Air Quality Impacts from Prescribed Forest Fires under Different Management Practices

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Large amounts of air pollutants are emitted during prescribed forest fires. Such emissions and corresponding air quality impacts can be modulated by different forest management practices. The impacts of changing burning seasons and frequencies and of controlling emissions during smoldering on regional air quality in Georgia are quantified using source-oriented air quality modeling, with modified emissions from prescribed fires reflecting effects of each practice. Equivalent fires in the spring and winter are found to have a greater impact on $PM_{2.5}$ than those in summer, though ozone impacts are larger from spring and summer fires. If prescribed fires are less frequent, more biofuel is burnt in each fire, leading to larger emissions and air quality impacts per fire. For example, emissions from a fire with a 5-year fire return interval (FRI) are 72% larger than those from a fire of the same acreage with a 2-year FRI. However, corresponding long-term regional impacts are reduced with the longer FRI since the annual burned area is reduced. Total emissions for fires in Georgia with a 5-year FRI are 32% less than those with a 2-year FRI. Smoldering emissions can lead to approximately 1.0 or 1.9 $\mu g/m^3$ of $PM_{2.5}$ in the Atlanta $PM_{2.5}$ nonattainment area during March 2002.

Introduction

Air pollutants from a prescribed fire about 80 km southeast of metro Atlanta on February 28, 2007 led to parts of the city being exposed to unhealthy levels of $PM_{2.5}$ for several hours. Observed 1-h $PM_{2.5}$ concentrations at several monitors in the city reached higher than 145 $\mu g/m^3$ (U.S. National Ambient Air Quality Standard (NAAQS) for 24-h $PM_{2.5}$ is 35 $\mu g/m^3$), increasing by over 100 $\mu g/m^3$ in two hours (*1*). In addition, as the plume hit, 1-h average ozone concentrations increased markedly from 63 to 95 ppb at one monitor.

Unlike wildfires, prescribed fires are intentionally ignited in order to maintain ecosystem health and minimize adverse impacts of long-term fire suppression while protecting property (*2–5*). About 2 million acres per year of federal forests were burned by prescribed fires from 1998 to 2006, in comparison to around 6 million acres of wildfires (*6*). Prescribed fires and wildfires together contributed about 20% of the fine particulate matter ($PM_{2.5}$) emissions in the United States (*7*). Results from both field measurements and numerical modeling have shown significant air quality degradation due to forest fire emissions (*8–10*).

Prescribed fires are usually planned for conditions that are not likely to lead to their becoming uncontrolled, and when feasible they are often planned to reduce impacts on populated areas. They are typically limited in extent, spatially and temporally. Therefore, emissions and corresponding air quality impacts from prescribed fires can be reduced by adopting smoke reduction techniques and choosing better dispersion conditions for burning, as suggested by both U.S. Environmental Protection Agency (EPA) and U.S. Forest Service (*4, 11*). Smoke reduction is usually achieved by reducing burned area and fuel consumption, and increasing combustion efficiency of fires. Such techniques include, but are not limited to, reducing forest fuels using mechanical and chemical methods, igniting back fires or aerial ignition, burning before precipitation or at high frequencies, using air curtain incinerators, and rapid mop-up (*4, 11*). Different technologies and their combinations can be chosen for different management goals. Though significant air quality impacts from application of different technologies are expected, such impacts are rarely quantified.

Increased application of prescribed fires is expected, given their characteristics of being controlled and requirements from ecosystem and air quality management (*4, 5*). Furthermore, a recent study showed that climate change led to increased wildfire activities in the western United States (*12*). Appropriate management practices, including prescribed fires, are increasingly required to reduce wildfire hazards. Therefore, understanding how forest management practices can change air quality impacts from prescribed fires should be addressed. Here, a source-oriented air quality model, capable of predicting air quality under different emissions and meteorological conditions, is employed. Historical air quality conditions are first reproduced using the actual prescribed fire emission patterns together with emissions from other sources as inputs. Emissions from prescribed fires are then modified to reflect the effects of various management practices.

Methods

Georgia, where forests cover more than 66% of the total land and prescribed fires have been widely used, is chosen for this case study. More than 92% of Georgia forestland is owned by private parties. Between 1994 and 2005, an average of 0.86 million acres per year of private and public forests were burned by prescribed fires in Georgia (*13*), in comparison with an average of 2 million acres per year in the United States on Federal forests. These fires mainly burn in the southern pine forests (Supporting Information, Figure 1), and consume understory fuels, such as grass, live shrubs, and needles, without significantly damaging trees (*2, 13, 14*).

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Air quality impacts from forest fires with different burning seasons and frequencies are evaluated in this paper, as well as air quality impacts from emissions during the smoldering combustion stage.

Historical air quality conditions during 2002 are simulated using the Community Multiscale Air Quality (CMAQ) model v. 4.3 (*15*), a three-dimensional, detailed photochemical atmospheric model. Meteorological conditions are simulated with the Pennsylvania State University (PSU)/National Center for Atmospheric Research (NCAR) Mesoscale Modeling System Generation 5 (MM5) (*16, 17*) and emissions are processed with the Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System v. 2.1 (*18*). The 2002 Visibility Improvement State and Tribal Association of the Southeast (VISTAS) emission inventories (*19*) are used with updated biomass burning emissions (*20*). Modeling performance is evaluated by comparing simulations with ozone observations from EPA's Aerometric Information Retrieval System (AIRS, http://www.epa.gov/ttn/airs/airsaqs/detaildata/downloadaqsdata.htm), and with total and speciated $PM_{2.5}$ observations collected as part of the Interagency Monitoring of Protected Visual Environments (IMPROVE, http://vista.cira.colostate.edu/improve/), the SouthEastern Aerosol Research and Characterization (SEARCH) (*21*), the Assessment of Spatial Aerosol Composition in Atlanta (ASACA) (*22*) and the Speciation Trends' Network (STN, http://www.epa.gov/ttn/airs/airsaqs) networks. Mean normalized bias for simulated ozone is within ±15%, and mean normalized error is less than 35% (*23*). Overall performance of simulated $PM_{2.5}$ is well within recent performance suggestions (*24*). Detailed information on the modeling system and performance can be found elsewhere (*20*).

Emissions from forest fires are calculated as the product of the burned area (A), fuel consumed per area (F_a) and an emission factor (E_f) (*25*):

$$E = A \times F_a \times E_f \tag{1}$$

Here, A is determined from current forest fire records (*13*), F_a is the amount of biomass consumed during a forest fire per area, and E_f is the ratio of the mass of pollutant emitted per unit mass of fuel consumed. F_a and E_f are functions of fuel condition (e.g., moisture content and availability) and meteorology. Forest managers choose to burn when fuel conditions are within specific limits, in order to sustain a burn, but minimize potential damage (e.g., to roots). Such fuel properties are chosen for simulation here.

Burning Season. Forest management issues involve choice of periods for prescribed fires, mainly depending upon the purpose of burning and ecosystem requirements. In Georgia, burning during winter and spring is most common (Supporting Information, Figure 2), as forests burned during summer and fall are more likely to die, and burning is harder to control due to commonly unstable atmospheric conditions in these periods (*26*). More than 86% of prescribed fires were scheduled between December and April according to records between 1994 and 2005, with 37% of the annual total occurring in March alone (*13*).

Four months in 2002, including January, March, May, and July, are selected to represent different burning seasons. March is chosen since it is the month with the most prescribed fires in Georgia. Burning in January is also frequent, with forest area burned about one-third of that in March. Natural wildfires are mainly ignited by lightning and occur in Georgia during May and June when lightning frequency is high and summer thunderstorms have not provided much moisture (*2, 4, 27*). Therefore, burning in May is also studied. Finally, burning in July is investigated with particular interest in corresponding air quality impacts during summer ozone seasons. Fall is not considered, because it is neither a naturally preferred season nor practical for prescribed fires. Simulations with and without prescribed fire emissions during respective months of 2002 are first conducted to investigate air quality impacts from existing fires. Emissions from March 2002 prescribed fires are also individually input into CMAQ for the other three months, together with the applicable emissions from other sources pertaining to the specific month, which vary according to time of year and meteorology. Such simulations are used to evaluate air quality impacts of the same fires during different burning seasons.

Burning Frequency. Burning frequencies (characterized by fire-return intervals, FRIs) influence fuel consumption. For a fixed burned area A, forest fire emissions change proportionally with F_a (1), which increases with longer FRIs. Prescribed fires in Georgia are currently applied to specific areas periodically in intervals of 2−5 years (*2, 14*), and would burn too severely if FRIs were longer than 5 years (*2*). The characteristic F_a (F_{aC}) for the prescribed fires in Georgia is 4 tons/acre for a 3.5-year FRI (the mean interval when considering 2 to 5 years), which has been used to develop the most recent emission inventory (*28*). Here, F_{aC} is used to calculate F_a values for 2-, 3-, 4-, and 5-year FRIs by multiplying with a relative ratio calculated for each FRI. These ratios are estimated using F_a values calculated by a fire behavior model, the First Order Fire Effects Model (FOFEM, http://fire.org) v 5.21. Default preburn fuel characteristics (such as relative abundance of particular fuelbed components and the condition of the fuel) for loblolly and slash pines (major forest types burned by prescribed fires in Georgia) at various ages are used as inputs. Since the default inputs in FOFEM do not represent fuel conditions in Georgia, the F_as calculated by FOFEM are used to scale the F_{aC} calculated for Georgia to each FRI.

The above estimates for a specific burned area are referred to as an "individual" fire impact, assuming FRIs for other prescribed fires in Georgia do not change. When FRI changes are applied to all forests in Georgia, the corresponding estimates are referred to as an "aggregate" fire impact. In this case, since FRIs influence not only F_a, but also yearly burned acreage (A), corresponding emissions do not simply increase with FRIs as does a single fire. For example, in Georgia, about 0.86 million acres of forests were burned per year by prescribed fires (average for 1994 to 2005 (*13*)). With the assumed 3.5-year FRI, the total forest area under management using fires is approximately 3 million acres (0.86 multiplied by 3.5). If a 2-year FRI were used, 1.5 million acres would be burned each year, and if a 5-year FRI were employed, 0.6 million acres would be burned. Here, annual emissions from prescribed fires with different FRIs ranging from 2 to 5 years in Georgia are calculated with respective A and F_a values.

Flaming and Smoldering. There are two combustion stages of forest fires: flaming and smoldering. Of the two, smoldering combustion is relatively incomplete with larger emissions per mass of fuel burned and lower heat release (*5, 29*). Due to the different heat release rate and timing, emissions during these two stages also have different dispersion behaviors in the atmosphere. Since flaming and smoldering emissions sometimes occur simultaneously, the flaming stage is defined, here, as emissions which are influenced by the strong convection associated with a flame front (*29, 30*). Thus, the portion of smoldering emissions which occurs during flaming is defined as a part of the flaming stage.

Air quality impacts from emissions during each combustion stage are simulated using CMAQ during March 2002, when prescribed fires are the largest in Georgia. Since prescribed fire emissions in the original emission inventory are total emissions from both stages, such emissions are split into each stage based on corresponding emission fractions.

Such fractions are estimated using two different methods. One method uses specific F_a values in combination with applicable emission factors for each combustion stage (*25, 31*). F_as are estimated using two fire behavior models, the Fire Emission Production Simulator (FEPS, http://www.fs.fed.us/pnw/fera/feps/) v1.0, and the Fire Characteristic Classification System (FCCS, http://www.fs.fed.us/pnw/fera/fccs). They can calculate separate F_a values during flaming, short-term smoldering, and residual smoldering combustion (RSC). According to the above definition of combustion stages, F_a values during both flaming and short-term smoldering in the two models are treated as flaming, and RSC is treated as smoldering. The other method is based on the diurnal temporal profile (showing hourly emission fractions) (*7*) and typical operation times for prescribed fires (*11, 13, 26*). The period between 10:00 and 17:00 is treated as flaming stage, and the rest as smoldering. Hourly emission fractions in the profile during the period designated for each stage are added to calculate emission fractions for each stage. In addition, the diurnal profile for total emission during both stages is split into two different profiles for flaming and smoldering. The hourly fractions during each stage defined above are renormalized to calculate the new diurnal profiles. The difference between simulations with and without specific emissions shows respective air quality impacts.

Results and Discussion

Burning Season. Different burning seasons feature varying meteorological conditions (ventilation, sunlight, and humidity) and levels of biogenic emissions. Monthly averages (and peaks for ozone) are calculated for $PM_{2.5}$ and ozone to compare air quality impacts during different burning seasons. $PM_{2.5}$ contributions from historical prescribed fires in 2002 averaged over the state of Georgia peak in March, being 4.8 $\mu g/m^3$ in March and 1.5 $\mu g/m^3$ in January, though local short-term contributions can be much higher. Corresponding contributions averaged for the Atlanta $PM_{2.5}$ nonattainment area are smaller, being 1.9 $\mu g/m^3$ in March and 0.7 $\mu g/m^3$ in January due to a longer distance from the prescribed fires. Source contributions of historical prescribed fires are negligible during May and July.

When emissions originally calculated for prescribed fires in March 2002 are applied to January, May, and July 2002, significant differences in their $PM_{2.5}$ contributions are simulated. The impacted regions and magnitudes diminish from January to July (Supporting Information, Figure 3). Such emissions lead to 7.3 $\mu g/m^3$ in January, 3.4 $\mu g/m^3$ in May, and 3.0 $\mu g/m^3$ in July of $PM_{2.5}$ averaged for the state of Georgia (Table 1). Impacts on $PM_{2.5}$ in the Atlanta nonattainment area are 2.0 $\mu g/m^3$ in January, 1.3 $\mu g/m^3$ in May, and 0.9 $\mu g/m^3$ in July. Decreased burning impacts during summer seasons can be explained by stronger vertical mixing and increased thunderstorm activity. Thunderstorms both increase ventilation and lead to pollutant rainout, evidenced, in part, by the increased rain in July versus May (11.8 cm versus 8 cm, http://www.sercc.com/climateinfo/monthly/state_avg_data/Georgia_prcp.html).

It is interesting to note a local discrepancy in the seasonal variation of $PM_{2.5}$ impacts from fires in the Okefenokee swamp, a Class I area located in the southeast of Georgia. When applying the same emissions from prescribed fires as in March 2002, the model shows fire contributions of 3.7 $\mu g/m^3$ of $PM_{2.5}$ in January, 1.4 $\mu g/m^3$ in March, 0.6 $\mu g/m^3$ in May, and 1.1 $\mu g/m^3$ in July (Table 1). This local difference (higher contribution during July than May) is partially explained by change of prevailing wind direction, and should be addressed in control strategy developments for protecting air quality in specific areas. In order to reduce $PM_{2.5}$ impacts, burning during summer seasons might be preferable for Georgia considering air quality impacts, alone, as tested for 2002.

TABLE 1. Source Contributions from Prescribed Fires in Georgia during January, March, May, and July 2002 Simulated with Two Sets of Emissions: (A) Simulations with Historical Prescribed Fires Emissions in the Respective Months and (B) Simulations with Historical March 2002 Emissions Applied to January, May, and July 2002 [a]

		January	March	May	July
A. Source contributions from historical prescribed fires in respective months of 2002					
$PM_{2.5}$	Georgia average	1.5	4.8	0.1	0.1
	Atlanta average	0.7	1.9	0.1	<0.1
	Okefenokee	2.7	1.4	<0.1	<0.1
Ozone	8-h average Atlanta	−0.01	0.30	0.02	<0.01
	8-h peak Atlanta	0.06	1.0	0.08	<0.01
	1-h maximum	2.2	16	0.73	0.98
B. Source contributions from the same prescribed fires emissions as in March 2002					
$PM_{2.5}$	Georgia average	7.3	4.8	3.4	3.0
	Atlanta average	2.0	1.9	1.3	0.9
	Okefenokee	3.7	1.4	0.6	1.1
Ozone	8-h average Atlanta	<0.01	0.30	0.40	0.27
	8-h peak Atlanta	0.18	1.0	2.4	0.48
	1-h maximum	12	16	21	23

[a] Monthly average $PM_{2.5}$ source contributions ($\mu g/m^3$) for Georgia and Atlanta refer to spatial averages of simulations for all grids within the state of Georgia and the Atlanta $PM_{2.5}$ nonattainment area, respectively. The Atlanta $PM_{2.5}$ nonattainment area includes 22 counties according to U.S. EPA designation on December 17, 2004. Values for Okefenokee refer to the simulations for the grid where the IMPROVE Okefenokee site (in Class I area) is located. Ozone source contributions are first calculated as monthly average and peak of daily maximum 8-h ozone (ppbv). Monthly average and peak ozone contributions are calculated for each grid cell, and then such contributions are averaged for all grids inside the Atlanta Metropolitan area (including 32 counties). They are referred to as "8-hr average Atlanta" and "8-hr peak Atlanta". Maximum 1-h ozone contributions in the whole modeling domain are also provided.

Prescribed fires have also been viewed as a source of ozone pollution during summer due to their NO_x, VOC, and CO emissions, and are addressed by different policies (e.g., burning bans in the Atlanta area during the summer O_3 season). In 2002, prescribed fires led to an increase of 1.0 ppbv during March in the monthly peak ozone concentrations averaged over the Atlanta metropolitan area (including 32 counties, a region with historical O_3 problems), with negligible contributions in May and July due to few fire activities (Table 1). Their ozone contributions in January are relatively small due to slow photochemical processes, though there were significant prescribed fires in that month. Slightly negative O_3 source contributions in January are observed when excess NO_x emitted from fires titrate O_3 and radicals.

When the same level of prescribed fires as in March 2002 is applied to other months, additional emissions lead to an increase of 0.18 ppbv monthly peak ozone in January averaged over Atlanta, 2.4 ppbv in May, and 0.48 ppbv in July (Table 1 and Supporting Information, Figure 3). Though O_3 formation potentials in July are the highest, less O_3 is formed by the additional prescribed fire emissions in July than in May and March. This is due to more rapid dispersion and reduced ozone sensitivities at high ozone levels in July. Since exceedance of the O_3 NAAQS is not observed during January and March in the Atlanta area, impacts of prescribed fires in these periods on O_3 are of less concern from a regulatory point of view, but may still have health implications.

TABLE 2. Typical Annual Burned Area (*A*), Fuel Consumption (F_a), and Emissions from Prescribed Fires with Different FRI in Georgia

FRI (year)	*A* (million acres)	F_a (tons/acre)	emissions (10^3 tons) CO	VOC	NO_x	NH_3	SO_2	PM_{10}	$PM_{2.5}$
2–5 (3.5)	0.86	4.0	519	24	11	2.3	3.1	51	43
2	1.51	2.9	668	31	14	3.0	4.0	65	56
3	1.00	3.5	535	25	12	2.4	3.2	52	45
4	0.75	4.4	498	23	11	2.2	3.0	48	42
5	0.60	5.0	455	21	10	2.0	2.7	44	38

The above ozone impacts averaged for the Atlanta area are relatively small, since ozone impacts of fires peak in their vicinity and most fires in Georgia are far from Atlanta. In nearby areas, historical fires in 2002 led to a maximum increase of 16 ppbv in 1-h ozone concentrations during March. When applying the same emissions from prescribed fires as in March 2002, the model shows maximum fire contributions of 12 ppbv during January, 21 ppbv during May, and 23 ppbv during July in 1-h ozone concentrations. The increasing trend from January to July agrees with corresponding photochemical potentials.

The same daily emissions from forest fires have been assumed in the simulations due to lack of data, even though not all meteorological conditions are equally preferred for burning. Impacts of prescribed fires on $PM_{2.5}$ concentrations vary significantly from day to day (Supporting Information, Figure 4). Violations of the 24-h $PM_{2.5}$ standard (35 $\mu g/m^3$) are simulated when source contributions from prescribed fires are large. Explicitly, probability of daily $PM_{2.5}$ source contributions from prescribed fires larger than 35 $\mu g/m^3$ is about 4% for grids in Georgia and days during January (Supporting Information, Figure 5). The probability is around 2% in March and very small in May and July. However, the probability of locations near a prescribed fire having such an exceedance is quite high. For the Atlanta $PM_{2.5}$ nonattainment area, March is the month with the highest daily $PM_{2.5}$ impacts. Exceedance days generally have poor dispersion characteristics or a wind direction toward the Atlanta area, and should be avoided in burning practice.

Relationships between air quality and forest fires during different seasons change with pollutants concerned, distance of fires and concerned regions, and wind directions. In order to meet requirements from varying air quality and ecosystem management goals, air quality impacts of both O_3 and $PM_{2.5}$ should be considered, along with other associated impacts on human health, visibility, climate, and ecosystem health. Different seasons are also associated with different fuel conditions, and thus corresponding emissions and air quality impacts vary. For example, fuel moisture contents are high during summer (the growing season). Higher fuel moisture contents are usually associated with less fuel consumption and more incomplete combustion (so higher emission factors). According to eq 1, emissions could either increase or decrease, as well as corresponding air quality impacts. Detailed information such as fuel moisture content by component is required to more fully understand air quality impacts of prescribed fires under different fuel conditions during different seasons. However, such information is rarely available. The results above are based on typical fuel conditions for prescribed fires, and mainly reflect impacts from different meteorological conditions during different seasons.

Burning Frequency. Ratios of fuel consumption (F_a) at different forest ages calculated by FOFEM are similar among different forest types and are further averaged to estimate

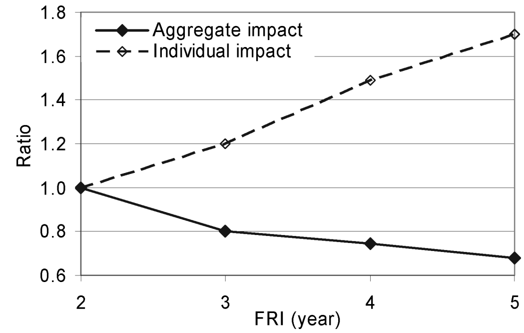

FIGURE 1. Trends of "individual" and "aggregate" prescribed fire emissions with different fire-return intervals (FRIs). "Individual impact" refers to the case when FRIs only changes for an individual fire and "aggregate impact" refers to the case when FRIs change for all forest area.

the ratios of F_a with different FRIs. While the F_a with the current average of 3.5-year FRI is 4 tons/acre, the F_a values with 2-, 3-, 4-, and 5-year FRIs are, respectively, 2.9, 3.5, 4.4, and 5.0 tons/acre. For an "individual" fire, emissions are proportional to F_a. Emissions from an individual forest fire with a 5-year FRI are approximately 72% larger than those from a fire of the same acreage if a 2-year FRI was employed. Their corresponding air quality impacts on local $PM_{2.5}$ have similar trends, since fire emissions mainly impact primary $PM_{2.5}$ species and are approximately linear to their impacts on $PM_{2.5}$ concentrations in current modeling. For an "aggregate" impact (Table 2), annual emissions from prescribed fires in Georgia with a 5-year FRI (38 thousand tons $PM_{2.5}$) are 32% less than those with a 2-year FRI (56 thousand tons $PM_{2.5}$), as less forest area is burned each year when a less frequent FRI is used. Less burned area offsets the increase of F_a per fire.

The opposing trends between "individual" and "aggregate" forest fire emissions and corresponding air quality impacts on $PM_{2.5}$ (Figure 1) pose a critical problem in forest and air quality managements in choosing an optimized FRI. Generally, a longer FRI is preferred to reduce long-term and regional air quality impacts, while a shorter FRI helps avoid intense short-term and local impacts. Specifically, a longer FRI can lower forest fire impacts on annual average $PM_{2.5}$ levels, however, increase chances of higher daily $PM_{2.5}$ levels. Thus, protecting acute exposure and responding to the new more stringent 24-h NAAQS, would suggest using more frequent burning (a smaller FRI), while attaining the annual standard would be more likely under less frequent burning strategies. In addition, the locations of forest fires are important for policy decisions. If forest fires are close to a sensitive area, short FRIs might be adopted to avoid acute deterioration of air quality though sacrificing longer term air quality. Longer FRIs might be employed to minimize long-term air quality impacts in relatively remote regions, where there is less concern about local episodic air quality impacts. Moreover, the increased risk of fire escaping with a longer FRI should also be considered in forest management.

Flaming and Smoldering. Prescribed fires emitted 560 tons/day $PM_{2.5}$ in Georgia during March 2002, using the VISTAS fire emission estimation method (*28*). Thirty percent (170 tons/day) of such emissions are released during smoldering, according to the diurnal temporal profile for prescribed fires and the designated periods for both combustion stages (Table 3). Corresponding $PM_{2.5}$ source contributions during both stages are mainly caused by primary $PM_{2.5}$ emissions. While some impacts on ozone from forest fires are simulated, such impacts are small in March and are not discussed here. Simulations with respective emissions

TABLE 3. $PM_{2.5}$ Emissions from Prescribed Fires in Georgia during Flaming and Smoldering and Corresponding Monthly Average $PM_{2.5}$ Source Contributions for March 2002 Using Two Different Methods: "Diurnal Profile" and "Specific F_a and E_f"[a]

	diurnal profile			specific F_a and E_f		
	total	flaming	smoldering	total	flaming	smoldering
emissions (tons/day)	560	390	170	560	250	310
Georgia	4.8	1.9	2.9	6.6	1.2	5.4
Atlanta	1.9	0.90	1.0	2.5	0.60	1.9

[a] The values for Georgia and Atlanta refer to spatial averages of simulations for all grids within the state of Georgia and the Atlanta $PM_{2.5}$ nonattainment area, respectively.

and diurnal temporal profiles (Supporting Information, Figure 6) during flaming and smoldering indicate that the total prescribed fires lead to 4.8 $\mu g/m^3$ of monthly average $PM_{2.5}$ for Georgia, 60% of which is caused by emissions during smoldering. In the Atlanta $PM_{2.5}$ nonattainment area, these fires lead to a monthly average $PM_{2.5}$ of 1.9 $\mu g/m^3$, with 53% from smoldering (Table 3).

When using specific F_a values and emissions factors for each combustion stage, we estimate that 60% of CO, 55% of VOC, 20% of NO_x, 70% of NH_3, 70% of SO_2, and 55% of $PM_{2.5}$ and PM_{10} emissions are from the smoldering stage. Explicitly, F_a values estimated by fire behavior models (FEPS and FCCS) for typical forest types in Georgia (e.g., loblolly pine and slash pine) indicate that approximately 38% of fuels are consumed during the smoldering stage. In comparison, fuel consumption during the smoldering stage was reported to be 38–44% in the Brazilian Amazon (*32*) and over 50% in temperate and boreal fires (*33*). Prescribed forest fires in Georgia mainly consume surface fuels; large woody and below-ground fuels are usually not consumed during smoldering. Therefore, less fuel is consumed during smoldering in Georgia, supporting the estimates by FEPS and FCCS. Even though estimated fractions of fuel consumption during the flaming stage are larger than those during smoldering, respective emission factors are much higher during smoldering for all pollutants except NO_x. As such, higher emission fractions during smoldering than flaming (except NO_x) are estimated. Such larger $PM_{2.5}$ emissions during smoldering (310 tons/day) increase estimated $PM_{2.5}$ source contributions from prescribed fires, by an additional 1.8 $\mu g/m^3$ averaged over Georgia and 0.6 $\mu g/m^3$ for the Atlanta area. The larger emissions during smoldering also lead to increased $PM_{2.5}$ contributions from smoldering: 81% for Georgia and 76% for Atlanta.

Large differences in estimated air quality impacts from forest fires during different combustion stages suggest the need to improve our understanding of emissions during the different stages. In addition, using the same diurnal profile for all fires is an approximation, recognizing that different fires will have different temporal characteristics. We have chosen a single one based upon the average found for prescribed fires. Information on fire-specific diurnal profiles is desired for further study. In spite of these uncertainties, air quality impacts per unit emissions during smoldering are larger than those during flaming, as dispersion is reduced during night when smoldering dominates. If techniques mentioned above (e.g., preprocessing fuels with a large potential to smolder using mechanical methods, keeping high moisture in large woody fuels, burning before precipitation, and rapid mop-up) are applied to reduce emissions during the smoldering stage, air quality impacts from forest fires can be significantly reduced. Due to an almost linear relationship between forest fire emissions and corresponding impacts on $PM_{2.5}$, a 50% reduction in smoldering emissions would lead to an approximately 1.5 or 2.7 $\mu g/m^3$ reduction in monthly $PM_{2.5}$ source contributions simulated for March 2002 in Georgia, using the two different methods. Similarly, such reduction can reduce approximately 0.5 or 1.0 $\mu g/m^3$ $PM_{2.5}$ in the Atlanta $PM_{2.5}$ nonattainment area.

Though impacts from other management practices or smoke reduction techniques are not discussed here, such impacts can be readily quantified using similar approaches. Different types of management practices can be applied at the same time and impact each other. For example, less frequent burning can lead to more fuels in larger sizes, which usually can not be consumed completely during flaming and contribute significantly to smoldering emissions. Impacts of controlling smoldering emissions are thus related to burning frequencies, and can be quantified using the approach developed in this study as long as there is information regarding fuel distributions by burning frequencies.

The quantified air quality impacts of prescribed fires in this study are for fires under typical fuel conditions and are based on meteorological conditions during 2002. Ignoring variability in fuel conditions and year-to-year variability in meteorological conditions can lead to uncertainties, however, the conclusion that air quality impacts of prescribed fires vary significantly with forest management practices, will not change. This conclusion is important for air quality management decisions. Due to the important role of fires in natural system and their significant impacts on air quality, cooperation between air quality and forest management specialists is crucial. This study provides information to bridge the two different areas, and highlights information that often is not available but would greatly enhance our understanding of air quality impacts from prescribed fires. Quantification of such impacts under different forest management practices is becoming critical to nonattainment designation, control strategy development, and effective air quality and ecosystem management. With the increased application of prescribed fires in forest management to reduce the risk of wildfires and improve ecosystem health, the methods and information provided can help avoid episodes leading to significant deterioration of air quality.

Supporting Information Available

Map of burned areas for prescribed forest fires in Georgia, daily $PM_{2.5}$ concentrations, and source contributions from prescribed fires and temporal diurnal profiles. This information is available free of charge via the Internet at http://pubs.acs.org.

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