Spring 2010

Emissions and Impacts from Prescribed Fire in a Longleaf Pine Ecosystem

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DOI: <https://doi.org/10.31979/etd.6tx2-2gjs>
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EMISSIONS AND IMPACTS FROM PRESCRIBED FIRE
IN A LONGLEAF PINE ECOSYSTEM

A Thesis

Presented to

The Faculty of the Department of Meteorology and Climate Science

San José State University

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

by

Scott J. Strenfel

May 2010

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The Designated Thesis Committee Approves the Thesis Titled

EMISSIONS AND IMPACTS FROM PRESCRIBED FIRE
IN A LONGLEAF PINE ECOSYSTEM

by

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APPROVED FOR THE DEPARTMENT OF METEOROLOGY
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May 2010

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ABSTRACT

EMISSIONS AND IMPACTS FROM PRESCRIBED FIRE IN A LONGLEAF PINE ECOSYSTEM

by Scott J. Strenfel

Prescribed fire is a frequently utilized land-management tool in the Southeastern US. In this study, effects of seasonal-related variations and of turbulence generation on emissions and impacts from prescribed fires were evaluated. High frequency *in situ* data were obtained from three summer (July 2008) and three winter (January 2009) fires within the active burn perimeter and downwind by use of a 10 m instrumented tower and 2 m tripod, respectively. Two cases were selected to evaluate the performance of the EPA-approved short-range regulatory dispersion model, AERMOD, for prescribed fire applications.

Results showed that summer fuels were much greener than winter fuels and did not burn as efficiently, thus resulting in enhanced particulate and CO emissions during summer fires compared to winter. Statistically significant correlations between turbulence, modified combustion efficiency, and pollutant concentrations were found. For both cases, AERMOD was able to reproduce the observed period and hourly averaged downwind particulate concentrations.

ACKNOWLEDGEMENTS

I would like to acknowledge the members of my thesis committee: Drs. Craig Clements, Robert Bornstein, and Frank Freedman. Serving as my advisor, Dr. Clements needs to be acknowledged for all the assistance and tireless help he provided. Dr. Bornstein was critical in the writing phase of the project as his notes from previous classes proved invaluable. Dr. Freedman is an expert in air dispersion modeling and I am grateful he allowed me to use up much of his valuable time to perfect the modeling section of this thesis.

Next, I would like to thank J. Kevin Heirs and the fire crew at the Joseph W. Jones Ecological Research Center at Ichauway, Georgia. They helped set up towers, calibrate instruments, download data, and were the most gracious of hosts. The field research I got to perform, and the experiences I learned, will and have undoubtedly prepared me for future field campaigns.

I would like to thank fellow colleagues Caroline Kiefer and Daisuke Seto for their assistance regarding work in the field. Their endless insights and conversation proved helpful for preparing this thesis. The faculty and students of San José State University deserve credit for their support and knowledge.

Finally I would like to thank the EPA for funding this work. Support for this research was provided by EPA grant # XA-96499308-0 and an additional grant from the Joseph W. Jones Ecological Research Center.

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1. Introduction

Biomass burning produces substantial amounts of trace gases and particulates that impact air quality and climate. Aerosols emitted from biomass burning reflect incoming solar radiation and can enhance the reflectivity of clouds, contributing an estimated net radiative forcing of about 2 W m^{-2} globally (Penner et al. 1992; Crutzen and Ramanathan 2003). Annually, CO_2 emissions from all fire types can equal up to approximately half of global CO_2 emissions from fossil-fuel combustion (Andreae and Merlet 2001; van der Werf et al. 2006; IPCC 2007; Schultz et al. 2008). Much of these emissions are from accidental wildfires and deforestation-related fires (Bowman et al. 2009); however, emissions from prescribed fires represent an important source of primary pollutants and secondary pollutant precursors (Lee et al. 2005). Prescribed fires are intentionally set to maintain ecosystem health, and to prevent destruction from high-intensity wildfires by reducing fuel loadings. It is estimated greater than one million acres are consumed annually by prescribed fire in Georgia, and eight million acres in southern states combined (Wade and Lundsford 1998; Lee et al. 2005).

The goal of any prescribed management program is the preservation of biodiversity. Since the natural biota has evolved under the historical fire regime, a prescribed fire program should be implemented to mimic the history of the region (Whelan 1995); however, a historical fire regime reconstruction is a difficult task since the temporal and spatial patterns and seasonality of fires must be evaluated. If the historical fire regime is unknown or cannot be replicated, then efforts should be made to evaluate how well the biota adapts to a departure from the historical regime.

Emissions from fires occur during both the flaming and smoldering stage. Prior to flaming, fuels are preheated and dried by radiation and convection from the fire front. The heat released from the flame enables highly volatile compounds, including hydrocarbons, to vaporize from the fuel (Ward and Hardy 1991). Flaming is initiated once an ignition source is present and the fuel-to-oxygen mixture reaches flammable proportions. The flammable vaporized compounds ejected from the surface material must rapidly oxidize to sustain the flame envelope. Chemical reactions during flaming rapidly liberate heat, which preheats adjacent fuels and causes further vaporization of low-vapor-pressure fuels and pyrolysis of solid fuels (Ward 2001). Pyrolysis is defined as a heat-induced chemical alteration of solid fuel. Fire, thus, is a self-sustainable reaction as the initial ignition heating provides the activation energy required for sustained ignition and propagation (Whelan 1995).

Smoldering combustion begins when there are insufficient fuel gases available to maintain the flame envelope. Oxygen must diffuse directly to the fuel surface to maintain combustion. Flaming and smoldering combustion have distinct emissions characteristics, chemical reactions, and Combustion Efficiencies (CEs, see Appendix for list of acronyms). Combustion efficiency is a measure of the oxidation capacity of the reactions and is evaluated by determining the amount of carbon released from the fuel in CO₂ form. During idealized complete combustion (i.e., CE = 100 %), all organic material is oxidized in the reactions to produce H₂O, CO₂, and heat. A simplified chemical equation for complete simple sugar (D-glucose) combustion is given by

where Q is the amount of heat released during the reaction. Chemically, plant material is much more complex, and naturally, complete combustion cannot occur due to oxygen deficiencies; thus, products of incomplete combustion, products that are incompletely oxidized, are formed (i.e., CO, CH₄, NH₃).

Combustion efficiencies are generally highest during flaming combustion and depend heavily on oxygen availability. Intense heating from the flame increases turbulent mixing, which advects oxygen into the oxygen-deficient regions. The influences of turbulence on emissions are poorly understood. During smoldering combustion, CEs decrease and more carbon is released as non-CO₂ compounds (i.e., products of incomplete combustion) relative to flaming.

During combustion, CO and CO₂ account for more than 95 % of the carbon released from the biomass (Ward and Hardy 1991). Typically less than 5 % of the carbon is released as Particulate Matter (PM) (Ward et al. 1992; Andreae and Merlet 2001), where PM is defined as solid or liquid particles suspended in the air. Biomass burning produces both solid and liquid particles. The majority of coarse mode PM particles (> 1 µm in diameter) are mechanically produced (Reid et al. 2005). These particles consist of the mechanical break up of plant material, soot, and crustal material lofted due to enhanced turbulence generation.

Fine PM mode (< 1 µm diameter) is divided into two modes: the nuclei-mode and accumulation mode (Winiwarter 2004). Extreme flame temperature causes compounds to vaporize from plant material. Substances with low vapor pressures condense to form nuclei-mode particles with diameters ranging from a few nm up to 0.1 µm. These

particles are quite numerous and act as Condensation Nuclei (CN) for other vaporized compounds; however, nuclei-mode particles account for only a fraction of PM total mass. After formation, nuclei-mode particles begin to grow by agglomeration and condensation into the accumulation mode.

Accumulation mode particles absorb other substances and are regularly found with heavy metals (e.g., Ward and Hardy 1991; Cachier et al. 1995; Lee et al. 2005). Particle formation and content varies depending on the fuel type, CE, oxygen availability, and turbulence to name a few (Radke et al. 1988). For example, Ward (1979) found the oxygen content of the fuel is inversely proportional to PM production. Janhall et al. (2009) found particle mass emissions were strongly dependent on fuel type, but could not establish the same relationship between fuel type and particle number emissions.

Reid et al. (2005) compiled all relevant literature on biomass burning particulates to examine if there was a consensus concerning the size of particulates emitted from fires. Results from numerous studies suggest the properties of particulates are well understood. Although numerous methods were applied in the literature to deduce the size properties of particulates, most reports yielded similar results. Count median diameters of particles in fresh smoke (< 4 hours) from biomass burning was determined to be in the 0.1 – 0.16 μm range, and the volume median diameters of smoke particles was in the 0.25 – 0.3 μm range.

In most studies larger particle sizes were observed during the smoldering phase of small to moderately sized fires, when CE decreases. It is theorized that a lower fraction of vaporized compounds are able to oxidize, leaving more available to condense

on CN. During some fires however, larger particles were observed during the flaming stage (Hays et al. 2002), where it is believed that extensive oxygen deprivation prevents fuel gases from oxidizing. Additionally, Reid and Hobbs (1998) demonstrated fire intensity, which is a measure of the heat liberated by combustion, may be as important a parameter as combustion efficiency in determining particle size properties of fresh smoke emissions.

Smoke particles have been shown in the literature to be made up of Particulate Organic Material (POM), Black Carbon (BC) or “soot”, and inorganic compounds. From all available data, Reid et al. (2005) found fresh-dry particulate mass consists of about 80 % POM, 5 - 9 % BC, and 12 - 15 % inorganic material. Additionally, fine-mode particle mass consists of about 50 - 70 % carbon, where about 55 % is Organic Carbon (OC) and about 8 % is BC. Organic carbon is defined as the carbon in POM.

Adverse health effects from PM exposure have been well documented (e.g., Dockery et al. 1993; US EPA 1997; MacNee and Donalson 1999; Pope 2000; Samet et al. 2000). It is estimated 40,000 premature deaths in the US are due to respiratory illness and heart attacks linked to PM exposure (US EPA 1997). Dockery et al. (1993) found a statistically significant increase in the mortality rate between adults living in the most polluted cities as compared with the least polluted. This study was expanded by Pope et al. (1995) by linking ambient pollution data from 151 US cities with greater than 500,000 adults residing in those cities. They used a multivariate analysis to control for smoking, education, and other risk factors and found a positive correlation between mortality and PM exposure.

Samet et al. (2000) collected PM data from 20 U. S. cities and concluded the estimated increase in the relative death rate from cardiovascular and respiratory causes was 0.68 % for each increase of $10 \mu\text{g m}^{-3}$ PM_{10} , where PM_{10} is defined as PM with aerodynamic diameters $10 \mu\text{m}$ or less. Pope (2000) compiled numerous studies and concluded short-term increases of $10 \mu\text{g m}^{-3}$ PM_{10} were associated with 0.5 - 1.5 % increases in daily mortality.

Particulates with aerodynamic diameters less than $2.5 \mu\text{m}$ ($\text{PM}_{2.5}$) cause more damage than larger sized particles because they penetrate deeper into lung tissue (US EPA 1997). Additionally, MacNee and Donalson (1999) suggest most of the adverse health effects from PM exposure are due to ultrafine particles having less than $0.1 \mu\text{m}$ diameters. Although there are many sources of PM (e.g., fossil fuel burning, mechanically generated dust, wood stove burning) wild and prescribed fires generate more than 600,000 Mg of PM per year, which have shown to be damaging to human health and can lead to premature death (US EPA 1995).

The Clean Air Act, last amended in 1990, requires the US Environmental Protection Agency (EPA) to set National Ambient Air Quality Standards (NAAQS) for PM. The EPA 24 h averaged NAAQS for PM_{10} is currently $150 \mu\text{g m}^{-3}$. Since $\text{PM}_{2.5}$ is more damaging to human health than PM_{10} , the standards are stricter. The EPA 24 h and annual averaged NAAQS for $\text{PM}_{2.5}$ are currently 35 and $15 \mu\text{g m}^{-3}$, respectively. The US EPA 2001 national emissions inventory suggests that in the US, about 35 % of $\text{PM}_{2.5}$ emissions are from biomass burning (Tian et al. 2009), and these emissions have also been shown to have significant impacts on ambient $\text{PM}_{2.5}$ concentrations (Marmur et al.

2005). The EPA regularly enforces NAAQS but does not directly regulate the use of fire within individual States (US EPA 1998). States are required to mitigate public health and welfare impacts from prescribed fires. In Georgia, for example, a burn permit must be obtained from the Georgia Forestry Commission (GFC) before a prescribed burn is implemented. Permits are issued by the GFC if meteorological conditions and regional air quality are acceptable.

Liu (2004) used a dataset of historical fire records collected by numerous US governmental agencies to estimate fire emissions over the contiguous US. An effort was made to quantify the spatial and temporal variability of these emissions. Results show both wild and prescribed fire emissions are greatest over the Pacific coastal states. In the southeastern coastal area prescribed fire emissions were greater than wildland fire emissions; however, a major shortcoming in this analysis is the burned-area data obtained only includes burns on federal lands. Burnings on state, private, and department of defense lands are not included; accordingly, there are significant underestimations of southeastern prescribed fire emissions because many prescribed burns in the region are conducted on private land. Moreover, the area held by private parties is much greater than federally owned land area. For example in Georgia, forests cover more than 66 % of total land and more than 92 % of forestland is owned by private parties (Tian et al. 2008).

Tian et al. (2009) used different emission inventories to evaluate impacts of PM_{2.5} from biomass burning in Georgia, US. Biomass burning data from the Visibility Improvement State and Tribal Association of the Southeast (VISTAS) 2002 inventory and the US EPA 2001 inventory were evaluated. The VISTAS 2002 contains biomass

burning data from wildfires, prescribed burning, agricultural burning, land clearing, and residential wood combustion, while EPA 2001 contains data from the first four sources only. Biomass consumption in VISTAS 2002 is estimated from burned area records obtained from state and federal agencies. Annual emissions from both inventories were processed to provide hourly gridded emissions. Emissions from biomass burning are nearly equally distributed among the four sources in EPA 2001, while prescribed fire represents about 70 % of all biomass emissions in VISTAS 2002. Again, there is most likely an underestimation in prescribed fire emissions as data from burns on privately held land are not included.

Tian et al. (2009) assessed air quality impacts in January, March, May, and July 2002 with the Community Multiscale Air Quality (CMAQ) model. Domain-averaged impacts from biomass burning emissions contribute 3.0, 5.1, 0.8, 0.3 $\mu\text{g m}^{-3}$ of $\text{PM}_{2.5}$ during January, March, May, and July 2002, respectively. This equates to 25 %, 40 %, 9 %, and 5 % of the total $\text{PM}_{2.5}$ during January, March, May, and July 2002, respectively. Moreover, $\text{PM}_{2.5}$ impacts from prescribed burning dominate total biomass burning impacts. Since prescribed fires are planned by forest managers, steps can be taken to reduce $\text{PM}_{2.5}$ impacts on local and regional air quality.

There are many ways managers can attempt to reduce emissions and impacts from fires. Tian et al. (2008) evaluated how emissions and impacts would change due to the Fire Return Interval (FRI). The FRI is defined as the amount of time between burnings, and an optimal FRI should be employed by managers to mitigate emissions. Emissions from fires with a 5 y FRI are approximately 72 % larger than equivalently-sized fires

with a 2 y FRI; thus, longer FRIs would reduce long-term regional pollutant concentrations while short FRIs would reduce short-term and local pollutant concentrations. Chances of increased daily $\text{PM}_{2.5}$ levels are higher with longer FRIs but annual averaged impacts would be lower than short FRIs. To adhere to 24 h NAAQS requires burning more frequently (i.e., smaller FRIs).

Southeastern prescribed burns are typically carried out in winter and spring, when fires are more easily controlled (Whelan 1995; Tian et al. 2008). From 1994 to 2005 prescribed fires consumed 0.86 million acres on average in Georgia, where about 86 % of these fires were scheduled between December and April. Most fires (≈ 37 %) occurred in March (Tian et al. 2008). Burning during summer is less common due to increased atmospheric instability.

Tian et al. (2008) used VISTAS 2002 in conjunction with CMAQ to evaluate 2002 historical air quality conditions. Emissions calculated during March 2002 were applied to January, May, and July 2002 to evaluate seasonal-influences on impacts from fires. Impacts were observed to decrease from January to July, most likely due to increased instability and vertical mixing. Additionally, an increased number of thunderstorms during summer months reduce impacts by increasing ventilation and pollution rainout.

Since managers can choose in which season to burn, evaluating the optimal burning season to mitigate emissions and impacts is crucial. The two major southeastern seasonal impacts are the change in atmospheric dispersion and fuel conditions. Korontzi et al. (2003) found seasonal emission factors, for African savanna fires, correlate linearly

with fuel moisture content. Emissions factors for products of incomplete combustion were higher when fuels were greener. During summer months in the southeast, the growing season, fuel moisture contents are higher compared to winter. Tian et al. (2008) do not incorporate the influence of fuel moisture content on emissions, but point out higher fuel moisture contents lead to less fuel consumption and increased incomplete combustion. Since emissions depend on the amount of fuel consumed and amount of pollutants produced, Tian et al. (2008) point out emissions could either increase or decrease; therefore, seasonal influences on emissions from southeastern prescribed fires must be quantified to answer these questions.

Whereas previous studies have evaluated emissions from prescribed fire, this study will focus on effects of seasonal-related variations and of turbulence generation on emissions and concentration impacts utilizing *in situ* data.

2. Methodology

In this study, effects of seasonal-related variations and turbulence generation on emissions and impacts from prescribed fires were evaluated. High frequency *in situ* data were obtained from three summer (July 2008) and three winter (January 2009) fires within the active burn perimeter using a 10 m instrumented flux tower at the Joseph W. Jones Ecological Research Center at Ichauway, Georgia. Permits were obtained from the GFC prior to each burn, and the Ichauway burn crew allowed us to monitor air quality and meteorology during each burn.

Data obtained using the Moderate Resolution Imaging Spectroradiometer (MODIS) suite have recently been used to estimate biomass burning emissions; however, no fires during this experiment were detected by MODIS although fire sizes were orders of magnitude larger than the minimum needed for detection. Most prescribed fires in Southeast long-leaf pine (*Pinus palustris*) forests consume understorey fuels (e.g., grass, live shrubs, and needles) with little to no tree scorch, and are not easily detected by remote sensing techniques; thus, *in situ* measurements are utilized to quantify turbulence, which cannot be ascertained by space-borne instruments, and seasonal influences on emissions from prescribed fire.

a. Instrumentation

Three wind components (u, v, and w) and temperature were sampled at 10 Hz using a 3D ultrasonic anemometer (R. M. Young 81000) and type e fine-wire thermocouples (Omega, Inc.), respectively. Data have been effectively collected from prescribed fires using sonic anemometers and represent a practical method, given

inherent errors, to observe atmospheric motions on small time scales (Clements et al. 2006, 2007). Tower configuration followed that used by Clements et al. (2006, 2007) where the sonic anemometry and gas analyzers are mounted on a cross arm pointed towards the upwind direction of the tower (Fig. 1).

Fig. 1. Schematic drawing of tower configuration and experimental design. A photograph of the interior 10 m tower following the first burn on 15 July 2008 is shown in the upper left.

Linking emissions to turbulence required use of quick-response air quality sensors. All interior-tower (hereafter referred to as tower) instruments were placed 10 m AGL. Concentrations of $PM_{2.5}$ were made with the DUSTTRAK (model 8520, TSI) aerosol monitor. Prior to each burn, the DUSTTRAK was calibrated and fitted with a cyclone impactor to screen out large particles ($> 2.5 \mu m$). The DUSTTRAK converts

scattered light into mass concentrations every 1 s; however, scattered light is dependent on particle size and is most dramatic for particles with diameters less than 0.25 μm . In addition, count median diameters of particles in fresh smoke (< 4 hours) from biomass burning are typically in the 0.1 – 0.16 μm range (Reid et al. 2005). As a consequence, DUSTTRAK derived concentrations may have some degree of error.

Carbon dioxide was sampled using a robust, open-path infrared sensor (Vaisala, Inc. GMP343 probe). The Vaisala probe has a wide temperature operating range and temporal resolution of 5 s. Carbon monoxide was sampled with the Onset HOBO H11-001 logger (hereafter referred to as HOBO). The HOBO has three operating channels, each of which having unique degrees of resolution and accuracy. Channel 1 covers CO concentrations from 0.2 – 124.3 ppm with 0.5 ppm resolution, while Channel 2 operates from 1 – 497.1 ppm with 2 ppm resolution. The HOBO operated in both channels allowing for a wider dynamic range without sacrificing resolution. Relatively small number of data fell outside the highest quality range (channel 1), and data from channel 2 were substituted in these instances. Channel 1 HOBO-accuracy is typically within ± 4.5 ppm and there may be an additional error that increases as temperatures increase above 20 °C. Although HOBO data most likely contain an absolute error, the HOBO performed well and was able to provide high-frequency measurements. The response time of the instrument was much better than indicated in the instrument manual and is most likely on the order of about 1 min.

A R. M. Young 05103-5 wind monitor was also installed 10 m AGL on the tower. This provided wind speed and direction measurements used to verify R. M. Young 81000 ultrasonic anemometer data.

During the three winter burns an additional air quality sensor was tower-mounted: a Magee Scientific Aethalometer (microAeth AE51). The AE51 is small and lightweight (250 g) and provided BC concentration measurements every 1 s. Measurement range, resolution, and precision are 0 – 1 mg BC m⁻³ for 15 min sampling period at 50 mL min⁻¹ flow rate, 0.001 µg m⁻³, and ± 0.100 µg m⁻³ for 1 min averages at 150 mL min⁻¹ flow rate, respectively.

The downwind receptor site consisted of a 2 m tripod, and was placed directly downwind of each burn-unit (hereafter referred to as unit) to assess plume particle concentrations and meteorological conditions. On the tripod, a Campbell Scientific, Inc. temperature and relative humidity probe (CS215), and R. M. Young 05103-5 anemometer were deployed to assess meteorological conditions, while HOBO (H11-001), DUSTTRAK (model 8520, TSI), and Magee Scientific Aethalometer instruments evaluated air quality. A cyclone impactor was installed on the Aethalometer inlet to screen out particles greater than 2.5 µm to evaluate BC_{2.5} concentrations. Prior to, during, and after each prescribed fire, atmospheric profiles were obtained using the Graw, Inc. DFM-06 radiosonde system. This provided a high vertical resolution dataset of boundary layer and lower tropospheric meteorological structure that determines local dispersion, plume transport, and plume heights.

b. Burn Implementation

Three summer burns were conducted between 15 – 17 July 2008, and represented typical burning conditions for southeastern growing season fires (Fig. 2). Three winter fires, representing winter burn conditions, were performed from 12 – 14 January 2009 (Fig. 3). The fuel during summer consisted of more live-green vegetation indicative of greater moisture content than winter vegetation. Data were collected from over 1 300 acres of fire consumption during ideal burning conditions at the Joseph W. Jones Ecological Research Center at Ichauway, Georgia (Fig. 4).

Fuel loadings of pine needles, grass fuels and shrub biomass were measured pre and post burn in all units. Consumption was estimated through clipping and weighing pre and post burn samples (Table 1).

Fig. 2. Photograph of moist wiregrass fuel and backing fire front on 16 July 2008 1045 EST.

Fig. 3. Photograph of dry wiregrass fuel and fire front on 12 January 2009 1258 EST.

Fig. 4. Joseph W. Jones Ecological Research Center at Ichauway, Georgia land-cover map. Units considered in this study are presented in the orange fill.

Table 1. Select fuel and fire characteristics of 2008 and 2009 experimental burns.

Burn Date	Unit Size (acres)	Loading (tons/acre)	Consumption (tons/acre)
7/15/2008	218	3.15	2.02
7/16/2008	105	3.36	2.19
7/17/2008	110	2.61	1.31
1/12/2009	260	2.72	0.85
1/13/2009	261	3.11	1.81
1/14/2009	257	3.60	1.90

A handheld GPS unit was used to obtain instrument coordinates and ESRI-ArcGIS software was utilized to visualize these locations. Experimental designs and instrument locations for data collected during three summer fires on the 15, 16, and 17 July 2008 are shown in Fig. 5a, b, and c, respectively. Experimental designs for data collected during three winter fires on the 12, 13, and 14 January 2009 are shown in Fig. 5d, e, and f, respectively.

Fig. 5. Experimental designs and instrument locations for data collected during three summer fires on the 15 (a), 16 (b), and 17 (c) July 2008, and during three winter fires on the 12 (d), 13 (e), and 14 (f) January 2009.

Each burn was implemented in a systematic fashion. Burn crews first evaluated the prevailing wind direction and proceeded to ignite the unit along the downwind edge. A backing fire was initiated and slowly progressed into the wind. After a substantial blackline had developed, the crew moved upwind, approximately 50 m, and ignited a line perpendicular to the wind direction. This created a short-lived head fire that propagates with the wind towards previously consumed fuels. For these burns, *in situ* data were collected at the tower, placed directly in the fuel bed, as a head fire propagates towards, under, and past the tower. A small area around the tower was cleared of fuel to prevent instrument damage. Adequate time was allowed to pass before the fire crew ignited further upwind allowing data collection during the smoldering stage. Once fire is set further upwind, the tower becomes a downwind receptor, and subsequent data collected represents a mixture from both flaming and smoldering combustion.

Prior to ignition, the two m tripod was placed directly downwind (150 – 900 m) of the unit (Table 2). It proved difficult to maintain an equivalent distance downwind for all burns as a truck was required to transport equipment. This limited possible tripod location choices, as the tripod needed to be placed near established-passable roads. During each burn, trucks were prohibited from driving upwind and near the tripod as this would contaminate the particulate data.

Table 2. Distance along mean wind direction observed during the burn duration from the most downwind edge of the unit to the tripod.

Date	Distance downwind (m)
7/15/2008	880
7/16/2008	150
7/17/2008	230
1/12/2009	300
1/13/2009	400
1/14/2009	700

c. Data Processing

Data were collected on numerous data loggers. Prior to each burn, all data logger clocks were synchronized as close to US official Eastern Standard Time (EST) as possible. Data were downloaded and stored following each burn for processing. If a data logger clock was not properly aligned to EST during an experiment, the data were synchronized during processing.

Ultrasonic anemometers were manually leveled in the horizontal and vertical plane; however, Dyer (1981) points out small alignment errors can cause large momentum flux errors due to cross contamination of velocities. To correct possible errors due to alignment issues, the 10 Hz *u*, *v*, and *w* wind components were tilt-corrected using a planar fit technique described in detail by Wilczak et al. (2001).

High-frequency turbulent measurements are subject to short-duration, large-amplitude fluctuations, defined as spikes. Spikes result from natural phenomenon or instrument noise and error. A quality control procedure is required to flag and remove data spikes (Brock 1985; Lee et al. 2004). Spikes are defined as points larger than four times the Standard Deviation (SD) of the time series. Lee et al. (2004) recommends processing high-frequency turbulence time series two to three times to flag and remove spikes. Usually, each point is tested against four times the SD of the time series. This approach proved inadequate for turbulence data gathered during a fire front passage, as fire generates turbulence significantly above background levels; thus, an algorithm was developed specifically for this dataset.

The algorithm calculates the SD of two min segment (1 200 points) and tests each point within the segment. Points greater than four times the segment-SD are characterized as spikes. A thorough visual and mathematical inspection is conducted to ensure points flagged by the algorithm are most likely instrument noise. Spikes are replaced with the mean value of the ten points preceding and following. The process is conducted three times to ensure erroneous spikes are removed.

Turbulent components of the flow were calculated by

$$u' = U - \bar{U}$$

$$v' = V - \bar{V}$$

$$w' = W - \bar{W},$$

where u' , v' , w' are turbulent components of the flow, $\overline{U}$, $\overline{V}$, $\overline{W}$ are 30 min mean velocities, and U , V , W are tilt-corrected-despiked instantaneous velocities (Stull 1988). Variances were calculated using turbulent components and fluxes were calculated from one min averaged covariances (Clements et al. 2006, 2007).

Turbulent Kinetic Energy (TKE) is defined as the mean kinetic energy of eddies in turbulent flow per unit mass, and is given by

$$TKE = \frac{1}{2} \left(\overline{u'^2} + \overline{v'^2} + \overline{w'^2} \right),$$

where $\overline{u'^2}$, $\overline{v'^2}$, and $\overline{w'^2}$ represent one min mean velocity variances. A one min TKE time series was produced for each burn to evaluate turbulent energy production and if a relationship between TKE generation and emissions exists.

d. Emissions

Tower concentration data of $PM_{2.5}$, CO_2 , and CO were averaged to one min. Data collected during the flaming and smoldering stage of the fire were separated to evaluate the difference in emissions from flaming and smoldering combustion (Table 3), where the data separation for all burns from the flaming and smoldering stage is shown Fig. 6.

Table 3. Start and end times (EST) of flaming and smoldering stage emissions for all burns.

Date	Flaming Stage		Smoldering Stage	
	Start	End	Start	End
7/15/2008	1241	1244	1245	1254
7/16/2008	1043	1053	1054	1112
7/17/2008	908	914	915	940
1/12/2009	1226	1228	1233	1238
1/13/2009	1220	1223	1224	1234
1/14/2009	1359	1405	1400	1414

Fig. 6. One min average concentration data on the 15 (a), 16 (b), and 17 (c) July 2008, and on the 12 (d), 13 (e), and 14 (f) January 2009. The dashed lines indicate where the data were separated from flaming stage emissions (F), and smoldering stage emissions (S).

To characterize emissions from each fire, *in situ* data were used to calculate Emission Factors (EFs) during the flaming (EF_f) and smoldering stage (EF_s). This calculation (Andreae and Merlet 2001) is given by

$$EF_x \cong \frac{[\Delta C_x]}{\sum ([\Delta C_{CO_2}] + [\Delta C_{CO}] + \{[\Delta C_{CO_2}] + [\Delta C_{CO}]\} * 0.025)} [C]_b ,$$

where ΔC_x is the concentration in the plume minus the ambient concentration for species x (g m^{-3}), $[C]_b$ is the carbon content of the fuel (g carbon / kg fuel consumed), and $\{[\Delta C_{CO_2}] + [\Delta C_{CO}]\} * 0.025$ is a conservative factor representing non- CO_2 and CO carbon emissions [e.g., hydrocarbons (CH_4), elemental and black carbon (Andreae and Merlet 2001; Ward and Hardy 1990)]. The mean pollutant concentrations observed before the fire were used to characterize ambient concentrations.

Since the fuel was determined to consist of 50 % carbon by weight, the carbon content of the fuel ($[C]_b$) used was 0.5. Instantaneous fuel consumption was calculated by

$$W_v = \frac{\sum C_n}{[C]_b} ,$$

where W_v is the amount of fuel consumed (g m^{-3}), C the carbon fraction of emissions, and n represents the various species emitted containing carbon (Hardy et al. 1996).

Calculation of the instantaneous fuel consumption was required to evaluate the effective EF (EF_t), which represents emissions during the fire duration (i.e., total fire). This was achieved using a weighted averaged procedure given by

$$EF_{t_j} = \frac{\overline{EF}_{f_j} * \sum W_f + \overline{EF}_{s_j} * \sum W_s}{\sum W_f + \sum W_s},$$

where $\overline{EF}_f$ is the mean EF during the flaming stage, $\overline{EF}_s$ the mean EF during smoldering, $\sum W_f$ fuel consumed during flaming, $\sum W_s$ fuel consumed during smoldering, and j is the species evaluated (Hardy et al. 1996).

The mean EF_s , EF_{s_s} , and EF_{s_t} for each burn were evaluated. To evaluate the precision and uncertainty of the means, the Standard Error (SE) of the means was calculated. The SE depends on the sample size, N, and the standard deviation of the sample, and decreases with increasing N. Since the EFs were found to be autocorrelated, the number of independent observations is fewer than N. The autocorrelations were found to be first-order (dependence on a time lag of one only) such that the effective sample size, N' , can be calculated by

$$N' = N \frac{(1 - r_1)}{(1 + r_1)},$$

where r_1 is the first-order autocorrelation coefficient (WMO 1966). Accounting for autocorrelation, the SE is then calculated by

$$SE = \frac{\sigma}{\sqrt{N'}},$$

where σ is the standard deviation of the sample. A schematic diagram is presented in Fig. 7, which shows the process used to derive the EFs.

Fig. 7. Schematic diagram of the process used to evaluate the EFs for each prescribed burn.

3. Results

a. Modified Combustion Efficiency

The instantaneous Modified Combustion Efficiency (MCE) for each case was evaluated by

$$MCE \equiv \frac{\Delta C_{CO_2}}{\Delta C_{CO_2} + \Delta C_{CO}} * 100\% ,$$

a measure of fire efficiency (Ward and Radke 1993). The MCEs were evaluated during smoldering (MCE_s), flaming (MCE_f) (Fig. 8), and for the effective fire (MCE_t) (Table 4). The MCEs observed during the winter were much less variable than MCEs observed during summer. Less efficient combustion produces relatively more products of incomplete combustion. The summer mean MCE_t was 94.4 ± 0.7 % and the winter mean MCE_t was 97.1 ± 0.7 %. The difference between the summer and winter mean MCE_t was found to be statistically significant to the 95 % confidence interval and suggests winter fires were more efficient in oxidizing the carbon contained in the fuel such that increased emissions of CO₂ and decreased emissions of CO and PM_{2.5} should be observed compared to summer.

Fig. 8. Flaming (a) and smoldering (b) MCEs during all burns. The red line indicates the mean of the calculated MCEs.

Table 4. Summer and winter Modified Combustion Efficiency (MCE).

MCE*			
Date	MCE_f	MCE_s	MCE_t
7/15/2008	97.2 ± 0.9	91.1 ± 1.0	96.3 ± 0.9
7/16/2008	95.8 ± 0.7	90.4 ± 1.1	94.9 ± 0.7
7/17/2008	94.7 ± 1.4	88.7 ± 1.1	93.2 ± 1.3
Summer	95.7 ± 0.7	89.7 ± 0.7	94.4 ± 0.7
1/12/2009	97.8 ± 0.7	96.0 ± 0.6	97.0 ± 0.7
1/13/2009	97.9 ± 0.3	94.8 ± 0.7	96.8 ± 0.4
1/14/2009	98.4 ± 0.3	94.3 ± 0.9	97.4 ± 0.4
Winter	98.1 ± 0.3	94.9 ± 0.4	97.1 ± 0.3

* Values are means ± one SE

b. Emissions

One min averaged tower PM_{2.5}, CO₂, CO, and BC concentrations, tripod PM_{2.5}, CO, and BC_{2.5} concentrations, and wind speed and direction measured during the fire front passage on the 15, 16, and 17 July 2008 are presented in Figs. 9, 10, and 11, respectively. The fire front passage is clearly seen in all tower time series as concentration of all species sharply increases. Data obtained during this brief period represent the flaming stage of combustion when MCEs are largest. Data were also obtained during the smoldering stage when MCE decreases. The segregation of data

between flaming and smoldering stages was confirmed using temperature, TKE, and MCE observed at the tower.

Similarly, one min averaged tower $PM_{2.5}$, CO_2 , CO , and BC concentrations measured during fire front passage on 12, 13, and 14 January 2009 are presented in Figs. 12, 13, and 14, respectively.

Fig. 9. One min averaged tower $PM_{2.5}$, CO_2 , CO , and BC concentrations, tripod $PM_{2.5}$, CO , and $BC_{2.5}$ concentrations, and tower and tripod wind speed and direction measured during hour when fire front moved passed tower on 15 July 2008.

Fig. 10. One min averaged tower $PM_{2.5}$, CO_2 , CO , and BC concentrations, tripod $PM_{2.5}$, CO , and $BC_{2.5}$ concentrations, and tower and tripod wind speed and direction measured during hour when fire front moved passed the tower on 16 July 2008.

Fig. 11. One min averaged tower $PM_{2.5}$, CO_2 , CO , and BC concentrations, tripod $PM_{2.5}$, CO , and $BC_{2.5}$ concentrations, and tower and tripod wind speed and direction measured during hour when fire front moved passed the tower on 17 July 2008.

Fig. 12. One min averaged tower $PM_{2.5}$, CO_2 , CO , and BC concentrations measured during hour when fire front moved passed the tower on 12 January 2009.

Fig. 13. One min averaged tower $PM_{2.5}$, CO_2 , CO , and BC concentrations measured during hour when fire front moved passed the tower on 13 January 2009.

Fig. 14. One min averaged tower PM_{2.5}, CO₂, CO, and BC concentrations measured during hour when fire front moved passed the tower on 14 January 2009.

Utilizing the *in situ* tower concentration data, EFs for PM_{2.5}, CO, CO₂, and BC (winter only) were determined. A visual inspection of each unit revealed a nearly homogenous fuel bed; thus, EFs derived from tower data are valid across the whole unit. These EFs are applicable for homogenous wiregrass longleaf-pine fuel with a FRI of approximately two yr. The EFs are presented in units of g of pollutant emitted per kg fuel consumed. To evaluate emissions from a prescribed burn falling within these burn parameters, one simply needs to estimate or calculate fuel consumption and then apply an EF. Tower and tripod maximum observed concentrations for each burn are presented in Table 5.

Table 5. Maximum tower (10 m) and tripod (2 m) concentrations measured during summer and winter burns.

Date	Interior				Downwind		
	PM _{2.5}	CO ₂	CO	BC	PM _{2.5}	BC _{2.5}	CO
	(mg m ⁻³)	(ppm)	(ppm)	(mg m ⁻³)	(mg m ⁻³)	(mg m ⁻³)	(ppm)
7/15/2008	52.0	2115	65.8	**	0.93	0.031	3.65
7/16/2008	65.1	2072	21.0	**	2.24	0.100	6.16
7/17/2008	91.3	3567	124.3	**	3.60	*	6.89
1/12/2009	20.5	1084	30.6	1.10	0.79	0.029	1.06
1/13/2009	28.8	2373	64.4	0.92	0.91	0.032	0.99
1/14/2009	42.6	2165	65.0	0.91	0.30	0.009	*

* Instrument error

** Not measured

Summer and winter derived PM_{2.5} EFs during flaming and smoldering combustion are shown in Fig. 15, and the mean EFs during flaming, smoldering, and for the total effective-fire are presented in Table 6. As expected, the EF_{s_f} for each burn were lower than EF_{s_s}. Due to increased MCEs, the PM_{2.5} EF_t for all winter fires were lower than summer fires; however, the surface concentration impact downwind from winter fires may be greater than summer fires due to increased atmospheric stability.

Summer and winter derived CO EFs during flaming and smoldering combustion are shown in Fig. 16, and the mean EFs during flaming, smoldering, and for the total effective-fire are presented in Table 7. Results are similar to EFs derived for PM_{2.5}, as

CO is also a product of incomplete combustion. Less CO was emitted in dryer fuel conditions of winter than during more moist fuel conditions of summer. Burning under dryer conditions can result in a reduction of about half, as seen in the winter versus summer mean $EF_{s,t}$. Summer and winter derived CO_2 EFs during flaming and smoldering combustion are shown in Fig. 17, and the mean EFs during flaming, smoldering, and for the total effective-fire are presented in Table 8. More CO_2 was emitted during all winter fires due to increased MCEs.

Fig. 15. Flaming (a) and smoldering (b) PM_{2.5} EFs during all burns. The red line indicates the mean of the calculated EFs.

Table 6. Summer and winter derived PM_{2.5} EFs.

PM _{2.5} EFs* [g kg ⁻¹]			
Date	EF _f	EF _s	EF _t
7/15/2008	8.8 ± 3.6	31.5 ± 9.2	12.1 ± 4.4
7/16/2008	12.4 ± 1.6	11.8 ± 1.3	11.9 ± 1.5
7/17/2008	9.9 ± 3.4	15.1 ± 2.2	11.1 ± 3.2
Summer	10.8 ± 1.7	17.0 ± 2.9	12.1 ± 2.0
1/12/2009	6.8 ± 0.5	9.4 ± 0.6	7.9 ± 0.5
1/13/2009	6.8 ± 1.9	12.5 ± 2.6	8.8 ± 2.2
1/14/2009	4.1 ± 0.8	11.6 ± 2.2	9.6 ± 1.1
Winter	5.4 ± 1.3	11.5 ± 1.2	7.3 ± 1.3

* Values are means ± one SE

Fig. 16. Flaming (a) and smoldering (b) CO EFs during all burns. The red line indicates the mean of the calculated EFs.

Table 7. Summer and winter derived CO EFs.

CO EFs* [g kg ⁻¹]			
Date	EF _f	EF _s	EF _t
7/15/2008	13.7 ± 4.4	43.6 ± 5.0	18.1 ± 4.5
7/16/2008	20.7 ± 3.3	46.7 ± 5.4	41.2 ± 3.6
7/17/2008	25.9 ± 6.7	55.3 ± 5.3	33.1 ± 6.5
Summer	21.1 ± 3.3	50.3 ± 3.6	27.2 ± 3.3
1/12/2009	10.5 ± 3.4	19.6 ± 3.0	14.4 ± 3.2
1/13/2009	10.1 ± 1.6	25.4 ± 3.3	15.4 ± 2.2
1/14/2009	7.8 ± 1.4	27.9 ± 4.2	22.6 ± 2.1
Winter	9.0 ± 1.4	24.9 ± 2.2	14.0 ± 1.7

* Values are means ± one SE

Fig. 17. Flaming (a) and smoldering (b) CO₂ EFs during all burns. The red line indicates the mean of the calculated EFs.

Table 8. Summer and winter derived CO₂ EFs.

EFs CO ₂ * [g kg ⁻¹]			
Date	EF _f	EF _s	EF _t
7/15/2008	1867 ± 27	1691 ± 28	1841 ± 27
7/16/2008	1825 ± 20	1675 ± 30	1706 ± 22
7/17/2008	1794 ± 40	1626 ± 30	1753 ± 39
Summer	1822 ± 20	1654 ± 20	1787 ± 20
1/12/2009	1886 ± 21	1831 ± 18	1862 ± 20
1/13/2009	1888 ± 10	1796 ± 20	1856 ± 13
1/14/2009	1903 ± 9	1781 ± 25	1814 ± 13
Winter	1895 ± 9	1799 ± 13	1865 ± 10

* Values are means ± one SE

Winter derived BC EFs are presented in Table 9. Unfortunately, the microAethalometer could not be secured in time for the summer burns. Burning in winter produced about 0.28 ± 0.05 g of BC per kg of fuel consumed. It is very reasonable to suspect BC EFs during the summer would be greater than winter due to decreased MCEs.

Table 9. Winter derived BC EFs.

BC EFs* [g kg ⁻¹]			
Date	EF _f	EF _s	EF _t
1/12/2009	0.40 ± 0.05	0.40 ± 0.01	0.40 ± 0.03
1/13/2009	0.21 ± 0.07	0.34 ± 0.04	0.25 ± 0.06
1/14/2009	0.23 ± 0.09	0.25 ± 0.04	0.24 ± 0.07
Winter	0.26 ± 0.06	0.32 ± 0.03	0.28 ± 0.05

* Values are means ± one SE

Direct EFs for BC_{2.5} could not be ascertained from measurements as the aethalometer could not be installed on the tower due to instrument size and weight; however, BC_{2.5} EFs were estimated using tripod BC_{2.5} and PM_{2.5} data, and PM_{2.5} measurements from the tower. Due to the proximity of the source location to the tripod we can assume the concentration ratio of BC_{2.5} to PM_{2.5} does not change during plume transport by a substantial amount. The mean BC_{2.5}/PM_{2.5} observed at the tripod during each burn was evaluated and used to estimate BC_{2.5} concentrations at the tower. Results are presented in Table 10. The statistical significance of the mean BC_{2.5}/PM_{2.5} to the 95 % confidence interval was tested using 1.96 times the SE of the means. For all burns, the mean BC_{2.5}/PM_{2.5} was found to be statistically significant to the 95 % confidence interval, which indicates the variability of the BC_{2.5}/PM_{2.5} is within an acceptable range to estimate BC_{2.5} tower concentrations. Tower concentrations of BC_{2.5} are estimated by

multiplying the mean $BC_{2.5}/PM_{2.5}$ and tower $PM_{2.5}$ concentrations. From these estimated $BC_{2.5}$ tower concentrations, $BC_{2.5}$ EFs are calculated (Table 11). Since $BC_{2.5}$ EFs were not directly derived, the values presented are means only. The $BC_{2.5}/PM_{2.5}$ observed during this experiment agree with results from Lee et al. (2005), where the $BC_{2.5}/PM_{2.5}$ was found to be ~ 0.04 from similar prescribed fires in Georgia.

Table 10. Summer and winter $BC_{2.5}/PM_{2.5}$.

Date	$BC_{2.5}/PM_{2.5}$^a	$BC_{2.5}/PM_{2.5}$ (std. dev)^b	$BC_{2.5}/PM_{2.5}$ (SE)^c
7/15/2008	0.032	0.014	0.000
7/16/2008	0.040	0.015	0.001
7/17/2008	*	*	*
1/12/2009	0.037	0.014	0.001
1/13/2009	0.033	0.010	0.001
1/14/2009	0.026	0.010	0.001

* Data were not collected

a. Values are arithmetic means computed at the downwind tripod during the burn

b. Values presented are one standard deviation

c. Values presented are one SE

Table 11. Summer and winter derived BC_{2.5} EFs.

BC _{2.5} EFs* [g kg ⁻¹]			
Date	EF _f	EF _s	EF _t
7/15/2008	0.28	1.01	0.39
7/16/2008	0.50	0.47	0.48
7/17/2008	**	**	**
Summer	0.44	0.67	0.47
1/12/2009	0.25	0.35	0.29
1/13/2009	0.22	0.41	0.29
1/14/2009	0.11	0.30	0.17
Winter	0.17	0.36	0.23

* Values are means

** Data were not collected

From laboratory experiments, McMeeking et al. (2009) found the MCE for the combustion of longleaf pine fuel was 94.4 ± 2.3 %. This agrees with results from this study where the summer mean MCE_t was 94.4 ± 0.7 % and the winter mean MCE_t was 97.1 ± 0.7 %. McMeeking et al. (2009) found the PM_{2.5} EF_t to be 38.3 ± 13.6 g kg⁻¹, which is much larger than the EF_s_t found during this experiment. Andreae and Merlet (2001) compiled and integrated all available literature and produced EFs for species emitted from various types of biomass burning. From data obtained from savanna and grassland, tropical forest, and extratropical forest burns, the PM_{2.5} EF_t was found to be

5.4 ± 1.5 , 9.1 ± 1.5 , and $13.0 \pm 7.0 \text{ g kg}^{-1}$, respectively. The $\text{PM}_{2.5} \text{ EF}_t$ observed during summer and winter experiment burns fall within the range published by Andreae and Merlet (2001) for extratropical forest.

A statistical analysis was performed for quality control, parameter prediction ability, and to measure the association of the relationship between variables. Table 12 presents the Pearson 2-tailed correlations between MCE, instantaneous fuel consumption, and one min averaged concentrations of CO_2 , CO , and $\text{PM}_{2.5}$ for the 15 July 2008 prescribed burn. All correlations for this case were found to be significantly significant to the 95 % confidence interval.

The predictive power of MCE for the $\text{PM}_{2.5} \text{ EF}_t$ for all burns was evaluated. A MCE- $\text{PM}_{2.5} \text{ EF}_t$ scatter plot for the 15 July 2008 burn is shown in Fig. 18, and linear and exponential regressions models were produced. The linear regression model is given by

$$\begin{aligned} \text{PM}_{2.5} \text{ EF}_t &= 410.1 - 4.15(\text{MCE}) \\ R^2 &= 0.61 \quad p = 0.001, \end{aligned}$$

and is statistically significant to the 99 % confidence interval. The MCE has the ability to capture the variance in the $\text{PM}_{2.5} \text{ EF}_t$, as seen in its reasonable R^2 value. The exponential regression model for this burn is given by

$$\begin{aligned} \text{PM}_{2.5} \text{ EF}_t &= 2.38 \times 10^9 e^{(-0.201 \cdot \text{MCE})} \\ R^2 &= 0.73 \quad p = 0.000, \end{aligned}$$

and is statistically significant to the 99 % confidence interval. The MCE has the ability to capture the variance in the EF_t , as seen in its reasonable R^2 value. Utilizing these equations could enable one to estimate the emissions from a prescribed burn in similar fuel types and under similar conditions from a MCE calculation alone.

Table 12. Pearson correlation coefficients between MCE, instantaneous fuel consumption (Fuel C) and one min averaged concentrations of CO_2 , CO, and $PM_{2.5}$ for burn on 15 July 2008.

	MCE	CO_2	CO	$PM_{2.5}$	Fuel C
MCE	1	0.76*	0.67*	0.58**	0.64**
CO_2	0.76*	1	0.91*	0.87*	0.92*
CO	0.67*	0.91*	1	0.97*	0.97*
$PM_{2.5}$	0.58**	0.87*	0.97*	1	0.99*
Fuel C	0.64**	0.92*	0.97*	0.99*	1

* Correlation is statistically significant to the 99 % confidence interval

** Correlation is statistically significant to the 95 % confidence interval

A MCE- $PM_{2.5}$ EF_t scatter plot for the 12 January 2009 burn is shown in Fig. 19, and linear and exponential regressions models were produced. The linear regression model is given by

$$PM_{2.5} EF_t = 165.1 - 1.613(MCE)$$

$$R^2 = 0.55 \quad p = 0.002,$$

and is statistically significant to the 99 % confidence interval. The exponential regression model for this burn is given by

$$PM_{2.5} EF_t = 1.965 \times 10^7 e^{(-0.151 \bullet MCE)}$$

$$R^2 = 0.57 \quad p = 0.001,$$

and is statistically significant to the 99 % confidence interval. In both cases an exponential regression provided the best fit for the data. Caution should be taken when estimating emissions from the exponential regressions when MCEs are small because the y-intercept values are unrealistic, and when MCEs approach 100 % because, in theory, the EFs for products of incomplete combustion should go to zero.

Fig. 18. Scatter plot of $PM_{2.5} EF_t$ versus MCE for the 15 July 2008 burn. The solid line shows the linear regression while the dashed line shows the exponential regression.

Fig. 19. Scatter plot of $PM_{2.5} EF_t$ versus MCE for the 12 January 2009 burn. The solid line shows the linear regression while the dashed line shows the exponential regression.

The two statistical examples above represent only one summer and one winter burn. All summer collected and derived data were combined and statistically tested to evaluate the variability between the three summer burns. Table 13 presents the correlations between MCE, instantaneous fuel consumption, and one min averaged concentrations of CO₂, CO, and PM_{2.5} for the combined summer data. The correlation of the combined data is smaller than for any individual burn; however, all correlations were found to be statistically significant except for the MCE-CO correlation. Differences in fuel moisture and fuel loading between burns may explain the enhanced variability in the combined data.

A MCE-PM_{2.5} EF_t scatter plot for all summer burn data is presented in Fig. 20, and linear and exponential regressions models were produced. The linear regression model is given by

$$PM_{2.5} EF_t = 95.9 - 0.883(MCE)$$

$$R^2 = 0.12 \quad p = 0.002,$$

and is statistically significant to the 99 % confidence interval, while the exponential regression model is given by

$$PM_{2.5} EF_t = 2900e^{(-0.059 \cdot MCE)}$$

$$R^2 = 0.21 \quad p = 0.000,$$

and is statistically significant to the 99 % confidence interval.

Table 13. Pearson correlation coefficients between MCE, instantaneous fuel consumption (Fuel C) and one min averaged concentrations of CO₂, CO, and PM_{2.5} for all summer burn data combined.

	MCE	CO₂	CO	PM_{2.5}	Fuel C
MCE	1	0.52*	0.20	0.39*	0.51*
CO ₂	0.52*	1	0.83*	0.87*	0.99*
CO	0.20	0.83*	1	0.92*	0.85*
PM _{2.5}	0.39*	0.87*	0.92*	1	0.89*
Fuel C	0.51*	0.99*	0.85*	0.89*	1

* Correlation is statistically significant to the 99 % confidence interval

Fig. 20. Scatter plot of PM_{2.5} EF_t versus MCE for all summer burn data. The solid line shows the linear regression, while the dashed line shows the exponential regression.

All winter collected and derived data were combined and statistically tested to evaluate the variability between the three winter burns. Table 14 presents the correlations between MCE, TKE, and one min averaged concentrations of CO₂, CO, BC, and PM_{2.5} for the combined data.

A MCE-PM_{2.5} EF_t scatter plot for all winter burn data is presented in Fig. 21, and linear and exponential regressions models were produced. The linear regression model is given by

$$PM_{2.5} EF_t = 178.3 - 1.76(MCE)$$

$$R^2 = 0.34 \quad p = 0.000,$$

and is statistically significant to the 99 % confidence interval. The exponential regression model for the combined data is given by

$$PM_{2.5} EF_t = 3.2 \times 10^9 e^{(-0.206 \bullet MCE)}$$

$$R^2 = 0.45 \quad p = 0.000,$$

and is statistically significant to the 99 % confidence interval.

Table 14. Pearson correlation coefficients between TKE, MCE, and one min averaged concentrations of CO₂, CO, PM_{2.5}, and BC for all winter burn data combined.

	TKE	MCE	CO₂	CO	PM_{2.5}	BC
TKE	1	0.51*	0.88*	0.70*	0.60*	0.44*
MCE	0.51*	1	0.54*	-0.01	0.02	0.57*
CO ₂	0.88*	0.54*	1	0.79*	0.60*	0.53*
CO	0.70*	-0.01	0.79*	1	0.85*	0.32**
PM _{2.5}	0.60*	0.02	0.60*	0.85*	1	0.48*
BC	0.44*	0.57*	0.53*	0.32**	0.48*	1

* Correlation is statistically significant to the 99 % confidence interval

** Correlation is statistically significant to the 95 % confidence interval

Fig. 21. Scatter plot of PM_{2.5} EF_t versus MCE for all winter burn data. The solid line shows the linear regression, while the dashed line shows the exponential regression.

c. Observed Fire-Atmosphere Interactions

The relationship between combustion efficiency and emissions is well understood (Ward and Hardy 1991; Ward et al. 1996; Ward 2001); however, the relationship between MCE, TKE, and emissions is much less known. A statistical analysis was performed to test the relationship between turbulence generation, TKE, and other measured quantities. The correlations between TKE and all variables, notably MCE, are statistically significant to the 99 % confidence interval, thus suggesting turbulence generation, fire efficiency, and emissions from fires, in this experiment, are related. For all combined data a TKE-MCE scatter plot was produced (Fig. 22), and a linear regression model was produced. The linear regression model is given by

$$\begin{aligned}MCE &= 94.9 + 0.61(TKE) \\ R^2 &= 0.26 \quad p = 0.001,\end{aligned}$$

and is statistically significant to the 99 % confidence interval.

Fig. 22. Scatter plot of MCE versus TKE for all winter burn data, where the solid line shows the linear regression.

Linear regression models for one min averaged tower concentrations of CO_2 (Fig. 23a), CO (Fig. 23b), $\text{PM}_{2.5}$ (Fig. 23c) and BC (Fig. 23d), as a function of TKE, yield R^2 values of 0.78, 0.50, 0.38, and 0.19, respectively; thus, TKE is able to explain most of the CO_2 , and CO variability. The CO_2 , CO, $\text{PM}_{2.5}$ and BC linear regressions are given by

$$\text{CO}_2 = -77.3 + 437(\text{TKE})$$

$$R^2 = 0.78 \quad p = 0.000$$

$$\text{CO} = 8.3 + 7.5(\text{TKE})$$

$$R^2 = 0.50 \quad p = 0.000$$

$$\text{PM}_{2.5} = 5.1 + 3.1(\text{TKE})$$

$$R^2 = 0.36 \quad p = 0.000$$

$$BC = 0.233 + 0.065(TKE)$$

$$R^2 = 0.19 \quad p = 0.005,$$

respectively, and are statistically significant to the 99 % confidence interval.

Fig. 23. Scatter plot of tower CO_2 (a), CO (b), $\text{PM}_{2.5}$ (c), and BC (d) concentrations versus TKE for all winter burn data, where the solid line shows the linear regression.

The correlation between TKE and CO_2 is strongest as represented by a large R^2 value. As seen in Fig. 23a, CO_2 concentrations increase when more turbulence is present. Enhanced turbulence generation may enable more oxygen entrainment into oxygen deficient regions allowing for increased carbon oxidization. On the other hand, complete combustion liberates more heat than incomplete combustion, which creates turbulence due to enhanced buoyancy. The impact of turbulence generation on fire efficiency and emissions needs to be more closely examined.

One min averaged TKE and friction velocity during the entire duration of each burn is shown in Fig. 24. The friction velocity is a measure of the vertical flux of horizontal momentum (Stull 1988). Prior to fire front passage, ambient TKE values were generally on the order of $1 \text{ m}^2 \text{ s}^{-2}$. During fire front passage, TKE values increase well above ambient. After passage, TKE values are generally higher than ambient due to weak convection during smoldering conditions. Since the tower becomes a downwind receptor after passage, subsequent plume impactions, and the turbulence associated with them, causes TKE to increase well above ambient again.

As seen in Fig. 24a, ambient TKE values are observed until about 1240 EST (= 1740 UTC) when TKE values sharply increase. This increase in TKE occurs during the fire front passage and plume impaction of the tower. Before this time, TKE is generated through mechanically generated shear and, to a lesser extent, through buoyancy. Gaps in data, seen in Figs. 24a, b, and c, correspond to the highest temperatures encountered, which caused the sonic anemometer to occasionally fail. As seen in Fig. 24d, e, and f, ambient TKE values are observed until they sharply increase, indicating fire front

passage and plume impaction. The heat released from the fire, in all cases, generates TKE well above the pre-passage ambient conditions.

Fig. 24. One min averaged TKE and friction velocity (Ustar) on the 15 (a), 16 (b), and 17 (c) July 2008, and on the 12 (d), 13 (e), and 14 (f) January 2009 during the entire duration of each burn.

One min averaged TKE, sensible heat flux, and 10 Hz temperature and u, v, and w wind components observed at 10 m during fire front passage at the tower on 15, 16, and 17 July 2008 are presented in Figs. 25, 26, and 27, respectively. The sensible heat flux (KW m^{-2}) is defined as the flux of energy from the surface that does not include phase changes of water.

On 15 July 2008 the plume impacted the tower just after 1240 EST. Sharp increases in TKE (Fig. 25a), u, v, and w wind components (Fig. 25b), sensible heat flux (Fig. 25c), and temperature (Fig. 25d) are clearly observed. The sonic anemometer failed from about 1245 to 1253 EST such that data during this period could not be collected. The thermocouple, placed near the anemometer at 10 m, reveals the maximum temperatures observed in the plume were about 125 °C. Since the thermocouple data did not fail, these data are used to verify when the plume impaction occurred. For this burn the anemometer collected data during the impaction and failed shortly after. In Fig. 25b, as the plume begins to impact the tower (about 1242 EST), a sustained updraft is observed. During this updraft period temperatures (Fig. 25d) start increasing. At approximately 1244 EST the updraft quickly transitions to a downdraft and temperatures return to near-ambient. This suggests that the updraft was observed inside the plume and as the plume propagates beyond the tower a downdraft was observed. Similar results were found by Clements et al. (2006, 2007, 2008) and seem to suggest the presence of the fire creates a circulation such that a downdraft is observed behind the propagating fire-front.

During the 16 July 2008 burn, the sonic anemometer seemed to fail during the plume impaction (Fig. 26b). As temperatures begin to peak (Fig. 26d) at about 1051 EST, the anemometer failed. Since TKE and sensible heat flux are both derived from 10 Hz wind components (Fig. 26b), these data are unavailable, thus, the maximum peaks in TKE, sensible heat flux, and the u, v, and w wind components could not be observed due to instrument failure; however, just before 1045 EST, a very weak plume seems to have impacted the tower as temperature (Fig. 26d) and TKE (Fig. 26a) increase well above ambient. Initially, an updraft is observed, and as the weak plume moves past the tower (about 1045 EST), the updraft transitions to a downdraft.

During the 17 July 2008 burn, the sonic anemometer failed before the plume impaction (Fig. 27b). It remains unknown why the anemometer failed well before temperatures (Fig. 27d) start increasing. Maximum temperatures observed in the plume were about 150 °C. Before the plume completely propagates past the tower the anemometer begins working (912 EST) and reveals an updraft, enhanced wind components (Fig. 27b), and enhanced TKE (Fig. 27a), from about 912 to 915 EST. As temperatures begin to return to ambient a weak downdraft is observed. As the case with the previous burn, maximum TKE and sensible heat flux could not be ascertained due to instrument failure.

Fig. 25. One min averaged tower TKE, turbulent variances, and vector wind components measured during hour when fire front moved passed the tower on 15 July 2008.

Fig. 26. One min averaged tower TKE, turbulent variances, and vector wind components measured during hour when fire front moved passed the tower on 16 July 2008.

Fig. 27. One min averaged tower TKE, turbulent variances, and vector wind components measured during hour when fire front moved passed the tower on 17 July 2008.

One min averaged TKE, sensible heat flux, 10 Hz u, v, and w wind components, and 1 Hz thermocouple temperature observed at 10 and 2 m during fire front passage at the tower on 12, 13, and 14 January 2009 are presented in Figs. 28, 29, and 30, respectively. During the winter burns the sonic anemometer did not fail allowing for complete data collecting during plume impaction.

The plume on the 12 January 2009 prescribed fire impacted the tower just after 1220 EST. Sensible heat flux (Fig. 28c), TKE (Fig. 28a), and u, v, and w wind components (Fig. 28b), increase well above ambient from about 1220 to 1223 EST during the impaction. Temperature was measured at 10 m and 2 m (Fig. 28d). The 10 m

temperature is collected at the same level as the sonic anemometer data. This reveals that the plume and increased temperatures are first observed at 10 m, since the wind causes the plume to bend as it rises. The fire front moves directly under the tower around 1230 as revealed in the 2 m temperature (Fig. 28d). Smoke from combustion directly under the tower does not impact the tower as the wind advects the smoke downwind before it can rise 10 m, thus, the data collected as the plume impacts the instruments placed at 10 m, occurs from combustion some distance directly upwind of the tower. Similar results were found by Clements et al. (2006, 2007).

The plume impaction at 10 m was associated by an updraft and was followed by a downdraft (Fig. 28b). The downdraft occurs as TKE (Fig. 28a), sensible heat flux (Fig. 28c), and 10 m temperature (Fig. 28d) return to near-ambient levels. The maximum temperature and sensible heat flux values observed were about 50 °C, and 18 KW m⁻², respectively.

The main plume impaction on the 13 January 2009 burn occurred from about 1220 to 1224 EST. Sensible heat flux (Fig. 29c) increased to about 20 KW m⁻², and TKE (Fig. 29a) increased to about 5 m² s⁻². These maximum values correspond to a maximum 10 m temperature of almost 100 °C (Fig. 29d). During the plume impaction an updraft is observed and is followed by a downdraft of about 2 m s⁻¹ (Fig. 29b). At approximately 1219 EST, the plume moves past the tower and TKE, sensible heat flux, and temperature return to near-ambient levels. Another sharp increase in TKE (Fig. 29a) is seen at 1255 EST. Coincidentally, there is a distinct wind shift as the u wind component transitions from positive to negative and the v component shifts from negative to slightly positive at

about 1300 EST (Fig. 29b). The wind shift and increased TKE values suggest a secondary plume impacted the tower from upwind flaming and/or smoldering combustion.

The last winter burn on 14 January 2009 was arguably the most intense as the largest TKE and sensible heat flux were recorded. At about 1357 EST 10 m temperatures begin to increase (Fig. 30d). From temperature observations it appears multiple plumes impacted the tower before the main plume impacted the tower at about 1402 EST. There appears to be three major spikes in thermocouple temperature data. The first occurs from about 1338 to 1401 EST, the second from about 1403 to 1404 EST, and the last and largest from about 1405 to 1408 EST. Prior to 1357 EST the u, v, and w wind components are fairly stable (Fig. 30b) and increase significantly above background values when the plume(s) impact the tower. Updrafts occurred during the three major temperature spikes and were followed by downdrafts. Sensible heat flux (Fig. 30d) and TKE (Fig. 30a) reach maximum values of about 78 KW m^{-2} and $7.7 \text{ m}^2 \text{ s}^{-2}$ at about 1404 EST, respectively. These peaks coincide with the observed maximum updraft (Fig. 30b) and temperature (Fig. 30d) of about 4 m s^{-1} and $100 \text{ }^{\circ}\text{C}$, respectively.

Fig. 28. One min averaged tower TKE, turbulent variances, and vector wind components measured during hour when fire front moved passed the tower on 12 January 2009.

Fig. 29. One min averaged tower TKE, turbulent variances, and vector wind components measured during hour when fire front moved passed the tower on 13 January 2009.

Fig. 30. One min averaged tower TKE, turbulent variances, and vector wind components measured during hour when fire front moved passed the tower on 14 January 2009.

4. Dispersion Modeling

Once the emission characteristics have been determined it is of interest to model the concentration impact of fire on global, regional, and local scales (Wiedinmyer et al. 2006). Numerous methods and models have been used to evaluate the emissions and concentration impacts from fire (e.g., Dennis et al. 2002; Clinton et al. 2006; Roy et al. 2007; Hu et al. 2008; Yongtao et al. 2008)

a. AERMOD

The American Meteorological Society (AMS) – EPA Regulatory Model (AERMOD) was accepted as the preferred short-range regulatory dispersion model on 9 December 2006 (US EPA 2005). The AERMOD modeling system is comprised of the AMS-EPA Meteorological preprocessing model (AERMET), a terrain data preprocessor (AERMAP), and a surface characteristics preprocessor (AERSURFACE) (US EPA 2004a).

Running AERMOD requires basic inputs such as source location(s), receptor locations, and two preprocessed AERMET meteorological data files. One file consists of surface parameters (e.g., roughness length, friction velocity, Bowen ratio), and the other file contains meteorological vertical profiles. AERMOD can simulate multiple point, volume, line, and area source-types. Source emission rates may be varied by hour, month, season, or can be treated as constant throughout the modeling period. User-specified receptor locations are highly flexible, and can also be placed above the surface (i.e., flagpole receptors).

The AERMET preprocessor is designed to process National Weather Service (NWS) hourly surface observations, NWS upper air sounding data, and on-site collected data (US EPA 2004b). These data are used to characterize Planetary Boundary Layer (PBL) parameters needed for AERMOD dispersion calculations. The PBL is defined as the region between the surface and the free atmosphere, which is not affected by the surface (Garratt 1992). Sensible and latent heat and momentum fluxes determine PBL evolution and structure. Pollutants are mixed, dispersed, and transported within the PBL, where the depth can be estimated from local-scale surface characteristics. The surface characteristics required by AERMET depend on land-use type (e.g., urban area, forest, grassland) and include the albedo, the Bowen ratio, which is the ratio of sensible to latent heat flux, and the surface roughness length, which is the theoretical height above the ground where the wind speed equals zero.

From meteorological and surface data AERMET calculates the PBL parameters that influence PBL evolution and pollutant dispersion. These parameters include the sensible heat flux, the surface friction velocity, a measure of the vertical transport of horizontal momentum, the Monin-Obukhov length, the height above which convectively driven turbulence dominates over mechanically produced turbulence, the daytime mixed layer height, and nocturnal layer height (US EPA 2004b).

Mixing heights in AERMET are determined by atmospheric stability. If the atmosphere is stable, AERMET computes the mechanical mixing height, which represents the layer formed by mechanically-induced stress, dependent on wind speed and surface characteristics, where pollutants are mixed. During unstable conditions,

usually during daytime hours when the flux of sensible heat is upwards, AERMET computes both the convective and mechanical mixing heights. Output from AERMET is used to drive the dispersion model, AERMOD.

The AERMOD plume model is steady-state and assumes the horizontal and vertical concentration distributions are Gaussian during stable conditions. During unstable conditions, the distribution in the horizontal is assumed Gaussian while the vertical distribution is treated as a bi-Gaussian probability density function (US EPA 2004c).

Depending on atmospheric stability, AERMOD calculates the dispersion factors, which govern the standard deviations of the lateral and vertical concentration distributions. Dispersion factors are calculated from measures of ambient turbulence and buoyancy.

Although AERMOD is probably not applicable for high-intensity wildfires, as increased turbulence and fumigation may not be captured, AERMOD may be able to reasonably reproduce the observed concentration distribution from low-intensity prescribed fires because prescribed fires are controlled, have smaller flame heights, and liberate much less heat than natural wildfire. Dispersion modeling is currently not required by the EPA for implementing a prescribed fire, but could be used as an important tool for land managers to evaluate the concentration impact downwind.

b. Model Setup

Two of the six prescribed fires were modeled with AERMOD. The goal of the simulations is to model the PM_{2.5} concentrations observed at the downwind tripod during

the 15 July 2008 and 12 January 2009 prescribed burns. Out of the three burns during the summer and winter these represent the most ideal experiments to model because the wind direction variability was generally low and the tripod was placed in an ideal location downwind.

Surface NWS data from Albany, GA (KABY), and NWS sounding data from Tallahassee, FL (KTLH) were obtained and input to AERMET. The data obtained represent the meteorological conditions for six days starting at 0000 EST two days prior to the simulation. This allows AERMET to compute the growth and decay of the boundary layer and boundary layer parameters for days prior to and after each simulation for comparison. Timeseries of AERMET-derived sensible heat flux, an important parameter used by AERMET to estimate the depth of the PBL, for the July and January simulation are shown in Figs. 31 and 32 respectively.

Wind direction and speed, temperature, and RH obtained at the tripod were utilized as on-site data. The tripod was in operation during the duration of each burn, thus employing these on-site data assures a more representative calculation of PBL parameters in the model domain. One min averaged wind speed and direction measured at the tripod during the duration of the burn on 15 July 2008 and 12 January 2009 are shown in Figs. 33 and 34, respectively. During both burns the prevailing winds were mainly north-easterly. All tripod meteorological data were averaged to one hour to be read by AERMET. The log-law was used to estimate 10 m wind speeds from 2 m tripod measurements (Arya 1999).

The AERSURFACE algorithm was applied to obtain realistic surface characteristic values of albedo, Bowen ratio, and surface roughness length. Given the location of the tripod, AERSURFACE reads in a 30 m resolution Georgia land cover dataset from the US Geological Survey (USGS) and computes the seasonal surface characteristic values for a specified area (US EPA 2008). The area chosen was the 90 deg-wide segment of a circle with a 1 km radius that encapsulates the tripod and the burn unit. The AERSURFACE-derived surface roughness lengths for the summer and winter simulations were 0.287 and 0.254, respectively. These values agree with Hicks et al. (1975) who found the surface roughness length of a pine forest to be 0.32. With NWS surface, profile, on-site, and surface characteristic data, AERMET was run and successfully produced meteorological input files for AERMOD.

Fig. 31. Timeseries of AERMET derived sensible heat flux from 13 to 18 July 2008.

Fig. 32. Timeseries of AERMET derived sensible heat flux from 10 to 15 January 2009.

Fig. 33. Wind speed and direction (averaged to 1 min) measured at the tripod during the duration of the burn on 15 July 2008.

Fig. 34. Wind speed and direction (averaged to 1 min) measured at the tripod during the duration of the burn on 12 January 2009.

The AERMOD simulations were conducted over the duration of the prescribed burns to ascertain the period and hourly averaged $PM_{2.5}$ concentration distributions. The burn duration was well documented with numerous data loggers and in field manuals. The tripod location was recorded with a GPS unit and is represented in the model by a 2 m flagpole receptor, the height of the $PM_{2.5}$ inlet.

Since the unit was burned from the most downwind edge to the most upwind edge, numerous area sources were created in the simulations to replicate this burning pattern. Although the exact location and time of ignition is uncertain, the unit was ignited from the most downwind to the most upwind side; thus, instead of modeling the emissions from the burns as one area source over the burn-duration, numerous area sources were utilized in the runs.

The location of the units, and the area sources derived from the unit locations, were determined from GIS data and Google Earth Pro software. The first hour of each burn required the development of a blackline, using a slow-propagating backing fire; accordingly, the first hour represents the smallest area source. The total remaining area of the units were divided by the remaining hours of each burn to obtain the average acreage burned per hour (Table 15). The location of the area sources, representing the area burned from hr 1 to N, are placed upwind of the previous source and do not overlap. The edge of each area source was assumed to be oriented perpendicular to the mean wind direction.

Table 15. AERMOD burn unit section number and size delineation for the 15 July 2008 and 12 January 2009 prescribed fires.

7/15/2008		1/12/2009	
Section	Size (acres)	Section	Size (acres)
1	23	1	49
2	47	2	75
3	47	3	75
4	47	4	47
5	47		

The burn on 15 July 2008 took five hours to complete. The tripod collected data for an additional hour, following the conclusion of flaming, during smoldering. The unit was divided into five sections (Fig. 35), and was simulated by initiating (i.e., “turning on”) the units from one to five. The 12 January 2009 burn took four hours to complete, and was divided into four sections (Fig. 36). The burn crew was initially not going to ignite the area to the southeast, but due to time, they were able to burn this area last; therefore, the fire was simulated by initiating units one to three over the first three hours, and the unit four, to the southeast, was initiated during the last hour of the burn.

Fig. 35. Five divided sections of the burn unit representing the 15 July 2008 prescribed burn. The fire is simulated as progressing from unit 1 to 5.

Fig. 36. Four divided sections of the burn unit representing the 12 January 2009 prescribed burn. The fire is simulated as progressing from unit 1 to 4.

Since emissions and impacts from flaming and smoldering are quite different, an effort was made to model both flaming and smoldering emissions. Each burn-area section corresponds to two area sources in AERMOD. Each section was represented with a flaming source and a smoldering source with the exact same dimensions such that the 15 July 2008 burn was modeled using 10 areas sources, two for each section. During the first hour of each simulation the most downwind section (section 1) is treated as both flaming and smoldering by “turning on” the flaming and smoldering area sources corresponding to that section. The emission rates of all other sources are set to zero during this hour. The next hour, the upwind adjacent section (section 2) is treated as both flaming and smoldering, while the previously flaming area source, active during the first hour in section 1, is “turned off”. Each section was allowed to smolder for three hours total, two hours of which are not accompanied by flaming emissions. The emissions from smoldering during the first two hours are equivalent while the smoldering emission rate during the third hour was assumed to be half of the emission rate of the first and second hours. By hour three of the simulation, section 1 and 2 are only smoldering while the section upwind and adjacent to section 2 (section 3) is flaming and smoldering. This procedure is repeated until the time period of the burn is complete.

The area emission rates, applied to each area source, were derived from calculated emission factors and pre and post clip-plot-derived fuel consumption by

$$AE = \frac{EF_t * FC}{t},$$

where AE is the area emission rate in $\text{g s}^{-1} \text{ m}^{-2}$, EF_t is the emission factor in g kg^{-1} , FC is the fuel consumption in kg m^{-2} , and t is the time in seconds, where the EF_t represents the emissions from both flaming and smoldering for each section. To separate flaming and smoldering emissions, W_v , summed during flaming and smoldering, was used. The modeled flaming emission factor was calculated by

$$EFm_f = \frac{\sum W_{vf}}{\sum W_{vf} + \sum W_{vs}} * EF_t,$$

where EFm_f is the modeled EF_f , $\sum W_{vf}$ is the sum of fuel consumption during the flaming stage, and $\sum W_{vs}$ is the sum of fuel consumption during the smoldering stage. The modeled smoldering emission factor was calculated by

$$EFm_s = \frac{\sum W_{vs}}{\sum W_{vf} + \sum W_{vs}} * EF_t,$$

where EFm_s is the modeled EF_s ; thus, the Area Emission rates due to flaming (AE_f) and smoldering (AE_s) are given by

$$AE_f = \frac{EFm_f * FC}{t}$$

$$AE_s = \frac{EFm_s * FC}{t}.$$

The EF_{ms} were divided by two to account for the smoldering emissions being applied over the first and second hour of smoldering. For example, the EF_t for the 15 July 2008 burn was determined to be $12.1 \pm 2.6 \text{ g kg}^{-1}$. Using the mean EF_t value, the EF_{mf} was calculated to be 10.32, and the EF_{ms} was $1.77/2 = 0.89$. If we apply smoldering over two hours the total of each section is $10.32 + 0.89 + 0.89 = 12.1 \text{ g kg}^{-1}$; however, since data captured during the smoldering phase could not be used to evaluate smoldering on periods longer than 20 mins, because the tower had become a downwind receptor, emissions from smoldering may be underestimated. To account for this possible underestimation, a third hour of smoldering emissions was allowed where the $EF_{ms} = EF_{ms}/4$. In the example, this would allow an extra 0.44 g kg^{-1} of smoldering-type emissions from each section. The new total, for the example, is $10.32 + 0.89 + 0.89 + 0.44 = 12.54 \text{ g kg}^{-1}$, which is well with the range of the calculated value of $12.1 \pm 2.6 \text{ g kg}^{-1}$.

One drawback in AERMOD is that the exit temperature and exit velocity of emissions from area sources cannot be defined. Even during a low-intensity prescribed fire, the heat liberated from flaming combustion will give the plume extra buoyancy and allow it to rise. The height to which the plume will rise is dependent on atmospheric stability, momentum, and buoyancy (Arya 1999). A simple way to account for plume rise is to specify a release height for flaming sources, some distance from the ground.

Two methods were used to estimate plume heights. During each burn a radiosonde was launched downwind of the unit into an established plume aloft. The radiosonde calculates height above the surface via GPS, as well as high-frequency

temperature and moisture data. As the radiosonde ascended through the plume, temperatures and moisture increased. Moisture is released during combustion, so one would expect to observe enhanced moisture values in a plume. The increased moisture values were corroborated with temperature data to ensure the signature of the phenomenon was due to the plume aloft.

Additionally, the Fire Emissions Production Simulator (FEPS) (Anderson et al. 2004) was used to model each case and estimate plume rise. The FEPS system provides dynamic simulations of fuel consumption, emissions, and plume buoyancy. Fuel loading and moisture, fire growth rate, fuel consumption, and meteorology were input to FEPS for each simulation. The Briggs maximum plume rise, was solved by FEPS for each case.

From radiosonde data, the plume signature aloft on 15 July 2008 and 12 January 2009 extends from 100 – 200 m, and 75 – 175 m AGL, respectively. Similarly, the Briggs maximum plume rise from FEPS on 15 July 2008 and 12 January 2009 was 202, and 187 m, respectively. The plume heights from these two methods are in agreement, however, FEPS-derived plume heights were used to specify the release height of the flaming area sources.

The release height of all smoldering sources was assumed to be 10 m. A sensitivity analysis was conducted to ensure the release height of the smoldering emissions does not affect the overall results by a significant margin. There was little change in the results when smoldering release heights were incremented from 0 – 20 m AGL. Emissions from smoldering do not tend to loft like flaming emissions but are subject to a more drift-like pattern.

c. Model Results

The period averaged PM_{2.5} concentration distribution at 2 m for the AERMOD prescribed fire simulation on 15 July 2008 is displayed in Fig. 37. Due to northeasterly winds, pollutants were transported to the southwest. The observed period-averaged PM_{2.5} concentration at the tripod during the burn was 119 $\mu\text{g m}^{-3}$, while the modeled period-averaged PM_{2.5} concentration at the tripod-receptor was 101 $\mu\text{g m}^{-3}$. Since the emissions were calculated by first subtracting ambient pre-fire concentrations to model fire-only impacts, pre-fire ambient concentrations were also subtracted from tripod concentrations observed during the fire. The period averaged concentrations were very well captured by AERMOD for this case.

Hourly averaged concentrations were also evaluated for the 15 July 2008 burn (Fig. 38). Modeled hourly averaged concentrations match very well to observations. The model seems to underestimate the concentration downwind during the first three hours of the burn and overestimate during the fourth hour. During Hour 6, when there is no emission due to flaming, AERMOD was able to do a reasonable job simulating the impact from smoldering, but the modeled value is a little more than half of the observed. This may indicate that emissions due to smoldering may still be underestimated. Overall, AERMOD was able to reproduce the observed hourly averaged concentration structure well.

Fig. 37. Period averaged PM_{2.5} concentration distribution at 2 m for the prescribed fire AERMOD simulation on 15 July 2008. The blue triangle represents the location of the tripod and the contours are in units of $\mu\text{g m}^{-3}$.

Fig. 38. Hourly averaged $PM_{2.5}$ observations and AERMOD output for the 15 July 2008 burn. The blue line represents the total modeled impact due to flaming emissions (red), and smoldering (brown). The green and purple lines depict the observed and modeled period average, respectively.

The period averaged $PM_{2.5}$ concentration distribution at 2 m for the AERMOD prescribed fire simulation on 12 January 2009 burn is shown in Fig. 39. The observed period-averaged $PM_{2.5}$ concentration at the tripod during the burn was $98 \mu\text{g m}^{-3}$, while the modeled period-averaged $PM_{2.5}$ concentration at the tripod was $89 \mu\text{g m}^{-3}$. The period averaged concentrations were well simulated by AERMOD for this burn.

The hourly averaged concentrations are shown in Fig. 40. Modeled hourly averaged concentrations do not seem to agree as well to observations as the first case. The model was able to capture the general pattern, where the maximum one hour concentration is observed during the second hour of the burn. The model overestimates

concentrations during the first and third hour, while severely underestimating the concentration during the second hour. Since the exact area that was burned per hour was unknown, the sources had to be divided up equally. It is quite possible the area modeled during the first and third hour were too large and that more acres were burned during the second hour. Overall, since the modeled period average is close to the observed, AERMOD seems to be able to evaluate the dispersion very well for this case.

Fig. 39. Period averaged PM_{2.5} concentration distribution at 2 m for the prescribed fire AERMOD simulation on 12 January 2009. The blue triangle represents the location of the tripod and the contours are in units of $\mu\text{g m}^{-3}$.

Fig. 40. Hourly averaged $PM_{2.5}$ observations and AERMOD output for the 12 January 2009 burn. The blue line represents the total modeled impact due to flaming emissions (red), and smoldering (brown). The green and purple lines depict the observed and modeled period average, respectively.

5. Conclusion

In this study, effects of seasonal-related variations and of turbulence generation on emissions and impacts from prescribed fires were evaluated. High frequency *in situ* data obtained from three summer (July 2008) and three winter (January 2009) prescribed fires were used to evaluate emissions and fire-atmosphere interactions. Two cases were selected to evaluate the performance of the EPA-approved short-range regulatory dispersion model, AERMOD, for prescribed fire. The following are the key findings of this study:

- MCE values during smoldering were lower than flaming, which suggests data segregation utilizing sensible heat flux, temperature, *in situ* concentrations, and MCE is applicable if a visual confirmation between flaming and smoldering cannot be made. MCE values during winter fires were larger than summer fires. This is most likely due to differences in fuel, as summer fuels were much greener and had higher moisture contents.
- The $PM_{2.5}$ EF_{st} for all winter fires were lower than summer fires. During the flaming stage, the mean winter EF_f was about half the mean summer EF_f . The mean winter EF_t was $7.7 \pm 1.3 \text{ g kg}^{-1}$, while the mean summer EF_t was $12.1 \pm 1.0 \text{ g kg}^{-1}$. $PM_{2.5}$ emissions were thus found to lower in the winter prescribed burns, as expected, since summer MCEs were lower.
- For the winter data, a statistical analysis was performed to test the relationship between turbulence generation, TKE, and other measured quantities. The

correlations between TKE and all concentration data, and notably, MCE, are statistically significant to the 99 % confidence interval, thus suggesting turbulence generation, fire efficiency, and emissions from fires, in this experiment, are related.

- During all burns, TKE, sensible heat flux, u, v, and w wind components, and temperature increased well above background levels when the plume impacted the tower. An updraft was also observed inside the plume and a downdraft was observed as the plume propagated beyond the tower. This corroborates results found by Clements et al. (2006, 2007, 2008).
- The AERMOD dispersion model was able to reproduce the observed period averaged concentration and the hourly averaged concentration structures for prescribed fire. AERMOD may be applicable for modeling short-range impacts from low-intensity prescribed fire, and could possibly be used as a tool for land managers who prescribe fire.

For future studies, it is recommended higher-quality pollutant samplers are used. Concentrations derived from the DUSTRAK are based on an internal calibration using coarser particles than those observed during biomass combustion. Likewise, the resolution and response time of the HOBO CO monitor was poorer than desired, and, as temperature increase above 20 °C, an additional temperature-induced error may exist. Temperature readings from the tower show temperature increases well above 20 °C, which could affect calculations of MCE, EFs, and fuel consumption.

To calculate emission factors precisely requires knowledge of all species emitted containing carbon. In this study only CO and CO₂ were measured, which thus required use of a factor to estimate the total amount of carbon contained in the other species. This introduces uncertainty in the calculations. To reduce this uncertainty, other carbonaceous species should be evaluated.

The size and location of emission sources for the AERMOD simulations had to be estimated. For future experiments, it is recommended burn crews mark their positions during ignition with a GPS unit, such that their location and the area burned per hour can be better estimated. AERMOD output could only be compared to one downwind receptor, the tripod. To better evaluate the applicability of AERMOD for prescribed fire, more experiments should be conducted with more downwind receptors.

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APPENDIX: ACRONYMS

AE_f	Area Emission rate due to flaming
AE_s	Area Emission rate due to smoldering
AERMAP	AMS-EPA terrain data preprocessor
AERMET	AMS-EPA Meteorological preprocessing model
AERMOD	AMS-EPA Regulatory Model
AERSURFACE	AMS-EPA surface characteristics preprocessor
AGL	Above Ground Level
AMS	American Meteorological Society
BC	Black Carbon
$BC_{2.5}$	Black Carbon particulates having an aerodynamic diameter less than 2.5 μm
CE	Combustion Efficiency
CMAQ	Community Multiscale Air Quality model
CN	Condensation Nuclei
EF	Emission Factor
EF_f	Emission Factor from flaming emissions
EF_s	Emission Factor from smoldering emissions
EF_t	Effective Emission Factor representing flaming plus smoldering emissions
EPA	Environmental Protection Agency
EST	Eastern Standard Time

F	Flaming
FEPS	Fire Emissions Production Simulator
FRI	Fire Return Interval
GFC	Georgia Forestry Commission
MCE	Modified Combustion Efficiency
MODIS	Moderate Resolution Imaging Spectroradiometer
NAAQS	National Ambient Air Quality Standards
NWS	National Weather Service
OC	Organic Carbon
PBL	Planetary Boundary Layer
PM	Particulate Matter
PM _{2.5}	Particulate Matter having an aerodynamic diameter < 2.5 µm
PM ₁₀	Particulate Matter having an aerodynamic diameter < 10 µm
POM	Particulate Organic Matter
S	Smoldering
SD	Standard Deviation
SE	Standard Error
TKE	Turbulent Kinetic Energy
USGS	US Geological Survey
VISTAS	Visibility Improvement State and Tribal Association of the Southeast