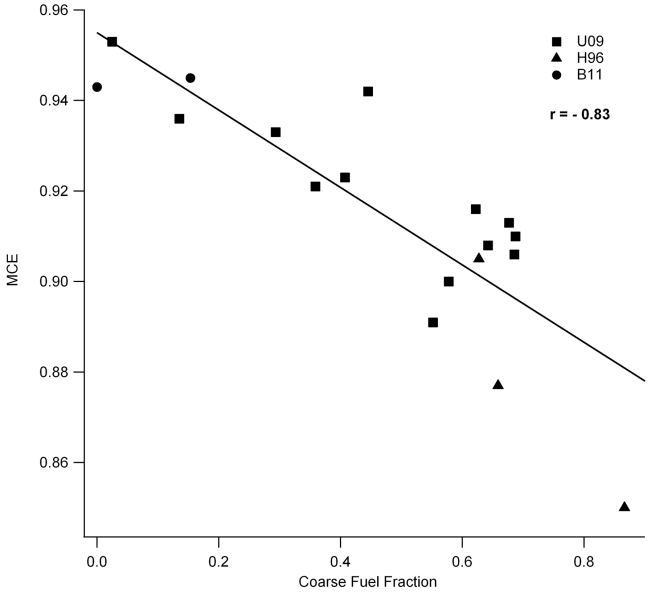

Fig. 5. Plot of MCE vs. coarse fuel fraction (the ratio of coarse fuel consumption to total fuel consumption) for 18 prescribed fires from previous studies (B11; U09; H96; Ottmar and Ward, 1996; Ottmar and Vihnanek, 1995; Reardon, 2012; Harrington, 2012). Coarse fuel is CWD and duff. See Sect. 3.4 and Appendix A for details.

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 two 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 three 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 ($r = -0.83$, $p = 1.7 \times 10^{-5}$), 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 that 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, $0.5^\circ \times 0.5^\circ$) 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 that the correlation they have observed between FTC–MCE reflects fuel composition, with FTC serving as a proxy for CWD loading.

4 Conclusions

Over eight days in August 2011, we sampled emissions from three 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 58 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 = 0.924) > 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 forests 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 mostly 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 can 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 fires measured in this work. The 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 northern Rockies 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 average 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 $PM_{2.5}$ is 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. We note that a recent study (Wigder et al., 2013) reported PM_1 and CO enhancements in smoke plumes from wildfires in eastern Oregon that were consistent with a $EFPM_{2.5}$ of only half that estimated in our study. This difference in estimated EFPM between our study and Widger et al. (2013) shows that the fires sampled in this work may not be applicable to all wildfires in the western US. Emission measurements for wildfires in other regions of the western US are needed.

Our study sampled four fires over eight days for a total of nine 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 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.

Appendix A

Qualitative reports indicate the fires sampled in our study burned in areas with significant loadings of CWD. In contrast, previous studies of prescribed burns in the southeastern US (B11, U09) measured relatively high MCE and anecdotal observations suggested these fires consumed mostly fine fuels with the consumption of CWD and duff being minimal. This pattern is not unexpected since fine fuels have a tendency to burn by flaming combustion, while CWD and duff favor smoldering combustion processes (Sandberg et al., 2002). Using previous studies of 18 prescribed burns for which detailed fuel consumption data was available, we tested for a relationship between fire average MCE and the composition of fuel consumed. Specifically, we tested for a significant correlation between the relative amount of flaming and smoldering combustion, quantified by MCE, and the relative amount of coarse fuel and fine fuel consumption. The later fire characteristic was quantified with the coarse fuel fraction (CFF), defined as the sum of CWD and duff

Table A1. Fire name, location, date, fuel consumption, fuel moisture, and MCE for 18 prescribed fires from previous studies.

Fire name	Location	Date mm/yyyy	MCE	Fuel consumption ($g\,m^{-2}$)			Fuel moisture[2] (%)		Reference
				Fine fuels[1]	CWD[1]	Duff	CWD	Duff	
Quinault	western Washington	09/1994	0.850	2940	8050	11000	25	120	a
Creamery	western Oregon	09/1994	0.905	2270	1320	2500	28	nm	a
Raymond	western Washington	10/1994	0.877	1980	1320	2500	28	nm	a
Camp Lejeune IA	North Carolina	02/2010	0.943	1541	0	0	nm	nm	b, c
Camp Lejeune ME	North Carolina	02/2010	0.945	1127	204	0	nm	nm	b, c
MT1	western Montana	05/1993	0.891	556	462	222	78	79	d, e
MT2	western Montana	05/1993	0.908	952	632	1076	45	17	d, e
MT3	western Montana	05/1994	0.913	649	439	918	50	30	d, e
MT4	western Montana	05/1994	0.910	536	632	545	50	30	d, e
OR1	eastern Oregon	05/1994	0.906	513	268	848	42	59	d, f
OR2	eastern Oregon	05/1994	0.900	1339	424	1406	75	75	d, f
OR3	eastern Oregon	05/1994	0.916	1004	223	1429	67	44	d, f
FL5	Florida	01/1996	0.933	923	0	383	nm	160	d, g
SC1	Georgia	02/1996	0.921	423	0	169	125	93	d, g
SC12A	Georgia	02/1996	0.942	374	0	300	125	93	d, g
SC12B	Georgia	02/1996	0.923	414	0	284	nm	98.5	d, g
MN4	Minnesota	04/1995	0.953	444	0	11	74	40	d, h
MN5	Minnesota	05/1995	0.936	884	0	138	113	43	d, h

[1] Fine fuels include litter, grasses, shrubs, and dead wood with diameter < 7.62 cm; CWD (coarse woody debris) is dead wood with diameter > 7.62 cm. [2] nm = not measured. References: (a) Hobbs et al. (1996); (b) Burling et al. (2011); (c) Reardon (2012); (d) Urbanski et al. (2009); (e) Harrington (2012); (f) Ottmar and Ward (1996); (g) Ottmar and Vihnanek (1997); (h) Ottmar and Vihnanek (1995).

fuel loading consumed divided by the sum of total fuel loading consumed. CFF is given by Eq. (A1) where C_i is consumption ($kg\,m^{-2}$) of fuel component i, fine fuels includes grasses, shrubs, foliage, litter, and fine woody debris (small diameter (< 7.62 cm) dead wood):

$$\mathrm{CFF} = \frac{C_{\mathrm{CWD}} + C_{\mathrm{duff}}}{C_{\mathrm{CWD}} + C_{\mathrm{duff}} + C_{\mathrm{fine\,fuels}}}. \quad \text{(A1)}$$

Name, location, date of the prescribed burns, MCE, fuel moisture, fuel consumption by class, and references for the 18 fires used in this analysis are provided in Table A1.

Supplementary material related to this article is available online at: http://www.atmos-chem-phys.net/13/7241/2013/acp-13-7241-2013-supplement.zip.

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