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Assessment of Biomass Burning Emissions and Their Impacts on Urban and Regional $PM_{2.5}$: A Georgia Case Study

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Biomass burning is a major and growing contributor to particulate matter with an aerodynamic diameter less than 2.5 μm ($PM_{2.5}$). Such impacts (especially individual impacts from each burning source) are quantified using the Community Multiscale Air Quality (CMAQ) Model, a chemical transport model (CTM). Given the sensitivity of CTM results to uncertain emission inputs, simulations were conducted using three biomass burning inventories. Shortcomings in the burning emissions were also evaluated by comparing simulations with observations and results from a receptor model. Model performance improved significantly with the updated emissions and speciation profiles based on recent measurements for biomass burning: mean fractional bias is reduced from 22% to 4% for elemental carbon and from 18% to 12% for organic matter; mean fractional error is reduced from 59% to 50% for elemental carbon and from 55% to 49% for organic matter. Quantified impacts of biomass burning on $PM_{2.5}$ during January, March, May, and July 2002 are 3.0, 5.1, 0.8, and 0.3 μg m^{-3} domainwide on average, with more than 80% of such impacts being from primary emissions. Impacts of prescribed burning dominate biomass burning impacts, contributing about 55% and 80% of $PM_{2.5}$ in January and March, respectively, followed by land clearing and agriculture field burning. Significant impacts of wildfires in May and residential wood combustion in fireplaces and woodstoves in January are also found.

Introduction

Biomass burning includes the combustion of biomass fuels through either natural (e.g., wildfires) or planned processes (e.g., prescribed burning and residential wood combustion in fireplaces and woodstoves) and can emit large amounts of air pollutants. In the United States, estimates suggest that about 35% of the primary fine particulate matter ($PM_{2.5}$, i.e., PM with an aerodynamic diameter less than 2.5 μm) emissions come from biomass burning (*1*), and a large portion of these emissions are carbonaceous (70−95%). Significant impacts of biomass burning on ambient $PM_{2.5}$ concentrations have been found (*2*−*4*).

PM can adversely affect human health (*5*), and there is growing evidence that the carbonaceous component may be of particular concern (*6, 7*). In response, the U.S. Environmental Protection Agency (US EPA) promulgated a National Ambient Air Quality Standard (NAAQS) in 1997 for $PM_{2.5}$ and designated nonattainment areas in April 2005. Recently, the 24-h $PM_{2.5}$ standard was reduced to 35 μg m^{-3}, a level at which a large number of areas are expected to exceed. Visibility impairment is also of concern. Regulators are now faced with identifying effective strategies to lower $PM_{2.5}$ levels, including biomass burning impacts.

Chemical transport models (CTMs) can be used to simulate impacts from biomass burning sources, though their accuracy is affected by the quality of emission inputs. In this study, impacts of uncertainties in biomass burning emissions (including magnitude, temporal and spatial distribution, and speciation) on $PM_{2.5}$ simulations are investigated. Emission estimate shortcomings are identified by comparing simulations with observations and results from a receptor model. Improved emission estimates are used to simulate biomass burning impacts on $PM_{2.5}$ during different seasons, with particular focus on the state of Georgia, where biomass burning emissions are large.

Methods

Biomass burning sources investigated here include wildfires, prescribed burning, agriculture field burning, land clearing, and residential wood combustion in fireplaces and woodstoves (RWC). Their air quality impacts during four months in 2002 (January, March, May, and July) are simulated using the Community Multiscale Air Quality (CMAQ) Model v. 4.3 (*8*), a state of the science CTM. CMAQ model performance has been evaluated in a number of studies, and the performance varies depending upon the species simulated (*9, 10*). The four months chosen have very different levels of biomass burning emissions, as well as different meteorological conditions. Using month-long simulations allows a number of synoptic meteorological conditions to be captured during each period, as opposed to concentrating on periods of highest impact alone.

Air-Quality Modeling. Air quality is simulated with CMAQ using the SAPRC-99 chemical mechanism (*8*). Initial and boundary conditions are supplied by simulations from a 36-km resolution grid covering the continental United States (Unified RPO modeling domain), and results of the first 2 days for each month are discarded. Major fixes, particularly for mass conservation problems, are included in the version used. The inner modeling domain covers the southeastern United States with a 12-km grid and has 19 vertical layers reaching to about 15 km vertically, with a 36 m bottom layer. Meteorological fields for the episodes are generated using MM5 (*11, 12*). Emissions are developed from a variety of sources. In particular, the base emissions are from the Visibility Improvement State and Tribal Association of the Southeast (VISTAS) 2002 inventory (hereafter referred to as VISTAS2002) (*13*). Development of alternative biomass emission estimates are discussed below. Emissions are

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processed using the Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System v. 2.1 (*14*). Detailed description of plume rise processing for biomass burning emissions is documented in a VISTAS technical report (available at http://www.vistas-sesarm.org/documents/ENVIRON_Air_Quality_Modeling_Technical_Support_Document_11-14-07.pdf). When there are detailed temporal and spatial records for fires, fire emissions are allocated vertically according to size. Otherwise, emissions are assumed to be released in the first modeling layer (about 36m), since fires without detailed records are usually small and their emissions rarely directly reach above the planetary boundary layer due to their limited heat release. $PM_{2.5}$ simulations are evaluated by comparing model results with observations collected from a variety of monitoring networks (*15, 16*) (Supporting Information, Figure 1). Organic carbon (OC) observations are converted to organic matter (OM) using a 1.4 multiplication factor, which is widely used, though recent studies suggest higher values (*17*).

Biomass Burning Emission Inventories. Biomass burning emissions are obtained from three different inventories. The first inventory is VISTAS2002, providing emissions from all five biomass burning sources. The second inventory is the US EPA 2001 inventory (hereafter referred to as EPA2001) (*1*), in which emission estimates for four burning sources in Georgia, including wildfires, prescribed burning, agriculture field burning and land clearing, are used. The biomass burning emissions in EPA2001 are similar to those in most EPA inventories (2001 and earlier), which have been widely used in previous air quality modeling studies (*3, 9*). The third inventory is developed for this study as described below. Emissions from biomass burning sources that are not included in the second and third inventories are supplemented by VISTAS2002.

Emissions from the four biomass burning sources in Georgia have been estimated using different data in VISTAS2002 and EPA2001. Generally, burning emissions are calculated as the product of the amount of biomass consumed and the associated emission factors (ratios of the mass of pollutants emitted per unit biomass on a dry basis). In the VISTAS2002 inventory, the amount of biomass consumed is estimated from burned area records obtained by surveying state and federal agencies. Such records are location-specific for wildfires and by county for other types of burning. In the EPA2001 inventory, the amount of biomass consumed is estimated by burned area or crop production at the state or regional level and then allocated to the county level according to specific spatial surrogates (e.g., forest area). Emissions from the two inventories differ significantly (Figure 1). In EPA2001, emissions are almost equally distributed among the four biomass burning sources. Differences among the burning sources in VISTAS2002 are large, with prescribed burning contributing about 70% of the total emissions from the four sources combined.

Emissions from prescribed, agriculture field, and land clearing burning in Georgia are inventoried as annual totals in both VISTAS2002 and EPA2001. Annual emissions are processed to give gridded hourly emissions using source-specific monthly, daily, and diurnal temporal profiles obtained from VISTAS (*13*) (monthly profiles are shown in Supporting Information, Figure 2). Since the temporal profiles are the same for all counties in Georgia, the difference in burning seasons between counties is not captured. The effect of this difference is explored by a simulation using emissions with improved temporal resolution, which is developed from detailed monthly burned area (*18*) and the same fuel consumption and emission factors as used in VISTAS2002. This inventory is hereafter referred to as "MONTHLY". Given the same annual burned area, the MONTHLY inventory has the same annual total emissions as the VISTAS2002, though the distribution of emissions is more consistent with burning records.

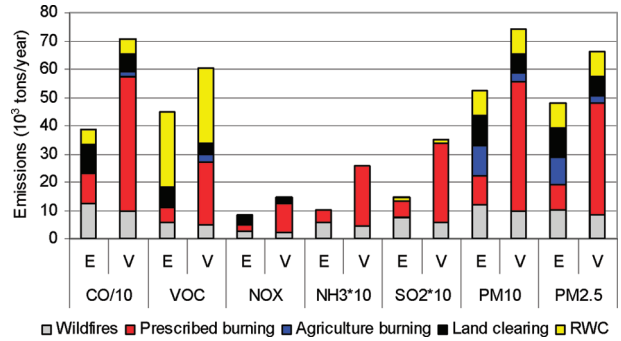

FIGURE 1. Biomass burning emissions in Georgia during 2002 (10^3 tons/year). CO emissions are divided by 10 and NH_3 and SO_2 emissions are multiplied by 10 for better visual effect. Emissions from all biomass burning sources except RWC are obtained from two inventories: EPA2001 (labeled as E) and VISTAS2002 (labeled as V). All RWC emissions are from the VISTAS2002 inventory. There are no estimates for NH_3 and SO_2 emissions from agriculture burning and land clearing or NH_3 emissions from RWC.

TABLE 1. $PM_{2.5}$ Speciation Profiles for Different Biomass Burning Sources[a]

	sources	POA	EC	SO_4^{2-}	NO_3^-	Other
wildfire and prescribed burning	EPA	0.770	0.160	0.020	0.002	0.048
	updated	0.898	0.056	0.001	0.015	0.030
residential wood combustion	EPA	0.566	0.108	0.004	0.002	0.321
	updated	0.865	0.108	0.004	0.003	0.020
agriculture field burning/land clearing	EPA	0.670	0.040	0.010	0.003	0.277

[a] Emissions of each species are calculated by multiplying the total $PM_{2.5}$ emissions and corresponding fractions in the speciation profile. Values from EPA are the profiles recommended by EPA in 2005 (http://www.epa.gov/ttn/chief/emch/speciation/). Updated values are recalculated using recent measurements.

Updated $PM_{2.5}$ Speciation for Biomass Burning Emissions. $PM_{2.5}$ emissions from biomass burning are speciated into five components [primary organic aerosol (POA), elemental carbon (EC), sulfate (SO_4^{2-}), nitrate (NO_3^-), and other unspecified mass] using speciation profiles obtained from US EPA (*19*). In the EPA profiles, EC and POA are the major components of biomass burning, accompanied by negligible sulfate and nitrate levels (Table 1). POA fractions in the EPA profiles have been calculated by multiplying the OC measurements by a factor of 1.2 to account for the other elements bound to C (*20, 21*). However, molecular level analyses of POA indicate that the POA/OC ratio for wood burning is about 1.9 (*17*). In addition, these analyses only measure less polar organics and do not account for water-soluble species, which comprise 20−80% of organic aerosols (*22*). Since water-soluble organic compounds tend to have higher POA/OC ratios than less water-soluble organic compounds, the POA/OC ratio for wood burning may be larger than 1.9.

Here, the fractions of EC (f_{EC}), SO_4^{2-} ($f_{SO_4^{2-}}$), NO_3^- ($f_{NO_3^-}$), and other unspecified mass (f_{other}) for wildfires/prescribed burning and RWC are updated using recent field and laboratory measurements (*21, 23−26*). The corresponding POA fractions (f_{POA}) are recalculated by a mass balance method:

$$f_{POA} = 1 - f_{EC} - f_{SO_4^{2-}} - f_{NO_3^-} - f_{other} \quad (1)$$

The updated fractions are presented in Table 1. Measurements for species other than EC, OC, SO_4^{2-}, and NO_3^- (e.g., potassium and chloride) are treated as other unspecified mass. The speciation profiles for agriculture field burning and land clearing are not updated due to lack of data.

Assessment of Biomass Burning Emissions and Their Impacts on $PM_{2.5}$. Biomass burning emissions are first assessed by comparing CMAQ simulations during January 2002 (a month when emissions from the various biomass burning sources are all significant) with ambient observations. Simulations using different biomass burning emission inventories and the updated $PM_{2.5}$ speciation profiles are investigated. Corresponding biomass burning impacts on $PM_{2.5}$, which are estimated by comparing CMAQ simulations with and without biomass burning emissions, are also compared with results from chemical mass balance (CMB) model 7.0 (*27*), a receptor model.

In CMAQ simulations with and without biomass burning emissions, the same meteorological conditions are assumed. As demonstrated by Hu et al. (*28*), plumes from two prescribed fires were captured successfully using the same system without accounting for potential feedbacks between the emissions and meteorology. The biomass burning emissions have a limited impact on regional meteorology and overall plume transport. This is, in part, due to their limited size. However, this is a topic for further study.

CMB results are not affected by the same emission uncertainties that impact CMAQ simulations (e.g., estimates of the total mass of emissions) and are used to help identify shortcomings in biomass burning emissions. For reference, it is noted that CMB results are affected by errors in the estimates of the composition of source emissions, observational errors, and atmospheric transformation. CMAQ simulations with the updated emissions suggested by CMB analysis are further compared with the observations to address uncertainties in the CMB analysis (*29*).

In the CMB modeling, 31 individual organic markers and three elemental species (EC, silicon, and aluminum) were employed both to characterize the emissions from the seven sources and to quantify their impacts at the receptors (*30*). CMB analysis results at JST (an urban SEARCH station at Jefferson Street, Atlanta, Georgia) during January 2002 are used here.

Adjusted biomass burning emissions based on the above assessment for January 2002 are then applied to the other three months (March, May, and July of 2002) to study corresponding biomass burning impacts. It is assumed that the biases in biomass burning emissions among these months are consistent. Impacts from each of the five individual biomass burning sources are computed by following source-specific POA emissions in one CMAQ simulation. These POA emissions are followed using model tracers and are treated as nonreactive species that go through similar physical processes as other primary carbonaceous aerosol species (*31*).

Results

$PM_{2.5}$ Simulations with Different Biomass Burning Emission Inventories. As expected from the amount of information used in the three inventories, the spatial distributions of POA (the major $PM_{2.5}$ component for biomass burning) emissions and concentrations are quite different during January 2002 (Figure 2a). POA emissions with EPA2001 are more intense in the Atlanta area than with VISTAS2002, which have denser emissions in southwestern Georgia (Figure 2a). POA concentrations simulated with EPA2001 are 0.7 $\mu g\ m^{-3}$ higher than those with VISTAS2002 for the Atlanta $PM_{2.5}$ nonattainment area, but 0.6 $\mu g\ m^{-3}$ lower than VISTAS2002 on average for Georgia. Average POA concentrations for Atlanta and Georgia are similar for simulations using VISTAS2002 and MONTHLY, despite the significant differences in the spatial distributions of POA emissions and concentrations (Figure 2a). Model performance statistics (*32*) find that OM concentrations simulated using VISTAS2002 agree better with the observations than those using EPA2001 (Figure 2b). Such statistics for simulations using VISTAS2002 and MONTHLY are similar. The negligible difference in model performance, in part, is because most observations are outside of the regions affected by the updated monthly emissions from biomass burning (Figure 2a and Supporting Information, Figure 1).

$PM_{2.5}$ Simulations with Updated $PM_{2.5}$ Biomass Burning Speciation Profiles. EC and OM concentrations simulated using the EPA profiles during January 2002 are high (Figure 2b, cases 1−3). Simulations with the updated speciation profiles led to decreased EC and increased OM concentrations. Model performance for EC is significantly improved, while OM deteriorates (Figure 2b, case 4). High sensitivities of model performance statistics to the speciation profiles for biomass burning also indicate the importance of developing more representative profiles. Time series of daily simulated and observed EC and OM concentrations during January 2002 show the same overestimation of EC and OM (Supporting Information, Figure 3)

Comparison with CMB Analysis. The CMB analysis for January 2002 suggests that wood burning contributes 2.76 $\mu g\ m^{-3}$ of $PM_{2.5}$ at JST, in comparison to CMAQ-simulated levels of 13.2 $\mu g\ m^{-3}$. High bias in biomass burning emissions is the only possible explanation for the large overestimation in burning contributions simulated by CMAQ. Simulated POA tracer concentrations for individual biomass burning sources show that RWC has the largest impact on $PM_{2.5}$ simulations, contributing about 90% (9.7 $\mu g\ m^{-3}$) of total POA concentrations from biomass burning at JST despite its small annual emissions (Figure 1). An approximate 90% reduction in Georgia RWC emissions is suggested so as to minimize the difference between simulated source contributions and results from CMB analysis. EC and OM simulations with the reduced RWC emissions agree better with observations (Figure 2b, case 5 and Supporting Information, Figure 3), though discrepancies between simulations and observations still remain due to other sources of error not investigated here. The abnormally large impact of RWC at JST is due to concentrated emissions in the Atlanta area. RWC emissions are calculated using regional wood consumption and spatially allocated to each county using the number of houses or fireplaces/woodstoves. As suggested by the above comparison, such spatial surrogates are not representative of actual RWC activities.

Performance Summary. Updated emissions suggested by the above assessment [including monthly county-level emissions for prescribed burning, agriculture field burning, and land clearing (MONTHLY), a 90% reduction of RWC emissions, and the improved speciation profiles] are applied to the other episodes (March, May, and July 2002). Using those updates, the overall performance of simulated $PM_{2.5}$ species during these episodes are well within recent performance suggestions (*32*), except for OM during May and July (Table 2 Supporting Information, Figure 4). Summertime low biases of OM are common in the current CMAQ model, and it is likely due to an underestimation of secondary organic aerosol (SOA) formation (*10*). Time series at JST and YRK (a rural SEARCH station at Yorkville, Georgia) show that simulated daily OM and EC follow observations well (Supporting Information, Figure 3).

Biomass Burning Impacts on $PM_{2.5}$. Biomass burning emissions contribute 3.0, 5.1, 0.8, and 0.3 $\mu g\ m^{-3}$ of $PM_{2.5}$ (averaged over the whole modeling domain), constituting 25%, 40%, 9%, and 4% of the total $PM_{2.5}$ during January, March, May, and July, respectively (Figures 3 and 4). Analyses using receptor models indicated similar seasonal trends (*33*).

TABLE 2. Air Quality Modeling Performance of Total and Speciated $PM_{2.5}$ during January, March, May, and July 2002[a]

month	species	NOBS (no.)	OBS_mean ($\mu g\ m^{-3}$)	SIM_mean ($\mu g\ m^{-3}$)	MB ($\mu g\ m^{-3}$)	ME ($\mu g\ m^{-3}$)	NMB (%)	NME (%)	MFB (%)	MFE (%)
January	EC	260	0.84	0.86	0.01	0.49	1.5	58.1	3.9	50.2
	OM	260	4.83	5.34	0.51	2.42	10.5	50.0	11.9	48.9
	SO_4^{2-}	255	2.29	2.20	−0.10	0.85	−4.3	37.0	−3.4	37.5
	NO_3^-	253	0.97	1.66	0.70	0.97	72.0	100.5	12.9	80.2
	NH_4^+	249	1.00	1.24	0.24	0.54	23.7	54.1	13.3	49.3
	$PM_{2.5}$	253	10.89	14.75	3.86	5.79	35.5	53.2	26.0	45.5
March	EC	266	0.58	0.55	−0.03	0.27	−5.6	46.1	−11.3	48.5
	OM	266	4.31	4.68	0.38	2.56	8.7	59.5	−11.4	58.3
	SO_4^{2-}	285	3.54	2.49	−1.05	1.28	−29.5	36.3	−32.4	43.3
	NO_3^-	284	0.75	1.19	0.44	0.82	58.3	109.3	−8.0	86.1
	NH_4^+	283	1.23	1.20	−0.03	0.47	−2.7	37.8	−5.4	40.8
	$PM_{2.5}$	279	11.64	11.88	0.24	4.40	2.1	37.8	−4.5	38.5
May	EC	253	0.62	0.40	−0.22	0.30	−35.0	48.5	−44.7	58.5
	OM	253	5.66	2.74	−2.92	3.17	−51.6	56.0	−69.9	77.1
	SO_4^{2-}	263	4.96	3.65	−1.31	1.83	−26.5	36.9	−34.5	45.0
	NO_3^-	256	0.54	0.23	−0.31	0.44	−57.2	81.8	−108.6	127.2
	NH_4^+	235	1.58	1.21	−0.37	0.59	−23.3	37.6	−30.9	45.8
	$PM_{2.5}$	252	15.71	10.15	−5.56	6.32	−35.4	40.2	−45.0	50.4
July	EC	333	0.54	0.42	−0.13	0.29	−22.9	52.3	−23.2	60.0
	OM	334	5.63	2.23	−3.40	3.58	−60.3	63.5	−86.2	91.5
	SO_4^{2-}	352	5.61	4.78	−0.83	2.18	−14.8	38.8	−18.1	40.4
	NO_3^-	346	0.43	0.06	−0.37	0.38	−86.9	87.7	−164.1	165.3
	NH_4^+	335	1.60	1.09	−0.51	0.68	−31.7	42.7	−35.6	49.5
	$PM_{2.5}$	342	16.97	10.67	−6.29	7.34	−37.1	43.2	−48.4	54.9

[a] NOBS, number of available observations; OBS, observations; SIM, simulations; MB, mean bias; ME, mean error; NMB, normalized mean bias; NME, normalized mean error; MFB, mean fractional bias; MFE, mean fractional error.

Biomass burning impacts during January and March are concentrated in southwestern Georgia, where large amounts of prescribed burning are conducted. Such impacts in the Atlanta $PM_{2.5}$ nonattainment area are 2.3 and 3.9 $\mu g\ m^{-3}$ during January and March, respectively (Figure 4).

More than 80% of $PM_{2.5}$ concentrations caused by biomass burning are primary (mainly POA), and the rest is secondary in origin (Figure 4). During January and March, biomass burning contributes about 66% and 86%, respectively, of the POA domainwide, and about 48% and 70% for the Atlanta $PM_{2.5}$ nonattainment area. Smaller impacts in the Atlanta area are results of longer distances to biomass burning emissions. NH_3 emissions from biomass burning also lead to increased NH_4^+, contributing about 2% of $PM_{2.5}$. Extra NH_3 and NO_X emissions from biomass burning led to increased NO_3^- as well (about 4% of $PM_{2.5}$). No significant increases in SO_4^{2-} levels are found, contrary to findings in a recent study in Texas (*34*). Spatial distributions of speciated $PM_{2.5}$ concentrations and source contributions from biomass burning during January 2002 are also provided (Supporting Information, Figure 6).

Individual Biomass Burning Impacts on $PM_{2.5}$. Simulated POA concentrations from individual biomass burning sources indicate that prescribed burning is the largest single biomass burning source over most of the modeling domain and the Atlanta $PM_{2.5}$ nonattainment area during most periods, except for May 2002 (Figure 5 and Supporting Information, Figure 7). The POA contributions from prescribed burning peak in March (2.9 $\mu g\ m^{-3}$ for the whole modeling domain), followed by January (1.2 $\mu g\ m^{-3}$), May (0.1 $\mu g\ m^{-3}$), and July (0.1 $\mu g\ m^{-3}$). Prescribed burning concentrates in the southwest of Georgia, though influencing much of the region. Overall impacts from wildfires are much smaller in spite of severe local impacts and peaks in May. Source contributions peak in March in southern Georgia for agriculture burning and are more spatially sporadic for land clearing. RWC has the largest impact in January and is centered in the Atlanta area, which is characterized with higher population densities.

Discussion

Model simulations of biomass burning impacts on $PM_{2.5}$ are crucial in understanding sources that cause elevated $PM_{2.5}$ concentrations and can help develop effective control strategies and smoke management plans (*35*). The simulated impacts are highly dependent on emission estimates, as indicated by simulations using the three inventories (EPA2001, VISTAS2002, and MONTHLY). Similar $PM_{2.5}$ emissions from each biomass burning source in EPA2001 lead to similar contributions; however, using more representative inventories (VISTAS 2002 and MONTHLY) leads to prescribed burning having the dominant impact. In addition, due to the inherently large temporal and spatial variation of biomass burning emissions, detailed information on not only the magnitude of emissions but also when and where burning actually occurs is important for simulating burning impacts. Here, MONTHLY has the most detailed information.

Simulated impacts suffer from likely errors in emission estimates, and both can be improved by comparing with findings from other analyses (e.g., CMB analysis, POA tracers, and EC/OC observations). While CMB analysis is not affected by the uncertainty in the mass emission rate that impacts CMAQ simulations, such analyses seldom differentiate between impacts from individual biomass burning sources, due to similar source profiles, and provide less information about the spatial and temporal distribution of impacts. Total burning impacts estimated by CMB are here used to help evaluate corresponding simulations. This comparison suggests about 90% reduction in biomass burning emissions impacting Atlanta in January. POA tracer analysis which is used to assess impacts from individual burning sources indicates that only RWC emissions (instead of total biomass burning emissions) should be reduced by 90%. This analysis assumes that wintertime biomass burning impacts on $PM_{2.5}$ in an urban area are mainly primary, though recent analyses suggest that SOA formation from forest fires may be significant, even in February (*28*). However, the processes leading to SOA formation involve release of isoprenoids from

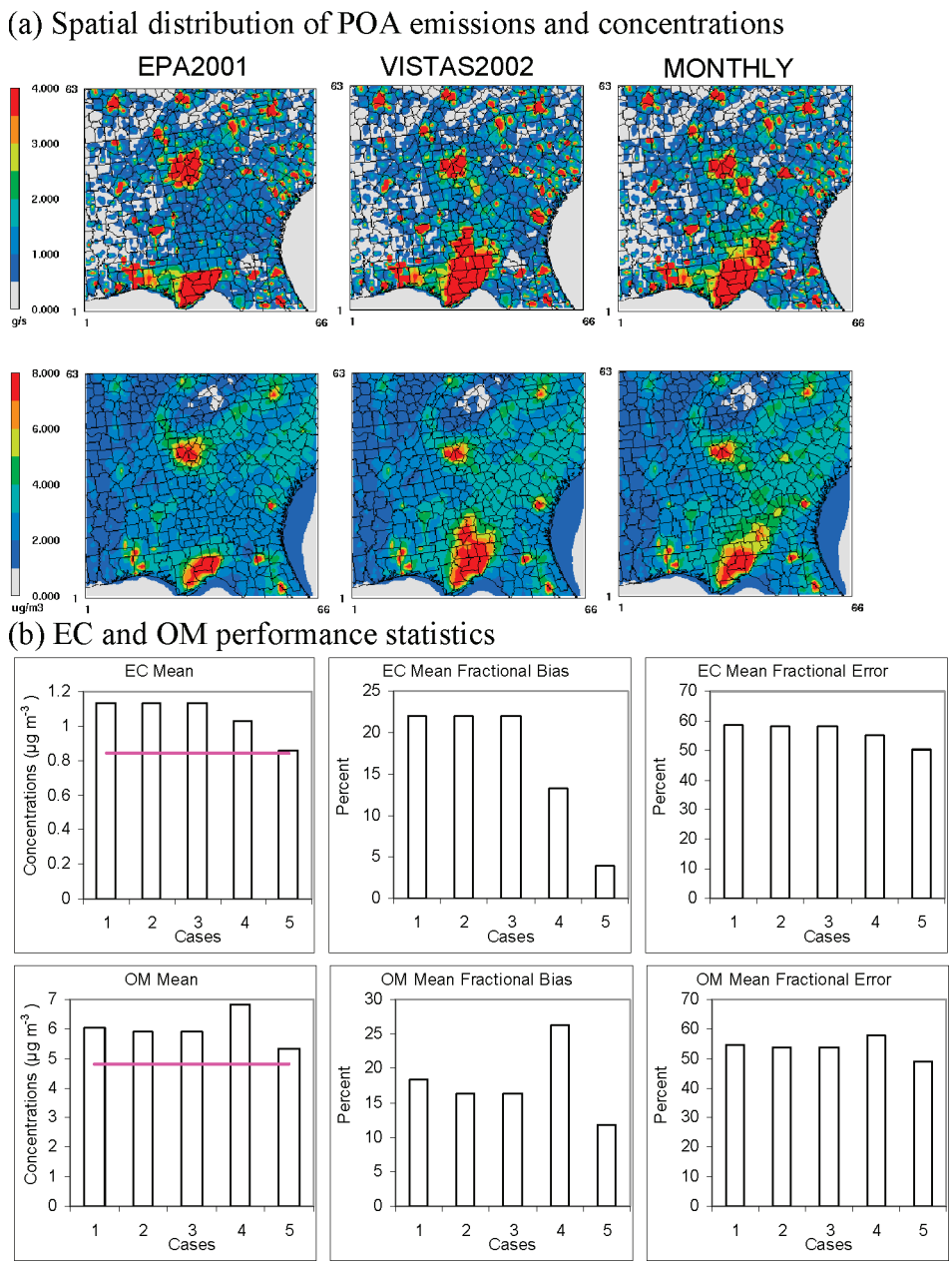

FIGURE 2. $PM_{2.5}$ simulations with different biomass burning emission inventories. (a) Monthly average POA emissions (upper row, g/s) and simulated concentrations (lower row, μg m^{-3}) using three different biomass burning emission inventories during January 2002. Monthly average values are used since allocation of emissions from prescribed burning, agriculture field burning, and land clearing are not day-dependent by default in SMOKE. (b) Mean, mean fractional bias, and error for EC and OM with different emission inventories and speciation methods during January 2002: case 1, simulations with the EPA2001 emission inventory; case 2, simulations with the VISTAS2002 emission inventories; case 3, simulations with the MONTHLY emission inventory; case 4, simulations with the MONTHLY emission inventory and updated speciation profiles; and case 5, simulation with the MONTHLY emission inventory and updated speciation profiles, as well as 90% reduction in RWC emissions. The horizontal lines are the mean of EC and OM observations.

leaves and needles during forest fires, while the excess OC found is linked to RWC, which laboratory studies suggest would not have similar emissions.

Impacts of biomass burning on annual $PM_{2.5}$ are estimated using the simulated impacts during the four months, assuming that impacts per unit emissions during February, April, and June are the average of those during adjacent modeled months, that unit impacts during November and December are similar to that of January, and that the summer months are similar to that of July. This leads to biomass burning causing 1.2 μg m^{-3} of $PM_{2.5}$ in the Atlanta $PM_{2.5}$ nonattainment area during 2002 (the annual $PM_{2.5}$ NAAQS is 15 μg m^{-3}). In addition, biomass burning can lead to a violation of 24-h $PM_{2.5}$ NAAQS, especially with promulgation of the new 24-h $PM_{2.5}$ standard (35 μg m^{-3}). For example, simulated impacts of biomass burning on 24-h $PM_{2.5}$ concentrations during March 2002 are higher than 35 μg m^{-3} in more than 20 counties in Georgia. Given that such $PM_{2.5}$ simulations are based on emissions from prescribed burning, agriculture field burning, land clearing, and RWC without day dependence, the possibility of violating 24-h $PM_{2.5}$ NAAQS might be higher. Due to the large impacts of biomass burning on $PM_{2.5}$, air quality and forest land managers are tasked with identifying practices to reduce related air quality impacts. Fortunately, most biomass burning impacts in the southeastern U.S. are from planned burning processes (e.g.,

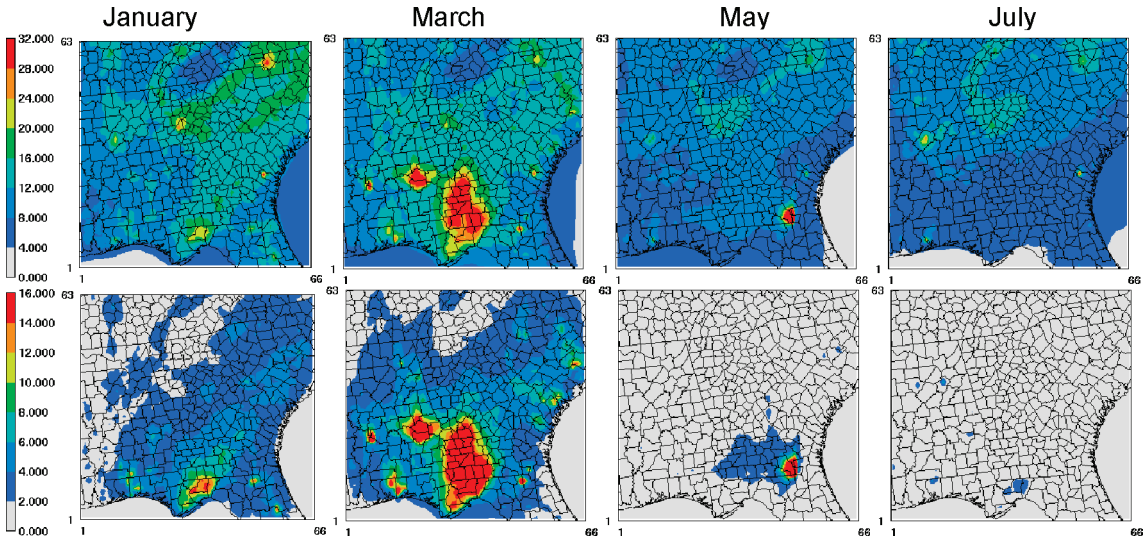

FIGURE 3. Monthly average $PM_{2.5}$ concentrations (first row, $\mu g\ m^{-3}$) and source contributions from biomass burning (second row, $\mu g\ m^{-3}$) during January, March, May, and July 2002. Source contributions are presented using different scales for more detailed interpretation (see also Supporting Information, Figure 5).

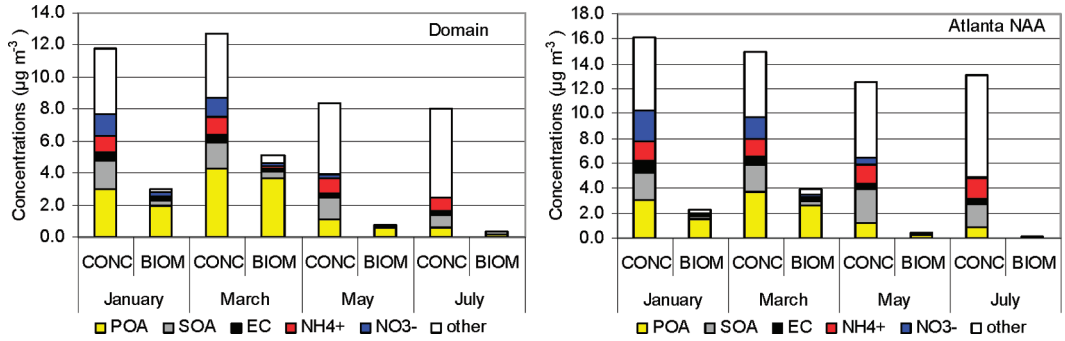

FIGURE 4. Monthly average speciated $PM_{2.5}$ concentrations (CONC, $\mu g\ m^{-3}$) and source contributions from biomass burning (BIOM, $\mu g\ m^{-3}$) during January, March, May, and July 2002. Domain refers to results averaged over all modeling grids and Atlanta NAA refers to results averaged over all grids within the Atlanta $PM_{2.5}$ nonattainment area.

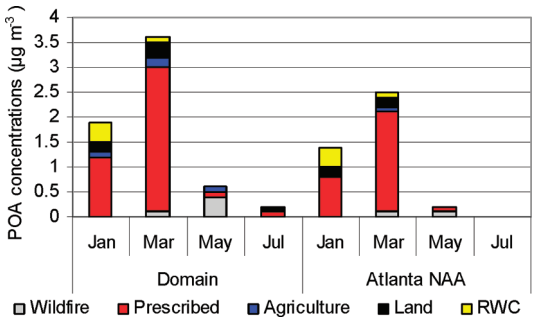

FIGURE 5. Monthly average POA source contribution ($\mu g\ m^{-3}$) from individual biomass burning sources during January, March, May, and July 2002. Domain refers to results averaged over all modeling grids and Atlanta NAA refers to results averaged over all grids within the Atlanta $PM_{2.5}$ nonattainment area.

prescribed burning). Unlike wildfires, prescribed burning is planned by forest managers and its air quality impacts can be reduced by adopting proper forest management practices (*36*). Practices like choosing appropriate burning periods and frequencies, as well as controlling smoldering emissions, are desired for better air quality.

State and Regional Planning Organizations (RPOs) in the U.S. currently use CTMs to project future air quality for regulatory applications (*37*), including performing attainment demonstrations for areas that are not currently meeting the annual $PM_{2.5}$ NAAQS. Inaccurate estimates of POA and EC emissions can have an impact on the regulatory decisions that are made to help bring nonattainment areas (such as Atlanta) back into attainment. According to EPA modeling guidance (*37*), future $PM_{2.5}$ concentrations are calculated by scaling the measured base-year speciated $PM_{2.5}$ concentrations by a species-specific relative response factor (RRF). Species-specific RRFs are calculated by taking the ratio of the future modeled concentration to the base-year modeled concentration for each individual $PM_{2.5}$ species. Lower RRFs will result in lower projected future $PM_{2.5}$ concentrations. If POA and EC emissions and modeled concentrations of OC and EC are overestimated, the resulting RRFs can be biased, and the simulated benefits of controls will be incorrect. Correcting the RWC emissions and speciation profiles will allow Atlanta to more accurately determine if the annual $PM_{2.5}$ NAAQS will be met in the future.

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Supporting Information Available

Maps of observation sites, model performance statistics, and detailed simulation results. This information is available free of charge via the Internet at http://pubs.acs.org.

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