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## 1. Abstract (one to three paragraphs)

Emissions from wildland (wild and prescribed) fires add to the burden of air pollution and can have adverse impacts on air quality and public health. Numerical models for dispersion and chemical transport, also known as air quality models, can be used to investigate the fire plume evolution and the smoke impacts. However, it is important that the predictive skills of smoke models be evaluated under a wide range of applicable conditions through systematic simulations of past events with existing data. Three models were evaluated in this research: CALPUFF, DAYSMOKE and CMAQ. Different prescribed burn and wildfire episodes occurring throughout the Southeastern US were simulated with the models to evaluate their performance. The abilities of the models to predict the observed PM<sub>2.5</sub> levels were evaluated in detail by pairing model predictions at monitoring locations with observations. Models were also assessed at a diagnostic level by analyzing whether they succeeded in predicting observed PM<sub>2.5</sub> levels for the right reason or, if they did not succeed, why they failed. The sensitivities of PM<sub>2.5</sub> predictions to fire emissions were estimated using brute-force and decoupled direct methods.

From the results of this study CALPUFF could not be determined to be a suitable model for simulating the air quality impacts of fires. DAYSMOKE matched the field measurements of plume tops and ground-level PM<sub>2.5</sub> concentrations quite well and can be used for injecting fire emissions into regional-scale air quality models. Model evaluation indicated that DAYSMOKE can be turned into a reliable a short-range smoke-impact prediction tool for land managers. On a regional scale, PM<sub>2.5</sub> impacts of prescribed burns and wildfires are best predicted by air quality models such as CMAQ. The standard version of CMAQ underestimated prescribed fire impacts and severely under-predicted wildfire impacts. CMAQ-APT, in its publicly available form, is not suitable for tracking fire plumes. AG-CMAQ requires that adaptation parameters be adjusted for adequate modeling of wildfires. Additionally, CMAQ predictions were found to be sensitive to the vertical distribution of emissions and the contributions calculated from first-order sensitivities fully explained the modeled PM<sub>2.5</sub> impacts. An uncertainty between 30-50% was determined for fire emissions estimated with existing tools and study results indicated that modeled air quality impacts are a linear function of emissions in this range of uncertainty.

## 2. Background and purpose

Emissions from wildland (wild and prescribed) fires add to the burden of air pollutants such as fine particulate matter (PM<sub>2.5</sub>), carbon monoxide (CO), volatile organic compounds (VOC) and nitrogen oxides (NO<sub>x</sub>). These pollutants are known to have adverse impacts on air quality and public health in urban areas and they can reduce visibility at national parks and vistas. Numerous field and laboratory studies have been carried out to measure the air pollutants emitted from various types of burning vegetation (e.g., *Urbanski et al., 2009; Burling et al., 2010*). Fire emissions can be estimated by utilizing emission factors (i.e., amounts of pollutants emitted per amount of material burned) derived from these studies in combination with other information such as the fuel load and consumption, the size of the burn, and its location and timing. An example of this process can be found in *Clinton, et al. (2006)* where a GIS based emissions estimation system was implemented to quantify emissions of 10 pollutant species from wildland fires that burned more than 200,000 hectares in Southern California, in 2003. Recently, satellite databases have also been used to provide fire-related information and improve fire emissions estimates. For example, *Wiedinmyer, et al. (2006)* developed a framework for estimating PM and gaseous fire emissions by utilizing the MODIS Thermal Anomalies Product (to get the burn location, size and timing

information) and the Global Land Cover Characteristics 2000 datasets and the MODIS Vegetation Continuous Fields Product (to get the fuel loading information).

Once the fire emissions are estimated, numerical models for dispersion and chemical transport, also known as air quality models, can be used to investigate the fire plume evolution and smoke impacts. *Junquera, et al.* (2005) used an air quality model to assess the wildland fire impacts on the CO and PM levels in southeast Texas. The concentrations predicted by the air quality model based on fire emissions estimates were within a factor of two of the observed values from aircrafts and surface sites. *Mason, et al.* (2006) compared two independently developed biomass-burning smoke plume models to simulate the evolutions of two plumes which originated from Alaskan boreal forest and African savanna, respectively. The differences between the two models' predictions were smaller than the uncertainties in available field measurements. *McKenzie, et al.* (2006) developed a modeling framework to predict the regional haze produced by smoke from wildfires and prescribed fires. However, there were large uncertainties in estimating smoke emissions and simulating the dispersion of the smoke plume

In order to build confidence in smoke models their predictive skills must be demonstrated under a wide range of applicable conditions. This requires simulations of past events and systematic evaluations of model predictions with existing smoke data. A model that performed well under a variety of conditions in the past is likely to do so in future applications. However, the level of confidence in a future prediction (e.g., a smoke forecast) is difficult to quantify. To be used in effective decision making the uncertainty in model predictions must be quantified. Uncertainties in fuel loads, fuel consumption, and emission factors limit our ability to provide the models with accurate emissions inputs. There are also various other uncertainties in meteorological inputs, and parameters related to modeling of smoke transport and dispersion. In addition, there are uncertainties in chemical reactions and phase transformations (gas to particle and vice versa) during the modeling of ozone and secondary particulate matter formation. A model that has strong skill in simulating dispersion may have weakness in chemistry and vice versa. It is therefore important to study a range of models, compare their performances, identify their strengths and weaknesses, quantify their uncertainties and try to make up for the weakness of one model with the strength of another for more accurate predictions of ultimate smoke impacts.

The first objective of this research is to evaluate a range of smoke models with existing datasets for prescribed burns and wildfires in the Southeast. The evaluation will provide information on the performances of these models in predicting past events. The second objective is to assess the uncertainties in model outputs stemming from uncertainties in emission inputs. The uncertainty information is critical for determining the reliability of future predictions by these models. Knowledge of uncertainties in model predictions can give decision-makers new perspectives for interpreting model results, which can lead to more effective decisions. Also, a ranking of the sources of uncertainty can help to set priorities for future research in emissions measurements.

The Monte Carlo method is the traditional way of investigating the impacts of parameter uncertainties on model outputs. In the past, it has been used to propagate uncertainty across certain dispersion models (e.g., *Bergin, et al.*, 1999; *Sax and Isakov*, 2003), and to estimate the contributions of variability and uncertainty from emissions, meteorology, and various model parameters. Applications of the Monte Carlo method in air quality modeling are limited. *Hanna, et al.*, (1998) carried out 50 runs of an ozone air quality model by random sampling of 109 parameters from their assumed distributions. *Rodriguez, et al.*, (2007) investigated the uncertainty and

sensitivity of ozone and PM<sub>2.5</sub> to variability in selected input parameters using a similar methodology with an air quality model. However, no studies have been conducted so far to investigate the uncertainty in fire impact predictions.

### 3. Study description and location

#### Models

Three models were evaluated in this research: CALPUFF, DAYSMOKE and CMAQ.

**CALPUFF** (*Scire et al.*, 2000) is a three-dimensional transient dispersion model that views the plumes as a series of puffs. The effects of changing meteorological conditions, both in time and in space, and the effects of topography around the source are considered during the dispersion of the puffs. During the simulation, the puffs may be split or merged due to wind shear. CALPUFF provides pollutant removal capability due to dry and wet deposition and limited chemical transformation capability, primarily for SO<sub>2</sub> to sulfate conversion as well as some limited NO<sub>x</sub> to nitrate conversion. CALPUFF is not recommended for dispersion applications beyond 100 km from the source. As implemented in the BlueSky Framework (<http://www.airfire.org/bluesky>), PM<sub>2.5</sub> is modeled with no chemical transformations to predict ground level concentrations due to smoke from fires.

**DAYSMOKE** (*Achtemeier et al.*, 2011) is a dynamic/stochastic plume model specifically developed for prescribed burns. The plume boundary is determined by a dynamic plume-rise algorithm. Deformation (e.g., looping and meandering) of this boundary is achieved through a large-eddy parameterization. Particles released from the burn site at the rate of one per kg of PM<sub>2.5</sub> emissions can randomly cross the plume boundary due to stochastic plume turbulence. Those that remain within the boundary continue to be followed by the model. Multiple plume boundaries can be defined to model multiple-core updrafts typical of prescribed burns. DAYSMOKE is a non-reactive model which assumes that the particles are chemically inert and it has no provisions for evaporation of particles or condensation of gases. Due to these limitations, DAYSMOKE should not be used for tracking plumes beyond 15-20 km from the source.

**CMAQ** (*Byun and Schere*, 2006), or Community Multiscale Air Quality, is an Eulerian grid model based on first principles for simulating the long-range transport and chemical transformation of plumes. When applied to large regions, typical grid sizes are not adequate for resolving the near-source dispersion of plumes. CMAQ has plume-rise algorithms and reactive sub-grid scale plume treatments that were designed for plumes from industrial stacks. These may or may not be suitable for prescribed or wildfire plumes. Most of the time the sub-grid scale plume models are turned off; in which case the fire plumes are immediately mixed into a vertical column of grid cells according to the fractions determined by the plume-rise algorithm. Also, terrain effects can only be resolved to the extent allowed by the grid sizes. Despite these limitations, CMAQ is the only model among the models considered here that can properly account for the complex chemical reactions, transformations between gas and aerosol phases and interactions from clouds. Considering the potential long-range impacts of prescribed burns and wildfires, especially on ozone and PM levels, CMAQ is a valuable model which can provide the secondary pollutant impacts that CALPUFF and DAYSMOKE cannot. It should be noted that about one quarter to one half of fire-related PM impact can be in the form of secondary organic aerosol that, as ozone, is not a directly emitted pollutant but is formed in the atmosphere.

To assure a consistent systematic evaluation, the same emissions and meteorological inputs were used with all the models. The Weather Research and Forecasting model (WRF; [wrf-model.org](http://wrf-model.org)) was used to generate the meteorological inputs. After comparing the performances of WRF and the PSU/NCAR mesoscale model (MM5; [www.mmm.ucar.edu/mm5](http://www.mmm.ucar.edu/mm5)) both with and without assimilation of reanalysis data, it was decided that WRF with reanalysis data provides the best meteorological input data set for the models. The largest uncertainty is in wind direction with  $\pi/8$  radians or more differences from observations being typical in hourly values. The loads for fuels in Fuel Characteristic Classification System (FCCS) were estimated using photo-series developed for Eastern fuels. CONSUME 3.0 ([www.fs.fed.us/pnw/fera/research/smoke/consume](http://www.fs.fed.us/pnw/fera/research/smoke/consume)) was used to estimate fuel consumptions. Fire emissions were estimated using the Fire Emission Production Simulator (FEPS; [www.fs.fed.us/pnw/fera/feps](http://www.fs.fed.us/pnw/fera/feps)) and emission factors derived from measurements in the southeast (Urbanski, et al. 199X) The amount of fire emissions were the same in all models but how these emissions were input to the models may be different depending on the processing of emissions by each model. All other (non-fire) emissions were generated from the 2005 National Emissions Inventory and projected to the actual year using the Sparse Matrix Operator Kernel Emissions (SMOKE; [www.smoke-model.org](http://www.smoke-model.org)) model.

## Episode Selection

In a related project sponsored by the Strategic Environmental Research and Development Program (SERDP), 11 burns were monitored during the 2008-2009 burn seasons at Ft. Benning Army Base (AB) near Columbus, Georgia. Out of these burns, 10 resulted in plumes directed away from the Columbus urban area. While this was certainly an operational success for the land managers, these burns did not fit our criterion for episode selection because they did not leave any trace at regional observational networks. Only one burn conducted on April 9, 2008 had a plume directed towards Columbus under east-southeasterly winds and had a possible impact at the PM monitor located at Columbus Airport. Around the evening hours there was a  $10 \mu\text{g m}^{-3}$  jump in hourly-averaged  $\text{PM}_{2.5}$  concentrations, which may have been due to the burn. A trajectory analysis revealed that the plume could have impacted the monitor and simple plume calculations suggested that the jump could be explained by the  $\text{PM}_{2.5}$  emissions from the burn. Therefore, this case was selected as an evaluation episode. The remaining 10 burns where  $\text{PM}_{2.5}$  measurements were made in the 1-5 km range from the burn sites were used for calibration and short-range evaluation of Daysmoke.

Since land managers are extremely careful to avoid smoking urban areas and affecting human health and given that observational networks are denser around those urban areas and sparse elsewhere, prescribed burns that impacted the  $\text{PM}_{2.5}$  network monitors are not very common. Accurate meteorological forecasts and smoke modeling tools such as Hysplit and V-Smoke help land managers to conduct successful burns. However, forecasts can occasionally be wrong. On February 28, 2007, smoke from two prescribed burns, one in the Oconee National Forest (NF) and the other in Piedmont National Wildlife Refuge (NWR) both about 80 km southeast of Atlanta, resulted in a remarkable episode. These burns totaling around 3000 acres were ignited that morning shortly before the winds shifted to blow from the southeast. Later in the afternoon, smoke covered the city of Atlanta and most of the surrounding metro areas. Monitored fine particulate matter ( $\text{PM}_{2.5}$ ) levels soared up to  $150 \mu\text{g m}^{-3}$  at several sites and ozone ( $\text{O}_3$ ) levels increased by 30 ppb. Preliminary analysis (*Hu et al.*, 2008) estimated that more than 1 million people were exposed to unhealthy ( $> 35 \mu\text{g m}^{-3}$ ) levels of  $\text{PM}_{2.5}$  for several hours. Asthma attacks apparently triggered by the smoke were reported by clinics receiving pulmonary patients the next day.

While it is difficult to find evaluation cases involving prescribed burns, there are sufficient cases where wildfire impacts are detected by routine observational networks. The 2007 Georgia-Florida wildfires burned about one half million acres from mid April through the first week of June. These fires have caused dozens of hazy days in cities across Alabama (AL), Florida (FL) and Georgia (GA). The following 5 cases were selected as wildfire smoke episodes to be used in model evaluation. The first case was a fire impact recorded at Macon, GA on May 12. On this day observed 1-hr  $PM_{2.5}$  concentrations increased to around  $350 \mu g m^{-3}$ . The second case selected was a smoke hit observed in Birmingham, AL a few days later on May 15. Here the impact on air quality was noticeably lower than during the previous case, yet  $PM_{2.5}$  levels exceeded  $80 \mu g m^{-3}$ . The following day, May 16, an impact at Atlanta, GA was registered with  $PM_{2.5}$  concentrations increasing above  $120 \mu g m^{-3}$ . An additional case was selected on May 16, when a strong hit was observed in Savannah, GA leading to  $PM_{2.5}$  concentrations greater than  $250 \mu g m^{-3}$ . Finally, a second impact at Birmingham, AL in which  $PM_{2.5}$  levels rose to more than  $150 \mu g m^{-3}$  was also chosen as the fifth study case.

There were also some episodes what were not selected since they would be extremely challenging for the models. For example, the May 22, 2007 episode in Atlanta occurred after the plume was transported into Alabama under southeasterly winds on May 21 and veered towards Atlanta the next day under westerly winds. This long-range transport event with nocturnal transport in the free-troposphere and fumigation the next morning is very difficult to simulate with CMAQ since the vertical resolution of the model is limited above the boundary layer.

## Model Evaluation

Model predictions of  $PM_{2.5}$  were compared to observations of hourly-averaged  $PM_{2.5}$  concentrations available from State and Local Air Monitoring Stations (SLAMS) network. These stations measure  $PM_{2.5}$  mass using the Federal Reference Method (FRM). Observations of  $PM_{2.5}$  composition (as sulfate, nitrate, organic and elemental carbon, and crustal material) are also available from networks such as the Speciated Trends Network (STN), the Interagency Monitoring of Protected Visual Environments (IMPROVE) and some special networks in the Southeast such as the Southeastern Aerosol Research and Characterization (SEARCH) network and the Assessment of Spatial Aerosol Composition in Atlanta (ASACA) network. Observations of chemical composition are ideal for the examination of the ability of the models to accurately simulate relevant atmospheric processes. Real-time  $PM_{2.5}$  mass data collected by portable smoke monitors during prescribed burns were used to evaluate dispersion models in the short-range. Aerosol optical depth data available from satellites such as GOES and MODIS were used in visualizing the trajectories of fire plumes.

The performances of the models in predicting the observed  $PM_{2.5}$  levels were evaluated in detail by pairing model predictions at monitoring locations with observations. Statistical measures such as normalized mean bias and error, fractional bias and error, and root-mean-square error were calculated for each model and the performances of the models were compared against each other. For each simulated case, the models were also evaluated at a diagnostic level by analyzing whether they succeeded in predicting observed  $PM_{2.5}$  levels for the right reason or, if they did not succeed, why they failed. Extensive three-dimensional visualizations of plume transport were used for this purpose.

## Sensitivity and Uncertainty Analysis

Sensitivities to emissions (meteorological inputs, initial or boundary conditions, or model parameters) can be determined by running the models with the base-case emissions and then with altered emissions and taking the difference between the results of these two simulations. This method is referred to as the “brute-force method” (BFM). This method can be tedious and expensive since it requires an additional model run for every sensitivity that needs to be calculated. The Decoupled Direct Method (DDM) is an efficient alternative that can calculate several sensitivities in a single model run. The implementation of DDM in air quality models was initiated by *Yang et al.* (1997) to study the sensitivities of ozone to NO<sub>x</sub> and VOC emissions. Recently, a DDM implementation for sensitivities of both gas and aerosol species was incorporated in CMAQ and evaluated for ozone and PM sensitivities to various emission types (*Napelenok et al.*, 2008). Here, both BFM and DDM were used to calculate the sensitivities of PM<sub>2.5</sub> predictions to fire emissions.

In this analysis, fire-related VOC, NO<sub>x</sub>, and PM emissions were lumped into a single parameter referred to as “fire emissions”. The reason for this is, when the sensitivities of PM<sub>2.5</sub> concentrations to the fire are calculated, to be able to capture the sensitivity of not only primary but also secondary components of PM<sub>2.5</sub> to fire emissions. Perhaps the most important secondary component of PM<sub>2.5</sub> formed from gaseous fire emissions is secondary organic aerosols (SOA). Fire emissions injected into different vertical layers were considered as different sensitivity parameters. Typically emissions are injected into 10-15 layers of the grid column above the fire location. Calculating sensitivities for 10-15 parameters is affordable for most of today’s computer systems, even with BFM. Time-varying 3-D sensitivity fields of PM<sub>2.5</sub> mass to each parameter were generated though only the ground-level sensitivities were analyzed.

DDM sensitivities calculated in CMAQ for PM<sub>2.5</sub> are first order sensitivities. By definition, a first order sensitivity is the local derivative of PM<sub>2.5</sub> concentration with respect to an emission (or other) parameter. Unless PM<sub>2.5</sub> concentration is a linear function of the parameter, DDM sensitivity can be useful only to calculate the PM<sub>2.5</sub> response to a small change in the parameter. BFM sensitivities were calculated for different levels of change in emissions parameter as follows:

$$S_j^{(1)} = \frac{\partial C}{\partial \lambda_j} = \frac{C(\lambda_j + \Delta\lambda_j) - C(\lambda_j - \Delta\lambda_j)}{2\Delta\lambda_j} \quad (1)$$

Here  $S_j^{(1)}$  is the first-order sensitivity,  $C$  is concentration,  $\lambda_j$  is the  $j^{\text{th}}$  emission parameter, and  $\Delta\lambda_j$  is the amount of perturbation used to calculate the change in concentration. The emissions parameters were changed (increased and decreased) by 10%, 30% and 50%, respectively. If the sensitivities calculated from each level of change are similar, this would indicate a linear relationship between PM<sub>2.5</sub> concentration and the emissions parameter. If they are different, this would imply a nonlinear relationship.

Sensitivities do not only determine which parameters are more important to the prediction but they also provide a basis for the uncertainty analysis. A simplified model that can be used in Monte Carlo analysis was built as follows.

$$C^* = C_0 + \sum_{j=1}^N S_j^{(1)} \Delta\lambda_j \quad (2)$$

Here  $C^*$  is the simplified model estimate for pollutant concentration after fire emission parameters are perturbed by  $\Delta\lambda_j$ ,  $C_0$  is the pollutant concentration predicted using nominal values of the parameters, and  $N$  is the total number of parameters. Note that this simplified model ignores the high order terms in the Taylor series expansion. Consequently, it becomes less accurate for larger perturbations (i.e., deviations from nominal values) as non-linear effects become important.

If the likely ranges of fire emission parameters can be determined and the simplified model is accurate within these ranges, the following Monte Carlo analysis can be used to determine the uncertainties in model predictions. First, the uncertainties of the parameters can be represented as probability density functions (PDFs). The data available for fire emission parameters are insufficient to determine the shapes of the PDFs but normal or lognormal distributions can be assumed as these are the most common types of functions used in the literature. Using these PDFs, random perturbation values can be selected for different emission parameters. Then, the simplified model above (Equation 2) can be used to generate estimates for  $PM_{2.5}$  concentration for each selected set of perturbed parameter values. After a large number of estimates are generated with the simplified model, a PDF of  $PM_{2.5}$  concentration would be obtained, and this would represent the uncertainty in  $PM_{2.5}$  predictions. Note that performing a sufficient number of simulations with the actual dispersion or air quality model for generating the PDF of  $PM_{2.5}$  concentration would be very time consuming and expensive; this is why a simplified model is sought.

## 4. Key findings with one-two paragraph discussion of each

### Model Evaluation

#### CALPUFF

**It could not be determined whether CALPUFF is a suitable model for simulating the air quality impacts of fires.**

The CALPUFF model was explored as a tool for simulating the air quality impacts of wildland fires. Although the model was originally intended for industrial pollution stacks, several options have been added in an attempt to extend its applicability to fires. These include treatment of buoyant area sources, plume rise algorithms, and a processor to use outputs from FEPS as inputs into CALPUFF. Although specific fire sources can be modeled, it is not possible to include all emissions sources within a domain of interest and therefore simulate the full interactions of a complete atmospheric field. A model postprocessor allows simulated pollutant concentrations to be added to a background concentrations field; however, such background concentrations fields must be provided in CALPUFF-specific format and are not readily available. Dry and wet deposition processes simulated by the model are limited. For particulate species, deposition rates are only available for  $PM_{10}$  and not for  $PM_{2.5}$ . Chemical transformation processes, including ozone chemistry and secondary organic aerosol formation, are not explicitly simulated; instead, they can only be treated with highly simplified parameterizations. In all fire simulations attempted with CALPUFF, the model consistently under-predicted ground-level pollutant concentrations. Modeled  $PM_{2.5}$  fire-attributable impacts at urban monitors were typically lower than  $10 \mu\text{g m}^{-3}$ , while observed concentrations at the sites frequently increased above  $100 \mu\text{g m}^{-3}$  during the episodes simulated. It may be possible to achieve better model performance with different model configurations or simulation setups.

## DAYSMOKE

### **DAYSMOKE matched the field measurements of plume tops and ground-level PM<sub>2.5</sub> concentrations quite well.**

As part of the SERDP project, DAYSMOKE was used to simulate the 11 prescribed burns monitored at Ft. Benning AB and was evaluated by comparing simulated plume heights and ground-level PM<sub>2.5</sub> concentrations to the measurements. Since DAYSMOKE is a stochastic model, it was run several times for each burn and the results of the ensemble were used for the comparisons. Sampling on the ground was conducted by mobile real-time PM<sub>2.5</sub> instruments positioned within 1-7 km downwind distance of the burn sites. DAYSMOKE plume tops were slightly lower than the measured plume tops. Simulations of ground-level PM<sub>2.5</sub> compared favorably with measurements. The difference between the DAYSMOKE ensemble average PM<sub>2.5</sub> and measurements over the period of the burns was  $\pm 5 \mu\text{g m}^{-3}$  for burns where ground-level PM<sub>2.5</sub> concentrations increased with distance from the burn and did not exceed  $\pm 10 \mu\text{g m}^{-3}$ . For burns with extreme decrease in ground-level PM<sub>2.5</sub> concentrations, agreement with measurements was greatly increased by triggering the highly tilted plume option of DAYSMOKE.

### **DAYSMOKE can be used for injecting fire emissions into regional-scale air quality models.**

DAYSMOKE is a plume rise and dispersion model and as such it has a limited application range. DAYSMOKE is driven with a vertical wind profile and not a 3-D wind field. The assumption of uniform winds horizontally may be valid for planes but it cannot be used for long-distances over complex terrain. Also, DAYSMOKE assumes that all PM<sub>2.5</sub> is primary (i.e., emitted as PM<sub>2.5</sub>) and does not allow for its evaporation. However, in wildland fire plumes, equilibrium between gas and particle phases can shift and semi-volatile organic compounds can start forming secondary PM<sub>2.5</sub> after a while. For these two reasons, application of DAYSMOKE to long-range plume transport is discouraged. The best uses of DAYSMOKE would be as a smoke impact predictor in short ranges and as an emission injector to air quality models. At a suitable distance from the source (e.g., 10-15 km) DAYSMOKE particles can be mixed as emissions into the corresponding vertical layers of an air quality model, where they would interact with other emissions, transform (particle to gas phase or vice versa), chemically react with other atmospheric species and get removed through deposition processes.

### **DAYSMOKE plume profiles somewhat matched Briggs profiles but significantly differed from FEPS profiles.**

For the Ft. Benning AB fires, Achtemeier et al. (2011) compared DAYSMOKE simulations with Briggs formulation and found remarkable similarities for “bent-over” plumes but considerable discrepancies for “highly-tilted” plumes characterized by weak buoyancy, low initial vertical velocity, and large initial plume diameters. Figure 1 shows the distribution of total fire emissions into the air quality model’s vertical layers as calculated by DAYSMOKE for the Oconee NF burn which led to the Atlanta smoke event on February 27, 2007, and compares it to the distribution by Briggs formulation and uniform distribution between plume tops and bottoms provided on an hourly basis by FEPS. Despite the difference in the amount of emissions that goes into each vertical layer, DAYSMOKE and Briggs profiles are similar in terms of plume height and the ranking of layers by fractions of emissions received, except for the surface layer. However, when FEPS plume tops and bottoms are used for distribution, emissions are allocated to more elevated layers and the resulting plume profile is significantly different in shape.

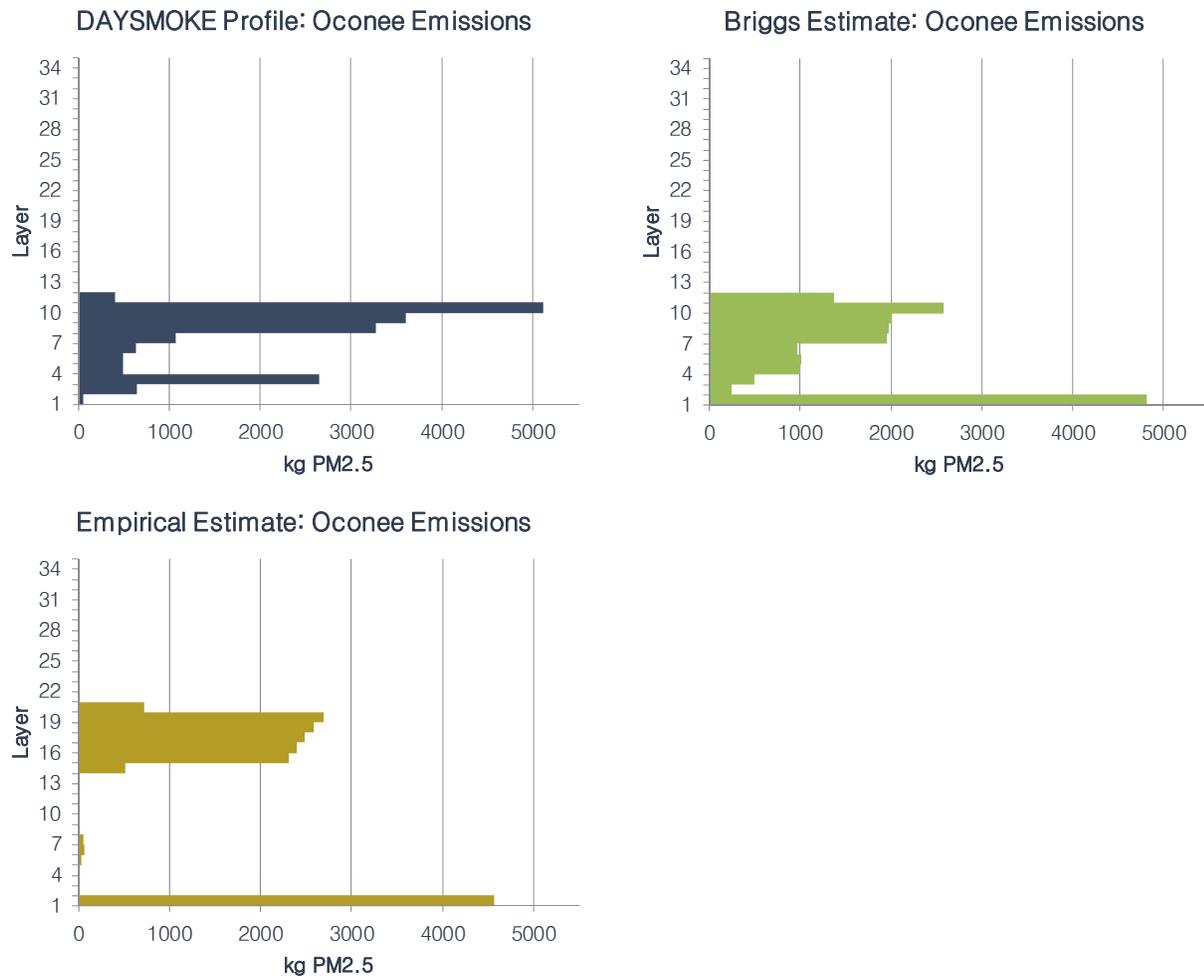


Figure 1. Vertical distribution of fire emissions: Comparison of DAYSMOKE to Briggs formulation and to FEPS empirical plume.

## CMAQ

Three variations of CMAQ were evaluated in this study: 1) Standard CMAQ, 2) CMAQ with advanced plume treatment (CMAQ-APT), and 3) Adaptive-Grid CMAQ (AG-CMAQ). The standard CMAQ is the most common version that is used by various regulatory agencies for assessing the effectiveness of emission control strategies. This version is also commonly used for air quality forecasting, regional climate and health effects of air pollution. Conformal with the emission source types available in CMAQ, biomass burns are treated either as area sources or elevated point sources. Small fires and the smoldering phase of the burns are treated as area sources. These emissions are distributed uniformly into the grid cell corresponding to the location of the fire, at the lowest model layer. The flaming phases of larger fires are treated as elevated point sources. With the help of a plume rise algorithm based on Briggs formulation, these emissions are distributed into the grid column over the fire location. As such, grid resolution becomes the single most important factor determining how well biomass plumes are distinguished from all the other sources in the same grid cell. Horizontal grid sizes are fixed, typically 12-km in regional applications and occasionally reducing to 4-km over certain urban areas, while vertical layer heights are variable, typically 20-50 m near the surface, and increasing rapidly with altitude. Inadequate resolution of fire plumes can lead to rapid dilution of fire plumes and mixing with surrounding ambient air through artificial diffusion. However, the computational intensity of CMAQ often limits the grid sizes that can be

used at regional-scale applications. CMAQ-APT (*Vijayaraghavan et al., 2006*) and AG-CMAQ (*Garcia-Menendez et al., 2010*) were developed to avoid the undesirable effects of limited grid resolution.

**Standard CMAQ underestimated prescribed fire impacts.**

The standard version of CMAQ was used by injecting the prescribed burn emissions according to three different vertical profiles for fire plumes as estimated by: 1) DAYSMOKE, 2) Briggs formulation, and 3) FEPS. Figure 2 shows the time evolution of PM<sub>2.5</sub> concentration at four monitoring sites in Atlanta that were impacted by the Oconee NP and Piedmont NWR burns on February 28, 2007. Among the different plume profiles used, the FEPS profile (marked “Empirical” in Figure 2), which distributes the flaming phase emissions uniformly between the plume tops and bottoms and injects the smoldering phase emissions into the first CMAQ layer above ground, resulted in the lowest fire impact predictions. Recall that, in Figure 1, the FEPS profile allocated the fire emissions to more elevated layers compared to the other profiles, resulting in deeper penetration of the plumes into the free troposphere. The impacts predicted by DAYSMOKE and Briggs profiles were very similar and higher by about 10 to 20 µg m<sup>-3</sup>. Despite this apparent increase in predicted impacts, standard CMAQ with DAYSMOKE or Briggs profiles still underestimated the peak PM<sub>2.5</sub> concentration observed at the sites by 80 to 100 µg m<sup>-3</sup>.

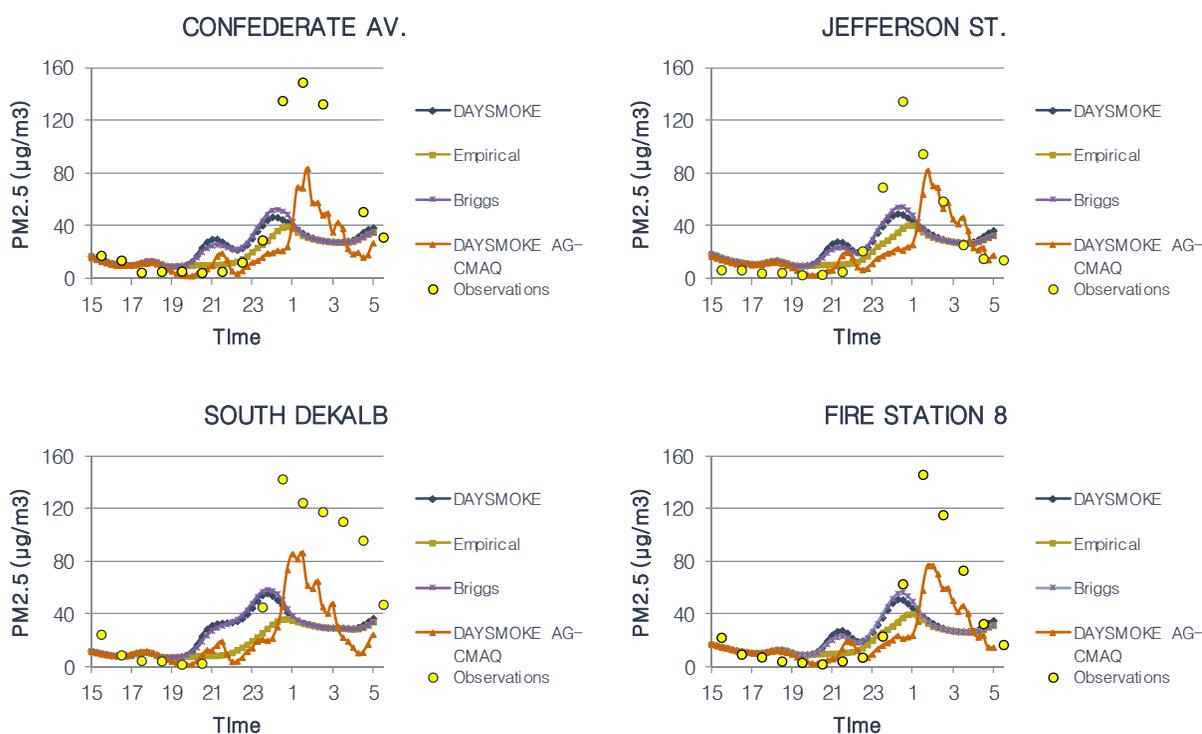


Figure 2. Observed and modeled PM<sub>2.5</sub> concentrations at 4 stations in Atlanta on February 28, 2007. The models are standard CMAQ with DAYSMOKE, Briggs, and FEPS plume profiles and AG-CMAQ with DAYSMOKE profile.

**CMAQ-APT, in its publicly available form, is not suitable for tracking fire plumes.**

CMAQ-APT is designed to simulate plumes from industrial smoke stacks. Therefore, input parameters required by CMAQ-APT related to industrial stacks were adapted to best represent biomass burning plumes, including stack height, stack diameter, exit temperature, exit velocity, and stack location (longitude and latitude).

The sensitivity of modeled concentrations to these user-defined inputs was also evaluated. Very limited sensitivity was observed to changes in the exit temperature and velocity, as well as stack height. Significant increases to pollutant concentrations were only produced by setting the stack diameter below 100 m, although burn acreages could be much larger. Additionally, the model processes only include gas-phase chemistry and no aerosol dynamics. CMAQ-APT simulations produced a series of pollutant puffs which lead to a poorly defined plume and unrealistic low pollutant concentrations. The frequency of puff release in CMAQ-APT is limited to one puff per hour. This release rate, while potentially appropriate for continuous sources, proved highly inadequate for wildland fire applications. For these reasons concentrations obtained from CMAQ-APT simulations were unsuccessful in representing realistic biomass burning plumes.

**AG-CMAQ performed better than CMAQ in simulating prescribed burn impacts.**

The adaptive grid version of CMAQ was also evaluated in prescribed burn applications. This version decreases the grid size around the fire plumes. It allows better resolution of fire emission by reducing the volume of the grid cells where emissions are being injected. It also improves the characterization of plume dispersion by reducing the numerical diffusion associated with transport in CMAQ.

Both standard CMAQ and AG-CMAQ were applied to the April 9, 2008 burn at Ft. Benning AB. Meteorological and emissions inputs to both models were the same; DAYSMOKE plume profiles were used to inject the fire emissions. The ground-level PM<sub>2.5</sub> fields shown in Figure 3 correspond to the instance when peak concentration was observed at Columbus Airport. Locations of the burn and Columbus Airport are indicated by red and white circles respectively. The standard CMAQ plume appears more diffused, relative to that produced with AG-CMAQ. It is also apparent that the impact at the airport site is not direct, but rather a tangential hit. The pollutant field from the AG-CMAQ simulation shows a more concentrated plume with higher pollutant levels near a core that has persisted longer into the simulation. Significant grid refinement occurs at the source of emissions as well as along the plume centerline. The area surrounding the airport site also experiences appreciable refinement.

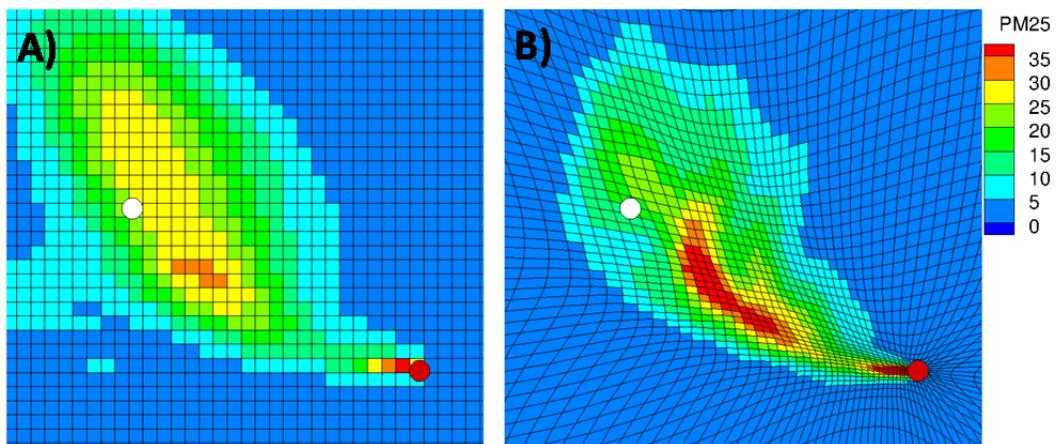


Figure 3. Simulated PM<sub>2.5</sub> concentrations (µgm<sup>-3</sup>) on April 9, 2008 at 19:45 GMT using (A) CMAQ; and (B) AG-CMAQ.

Figure 4 shows the time evolution of PM<sub>2.5</sub> concentration at the Columbus Airport monitoring site about 30 km from the location of the prescribed burn for both the standard CMAQ and AG-CMAQ runs, as well as available observational data. CMAQ overestimates the peak concentration while AG-CMAQ underestimates this

value. However, the magnitude of the error in the maximum  $PM_{2.5}$  level for both models is approximately the same ( $4 \mu\text{g m}^{-3}$ ). A sharp increase in concentration is perceived in the observations after 21:00 UTC. Similarly, rapid increases are perceived in the standard CMAQ and AG-CMAQ simulations, although occurring at earlier times.  $PM_{2.5}$  concentrations of standard CMAQ simulation fall abruptly after peaking, while the decrease is gentler with AG-CMAQ and more closely resembles that seen in the observations. Throughout the simulation, the mean fractional error in the modeled results relative to station observations was reduced by 17% on average using AG-CMAQ compared to standard CMAQ. The timing mismatch between observed and modeled peak pollutant levels is partially due to the fact that emissions inputs to the models are hourly, starting at the top of the hour, and ignition actually occurred 30 minutes past 16:00 GMT.

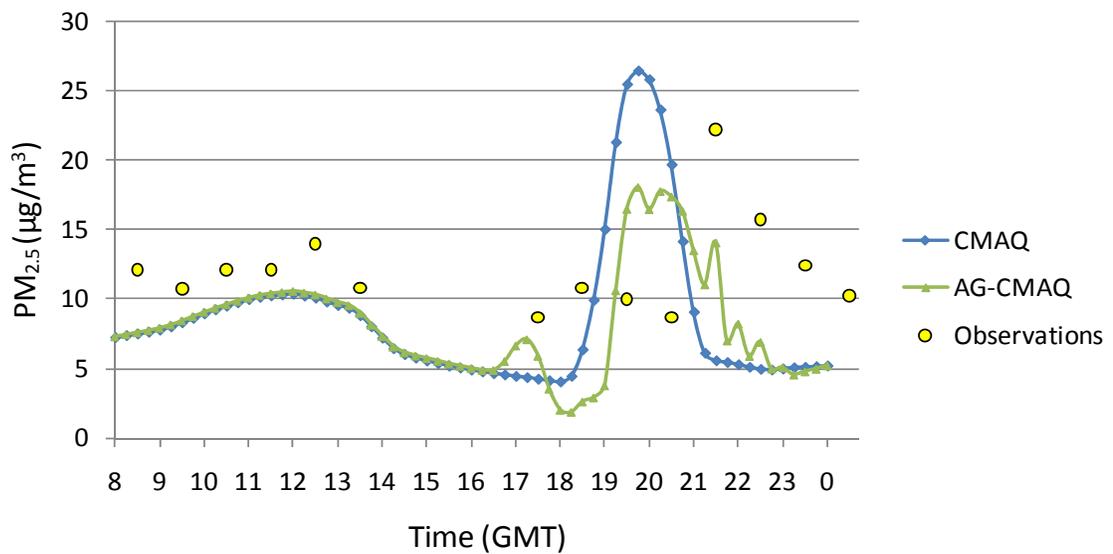


Figure 4.  $PM_{2.5}$  concentrations as observed and modeled by CMAQ and AG-CMAQ at Columbus Airport on April 9, 2008.

Application of AG-CMAQ to the February 28, 2007 Atlanta smoke event produced similar results. The evaluation showed that AG-CMAQ successfully reduced the artificial diffusion inherent to CMAQ and produced better defined plumes compared to the standard CMAQ. Most importantly, AG-CMAQ allowed the plumes from Oconee NF and Piedmont NWR burns to be distinctly observed and fire impacts at certain locations to be attributed to one burn or another while the plumes were merged almost immediately in the standard CMAQ simulation. AG-CMAQ predicted  $PM_{2.5}$  concentrations with less error than CMAQ at most monitoring sites affected during the incident (Figure 2). The mean fractional error was reduced by 15% on average, indicating significantly better agreement with site measurements. Furthermore, 3-D visualizations of plumes tracked by AG-CMAQ provided forensic insight into atmospheric processes that led to the incident well beyond what can be gained from standard CMAQ simulations.

#### Standard CMAQ severely underestimated wildfire impacts.

CMAQ was applied to the 2007 Georgia-Florida wildfires. For these simulations, fire emissions were estimated using the Satellite Mapping Automated Reanalysis Tool for Fire Incident Reconciliation (SMARTFIRE) and the BlueSky modeling framework. Background emissions from non-fire sources were prepared with the Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System. Meteorology was prepared with the Weather Research & Forecasting model (WRF). Additionally 34 vertical layers and 3 nested domains at 36 km, 12

km, and 4 km horizontal resolution were used for the CMAQ domain. The results of the Savannah case simulation on May 17 along with monitoring site observations for the period are shown in Figure 5. A visual analysis of modeled concentrations indicated that the pollutant plume from the wildfires does reach the Savannah site. However, comparison of simulated and observed  $PM_{2.5}$  concentrations demonstrates there is a severe under-prediction of the wildfire impacts on air quality. Nonetheless, the modeled pollutant plume does follow a trajectory towards a timely hit at the site. Simulation results demonstrate an important sensitivity to horizontal grid resolution, appreciable in Figure 5. The finest resolution, 4 km, produced  $PM_{2.5}$  concentrations that were over twice as large as those attained under the coarsest 36 km grid. Visualization of surface concentrations showed that the finer grid resolution leads to a less diffused and better structured pollutant plume which results in a larger impact. A three-dimensional visualization of the modeled plume structure also showed that during the instant the site is impacted, a significant amount of  $PM_{2.5}$  is concentrated aloft in upper CMAQ layers and does not affect surface concentrations.

Under-prediction of  $PM_{2.5}$  concentrations in all cases simulated with CMAQ was significant. Furthermore, the Macon, GA simulation on May 12 showed no impact in the modeled results at the site. Generally, the difference between modeled and observed concentrations heightens as the distance between emissions sources and receptors is increased. On the other hand, plume trajectories predicted by CMAQ are relatively accurate, producing well-timed hits at the expected locations. The sensitivity to grid resolution was confirmed in all study cases. Higher resolutions produced better defined plumes and larger pollutant concentrations. It is important to note that higher grid resolution also increases the sensitivity of modeled surface concentrations to meteorological inputs such as wind direction and speed, strengthening the probability of fully missing smoke plume hits under uncertain or coarse meteorological fields. A three-dimensional visualization showed that for the Atlanta, GA case on May 16, significant pollution is concentrated in upper model layers and does not impact surface concentrations, similar to the results of the May 17 Savannah case. In the Macon and Birmingham simulations, only weak and diffuse smoke plumes are perceived throughout the simulations and an under-prediction of fire emissions may be more apparent.

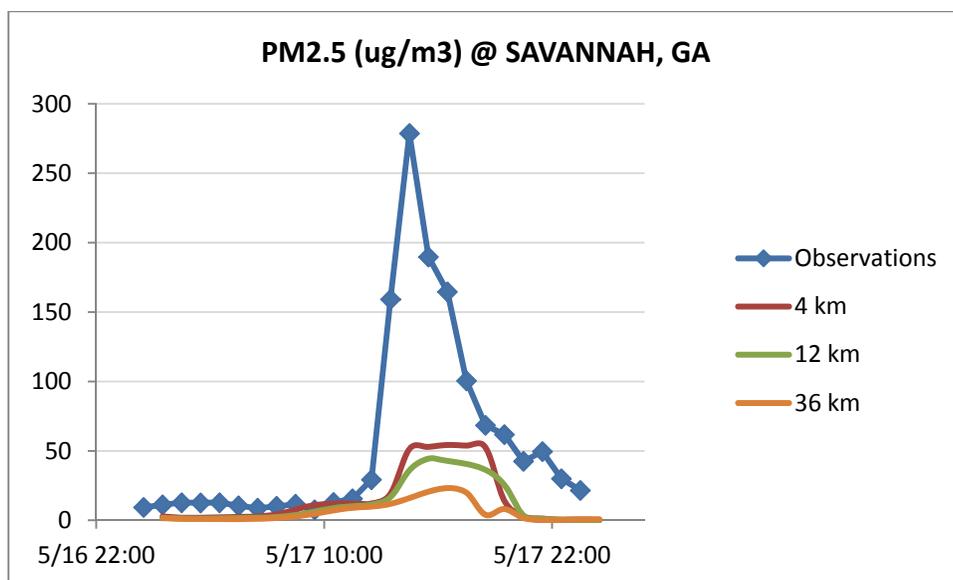


Figure 5.  $PM_{2.5}$  concentrations as observed and modeled by CMAQ using 36, 12, and 4 km grids at Savannah, Georgia on May 17, 2007.

**Grid adaptation parameters of AG-CMAQ need to be adjusted for wildfires.**

AG-CMAQ was also applied to the simulation of the 2007 Georgia-Florida wildfires. Grid adaptation to wildfire plumes was excessive, leading to high clustering of grid cells around the source and coarsening of the grid over downwind receptor areas. This degraded the performance of the model in tracking the downwind impacts. Also, the multiday nature of the simulation and modeling of multiple plumes from a large area, as opposed to one or two plumes in prescribed fire applications, led to performance degradations. The grid adaptation parameters must be readjusted for typical wildfire plumes before AG-CMAQ can be used effectively in wildfire applications.

**Sensitivity of PM<sub>2.5</sub> predictions to fire emissions**

The standard CMAQ model was used for this analysis since the other versions of CMAQ are not equipped with DDM for sensitivity calculations. DAYSMOKE was used to determine the vertical distributions of fire emissions. Emissions were injected to the vertical layers of the CMAQ grid column directly above the burn sites according to the DAYSMOKE plume profile at 11 km downwind of the sites. A better approach might be to continuously search for the CMAQ grid column that corresponds to 11 km downwind distance along the DAYSMOKE plume centerline but this was not pursued here.

**CMAQ prediction is sensitive to the vertical distribution of emissions.**

Using the brute force method (BFM), the first order sensitivities, i.e.  $S_j^{(1)}$  in Equation 1, were calculated for 10% perturbations ( $\Delta\lambda_j = 10\%$ ) of the fire emissions injected into each CMAQ layer. These sensitivities at a downtown Atlanta monitor during the Atlanta smoke event are shown in **Error! Reference source not found.**, starting at 15:00 GMT on February 28, 2007. In CMAQ, Layer 1 is the layer closest to the ground and the fire emissions were confined to the first 11 layers in this simulation. Note that the sensitivities are in units of  $\mu\text{g m}^{-3}$  per ton of emissions (left axis). Also shown are the observed hourly-averaged concentrations in units of  $\mu\text{g m}^{-3}$  (right axis). When the observed concentrations start increasing at the monitor, the greatest sensitivity of PM<sub>2.5</sub> is to fire emissions injected into Layers 8, 9 and 7, respectively. Around the same time as when the peak concentration is observed at the monitor, the sensitivity to Layer 10 emissions is the largest sensitivity. Later on, as observed concentrations decrease, the largest sensitivity is to the Layer 3 and Layer 2 emissions. Additionally the results indicate that surface concentrations are not sensitive to emissions from layers above layer 10, or 1200 m.

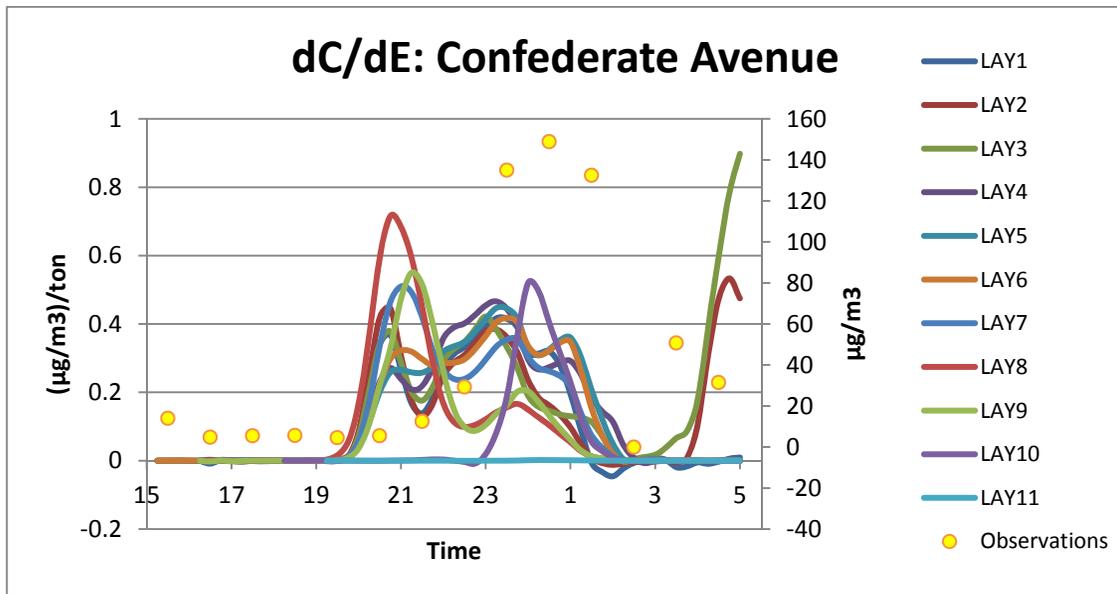


Figure 6. PM<sub>2.5</sub> sensitivities to emissions injected into different vertical layers of CMAQ.

The sensitivity analysis above suggests that the fire impacts predicted by CMAQ are sensitive to the distribution of emission into different vertical layers. One implication of this is that the underestimation of fire impact on PM<sub>2.5</sub> concentrations can be reduced by injecting the emissions into certain layers. In fact, the vertical distribution of emissions can be optimized to make the PM<sub>2.5</sub> concentrations predicted by CMAQ as close to the observations as possible, at all of the monitors, without changing the total amount of fire emissions. Then, DAYSMOKE can be recalibrated to produce that optimal vertical emission profile. In solving the optimization problem, the changes in layer emissions should be constrained to 30-50% of their nominal values since this is the range for which the sensitivities were calculated. After optimizing the vertical distribution of emissions, the remaining differences between predictions and observations can be associated to an underestimation of emissions, errors in wind speed and direction, or artificial diffusion of the plume in the model among other possibilities.

**The contributions calculated from first-order sensitivities fully explain the modeled impact.**

A simulation was conducted with all the emissions except those from Ocone NF and Piedmont NWR fires. Since the PM<sub>2.5</sub> impacts of these two fires are at issue here, the results of the simulation without those fire emissions will be referred to as “background” PM<sub>2.5</sub> concentrations. Further, the contributions of fire emissions were calculated by multiplying the sensitivities above with the amounts of emissions injected into each vertical layer. Figure 7 shows the time history of cumulative contributions from fire emissions injected into each CMAQ layer plus the background, which is the contribution of non-fire emissions. Also shown is the time history of concentration modeled with all of the emissions (dashed line). Note that there is almost a perfect match with the concentration and the sum of the contributions. This has several implications. First, it means that the sensitivities calculated by BFM using a 10% perturbation are accurate. This also implies that the response of modeled PM<sub>2.5</sub> concentrations is linear for 10% perturbation of emissions around their nominal value. Finally, it means that the modeled concentrations are fully explainable in terms of background concentrations and the emissions from the two fires in question.

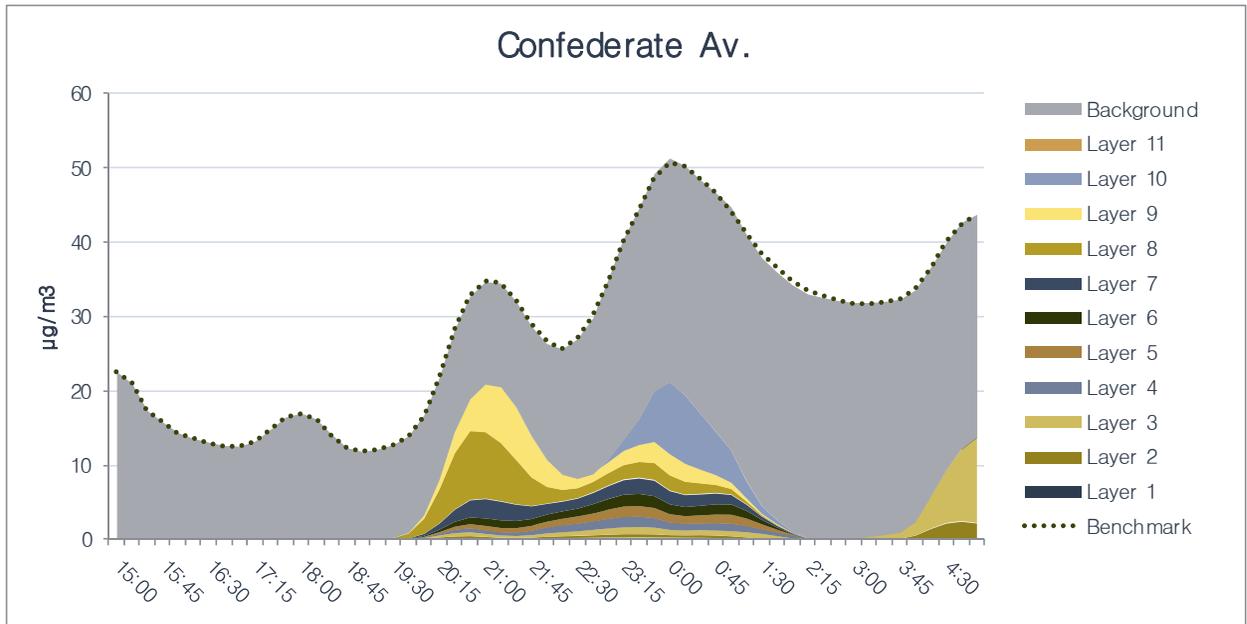


Figure 7. Contributions of fire emissions by layer to modeled PM<sub>2.5</sub> concentrations.

**Sensitivities calculated by DDM are similar to those calculated using BFM with 10% perturbations.**

The contributions of emissions injected into each CMAQ layer were calculated a second time, this time using the first order sensitivities obtained from CMAQ-DDM. They are shown in Figure 8 as hourly average contributions at another downtown Atlanta monitor. Similar to the contributions calculated from BFM sensitivities at the previous downtown monitor (compare Figure 8 to Figure 7), the largest contribution is from emissions injected into Layer 8 at 21:00 GMT, Layer 10 at 0:00 GMT, and Layer 3 at 4:00 GMT. The only difference is the rather large contribution of Layer 2 at 0:00 GMT and 1:00 GMT. This difference led to concerns about the accuracy of the DDM sensitivities. Otherwise, the BFM sensitivities and DDM sensitivities are in good agreement. Since the calculation of DDM sensitivities is independent from the calculation of concentrations in CMAQ, hence from BFM sensitivities, this agreement increases the confidence in the accuracy of the sensitivities. Further, the similarity of DDM sensitivities to BFM sensitivities implies that the more efficient DDM can be used to calculate the sensitivities.

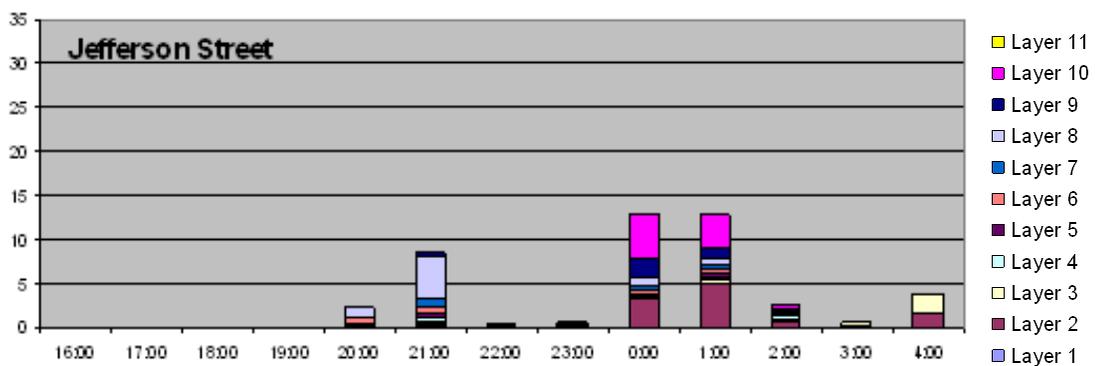


Figure 8. Contributions of fire emissions to PM<sub>2.5</sub> by layer, as calculated from DDM sensitivities

## Uncertainty of emissions and PM<sub>2.5</sub> predictions

The emissions uncertainty estimates here are based on the data collected during the Rx-CADRE field study conducted at Eglin Air Force Base (AFB) near Pensacola, Florida in February 2011.

### The uncertainty in fuel load estimates is about 20%.

An estimate of the fuel load at a burn site is the starting point for estimating the emissions from a wildland fire. Detailed measurements of fuel loads are time consuming to obtain and are therefore not done as part of a land manager's normal burn planning. However, during Rx-CADRE, detailed measurements were conducted at the sites prior to the burns, as well as after the burns (*Ottmar, 2011*). Here, those measurements were used in determining the uncertainty in fuel load estimates. Typically, wildland fuel photo series are used to estimate fuel loads by comparing site conditions against a collection of photographs with known fuel loadings. Here, a combination of photo series developed for ecosystems similar to Eglin AFB were used (*Scholl and Waldrop, 1999; Ottmar et al., 2000, 2003*). The sites were visually surveyed and then described in terms of the photo series fuel types. The descriptions were then translated into fuel loadings by taking weighted averages of the fuel bed components for the selected photos.

Figure 9 compares the estimates obtained by using the photo series to the measured fuel loads for applicable fuel bed components in FCCS. Note that the majority of the fuel by weight is conifer litter, which is estimated very well due to its relatively uniform distribution throughout the site. The use of photo series tends to overestimate the fuels except for the 100-hour fuels. The measured total fuels were 2.7 tons per acre versus the 3.2 tons per acre estimated from the photo series.

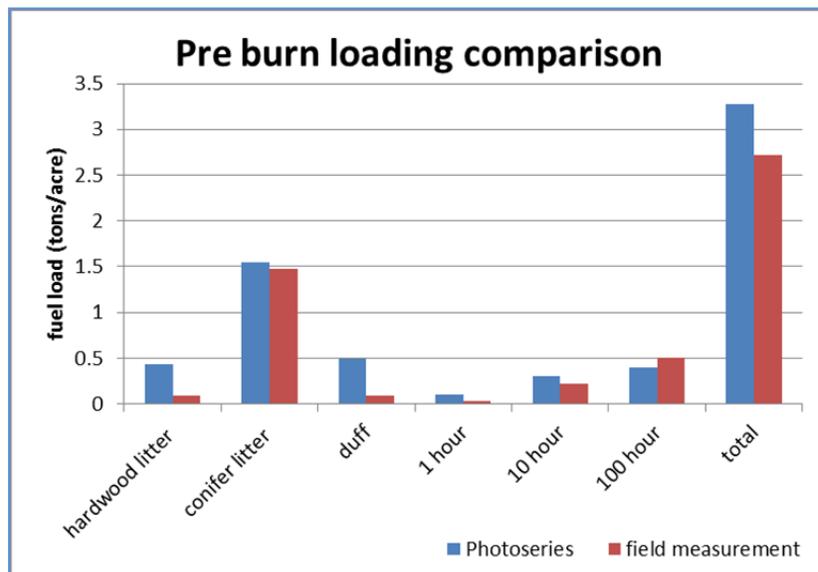


Figure 9. Comparison of estimated fuel loads to measured fuels during Rx-CADRE.

### The uncertainty in fuel consumption estimates is about 10%.

Fuel consumptions for the Eglin AFB burns were estimated using version 3.0 of the CONSUME model. These estimates were compared to the consumption measurements derived from the difference of pre- and post-

fire fuel amounts (Ottmar, 2011). Figure 10 shows the comparison of estimated and measured consumption by fuel type and for two different burns (sites 608A and 703C).

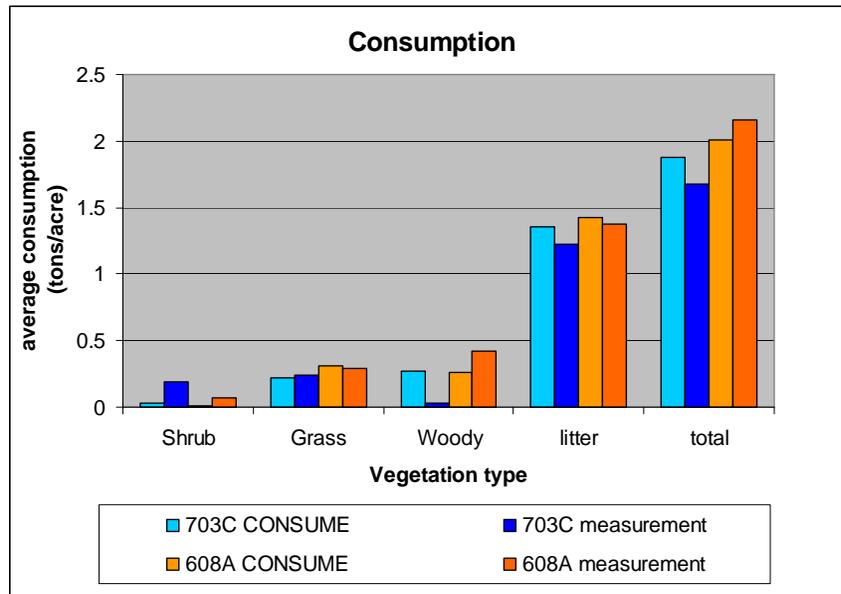


Figure 10. Comparison of estimated fuel consumption to measured consumption during Rx-CADRE.

Litter and grass consumptions are estimated well with CONSUME while shrub and woody fuel consumptions are highly uncertain. Estimates for shrub and woody fuel consumptions are in better agreement with measurements at one site (608A) than the other (703C). Also, while the consumptions of shrubs are underestimated at both sites, woody fuel consumption is overestimated at one site (703C) but underestimated at the other (608A). The total fuel consumption estimates are approximately within 10% of the measurements, overestimated by 12% at one site (703C) and underestimated by 7% at the other (608A).

**The uncertainty in PM<sub>2.5</sub> emission factor is about 20%.**

Figure 11 compares PM<sub>2.5</sub> emission factors used in this study (Urbanski et al., 2009) to those in AP-42 tables and recent laboratory and field measurements most relevant to the simulated burns. AP-42 is US EPA’s compilation of air pollutant emission factors. The AP-42 factors shown in Figure 11 are those listed for prescribed burning of long-needle conifer fuels. The fuels collected from one of the burn sites at Ft. Benning AB were burned at the Fire Sciences Laboratory in Missoula, Montana and their emissions were measured (Burling et al., 2010). Those laboratory measurements are labeled SMRFS in Figure 11. Finally, emissions from one of the burns at Eglin AFB were measured on site by means of a balloon lofted and tether maneuvered instrument package (Gullett, 2011). Those field measurements are marked Aerostat 608A in Figure 11.

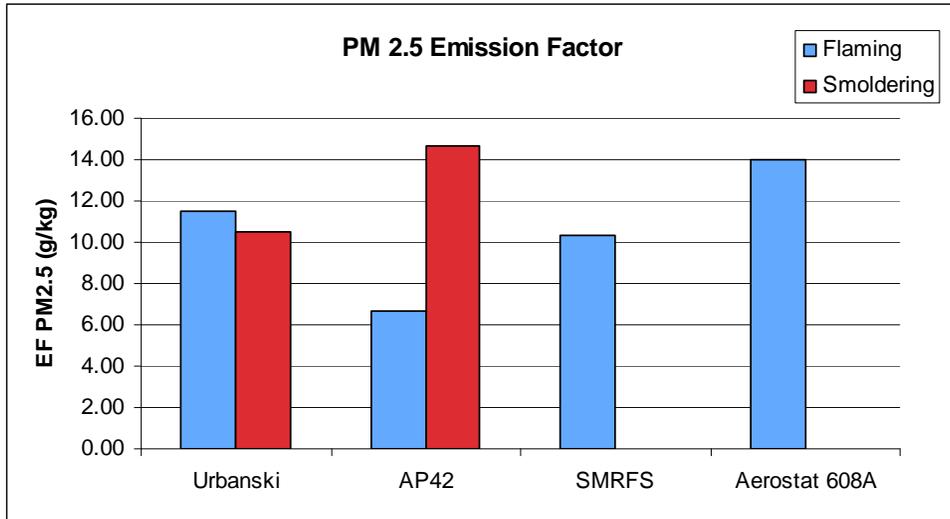


Figure 11. Comparison of emission factors.

The emission factor from the AP-42 table is not very specific in terms of ecology and fuel; therefore, it was not included in the uncertainty estimate. The laboratory and field measurements are within 31% of each other. Compared to the  $PM_{2.5}$  emission factors of *Urbanski et al.* (2009), the laboratory measurements are 2% lower than the smoldering phase emission factor and the field measurements are 22% higher than the flaming phase emission factor. Therefore, the uncertainty of emission factors used in this study is estimated to be about 20%.

**The uncertainty in  $PM_{2.5}$  emission estimates is 30% to 50%.**

Emissions are calculated by multiplying the amount of fuel consumed by the emission factor. The relative uncertainty of the product of a multiplication is equal to the sum of the relative uncertainties of the individual terms. Therefore, the uncertainty in emissions is equal to the sum of the uncertainty in the amount of fuel consumed and the uncertainty in the emission factor. The uncertainty in fuel consumption estimation by CONSUME will grow when the estimation starts with the estimation of the fuel load. Therefore, we place the uncertainty in the amount of fuel consumed between 10% (the uncertainty in CONSUME) and 30% (10% plus the uncertainty in photo-series based estimate of fuel load or 20%). When the uncertainty in emission factor (20%) is added to this bracket, the uncertainty in  $PM_{2.5}$  emission estimates would be between 30% and 50%.

**The simple model proposed as a surrogate to CMAQ is accurate in the 30-50% fire emission perturbation range.**

The first order sensitivities were recalculated using perturbations of 30% and 50% for the fire emission parameters. Recall that the parameters were emissions injected into different vertical layers of the CMAQ model. Figure 12 shows the difference between the background plus the sum of each layer's contribution and the simulated concentration at a receptor location about 80 km downwind of the fire. The difference is due to the non-linearity of the relation between downwind concentrations and fire emissions in CMAQ. The difference was negligible when the perturbation was only 10% (Figure 7). As can be seen in Figure 12, the difference does not change when the perturbation level is increased to 30% and to 50%. This is indicative of the strongly linear relationship between prescribed burn emissions and  $PM_{2.5}$  predictions in CMAQ. Therefore, the simple model in

Equation 2 can be safely used as a surrogate to CMAQ for propagating the uncertainty in fire emission parameters.

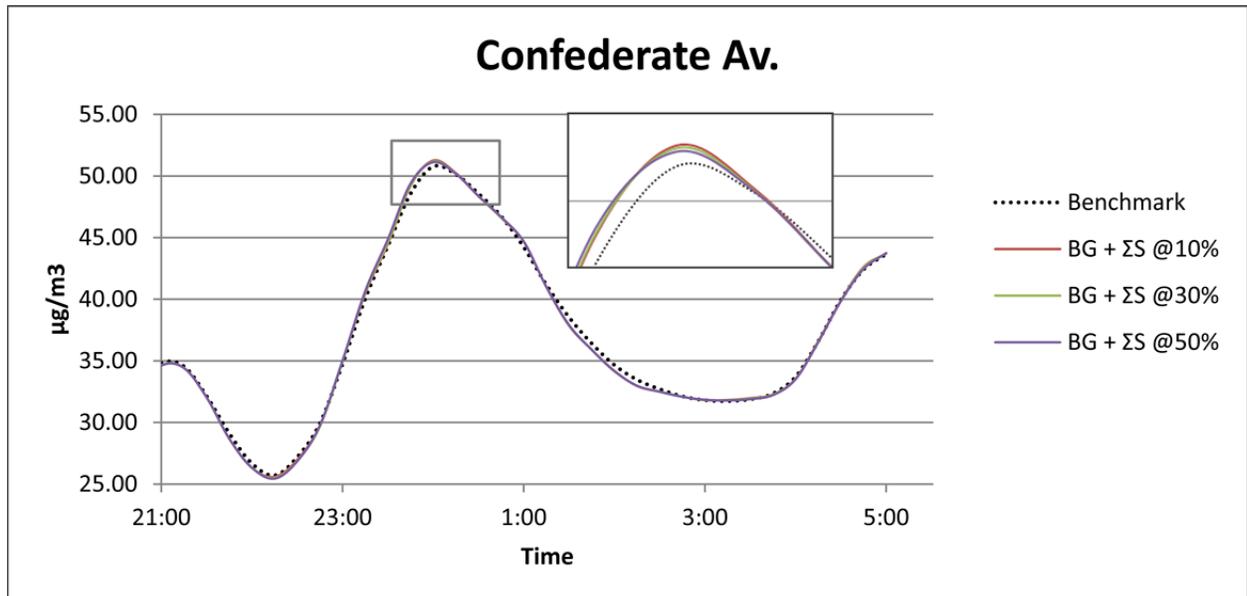


Figure 12. Non-linearity of the response of  $PM_{2.5}$  concentration to changes in fire emissions.

## 5. Management implications

The management implications of the findings of this research are as follows.

### **DAYSMOKE can be turned into a reliable smoke-impact prediction tool for land managers.**

DAYSMOKE reproduced measured plume top heights and ground-level  $PM_{2.5}$  concentrations up to 7-km downwind distances with reasonable accuracy for weak prescribed burn plumes in the Southeastern US. Therefore, it can be used reliably as a short-range smoke impact predictor. However, it should be used with caution beyond this tested range. Over complex terrain, using DAYSMOKE predictions for downwind distances more than 15 km from the burn is not recommended.

DAYSMOKE can be a very valuable tool for managers because of its accuracy in the short range, especially for use in lands surrounded by urban sprawl. Different to other smoke models, DAYSMOKE can provide information about the variability inherent to the turbulent nature of the atmosphere. From a management point of view, this variability can be interpreted as the most likely outcome (smoke impact) and probable range at any given location. During the evaluation of the DAYSMOKE several areas of improvement were identified. The following improvements would be helpful to land managers.

- 1) A graphical user interface is recommended for easy entry of user specified parameters and input data.
- 2) A knowledge base should be added to determine the most appropriate number of updraft cores for a given burn after querying the user.
- 3) DAYSMOKE should be programmed to automatically generate ensemble predictions.

- 4) The graphical output should be enhanced for presenting the variability in the predictions, for example with plume footprints and histograms of the probability of occurrence at specific receptor locations.

Linking DAYSMOKE with a forecasting system would be extremely useful. One such system is the Hi-Res air quality forecasting system operated by Georgia Institute of Technology (<http://forecast.ce.gatech.edu>). The forecasted meteorological fields can provide DAYSMOKE with necessary inputs to perform smoke forecasts.

**PM<sub>2.5</sub> impacts of prescribed burns and wildfires are best predicted by air quality models such as CMAQ.**

On a regional scale, the PM<sub>2.5</sub> impacts can best be predicted with air quality models that can simulate the transport and chemistry of fire emissions. Models like CMAQ consider all sources of emissions; therefore, they can segregate the impacts of different sources and apportion the contribution of wildland fires to receptor areas. This is particularly useful for air quality managers that are trying to develop regional control strategies. However, when making decisions based on predictions of air quality models, it is important to remember that the models are not perfect. For example, the CMAQ model used in this research was found to under predict the peak impacts of prescribed burns. Additionally, the model spreads the impacts over a larger area than what is actually affected by the fires. Under prediction of the peak impacts is even larger for wildfires. It should be kept in mind that the grid size used in air quality model applications is very important. In this research, CMAQ predictions became increasingly accurate as the grid sizes decreased from 36 to 12 to 4 km. Alternate versions of CMAQ such as AG-CMAQ that provide high resolution of five plumes over regional domains are promising but need further development. This research also found that predicted PM<sub>2.5</sub> impacts were very sensitive to the vertical distribution of fire emissions. The best results were obtained by using DAYSMOKE as an emissions injector to CMAQ.

**Fire emissions can be estimated with 30-50% uncertainty with existing tools.**

This research used a combination of wildland fuel photo series by *Scholl and Waldrop* (1999) and *Ottmar et al.* (2000; 2003) to estimate fuel loads, CONSUME 3.0 to estimate fuel consumptions, and emissions factors published by *Urbanski, et al.* (2009) to estimate emissions from consumed fuels. By comparing the estimates to measurements during Rx-CADRE 2011, a study conducted at Eglin AFB, this research has found that the uncertainties in estimates of fuel loads, fuel consumptions, and emissions factors are about 20%, 10%, and 20% respectively. The uncertainty in fire emissions of PM<sub>2.5</sub> estimated using these tools is estimated to be in the range of 30% to 50%. Air quality managers should keep this uncertainty in mind when making decisions based on the amounts of emissions from prescribed fires.

**It is safe to assume that the impacts are a linear function of emissions in this range of uncertainty (30-50%).**

When fire emissions were perturbed by amounts within the range of their uncertainties (30-50%) the PM<sub>2.5</sub> impacts predicted by CMAQ responded linearly. This implies that managers can apply margins of safety to model predicted impacts by the same relative amounts (30-50%). It also implies that a simple model can be built to determine the probability distributions of the impacts from the probability distributions of various emissions parameters. This research initiated the development of such a model. Once the model is fully developed and the probability of occurrence of an air quality model's impact predictions are determined, management questions like 'what is the likelihood of exceeding the PM<sub>2.5</sub> standard due to prescribed burning?' can be answered.

## 6. Relationship to other recent findings and ongoing work on this topic

The findings of the JFSP projects 08-1-6-10, Creation of a Smoke and Emissions Model Intercomparison Project (SEMIP) and Evaluation of Current Models, by N. K. Larkin, and 08-1-6-06, Evaluation and Improvement of Smoke Plume Rise Modeling, by Y. Liu are very relevant to this research. SEMIP, as well as earlier work by *Larkin et al.* (2009) are complementary by providing evaluation results for models that were not evaluated here. SEMIP also provided important information on the uncertainty of both prescribed and wild fire emissions that were input to the models evaluated here. The plume rise modeling of *Liu et al.* (2010) with DAYSMOKE had direct inputs into this research. Also, products and findings from SERDP project, SI/RC-1647, Characterization of Emissions and Air Quality Modeling for Predicting the Impacts of Prescribed Burns at DoD Lands, by M. T. Odman were utilized in this research.

The performances of the CMAQ model and its variations in this research have been continuously evaluated against performances reported by others, for CMAQ and other air quality models, in similar applications. The evaluation by *Junquera, et al.* (2005) of the Comprehensive Air quality Model with extensions (CAMx) in an application to wildland fires in southeast Texas and the application of CMAQ to the 2007 Georgia-Florida wildfires by *Yang et al.* (2008; 2011) were used as benchmarks. The performances obtained here in all CMAQ applications exceeded those benchmarks.

In this research, the estimates of uncertainty in prescribed burn emissions are limited to the measurements made during the Rx-CADRE campaign at Eglin AFB during, in February 2011. SEMIP provides estimates based on a wider array of measurements including satellites (*Larkin et al.*, 2011). The uncertainty estimated to be introduced by the fuel consumption model is in agreement with the estimate by *Larkin et al.* (2011). However, there are important differences between the uncertainties estimated for the fuel loads in the two studies. *Larkin et al.* (2011) conclude that the uncertainty in fuel loads are larger than what is reported here, especially in western US. Also, the uncertainty in canopy fuels involved in wildfires is much larger than the understory fuels according to *Larkin et al.* (2011). Therefore, the underestimation of fire impacts predicted by the CMAQ model in 2007 Georgia-Florida wildfire simulations may be largely due to the uncertainty in satellite-based emissions estimates.

Another important uncertainty in model simulations was found to be the vertical plume distribution. Recent and ongoing research on measuring/estimating plume heights will be very helpful in determining these uncertainties and, eventually, reducing them by improving the models. Ground-based plume measurements by LIDAR are being used currently in evaluating modeled plume heights (*Liu et al.*, 2011). When ground-based measurements are not available, satellite based information can be very useful. Plume injection heights have been calculated from multi-angle and multispectral image produced by Multi-angle Imaging SpectroRadiometer (MISR) by *Khan et al.* (2007). Satellite-based data have certain limitations; for example, observations can be affected by cloud attenuation, or short lived fires can be missed due to the satellite's rotation rate for polar orbiting satellites such as Terra, which carries MISR.

There is potential synergy in combining data from different satellites. The combination of fire radiative power (FRP) from MODIS and aerosol optical thickness (AOT) from MISR (*Van Martin et al.*, 2010) or CALIPSO (*Amiridis et al.*, 2010) were used together to calculate smoke injection heights. Significant improvements in

correlation of injection heights with PBL heights were reported as a result of these efforts. Another benefit of satellite observations is reconciling fire activity data to be used in regional-scale simulations. Roy, *et al.* (2007) utilized the MODIS observed fire activity to refine the National Emissions Inventory fire emissions for major wildfire events. The reallocation of fire emissions in this manner, both temporally and spatially, can significantly improve the PM<sub>2.5</sub> predictions by air quality models like CMAQ.

LIDAR measurements contain important information to characterize the structure and density of the smoke plume. The continuous vertical profiles of aerosol backscatter provided by the LIDAR can serve as a proxy for PM<sub>2.5</sub> as shown by Muenkel *et al.* (2007). While this method can provide reliable PM<sub>2.5</sub> values near the plume bottom, the backscatter is subject to attenuation at greater heights by possibly intervening layers of smoke. Without an external reference source, attenuation cannot be easily distinguished from backscatter. This is a well-known limitation of elastic backscatter LIDAR. The best way of dealing with attenuation is to calibrate the the LIDAR backscatter with *in-situ* measurements of PM<sub>2.5</sub>. *In-situ* measurements can be collected by aircraft (Akagi *et al.*, 2011) or by a tethered balloon, like the one used at Eglin AFB during Rx-CADRE (Gullett, 2011).

This research laid out a method for propagating the uncertainties associated with partitioning fire emissions into the vertical layers of an air quality model. As related research continues to determine the uncertainties in plume heights, structure, and density, future applications of this method are expected to demonstrate how these uncertainties reflect on the accuracy of air quality model predictions.

## 7. Future work needed

The findings of this study indicate that successful simulation of wildland fire impacts on air quality with numerical models requires that further research be performed on multiple elements of the fire modeling process, which at present may be limiting model performance. The degree to which these elements hinder simulation accuracy and the significance of each relative to the others is perhaps the most critical issue. The sensitivity and uncertainty analyses outlined here should be expanded to prioritize the concerns with different elements of the modeling process. In this study, several pressing research needs essential for functional numerical prediction of the air quality impacts of wildland fires have been identified and are described below. Future research should focus on these topics and investigating them in greater detail than what was possible here.

It is clear that additional research is needed to better characterize the pollutant emissions from prescribed burns and wildfires. Available information describing fuel loads, consumption estimates, and emission factors is limited, and the uncertainties in each can have a significant effect on predicted downwind pollutant concentrations. The results of the analyses on wildfire simulations undertaken as part of this study indicate that satellite-derived emissions estimates for wildfires may be significantly under-predicting pollutant releases and related research points to the canopy fuel load as the major source of uncertainty. What is clear now is that accurate emissions will be an essential component of reliable air quality simulations in the future.

A need to better describe and represent plume rise phenomena has also been identified from the results of this study. Plume rise and the subsequent vertical distribution of fire emissions in gridded air quality models have proven to significantly impact concentration estimates. Many current plume rise estimates only provide limited information and do not offer realistic vertical plume structures. However, accurate representations of the vertical distribution of fire-related pollutant emissions are a crucial aspect of smoke episode numerical simulations. One plume-rise model evaluated here, DAYSMOKE, provided improved predictions of the plume

structure and density but it needs further evaluations and possible improvements to be applied to a broader range of fires.

In previously reported research, little attention has been given to secondary aerosol formation in air quality modeling of wildland fires. Secondary organic aerosol (SOA) production might be significant in fire-related episodes. However, realistic simulation of the process requires adequate emissions estimates for precursors to SOA formation and appropriate chemical transformation mechanisms. Available emissions factors and estimates for the species of greatest interest to SOA formation are insufficient. Reactive plume models are lacking; for example, DAYSMOKE treats PM<sub>2.5</sub> as an inert species. Chemical mechanisms that treat the process of secondary organic aerosol formation are becoming available in the latest versions of air quality models such as CMAQ, but need further development and continuous refinements. Also, some of the emissions and processes that lead to SOA are still relatively unknown. Indicators derived from special measurements offer great new opportunities for better understanding of the SOA formation processes associated with biomass burning.

From the results of this project the potential effects of meteorological inputs to air quality models has become apparent in wildland fire applications. Predicted concentrations might be highly responsive to the accuracy and the spatial and temporal resolution of key meteorological fields, including wind speed, wind direction, and mixing layer height. In future work, the techniques presented within this study to estimate air quality sensitivities will be applied to quantify the responsiveness to these meteorological variables. It is necessary to determine the degree to which the accuracy of air quality simulations might be bounded by meteorological information normally provided by weather forecasting models. Another topic of future research should be the impacts of fire plumes on boundary layer dynamics. Some air quality models such as WRF-Chem and WRF-CMAQ (available with version 5.0 of CMAQ) already consider the radiative feedbacks of smoke plumes by taking a coupled meteorology and air quality modeling approach.

The importance of grid resolution in modeling fire-related impacts on air quality with gridded air quality models has been demonstrated. High resolution simulations are possible with today's computational resources and adaptive grid methods. However, the finest scales applied in modeling applications necessitate new or resolution-dependent parameterizations for processes that cannot be explicitly resolved. Parameterizations designed for coarse grid resolution must be reconsidered as grid resolution is increased. The potential benefits of reformulating these parameters have not yet been quantified, but should clearly be investigated in future research efforts.

Finally, to benefit from all these models in answering policy questions like "What is the contribution of wildland fires to PM levels in the US and how important that contribution would be if the PM standards were to be tightened?" the uncertainties in model predictions must be determined. Knowing the uncertainty of a prediction is critical for the decision making process. While the uncertainties in various elements of the modeling process are being determined, it is also important to investigate how those uncertainties interact with each other and contribute to the uncertainty in the final result, i.e., PM concentration predictions. Conducting Monte Carlo uncertainty analyses by using simplified surrogates of the air quality model, as outlined in this report, is a promising approach. Future research should focus on deriving the probability of occurrence for the predicted PM and other pollutant outcomes from the models.

## 8. The deliverables crosswalk

Proposed	Delivered	Status
Project web site	<a href="http://sipc.ce.gatech.edu/">http://sipc.ce.gatech.edu/</a>	Updated as needed
Data set	Data sets including all model inputs (emissions and meteorology) were uploaded to the project website. The upload of model output data (concentrations and sensitivities) continues.	In progress
Software	Links to computer models used in this research (CALPUFF, DAYSMOKE and CMAQ) were provided on the project website. The AG-CMAQ code is being updated for compatibility with version 5.0 of CMAQ released on November 2011.	In progress
Publications	<p>Two refereed publications: one on model evaluation and another on sensitivity/uncertainty analysis were promised.</p> <p>Three papers on model evaluation</p> <ul style="list-style-type: none"> <li>(1) Garcia-Menendez et al. (2010)</li> <li>(2) Achtemeier, et al. (2011)</li> <li>(3) Garcia-Menendez et al. (2012)</li> </ul> <p>Two papers on sensitivity are uncertainty analysis</p> <ul style="list-style-type: none"> <li>(1) Garcia-Menendez et al. (2011)</li> <li>(2) Yano, et al (2012)</li> </ul> <p>One review paper on air quality models:</p> <ul style="list-style-type: none"> <li>(1) Garcia-Menendez and Odman (2011)</li> </ul>	<p>Completed</p> <p>Completed</p> <p>In progress</p> <p>In progress</p> <p>In progress</p> <p>Completed</p>
Publication	A manager's guide to interpreting smoke model output was promised. With the help of the Consortium of Appalachian Fire Managers and Scientists (CAFMS) users are integrated into the development of the guide through a series of workshops. The first was in August 2011 in Virginia and the second will be in North Carolina in spring 2012. The workshop is focused on smoke management tools for land managers, their limitations and practical applications. Closely integrating users in the development by teaming with CAFMS has introduced delays into the development of the guide, but the end product will be much better geared towards manager needs than it would have otherwise. After the second workshop the guide is expected to be complete by summer 2012 and published as a Forest Service GTR.	In progress
Presentations	Nine conference/symposium presentations	Completed

	Two conference presentation	In progress
Annual Progress Report	An annual progress report was submitted on August 2010.	Completed
Final Report	This document is the final report.	Completed

## Detailed List of Deliverables

### Refereed Publications

- (1) Garcia-Menendez, F., A. Yano, Y. Hu, and M. T. Odman (2010), An adaptive grid version of CMAQ for improving the resolution of plumes, *Atmospheric Pollution Research*, 1, 239-249.
- (2) Achtemeier, G. L., S. L. Goodrick, Y. Liu, F. Garcia-Menendez, Y. Hu, M. T. Odman (2011), Modeling Smoke Plume-Rise and Dispersion from Southern United States Prescribed Burns with Daysmoke, *Atmosphere*, 2, 358-388.
- (3) Garcia-Menendez, F. and M. T. Odman 2011: Adaptive Grid Use in Air Quality Modeling, *Atmosphere*, 2, 484-509.
- (4) Garcia-Menendez, F., D. G. Tokgoz, and M. T. Odman (2011), Analysis of the impacts of fire emissions and plume-rise in air quality simulations, in preparation.
- (5) Garcia-Menendez, F., Y. Hu, and M. T. Odman 2012: Modeling the air quality impacts of wildfires, in preparation.
- (6) Yano A., M. T. Odman, R. D. Ottmar, S. L. Goodrick, Y. Liu, G. L. Achtemeier, and B. Gullett (2012), Uncertainties in prescribed fire emissions and their impact on smoke dispersion predictions, in preparation.

### Technical Abstracts/Presentations

- (1) Odman, M. T., Y. Hu, A. Yano, F. Garcia-Menendez, G. L. Achtemeier, S. L. Goodrick, Y. Liu, D. S. Mcrae, L. Naeher, Development of a modeling system for prescribed burn emissions and air quality impacts, 16<sup>th</sup> Joint Conference on the Applications of Air Pollution Meteorology with the A&WMA, 17-21 January 2010 at Atlanta, GA.
- (2) Garcia-Menendez, F., Y. Hu, and M. T. Odman, Evaluation of air quality models applied to wildland fire impact simulation, 9th Annual CMAS Conference, Chapel Hill, NC, October 11-13, 2010.
- (3) Yano, A., F. Garcia-Menendez, Y. Hu, M. T. Odman, D. S. McRae, and G. L. Achtemeier, Modeling biomass burnings by coupling a sub-grid scale plume model with Adaptive Grid CMAQ, 9th Annual CMAS Conference, Chapel Hill, NC, October 11-13, 2010.
- (4) Odman, M. T., F. Garcia-Menendez, A. Yano, Y. Hu, G. L. Achtemeier, and D. S. McRae, A High-Resolution Modeling System for Regional Carbonaceous Aerosols, 29<sup>th</sup> Annual Conference of the American Association for Aerosol Research, Portland, OR, October 25-29, 2010.
- (5) Yano A. and M. T. Odman, Modeling biomass burnings by coupling a sub-grid scale plume model with Adaptive Grid CMAQ, American Meteorological Society's 9th Symposium on Fire and Forest Meteorology, Palm Springs, CA, October 18-20 2011.
- (6) Yano A., M. T. Odman, R. D. Ottmar, S. L. Goodrick, Y. Liu, and B. Gullett, Uncertainties in prescribed fire emissions and their impact on smoke dispersion predictions, American Meteorological Society's 9th Symposium on Fire and Forest Meteorology, Palm Springs, CA, October 18-20 2011.

- (7) Garcia-Menendez, F., Y. Hu, and M. T. Odman, Modeling the air quality impacts of wildfires with CMAQ, 10th Annual CMAS Conference, Chapel Hill, NC, October 24-26, 2011.
- (8) Garcia-Menendez, F., D. G. Tokgoz, and M. T. Odman, Analysis of vertical fire emissions distribution in CMAQ, 10th Annual CMAS Conference, Chapel Hill, NC, October 24-26, 2011.
- (9) Yano A. and M. T. Odman, Coupling a subgrid scale plume model for biomass burns with the adaptive grid CMAQ model, 10th Annual CMAS Conference, Chapel Hill, NC, October 24-26, 2011.
- (10) Garcia-Menendez, F., D. G. Tokgoz, Y. Hu and M. T. Odman, Sensitivity of air quality to meteorology inputs in forest fire simulations, 14th Conference on Atmospheric Chemistry, New Orleans, LA, January 22-26, 2012.
- (11) Garcia-Menendez, F., Y. Hu and M. T. Odman, Improved Modeling of Wildland Fire Plumes with CMAQ, 17th Conference on Air Pollution Meteorology with the A&WMA, New Orleans, LA, January 22-26, 2012.

## Reports

- (1) Odman, M. T. and S. L. Goodrick (2011), Evaluation of Smoke Models and Sensitivity Analysis for Determining their Emission Related Uncertainties, Annual Progress Report (JFSP Project Number 08-1-6-04), August 24, 2010, Atlanta, Georgia.
- (2) Odman, M. T., F. Garcia-Menendez, A. Yano and S. L. Goodrick (2011), Evaluation of Smoke Models and Sensitivity Analysis for Determining their Emission Related Uncertainties, Final Project Report (JFSP Project Number 08-1-6-04), November 11, 2011, Atlanta, Georgia.

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