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Emission factors from wheat and Kentucky bluegrass stubble burning: Comparison of field and simulated burn experiments

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Abstract

Emission factors (EFs) of PM_{2.5}, CO, elemental carbon (EC), particulate organic carbon (OC), polycyclic aromatic hydrocarbons (PAHs) and methoxyphenols (MPs) from post-harvest burning of wheat and Kentucky bluegrass (KBG) stubble were evaluated in a series of field burns. Integrated smoke samples were collected at ground level, upwind and downwind of the fires, and EFs were determined with the carbon balance method (validated during previous chamber experiments). These EFs were compared against EFs evaluated from previously conducted chamber burns, to determine how well the latter represent field scenarios. In general, when combustion efficiency (CE) differences were taken into account, a reasonable degree of agreement was observed between emission factors measured in the field and in the chamber, except for EC and solid + vapor phase PAHs, both from wheat burns. EC and PAHs from wheat burns were seen in higher amounts in the chamber, although the PAH data are in agreement at CEs>90%. EC overestimates might be due to a misassigned EC-OC split in the heavily loaded quartz filters from chamber burns. Poor EC and OC EF-CE correlations in KBG chamber data make the comparison with field data difficult. The particulate organic matter/OC ratios (2.1 ± 1.3) for wheat and 1.9 for KBG) were higher than those observed during chamber experiments (1.5 for both wheat and KBG). Overestimates of EC in the chamber and possibly the condensation of oxygenated species in the field may be responsible for this difference. Though CO and CH4 EFs evaluated from ground-based samples differed from those collected on board a light aircraft, EF-CE relationships were similar. This underscores the importance of determining both the CEs and EFs simultaneously. © 2006 Elsevier Ltd. All rights reserved.

Keywords: Wheat; Kentucky bluegrass; Field burn; Combustion efficiency; Emission factor

1. Introduction

The post-harvest burning of agricultural residue produces several atmospheric pollutants. Recently, we reported emission factors (EFs) of $PM_{2.5}$, CO, hydrocarbons (Dhammapala et al., 2006a), elemen-

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tal and particulate organic carbon (EC and OC), selected polycyclic aromatic hydrocarbons (PAHs) and methoxyphenols (MPs) (Dhammapala et al., 2006b), emitted from wheat (*Triticum aestivum* L.) and Kentucky bluegrass (KBG, *Poa pratensis* L.) stubble burning. Those experiments were conducted in a test burn chamber.

In contrast with combustion in burn chambers, combustion in the field is affected by several additional variables including local meteorology,

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terrain, cropping and burning practices, stubble moisture content and fire characteristics such as temperature, intensity, spread and burn rate. Since several of these factors could affect the combustion efficiency (CE) of the fire and therefore EFs (Air Sciences Inc. (A.S.I.), 2003; Jenkins et al., 1996; Carroll et al., 1977), it is necessary to understand how well chamber burns represent field scenarios.

Data from our above-mentioned chamber experiments cannot be readily compared against field data in literature, since few field studies have been conducted to determine EFs from wheat and KBG burning (Darley et al., 1974; Boubel et al., 1969; A.S.I., 2003; Air Sciences Inc. (A.S.I.), 2004). These few studies focus mostly on particulate matter, CO and hydrocarbon emissions and the A.S.I. studies provide very limited data on solid phase PAH emissions. We are unaware of field studies reporting EFs of EC, OC, MPs (solid or vapor phase) or vapor phase PAHs, from wheat and KBG burns.

The purpose of this work is to compare the EF–CE relationships for $PM_{2.5}$, CO, EC, OC, MPs and PAHs from wheat field burns (conducted in eastern Washington) and KBG field burns (conducted in northern Idaho), with corresponding measurements previously reported from the burn chamber. Ground-based smoke samples collected during field studies may be biased toward smoldering emissions on account of lower plume buoyancy (Hurst et al., 1994; Ward et al., 1992). Ideally smoke samples need to be representative of the entire plume. Therefore, a secondary objective of this work is to compare EFs calculated from ground-based field samples, against EFs evaluated by collecting samples on board a light aircraft.

2. Experimental

Six wheat and two KBG field burns were conducted between August 2004 and August 2005. Field sizes ranged from 13 to 80 ha. Samples of preburn stubble and post-burn ash were collected from at least 10 randomly chosen locations in the field, finely ground, composited and analyzed for carbon content using a CNS2000 total carbon analyzer (LECO Corporation, St. Joseph, MI). Ambient temperature and pressure during field burns were recorded.

Integrated smoke samples were collected at ground level, upwind and downwind of the fires. Downwind sampling commenced at the time of ignition and continued until smoldering had almost ended. Sampling times ranged from 40 min to 2 h. Downwind sites were typically a few meters away from the field edge of the burn area. Upwind samplers, located at least 200 m away from the burn, started before and ended after the fire. The concentration of pollutants generated by the fire is the difference in concentrations between upwind (background) and downwind samples.

Integrated gaseous samples (for CO₂, CO and CH₄ analysis) were collected in 30 mL plastic syringes, mounted on ground-based syringe samplers and filled at a constant rate over 20 min. PM_{2.5}, PAH, MP, EC and OC sampling methods are described in detail in Dhammapala et al. (2006a, b). Briefly, integrated $PM_{2.5}$ and solid phase semivolatile organic compounds (SVOCs; includes PAHs and MPs) were sampled on Teflon filters, while vapor phase SVOCs were trapped on polyurethane foam (PUF) plugs placed downstream of Teflon filters. PM_{2.5} samples were also collected onto quartz filters (abbreviated QF), to determine the EC and OC fractions therein. All OC data reported here have been corrected for the adsorption artifact on QF (Turpin et al., 1994), using a separate quartz filter placed behind a Teflon filter (abbreviated TQ).

A single engine Cessna C-172 aircraft was used to obtain smoke samples for calculation of EFs from airborne measurements. Grab samples were manually drawn into syringes while flying through the plume, between 100 and 2100 m (median 1100 m) above ground level. $PM_{2.5}$ measurements from the aircraft were not possible since an isokinetic sampling inlet was unavailable. To minimize secondary atmospheric processes and stay well above background concentrations, aircraft-based sampling was conducted relatively close to the fire. Table 1 lists pertinent experimental details.

All syringe samples were analyzed for CO₂, CO and CH₄ using a Carle SX 270 Gas Chromatograph (1 m Molecular Sieve 5A 45/60 mesh column at 90 °C, N2 carrier gas flowing at 35 mL min^{-1}) equipped with a methanizer (400 °C) and a flame ionization detector (FID). Analytical methods for PM_{2.5}, PAH, MP, EC and OC are described in detail in Dhammapala et al. (2006a, b). Briefly, PM_{2.5} mass was determined gravimetrically. Teflon filter and PUF extracts were analyzed by gas chromatography–mass spectrometry (GC–MS) for PAHs and MPs. Teflon filter extracts were also analyzed for levoglucosan, and the results are reported in Jimenez et al. (2006). Quartz filters were

Table 1	
Inventory of field burns conducted and samples collected	

Date	Location	Upwind and downwind samples			
		CO ₂ , CO, CH ₄	PM _{2.5}	QF and TQ	PUF
20 Aug and 8 Sep '04	Nez Perce County, ID (KBG)	4	1	1	
29 Sep '04 and 18 Mar '05	Dayton, WA (wheat)	4 ^a	2 ^a	1^{a}	1^{a}
22 Mar '05	Colfax, WA (wheat)	4 ^a	2 ^a	1^{a}	1 ^a
23 Apr '05	Palouse, WA (2 wheat burns)	4^{a}	2^{a}	1^{a}	1^{a}
2 Aug '05	Connell, WA (wheat)	4 ^b	1 ^b	1 ^b	—

^aTwo sets of field duplicates were obtained.

^bThree sets of upwind samples were obtained. In addition 21 Grab samples were collected from an aircraft by drawing smoke into syringes.

analyzed for EC and OC on a thermal optical transmittance (TOT) analyzer.

As reported in Dhammapala et al. (2006a, b), the chamber burns were conducted using 750 g wheat or KBG stubble, arranged to mimic field orientations in eastern Washington (WA) and northern Idaho (ID), respectively.

2.1. Calculation of EFs

The method used for calculating EFs in the field (A.S.I., 2003; Hurst et al., 1994; Andreae and Merlet, 2001; Radke et al., 1988; Nelson, 1982) is presented in Eq. (1). The method is based on the conservation of C in the biomass, and does not require pre- and post-burn weighing of biomass.

 $EF_x(g kg^{-1})$

$$= \frac{\Delta C_x \times (1000 \text{ g kg}^{-1}) \times C_{\text{fraction}}}{(\Delta \text{CO}_{2-C} + \Delta \text{CO} - C + \Delta \text{CH}_4 - C + \Delta \text{PM}_{2.5} - C)},$$
(1)

where EF_x is the emission factor in g of pollutant $X \text{ kg}^{-1}$ dry stubble burned, ΔC_x is the concentration of pollutant X in excess of the background in mg m^{-3} , ΔCO_{2-C} (mg m⁻³) the [mass fraction of C (w/w) in CO₂ (i.e. 12/44)] × [average background-corrected CO₂ concentration (mg m⁻³)], etc. for all species and C_{fraction} is the mass fraction of C in dry stubble.

This method assumes the following:

 Physico-chemical processes between emission and sampling are considered to have negligible effect on the pollutant concentrations. All pollutants are present in a unit volume of air sampled, in the same proportion in which they were produced by the fire (Nelson, 1982; Radke et al., 1988). Since the time between pollutant generation and field sampling ranges from seconds to minutes, it is unlikely that secondary processes cause a significant impact.

- 2. Pre- and post-burn biomass have similar $C_{\text{fractions}}$ (Nelson, 1982; A.S.I., 2003). Though the C_{fraction} in ash was found to be $5.8 \pm 4.3\%$ less than that of stubble, data in A.S.I. (2003), and Dhammapala et al. (2006a) show that unburned stubble accounts for 95% (wheat) and 94% (KBG) of the post-burn biomass. Hence the C_{fraction} in postburn biomass is approximately equal to that of pre-burn stubble.
- 3. All carbon in the biomass consumed by fire is released into the atmosphere during combustion, and can be accounted for by measuring concentrations of CO₂, CO, total hydrocarbons and PM_{2.5} in the plume (Radke et al., 1988; Andreae and Merlet, 2001). This assumption was validated by performing a carbon balance closure during the burn chamber experiments (Dhammapala et al., 2006a). For the field experiments, we were only able to measure CH₄ instead of total hydrocarbons. But we determined that this substitution altered the carbon balance, CEs and EFs by < 3%.
- 4. The background concentrations measured upwind are representative of ambient air impacting the field. During the wheat field burns in Connell, WA, the temporal and spatial variation of the background was assessed by placing three sets of samplers around the field at different upwind locations. The average relative standard deviation (RSD) of background CO₂, CO, CH₄ and

 $PM_{2.5}$, measured over 2 h was 2.8%. Further, the average RSD of CO₂, CO and CH₄ in all airborne and ground-based background samples was 2.7%. Therefore this assumption was deemed valid.

2.2. QA/QC measures

Ten syringes were pre-filled with CO_2 , CO and CH_4 calibration standards, transported to and from the field, stored for 2 months and analyzed. The change in CO_2 , CO and CH_4 concentrations averaged 1.9%. The precision of field duplicates for each parameter is shown in Table 2.

PAHs and MPs are subject to larger uncertainties than the other pollutants due to the increased complexity of analytical procedures for these analytes. PAH and MP samples were spiked with deuterated mixtures of PAHs and MPs prior to analysis. All PAH and MP results reported here have been corrected for losses during analysis, using the fractional recoveries of the spiked deuterated compounds.

When comparing ground and aircraft based EFs, the combustion efficiencies (CE) and emission factors (EFs) were calculated with CO₂, CO and CH₄. Although PM_{2.5} was monitored on the ground, it is omitted from the calculations to facilitate comparison with aircraft measurements. The omission of non-methane hydrocarbons and PM_{2.5} results in the CEs and EFs being overestimated by no more than 5%.

Atmospheric O_3 has been observed to react with PAHs on samples collected on filter substrates (Pöschl et al., 2001; Peltonen and Kuljukka, 1995; Finlayson-Pitts and Pitts, 2000), resulting in under-

Table 2		
Precision	of field	duplicates

	Precision ^a , (%)	No. of duplicate samples
PM _{2.5}	9.9	14
CO_2 (CO) [CH ₄]	1.3 (3.5) [2.2]	12 each
EC (OC)	10.9 (7.2)	2 each
PAH solid (vapor) phase ^b	19.5 (24.8)	5 (2)
MP solid (vapor) phase ^b	26.6 (18.7)	2 (2)

 $^{\rm a} Precision = absolute difference between two field duplicates/ mean.$

^bBased on average of all individual SVOCs analyzed. The RSD of replicate laboratory analyses for PAHs was 14.2% (n = 12) and 9.5% for MPs (n = 4).

estimation of PAH EFs. However it is unlikely that sufficient O_3 would be formed in the plume, during the short time between emission and sampling. Since biomass fires release NO (Hurst et al., 1994), it is likely that ambient O_3 present would be removed by reacting with NO.

Because benzo[a]pyrene (b[a]p) is considered to be more photochemically reactive than benzo[e]pyrene (b[e]p), the b[a]p/b[e]p ratio has been used to assess if smoke is subject to photochemical degradation during transport (Finlayson-Pitts and Pitts, 2000). We found that b[a]p/b[e]p ratios in the chamber and field were 1.4 ± 0.2 and 1.5 ± 0.2 ,respectively. This suggests that PAHs were not degraded appreciably during the short time between emission and field sampling. MP degradation during transport was assumed to be negligible.

For field experiments, based on propagation of errors, $PM_{2.5}$, CO, CH₄, EC and OC EFs are estimated to be within $\pm 20\%$, while SVOC EFs are within $\pm 30\%$.

3. Results and discussion

The trendlines and R^2 values shown in the following figures pertain to the chamber experiments described in our previous manuscripts (Dhammapala et al., 2006a, b), and cover the CE range seen therein. The CEs encountered in the field appear lower than those seen during chamber burns. Chamber EF-CE data were extrapolated by way of linear regression, to obtain EFs at the CE's encountered in the field. Extrapolated EFs expressed as a percentage of EFs measured in the field are presented in Table 3. Statistical significance testing was not conducted on account of the few data points. KBG data should be interpreted with caution since only two field burns could be conducted. The concentrations of each pollutant measured upwind and downwind of burns are presented in Table A1 of the Electronic Supplementary Material A.

3.1. CO and $PM_{2.5}$

The CO EF–CE relationships (Fig. 1a) observed for both wheat and KBG in the field are similar to the CO EF–CE relationship observed for wheat in the chamber. The CO data from KBG chamber experiments were not considered since these EFs did not line up with the corresponding wheat data (see Fig. 1a). No obvious reason for the discrepancy was

	$\left(\frac{\text{Chamber EFs extrapolated to field CEs}}{\text{Field EFs}}\right), (\%)$			
	Wheat	KBG ^a		
СО	104 ± 4	105, 110		
PM _{2.5}	78 ± 33	80, 87		
EC	229 ± 94	145, 83		
OC	91 ± 12^{b}	176 ^b		
Solid + vapor phase PAH	169 ± 112	No vapor phase PAH data		
Solid + vapor phase MP	70 ± 25	No vapor phase MP data		

Table 3 Comparison of emission factors evaluated in chamber and field experiments

^aSince n = 2, EF ratios of both data points rather than their average, are shown.

^bSamples with high backgrounds omitted (see text).



Fig. 1. EFs vs. CE relationships determined in chamber and field burns for (a) CO and (b) PM_{2.5}. The trendlines and R^2 values superimposed on these charts were obtained from our previous chamber studies (Dhammapala et al., 2006a).

found, although data were thoroughly checked for integrity. On average the extrapolated wheat chamber CO EFs were 104% of wheat field EFs, and 108% of KBG field EFs.

The PM_{2.5} EF–CE relationships are shown in Fig. 1b. On average, the extrapolated chamber EFs were 78% of field EFs for wheat, and 83% for KBG.

3.2. EC and OC

EC EFs from wheat field burns are lower than the chamber (Fig. 2a; extrapolated chamber EFs are 229% of field EFs). These differences may be partly caused by the TOT incorrectly assigning the EC–OC split in samples with high EC loadings (Reid et al., 2005), and could lead to the overestimation of chamber EC EFs. Since OC EFs are larger than EC EFs, the former may not be significantly affected by the mis-assigned carbon.

The artifact corrected OC EF-CE relationships (Fig. 2b) for wheat in the chamber and field are

comparable except for two field measurements (circled in Fig. 2b). These were traced to two background samples with high OC concentrations, possibly caused by contamination. When these two data points were omitted, extrapolated wheat chamber OC EFs were 91% of field EFs.

For KBG, when one data point (circled in Fig. 2b) with a high OC background concentration was omitted, the extrapolated chamber EFs were 176% and 114% of the field EFs for OC and EC, respectively. However KBG OC and EC data should be interpreted with caution because of the few data points as well as the poor EF-CE correlations seen during chamber experiments.

Since average ambient temperatures measured during field burns were lower than those in the chamber, field OC measurements reported above may include additional condensed organic material. The OC/EC ratio is negatively correlated with average ambient temperature measured during the field burns (Fig. 3; three high OC background



Fig. 2. (a) EC EFs vs. CE and (b) OC EFs vs. CE relationships determined in chamber and field burns. Circled OC data points had high backgrounds (see text). The trendlines and R^2 values superimposed on these charts were obtained from our previous chamber studies (Dhammapala et al., 2006b).



Fig. 3. OC/EC ratio as a function of average ambient temperature measured during field burns. Circled data points have not been corrected for OC backgrounds (see text).

samples—circled in figure—were not subtracted from downwind measurements). Although a negative correlation between the OC/EC ratio and ambient temperature would be expected for the OC sampling artifact, our data have been corrected for this artifact. Therefore we postulate that adsorption of OC to $PM_{2.5}$ during atmospheric transport is the likely reason for the trend observed in Fig. 3. It is unlikely that secondary OC formation would be responsible for the trend in Fig. 3, due to the short time between pollutant generation and sampling.

 $PM_{2.5}$ is comprised of particulate organic matter (POM), EC and inorganic matter. Some $PM_{2.5}$ samples were analyzed for 38 elements using X-ray fluorescence spectroscopy. Of these K, Cl, S, Si, Fe, Al, Ca and Ti accounted for >99% of the elemental mass. Inorganic mass was calculated by considering the ions associated with these elements (Malm et al.,

1994). The POM (i.e. $PM_{2.5}$ -EC-inorganic mass)/ artifact corrected OC ratios from field experiments (1.9 for KBG and 2.1 ± 1.3 for wheat) were higher than the POM/OC ratios from the chamber experiments (1.5 ± 0.1 for KBG and 1.5 ± 0.2 for wheat). While field data are based on few samples with much variability, possible reasons for differences could include the aforementioned overestimation of chamber EC. Further, POM accompanying the condensing OC (in field samples) might contain heavier (e.g. oxygenated) species. Higher POM/OC ratios might also explain why field $PM_{2.5}$ EFs are slightly higher than chamber burns.

3.3. PAHs

Each gram of $PM_{2.5}$ emitted during wheat field burns consists of $0.2\pm0.2 \text{ mg}$ PAHs (solid phase PAHs include those with a molecular weight 202 g mol^{-1} —see Dhammapala et al., 2006b). This is lower than the $0.7\pm0.1 \text{ mg g}^{-1}$ measured during chamber experiments. PAH content in PM_{2.5} from KBG burns are similar when confidence intervals are taken into account (field 0.2, chamber $0.3\pm0.1 \text{ mg g}^{-1}$). However the effects of plume dilution on phase partitioning (Lipsky and Robinson, 2006) make comparisons of solid phase SVOCs less straightforward.

To remove the effect of phase partitioning, the solid+vapor phase PAH EFs were compared in Fig. 4. Relatively good agreement was observed for solid+vapor phase PAH EFs from wheat chamber burns at CEs >90%, however agreement was poor at lower CEs. Overall the extrapolated solid+vapor phase PAH EFs from chamber burns was 169% of the field burns, for wheat. Individual PAHs in both

phases from field and chamber experiments were compared (by extrapolating EFs; data not shown) but no distinct trend was seen. No data are shown



Fig. 4. Solid+vapor phase PAH EF vs. CE relationship determined in chamber and field burns, for wheat stubble. The superimposed trendline and R^2 value was obtained from our previous chamber study (Dhammapala et al., 2006b).



Fig. 5. Solid + vapor phase MP EF vs. CE relationship determined in chamber and field burns, for wheat stubble. The superimposed trendline and R^2 value was obtained from our previous chamber study (Dhammapala et al., 2006b).

for KBG because vapor phase PAH data were not collected for KBG burns.

3.4. Methoxyphenols

Species-specific solid phase MP/PM_{2.5} ratios are presented in Jimenez et al. (2006) and are not discussed here. To remove the effects of phase partitioning, the solid + vapor phase MP EFs from field and chamber burns were compared (Fig. 5). Though no vapor phase MP data are available for KBG, extrapolated solid + vapor phase MP Efs from wheat chamber burns were 70% of the wheat field burns. The comparison of individual MPs in both phases from field and chamber experiments (by extrapolating EFs; data not shown) showed no distinct trend.

3.5. Comparison of ground and airborne data

Figs. 6a and b show the comparison of EFs for CO and CH₄ as a function of CE, evaluated from ground and aircraft based samples, all collected during the wheat field burn on 2 August 2005. Aircraft based sampling captures the more buoyant sections of the plume (i.e. predominantly from the flaming phase with higher CEs and lower EFs). Ground measurements may be weighted toward less buoyant smoldering emissions (lower CEs, higher EFs) (Ward et al., 1992; Hurst et al., 1994). The ground-based sample with the highest CE (94.9%) was collected over the 20 min following field ignition. Five airborne grab samples taken over the same period had an average CE of 96.1%.

Based on data in Fig. 6, it appears that ground based fixed-point measurements may not represent the CE of the whole plume, beginning from ignition to flameout. Yet since ground and airborne data appear to have similar EF–CE relationships, one



Fig. 6. Comparison of EF-CE relationships from ground and aircraft based samples for (a) CO and (b) CH₄.

data set could be extrapolated to obtain a reasonable estimate of the other. The importance of reporting CEs alongside EFs is underscored.

A more accurate estimate of the plume-wide EFs is likely to involve combustion phase-specific EFs, weighted by the fraction of stubble mass consumed during the respective phase (Ward et al., 1982, 1992, Ortiz de Zarate et al., 2000). This exercise requires a knowledge of emissions and biomass consumption during each combustion phase, and cannot be attempted with the integrated measurements made here.

4. Conclusions

It must be borne in mind that EFs from chamber burns are subject to uncertainties similar to those pertaining to field burns (Dhammapala et al., 2006a, b). As a result, based on the propagation of errors, extrapolated chamber/field EF ratios for $PM_{2.5}$, CO, EC and OC are expected to be within $100 \pm 30\%$ and $100 \pm 45\%$ for SVOCs.

Therefore chamber experiments conducted previously appear to provide reasonable representations of CO and $PM_{2.5}$ EF–CE relationships for wheat and KBG field burns, and particulate OC and solid + vapor phase MPs from wheat field burns. For wheat burns, CE differences do not explain the variability of PAH EFs in both phases, possibly suggesting that other factors play a role. Though field EC EFs appear lower than the chamber, these differences may be partly caused by the TOT's inability to properly resolve the EC-OC split in chamber samples with high EC loadings. More KBG field burns are needed to facilitate better comparisons.

Utilizing more stubble than the 750 g used in chamber burns might help better mimic flame structures, temperatures, intensities and burn rates seen during field burns. This may also improve the comparisons between chamber and field EFs. Average EFs evaluated from chamber experiments may be less biased toward either smoldering or flaming combustion than field measurements, since the chamber is well mixed. Chamber experiments might also be more amenable to calculating weighted emission factors, since the separation of combustion phases might be easier to accomplish than in the field.

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Appendix A. Supplementary Materials

Supplementary data associated with this article can be found at doi:10.1016/j.atmosenv.2006.10.008.

References

- A.S.I. (Air Sciences Inc.), 2003. Final Report: Cereal Grain Crop Open Field Burning missions Study. Project 152-02. Available online at <htp://www.ecy.wa.gov/programs/air/pdfs/ FinalWheat_081303.pdf> (accessed 15 Mar 06).
- A.S.I. (Air Sciences Inc.), 2004. Quantifying post-harvest emissions from bluegrass seed production field burning. Available online at <htp://www.ecy.wa.gov/programs/air/ pdfs/bluegrass_final_report.pdf> (accessed 15 Mar 06).
- Andreae, M.O., Merlet, P., 2001. Emission of trace gases and aerosols from biomass burning. Global Biogeochemical Cycles 15, 955–966.
- Boubel, R.W., Darley, E.F., Schuck, E.A., 1969. Emissions from burning grass stubble and straw. Journal of the Air Pollution Control Association 19, 497–500.
- Carroll, J.J., Miller, G.E., Thompson, J.F., Darley, E.F., 1977. The dependence of open field burning emissions and plume concentrations on meteorology, field conditions and ignition technique. Atmospheric Environment 11, 1037–1050.
- Darley, E.F., Miller, G.E. Jr., Goss, J.R., Biswell, H.H., 1974. Air Pollution from Forest and Agricultural Burning. California Air Resources Board Project No. 2-017-31, University Of California, Davis, CA. Available at <ftp://ftp.arb.ca.gov/ carbis/research/apr/past/2-017-1.pdf> (accessed 15 Mar 06).
- Dhammapala, R.S., Claiborn, C.S., Corkill, J., Gullett, B.K., 2006a. Particulate emissions from wheat and Kentucky bluegrass stubble burning in eastern Washington and northern Idaho. Atmospheric Environment 40, 1007–1015.
- Dhammapala, R.S., Claiborn, C.S., Simpson, C., Gullett, B.K., Paulsen, M., 2006b. Emission factors of PAHs, methoxyphenols, elemental carbon and organic carbon from simulated wheat and Kentucky bluegrass stubble burns. Atmospheric Environment, accepted.
- Finlayson-Pitts, B.J., Pitts Jr., J.N., 2000. Chemistry of the upper and lower atmosphere: theory, experiments and applications, first ed. Academic Press, New York.
- Hurst, D.F., Griffith, D.W., Cook, G.D., 1994. Trace gas emissions from biomass burning in tropical Australian Savannas. Journal of Geophysical Research 99, 16441–16456.
- Jenkins, B.M., Turn, S.Q., Williams, R.B., Goronea, M., Abd-al-Fatah, H., Mehlschau, J., Raubach, N., Chang, D.P.Y.,

Kang, M., Teague, S.V., Raabe, O.G., Campbell, D.E., Cahill, T.A., Pritchett, L., Chow, J., Jones, A.D., 1996. Atmospheric pollutant emission factors from open burning of agricultural and forest biomass by wind tunnel simulations: final Report, vol. I. Prepared for the California Air Resources Board, Sacramento, CA. CARB Project No. A932-126: Available online at http://www.arb.ca.gov/research/ abstracts/a932-126.htm > (accessed 15 Mar 06).

- Jimenez, J., Claiborn, C., Simpson, C., Dhammapala, R., 2006. Developing a source fingerprint for burning of wheat and Kentucky bluegrass stubble in eastern Washington and northern Idaho. Environmental Science and Technology, submitted to.
- Lipsky, E.M., Robinson, A.L., 2006. Effects of dilution on fine particle mass and partitioning of semivolatile organics in diesel exhaust and wood smoke. Environmental Science and Technology 40, 155–162.
- Malm, W., Sisler, J., Huffman, D., Eldred, R., Cahill, T., 1994. Spatial and seasonal trends in particle concentration and optical extinction in the United States. Journal of Geophysical research 99, 1347–1370.
- Nelson Jr., R.M., 1982. An evaluation of the carbon balance technique for estimating emission factors and fuel consumption in forest fires. Research Paper SE-231. United States Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, NC.
- Ortiz de Zarate, I., Ezcurra, A., Lacaux, J.P., Van Dinh, P., 2000. Emission factor estimates of cereal waste burning in Spain. Atmospheric Environment 34, 3183–153193.

- Peltonen, K., Kuljukka, T., 1995. Air sampling and analysis of polycyclic aromatic hydrocarbons. Journal of Chromatography A 710, 93–108.
- Pöschl, U., Letzel, T., Schauer, C., Niessner, R., 2001. Interaction of ozone and water vapor with spark discharge soot aerosol particles coated with benzo[a]pyrene: O₃ and H₂O adsorption, benzo[a]pyrene degradation, and atmospheric implications. Journal of Physical Chemistry A 105, 4029–4041.
- Radke, L.F., Hegg, D.A., Lyons, J.H., Brock, C.A., Hobbs, P.V., Weiss, R., Rasmussen, R., 1988. Airborne measurements on smokes from biomass burning. In: Hobbs, P.V., McCormick, P. (Eds.), Aerosols and Climate. Deepak Publishing, Virginia, pp. 411–422.
- Reid, J.S., Koppmann, R., Eck, T.F., Eleuterio, D.P., 2005. A review of biomass burning emissions Part II: intensive physical properties of biomass burning particles. Atmospheric Chemistry and Physics, 799–825.
- Turpin, B.J., Huntzicker, J.J., Hering, S.V., 1994. Investigation of organic aerosol sampling artifacts in the Los Angeles basin. Atmospheric Environment 28, 3061–3071.
- Ward, D.E., Hardy, C.C., Ottmar, R.D., Sandberg, D.V., 1982. A sampling system for measuring emissions from west coast prescribed fires. In: Proceedings, Annual Meeting of the PNWIS/AWMA, 15–17 November, Vancouver, BC.
- Ward, D.E., Susott, R.A., Waggoner, A.P., Hobbs, P.V., Nance, J.D., 1992. Emission factor measurements for two fires in British Columbia compared with results for Oregon and Washington. In: Proceedings, 29th Annual meeting of the PNWIS/AWMA, Bellevue, WA, 11–13 November.