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Atmospheric Environment 39 (2005) 5787–5798
ATMOSPHERIC ENVIRONMENT
www.elsevier.com/locate/atmosenv

Airport related emissions and impacts on air quality: Application to the Atlanta International Airport

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Received 3 January 2005; accepted 9 May 2005

Abstract

In the last decade, air traffic has increased dramatically with a significant increase in emissions. Our goal is to quantify the impact of aircraft emissions on regional air quality, especially in regards to $PM_{2.5}$ and ozone. Here the focus is on Hartsfield–Jackson Atlanta International Airport which is the busiest airport in the world based on passenger traffic.

First, aircraft $PM_{2.5}$ emissions are estimated based on the Smoke Number (SN) by using the "first order" method. The Emissions and Dispersion Modeling System (EDMS) is used for gaseous species. $PM_{2.5}$ emissions are estimated once based on the characteristic SN and a second time using the mode-specific SN. Further, aircraft emissions are processed in two ways: (1) allocating the emissions at the airport itself, and (2) by accounting for flight paths, mode, and plume rise.

When the more conservative emission estimates are used (i.e, the characteristic SN estimates allocated to the airport), results suggest that Hartsfield–Jackson airport can have a maximum impact of 56 ppb on ozone with a 5 ppb average impact over most of the Atlanta area. $PM_{2.5}$ impacts are also estimated to be quite large with a maximum local impact of $25\,\mu g\,m^{-3}$. Impacts over most of the Atlanta area are less than $4\,\mu g\,m^{-3}$. The second set of emissions with detailed spatial allocation leads to a less intense ozone impact with a maximum of 20 ppb and an average of less than 1 ppb. $PM_{2.5}$ impacts, in this case, are about $1\,\mu g\,m^{-3}$ within a radius of 16 km around the airport. The difference in these two results shows the importance of how aircraft emissions are treated. The impacts on ozone and $PM_{2.5}$ of ground support equipment at the airport are smaller compared to the aircraft impacts, with a maximum impact of 2 ppb for ozone and $9\,\mu g\,m^{-3}$ for $PM_{2.5}$.

Keywords: Aircraft emissions; Regional air quality; Ozone; Fine particulate matter

1. Introduction

Air transportation plays a substantial role in global economic activity and in the last decade air traffic has increased dramatically: 47 percent between 1991 and 2000 (DOT, 2003). The US environmental protection agency (EPA) estimated that airport emissions (i.e., aircraft and ground support equipment) in 1999 as compared to 1970 are up more than 80 percent for volatile organic carbon (VOC) and nitrogen oxide (NO_X) emissions doubled (EPA, 2004). Airport

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1352-2310/$ - see front matter
doi:10.1016/j.atmosenv.2005.05.051

emissions now make up about 2 percent of total nonroad emissions (EPA, 2004).

In the last decade, research has focused on quantification of the impact of aircraft emissions on the ozone layer, greenhouse gases, and the climate impact of aerosols (Brasseur et al., 1998; Schroder et al., 1998; IPCC, 1999; Brock, 2000; Kentarchos and Roelofs, 2002). The Intergovernmental Panel on Climate Change (IPCC) concluded that while the impact of aircraft emissions on stratospheric ozone depletion is modest, the impact on ozone formation in the upper troposphere and contrail and cloud formation might be significant (IPCC, 1999).

There have been a few studies of aircraft emissions. Some of these studies focused on measurement of aircraft emissions using remote sensing and Fourier-transform infrared (FTIR) emission spectroscopy (Heland and Schafer, 1998; Popp et al., 1999; Schafer et al., 2003). Other studies focused on modeling the local and regional impact of aircraft emissions. Yu et al. (2004) utilized the nonparametric regression method to estimate the average concentration of pollutants such as sulfur dioxide (SO_2) and carbon monoxide (CO) as a function of wind direction and speed near Hong Kong and Los Angeles airports based upon observational data. However, their monitoring sites were very close to highways and were most likely affected by roadway vehicle emissions. Although their method might be useful to determine the impact of pollutants emitted solely from airport operations, it would not be accurate enough for others particularly if chemical reactions play an important role.

In another study, Moussiopoulos et al. (1997) quantified the potential impact of emissions from a planned airport on the Athens basin using an Eulerian dispersion model. They showed an increase in NO_X concentrations in the vicinity of the airports, though the contribution to regional air quality was found to be minimal. Pison and Menut (2004) quantified the impact of aircraft emissions on ozone concentrations over the Paris area. For this purpose they used a mesoscale air quality model, CHIMERE, with $150 \times 150\,km^2$ resolution and a vertical extension of 3100 m. They found that, during the night time, ozone levels decrease as much as 10 ppb in the vicinity of airports due to the titration effect of NO_X emissions from aircraft. During the daytime, ozone levels increase as much as 10 ppb in NO_X-limited areas. None of these previous studies quantified the impact of aircraft emissions on fine particulate matter (i.e., particulate matter with aerodynamic diameters less-than 2.5 μm, $PM_{2.5}$).

The goal of this study is to quantify the impact of aircraft emissions on regional air quality, especially in regards to $PM_{2.5}$ and ozone. Here we focus on the Hartsfield–Jackson Atlanta International Airport which is the busiest airport in the world based on passenger traffic (Airports Council International, 2003). Hartsfield–Jackson serves the metropolitan Atlanta area where air quality continues to violate national standards and will likely remain in non-attainment in the near future for both ozone and $PM_{2.5}$. Emissions from mobile, industrial and utility sectors along with regionally high levels of ozone and $PM_{2.5}$, are the major contributors to the area's air pollution.

2. Methodology

A detailed modeling approach was used to quantify the impact of aircraft and ground support equipment on air quality from Hartsfield–Jackson International Airport, which is located south of Atlanta between Fulton and Clayton counties (Fig. 1). First, a detailed inventory was developed for aircraft and other emissions. Then, air quality simulations were performed to relate these emissions to regional air quality around Atlanta using the Community Multi-scale Air Quality Model (CMAQ) (Byun and Ching, 1999). August 11-20 2000 was selected as the focus episode because prior modeling identified this period to be critical for planning purposes (Hu et al., 2003, 2004a). This episode has a number of days characteristic of high air pollution levels. Meteorological data for this episode as well as emission data for sources other than aircraft were already available (Hu et al., 2003; Unal et al., 2003).

2.1. Data preparation

Emissions inventories have long been noted for being one of the most, if not the most, uncertain aspect of air quality modeling (e.g., Sawyer et al., 2000). This uncertainty inhibits accurate air quality modeling (e.g., Hanna et al., 1998), effective air quality management, and detailed understanding of the mechanisms impacting the formation and fate of particulate matter in the atmosphere. For example, in modeling studies, inaccurate emissions can lead to either poor model performance or, worse, to the introduction of compensatory errors being introduced (NARSTO, 2000). Many air quality management programs, such as trading, control strategy assessment and permitting depend on accurate knowledge of source emissions rates. Understanding the formation and transport of pollutants requires knowing the properties and rates of source emissions.

Emissions from aircraft can be estimated based on the number of landing and takeoff cycles (LTO). Aircraft engines emit pollutants at different rates during the various phases of operation, such as: idling, taxing, takeoff, climbing, cruising, and approach for landing. Different emissions estimates must be employed for commercial air carrier, air taxi, general aviation, and military aircraft. The Emissions and Dispersion

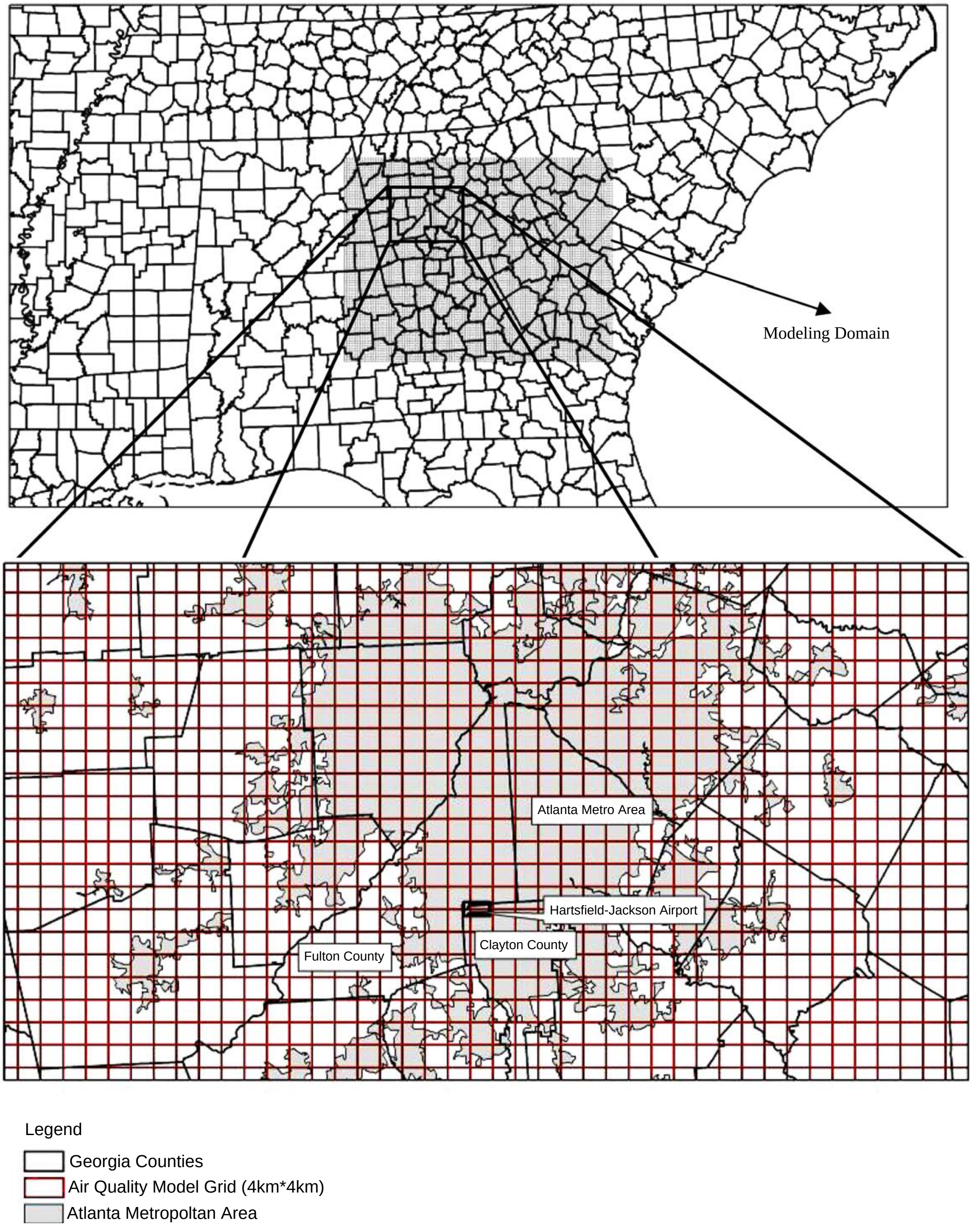

Fig. 1. The modeling domain (shaded area) and Atlanta Hartsfield–Jackson International Airport.
Simulations with coarser grid resolutions (36 km over the Eastern US and 12 km over the region shown) provided boundary conditions (Hu et al., 2004a).

Modeling System (EDMS) Version 4.01 (FAA, 2001) is widely used for estimating emissions from aircraft and ground support equipment (GSE). This model can calculate emissions for volatile organic compounds (VOCs), nitrogen oxides (NO_X), carbon monoxide (CO), and sulfur oxides (SO_X). Currently, EDMS does not calculate particulate matter (PM) emissions, coarse or fine, for aircraft although it estimates PM emissions for ground support equipment (FAA, 2001). This is expected to change as of Version 4.3.

As part of the Fall Line Air Quality Study (FAQS), an emissions inventory was prepared for each source category in Georgia (Unal et al., 2003). In this work VOC, NO_X, CO, and SO_X emissions for commercial aircraft were calculated using EDMS Version 4.01 and PM emissions were scaled to fuel use.

We reviewed the literature to improve the estimates of fine PM emissions from aircraft. Recently, the Federal Aviation Administration (FAA) has developed a first-order approximation (FOA) method (Wayson et al., 2003) for estimating $PM_{2.5}$ emissions from aircraft. In this method a relationship was developed that relates $PM_{2.5}$ emissions to Smoke Number (SN) and fuel flow rate (FF) as follows:

$$\mathrm{EI} = 0.6 \times (\mathrm{SN})^{1.8} \times (\mathrm{FF}), \tag{1}$$

where EI is the emission index in mg of $PM_{2.5}$ emitted per second, SN the Smoke Number, and FF the fuel flow rate in kg s^{-1}.

SN is a dimensionless number that identifies the smoke level and is determined by means of the loss of reflectance of a filter used to trap smoke particles from a prescribed mass of exhaust per unit area of filter (Wayson et al., 2003). The data leading to the FOA formula were collected by probes placed at fixed distances behind the engines. It is possible that emissions that are in the vapor phase at the probe orifice may condense into particles farther away. If this is the case, the amount of particles measured by the probe would be less than the total amount of particles caused by the aircraft engine. Therefore FOA may lead to an underestimation of $PM_{2.5}$ emissions from aircraft. An important part of applying Eq. (1) is to find the correct SN and FF for individual engines in each operational mode (i.e., idling, takeoff, climb-out, and approach). Here the International Civil Aviation Organization (ICAO) database was utilized to determine the SN and FF for each aircraft and engine type as available. In the ICAO database different statistics, such as average, standard deviation and characteristic value are provided for SN. Characteristic value for SN is the mean of the values of all the engines tested and corrected to the reference standard engine and reference ambient conditions divided by the coefficient corresponding to the number of engines tested as explained by the ICAO manual (Wayson et al., 2003). We have estimated total PM emissions at Hartsfield–Jackson Airport using two different methods. In the first method we utilized only the characteristic value SN. For aircraft which do not have the corresponding characteristic value in the ICAO database, we utilized a database average value. $PM_{2.5}$ emissions calculated with this method (i.e., characteristic value method) are 70 tons yr^{-1}.

In the second method we utilized the mode specific SN values. For most aircraft SN is only measured for the takeoff mode. We developed a statistical relation, through linear regression, between the takeoff mode and other modes using the data for the engines tested in other modes, and utilized this relation to estimate the SN for modes other than takeoff. The regression equations between the takeoff and other modes are as follows:

$$\mathrm{SN}_{\text{Climb-Out}} = 0.86 \times \mathrm{SN}_{\text{TakeOff}}, \qquad R^2 = 0.91, \tag{2}$$

$$\mathrm{SN}_{\text{Approach}} = 0.51 \times \mathrm{SN}_{\text{TakeOff}}, \qquad R^2 = 0.57, \tag{3}$$

$$\mathrm{SN}_{\text{Idle}} = 0.41 \times \mathrm{SN}_{\text{TakeOff}}, \qquad R^2 = 0.37, \tag{4}$$

For aircraft which do not have the corresponding takeoff SN in the ICAO database, we utilized the database average value. $PM_{2.5}$ emissions calculated with this method (i.e., Mode Specific Method) are 27 tons yr^{-1}. It is estimated that the highest contributor to total emissions is the climb-out mode with 65 percent. Takeoff and idling modes come after climb-out with 17 and 12 percent, respectively. The approach mode makes about 6 percent of the total emissions.

It should be noted that Eq. (1) provides aircraft emissions per LTO. We estimated annual emissions by utilizing aircraft specific LTO data for Hartsfield–Jackson airport in the year 2000 (Nissalke, 2003); the total was 423,423 LTOs.

For our assessment of airport-related air quality impacts, we prepared four different sets of emissions inventories; these are: (i) without aircraft emissions; (ii) with aircraft emissions estimated using the characteristic value method; (iii) with aircraft emissions estimated using the mode specific method; and (iv) with mode specific aircraft emissions but without GSE emissions (Table 1).

To put the Hartsfield–Jackson airport emissions in perspective, we compared aircraft and GSE emissions with total emissions from other sources in the Atlanta non-attainment region (i.e., 13-county region consisting of Cherokee, Clayton, Cobb, Coweta, DeKalb, Douglas, Fayette, Forsyth, Fulton, Gwinnett, Henry, Paulding and Rockdale counties). For each pollutant, contributions of each source category and the total over the non-attainment region are listed in Table 2. The largest contribution of aircraft emissions is 2.6 percent for NO_X. For $PM_{2.5}$, the contribution is 0.13 percent when aircraft emissions are estimated with the characteristic value method and 0.05 percent with the mode specific method. GSE contribution is 0.19 percent for NO_X, and it is 0.05 percent for $PM_{2.5}$.

Table 1
Emissions at Hartsfield–Jackson international airport used in this study

Pollutant	No-aircraft ($tons\,yr^{-1}$)		Characteristic value ($tons\,yr^{-1}$)		Mode specific ($tons\,yr^{-1}$)		No-GSE ($tons\,yr^{-1}$)	
	Aircraft	GSE	Aircraft	GSE	Aircraft	GSE	Aircraft	GSE
CO	0	584	5204	584	5204	584	5204	0
NO_X	0	343	4910	343	4910	343	4910	0
SO_2	0	46	473	46	473	46	473	0
VOC	0	43	1013	43	1013	43	1013	0
PM_{10}	0	30	101	30	62	30	62	0
$PM_{2.5}$	0	27	70	27	27	27	27	0

Table 2
Percent contribution of different source categories in Atlanta non-attainment region

	EGU[a] point (%)	Non-EGU point (%)	Area (%)	Non-road (%)	On-road[b] (%)	Biogenic (%)[b]	Hartsfield Airport (%)			Total ($tons\,yr^{-1}$)
							Aircraft		GSE	
							Characteristic value	Mode specific		
CO	0.1	0.3	14.5	31.8	53.4	0.0	0.45	0.45	0.05	1,147,438
NH_3	0.1	0.0	63.1	3.2	33.6	0.0	0.00	0.00	0.00	12,034
NO_X	8.6	6.3	10.4	18.9	55.4	0.5	2.65	2.65	0.19	185,374
PM_{10}	0.5	0.5	96.1	1.5	1.4	0.0	0.06	0.03	0.02	181,601
$PM_{2.5}$	0.8	1.2	90.2	4.5	3.3	0.0	0.13	0.05	0.05	54,210
SO_2	81.9	6.2	3.2	4.6	4.0	0.0	0.63	0.63	0.06	75,276
VOC	0.0	2.2	12.5	6.0	14.5	64.8	0.24	0.24	0.01	430,277

[a]EGU stands for electric generating utilities.
[b]These values are estimated from daily emission totals.

2.2. Emissions modeling

An air quality model like CMAQ needs hourly, gridded, and speciated emissions. Here Sparse Matrix Operator Kernel Emission (SMOKE) (CEP, 2003) is used for spatiotemporal distribution and speciation. Historically SMOKE treated aircraft emissions as point sources and did not distribute them spatially. However, recently SMOKE started to give the user an option to spatially allocate airport-emissions to grid cells based upon a ''point location'' (CEP, 2003; Strum et al., 2004). Furthermore, SMOKE uses a default temporal profile for all aircraft source category code (SCC) types. In order to better resolve and distribute emissions from aircraft, we developed a new emissions processing framework. This framework involves the following parts: temporal distribution; three-dimensional (3-D) spatial distribution; and speciation.

Temporal distribution: Hourly emissions profiles were developed from activity profiles for Hartsfield–Jackson airport (Nissalke, 2003). There were three different temporal profiles: monthly; weekly; and diurnal. Some of these temporal profiles differ significantly from default temporal profiles used in SMOKE. Fig. 2 compares the default diurnal profile of aircraft activity used in SMOKE to the actual Hartsfield–Jackson airport data. The default profile assumes more flight activity between midnight and 9:00 am. On the other hand, the actual evening flight activity remains high from 17:00 until midnight while the SMOKE default profile assumes a decreasing trend in evening flight activity.

Spatial distribution: Emissions from aircraft are distributed in 3-D space especially in takeoff, climb-out, and approach modes. However, they are generally put into the first layer of the air quality model in a single grid cell that coincides with the airport location. Here, emissions were distributed using two different methods. In the first method emissions were injected into the three first-layer cells where Hartsfield–Jackson airport run-ways are located. In the second method, actual aircraft location data were used to distribute the emissions horizontally and vertically. The location data consisted of 3-D coordinates typical for different aircraft types during takeoff and landing in August 2000. For example, according to the path shown in Fig. 3, a typical DC-9 aircraft elevates very fast during takeoff. Assigning all of its takeoff emissions to the first layer,

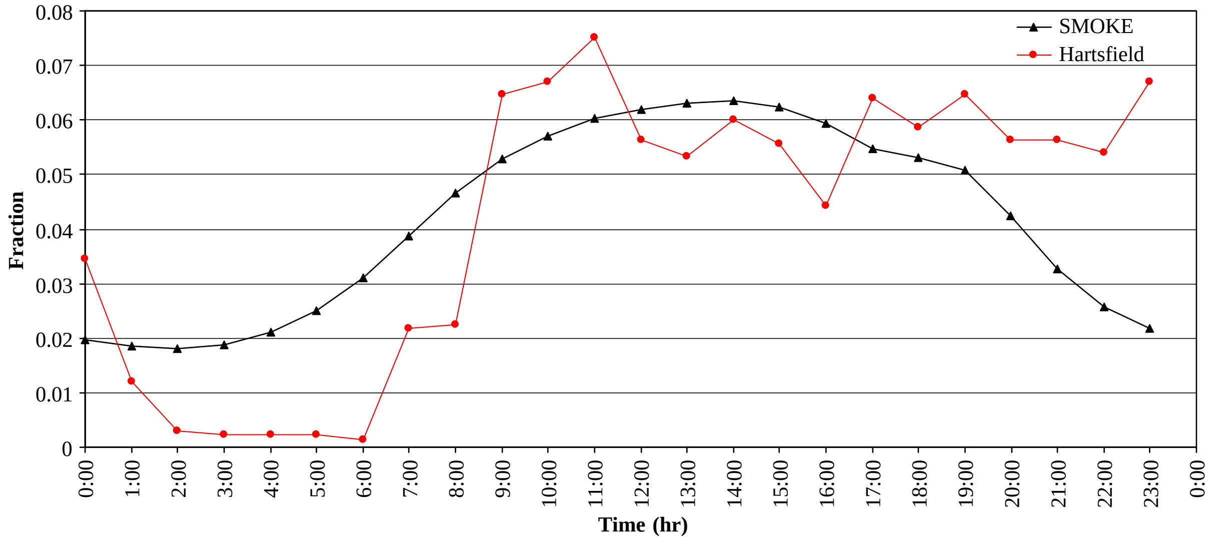

Fig. 2. Diurnal flight activity profile for Hartsfield–Jackson airport and the default profile used in SMOKE for commercial aircraft (SCC = 227502000).

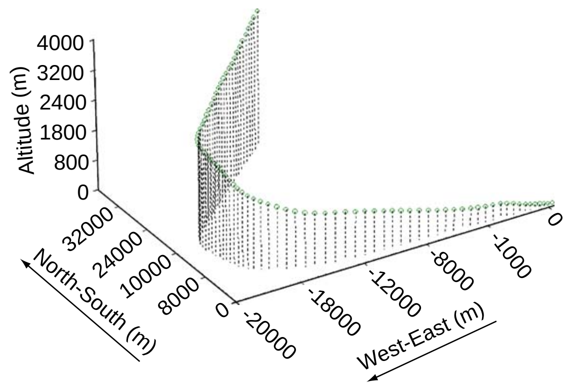

Fig. 3. Typical 3-D path for DC-9 s for takeoff and climb-out modes at Atlanta Hartsfield–Jackson airport in August 2000 (Zero indicates the location where the runway ends).

which is 19 m from the ground, and in one horizontal cell (4 km × 4 km) would be an oversimplification of the actual emission distribution. Here a more realistic spatial distribution is obtained by utilizing a different typical pathway for each aircraft. Plume rise is also considered by adding a fixed plume rise factor of 12 m, as suggested by Wayson et al. (2004), to the aircraft altitude. 3-D spatial distribution was applied only to mode specific emissions.

Speciation: In order to speciate PM emissions we utilized the default profiles for aircraft emissions used in SMOKE (CEP, 2003). In these profiles, most of $PM_{2.5}$ emissions are assumed to be elemental carbon (66%) followed by organic carbon (29%). There is a small amount of sulfate (4.6%) and trace amount of nitrate (0.32%).

3. Results

In this study, Version 4.3 of CMAQ is used (Byun and Ching, 1999). However, improvements to the aerosol module ISORROPIA (i.e., equilibrium in Greek) that were introduced in Version 4.4 by EPA are also used. In addition, a mass conservation and advection correction scheme were applied (Hu et al., 2004b). The grid consisted of 102 columns by 78 rows of $4 \times 4\,km^2$ grid cells, covering the State of Georgia. There were 13 layers in the vertical. Detailed information on modeling configuration can be found in Hu et al., (2004a). In this study we have conducted air quality model runs for five different scenarios. These scenarios are summarized in Table 3. It should be noted that Scenario 1 is selected as the basecase for determining the impact of aircraft emissions. Other scenarios are utilized to determine the impact of 3-D spatial distribution, airport specific temporal profile and different emissions estimates. For GSE we utilized Scenario 5 as the basecase.

Results from applying CMAQ with the four sets of emissions to the 11–20 August 2000 episode show the estimated impact of Hartsfield–Jackson airport (Figs. 4–6). Areas impacted by Hartsfield–Jackson's aircraft emissions and the level of impact changes every hour due to changes in meteorology, flight paths, and other parameters that affect O_3 and $PM_{2.5}$ levels. Though sensitivities were calculated for each hour and visually analyzed, only the maximum and average sensitivities for each grid cell are shown here. Maximum of different cells do not necessarily occur at the same time. For example, while the maximum for one cell may occur on August 16 the maximum of another cell may occur on August 18.

Table 3
Air quality modeling scenarios used in this study

Scenario	Aircraft emissions	GSE emissions	Spatial distribution for aircraft	Temporal distribution for aircraft
1[a]	No Aircraft	GSE	—	—
2	Characteristic value	GSE	First layer	Hartsfield Specific
3	Mode specific	GSE	First layer	Hartsfield Specific
4	Mode specific	GSE	3-D	Hartsfield Specific
5[b]	Mode specific	No GSE	3-D	Hartsfield Specific

[a]Basecase for determining the impact from aircraft emissions.
[b]Basecase for determining the impact from GSE emissions.

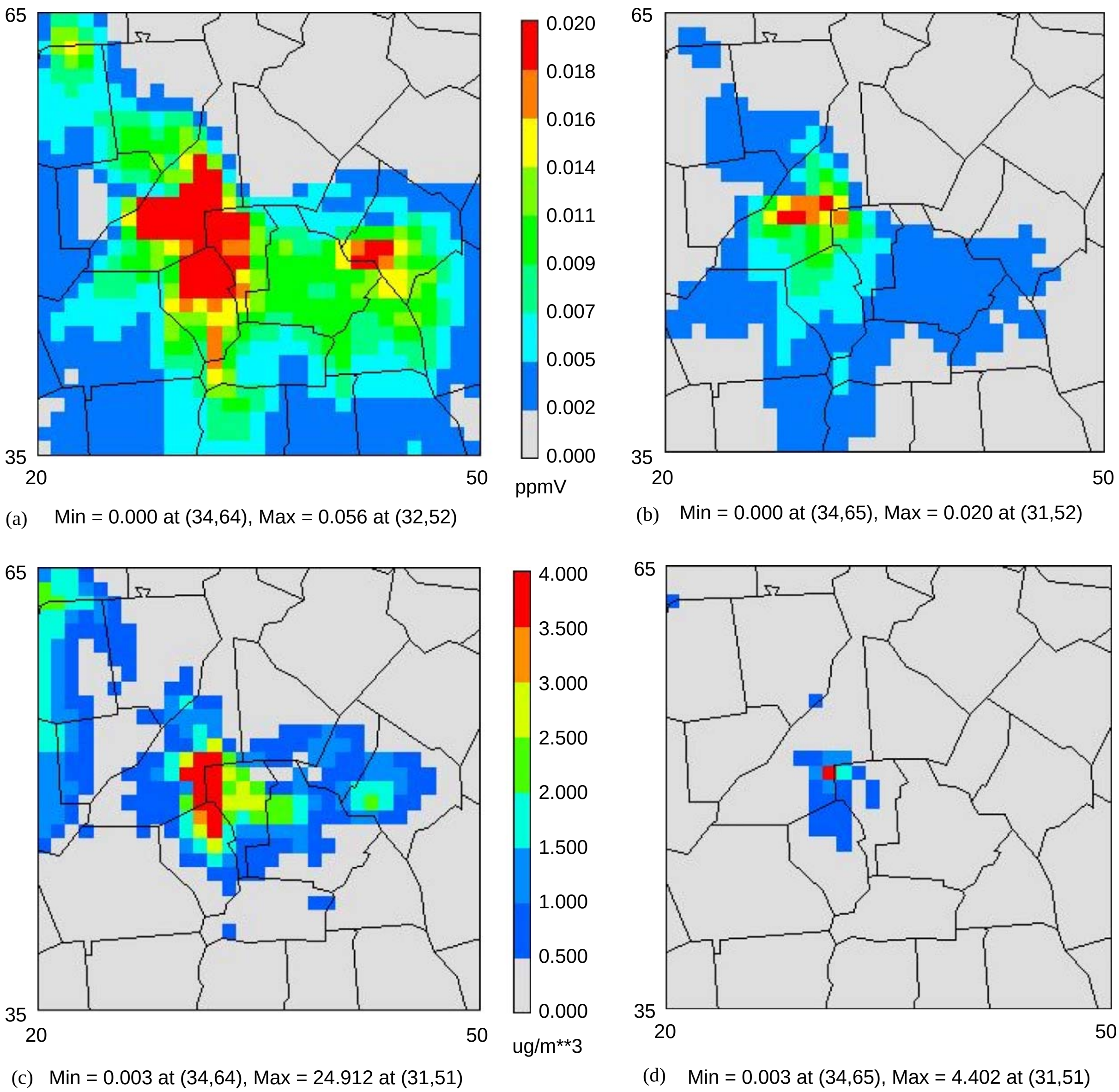

Fig. 4. Maximum sensitivity of regional concentrations to aircraft emissions from Hartsfield–Jackson Atlanta International Airport during the August 11–20, 2000 period: (a) O_3 to characteristic value emissions; (b) O_3 to mode specific emissions; (c) $PM_{2.5}$ to characteristic value emissions; and (d) $PM_{2.5}$ to mode specific emissions.

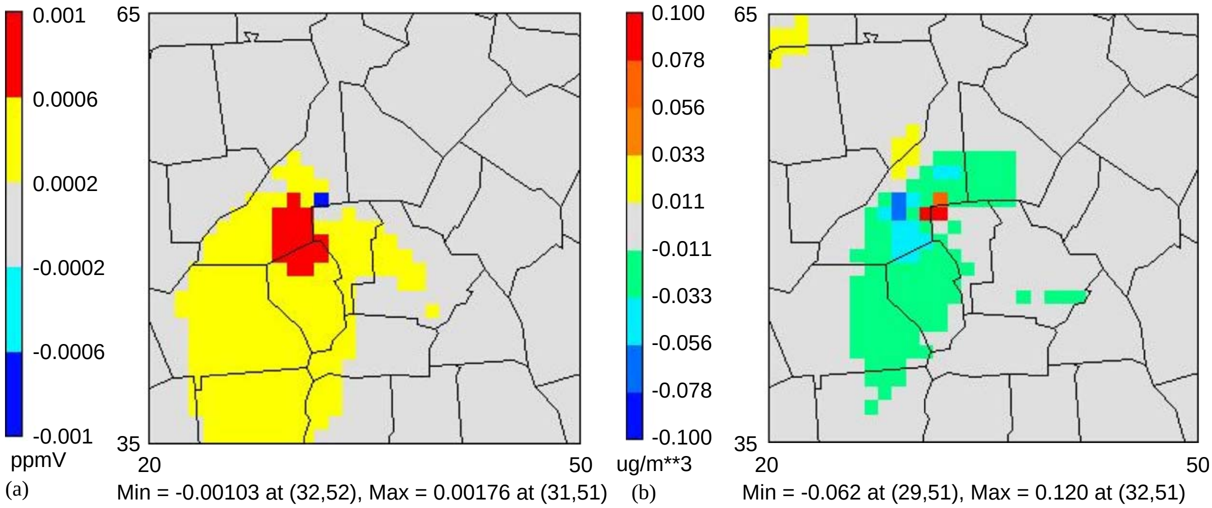

Fig. 5. Average sensitivity of regional concentrations to mode specific aircraft emissions from Hartsfield–Jackson Atlanta International Airport during the August 11–20, 2000 period: (a) O_3; (b) $PM_{2.5}$.

For the characteristic value emissions, assigning all of the emissions to the first layer cells at the airport, the maximum increase in ozone is 56 ppb in the grid cell directly to the north of Hartsfield–Jackson airport (Fig. 4a). Cell-maximum increases in ozone values are more than 20 ppb in the vicinity of Hartsfield airport (i.e., within 12-km distance to the north and 20-km distance to the west and south of the airport). Within a 32-km radius, there are areas to the southwest and southeast where ozone concentrations increase by as much as 10 ppb. Ozone levels increase by 5 ppb or more in the southern part of Metropolitan Atlanta.

While the total emissions of ozone precursors are the same their horizontal and vertical distribution is different using the mode-specific inventory, leading to marked differences in simulated O_3 impacts. In this case, there is little impact to the northwest and a smaller impact (as compared to characteristic value case) to the south (Fig. 4b). A cell-maximum impact of 15 ppb or more is found within a distance of 16 km to the west of the airport. Approximately within an 8-km distance to the north, south, and southwest of the airport the impact of aircraft emissions is about 10 ppb.

Using the characteristic value emissions led to a maximum increase in $PM_{2.5}$ of 25 $\mu g\,m^{-3}$ in one of the three Hartsfield–Jackson airport cells (Fig. 4c). $PM_{2.5}$ increases more than 3.5 $\mu g\,m^{-3}$ in the vicinity of the airport (i.e., within a 12 km distance to the south and 4 km distance to the west of the airport). Within an 8-km distance to the east and southeast of the airport, there are areas where $PM_{2.5}$ increases by as much as 3 $\mu g\,m^{-3}$. There is also an increase of more than 2 $\mu g\,m^{-3}$ in $PM_{2.5}$ concentrations to 64-km northwest of the airport.

In the mode-specific inventory, both the magnitude and the spatial distribution of aircraft $PM_{2.5}$ emissions are different. Recall that the mode-specific $PM_{2.5}$ emissions were about 2.5 times less than the characteristic-value emissions (Table 1). As a result, there is much less impact in all directions (Fig. 4d). Maximum impact, 4.4 $\mu g\,m^{-3}$, occurs at the same cell as the characteristic-value case (i.e., one of the three Hartsfield–Jackson airport cells). There is an increase of 1 $\mu g\,m^{-3}$ to the south and southwest of the airport within a distance of 16 km.

Average impact of aircraft emissions during the episode (10 days) was also computed for O_3 and $PM_{2.5}$ with mode-specific inventory. Overall there is an increase of approximately 1 ppb of O_3 to the southwest of the airport up to a distance of approximately 20 km (Fig. 5a). Within a distance of approximately 50 km to the south and southwest of the airport there is an average increase of 0.2 ppb or more. There is a slight decrease in ozone in the grid cell just to the north of the airport, which is due to the nighttime decrease in ozone levels. There is an average increase of about 0.1 $\mu g\,m^{-3}$ in $PM_{2.5}$ in grid cells where Hartsfield–Jackson airport is located (Fig. 5b). There is a slight decrease in $PM_{2.5}$ to the southwest, north and west of the airport, mainly due to the reduction in the oxidation of NO_2 and SO_2 as increased NO_X scavenges O_3 and OH.

Here we also quantified the impact of 3-D spatial distribution of emissions by simulating a case where mode-specific emissions are injected in the first layer cells at the airport and subtracting from its results those of the spatially distributed case (Fig. 6). Both maximum and average impacts are estimated. Maximum ozone

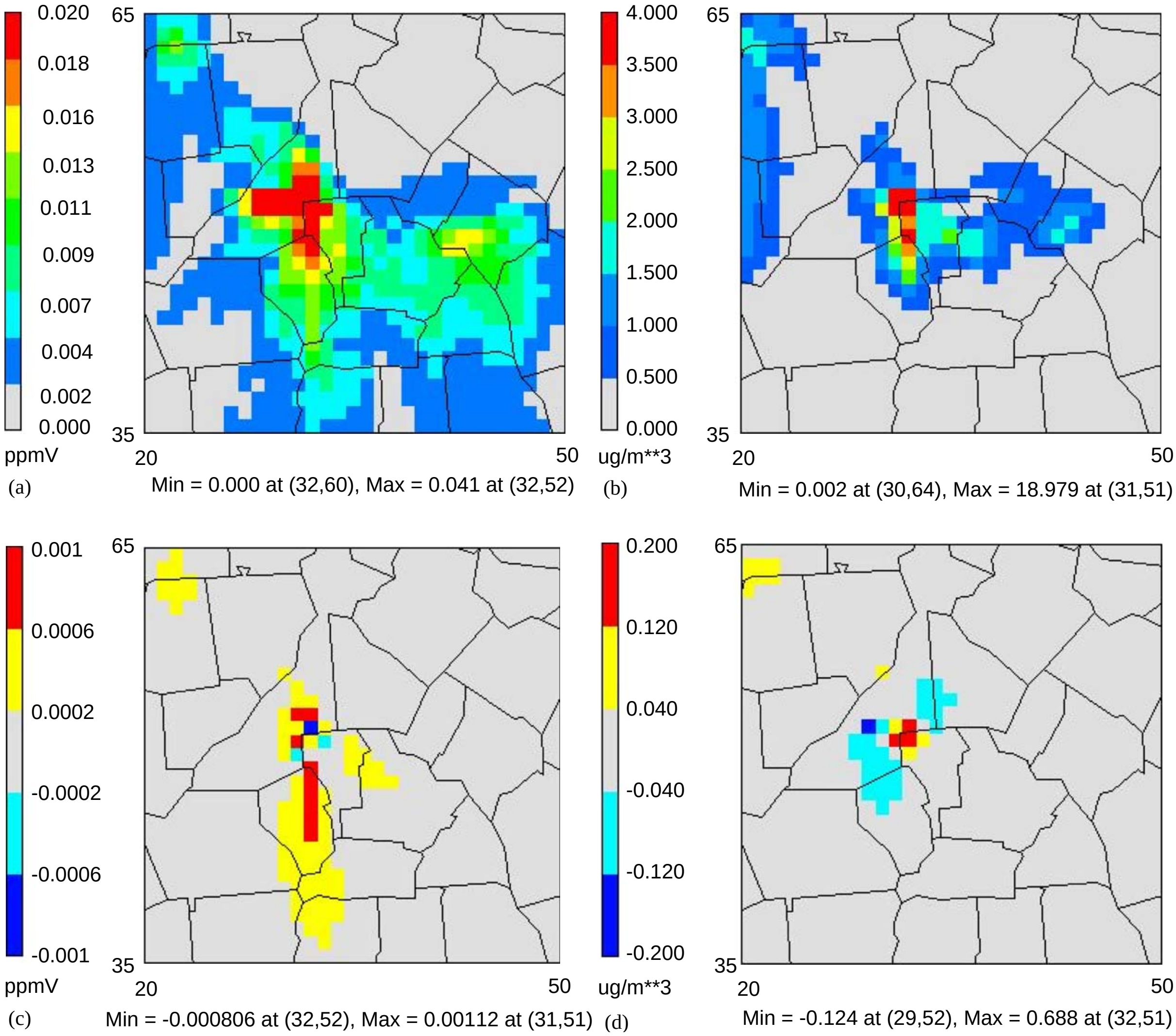

Fig. 6. Difference of injecting aircraft emissions (mode specific) into the first-layer airport cells from distributing them in 3-D space: (a) maximum O_3; (b) maximum $PM_{2.5}$; (c) average O_3; (d) average $PM_{2.5}$.

and $PM_{2.5}$ levels are significantly lower in the case of 3-D spatial distribution since emissions are considerably diluted. For ozone, the maximum difference of 41 ppb occurs at one of three airport cells. There is a difference of 10 ppb or more in ozone within a distance of 12 km in west, north and south directions. For the spatially distributed emissions ozone levels are less than first layer injected emissions case by as much as 5 ppb to the south within a distance of approximately 64 km. The maximum difference in $PM_{2.5}$ is 19 $\mu g\,m^{-3}$ and occurs at one of three airport cells. Within a distance of approximately 20 km to the south, $PM_{2.5}$ in 3-D case is lower by as much as 2 $\mu g\,m^{-3}$ compared to the first layer case. Compared to the 3-D case, injecting emissions into the first layer leads to higher average ozone levels approximately by 1 ppb to the north, south and southeast of the airport. There are three cells near the airport where ozone levels are lower in the first layer case, by as much as 1 ppb, due to NO_X scavenging. $PM_{2.5}$ levels at the airport are higher, by as much as 0.7 $\mu g\,m^{-3}$ on average, using first layer injection of emissions compared to the 3-D spatial distribution. However, 3-D spatial distribution leads to higher average $PM_{2.5}$ levels in the vicinity, especially to the northeast and southwest, of the airport.

Maximum simulated impact of ground support equipment (GSE) on O_3 and $PM_{2.5}$, respectively, are up to 2 ppb and 9 $\mu g\,m^{-3}$ (Fig. 7). Mode-specific aircraft emissions were used in simulations both with and without GSE emissions. Impact of GSE on ozone is smaller than 1 ppb outside a 4-km radius around the

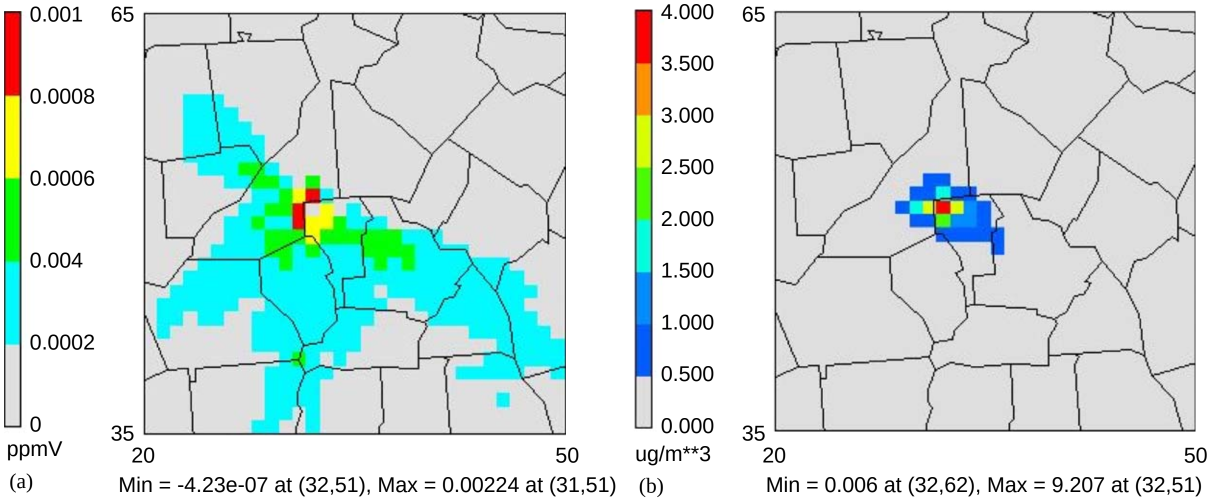

Fig. 7. Maximum sensitivity of regional concentrations to ground support equipment (GSE) emissions from Hartsfield–Jackson Atlanta International Airport during the August 11–20, 2000 period: (a) O_3; (b) $PM_{2.5}$.

Table 4
Summary of model performance

Statistics	Ozone		$PM_{2.5}$		PM_{10}	
	SAMI[a]	FAQS	SAMI[b]	FAQS	SAMI	FAQS
MNB (%)	−20.41–5.94	−0.51	−22.6	27.99	−6.4	19.73
MNE (%)	17.19–22.03	19.08	36.8	38.55	36.9	35.35

[a]Range of statistics for seven episodes (Odman et al., 2002a).
[b]Over nine episodes (Odman et al., 2002b).

airport. For $PM_{2.5}$, the impact of GSE is notable however the area of impact is smaller as compared to the aircraft emissions impact area.

Air quality model results were compared to ozone and particulate matter observations at various observation stations around Hartsfield–Jackson International Airport. Detailed performance statistics can be found in Hu et al. (2004a) for the FAQS study in which all emissions other than the aircraft emissions, and the meteorological data were the same as those used here. Model results in FAQS study agreed better with ozone and almost the same with PM observations in comparison to an earlier study (Southern Appalachian Mountains Initiative, SAMI) (Odman et al., 2002a,b). Table 4 lists the mean normalized bias (MNB) and mean normalized error (MNE) values for ozone and PM (both $PM_{2.5}$ and PM_{10}) from the FAQS and SAMI studies. Note that both studies' ozone results are within EPA's guidance values (i.e., 15 percent for bias and 30 percent for error) (EPA, 1991). The changes made to aircraft emissions in this study did not significantly change model performance achieved during FAQS, except at nearby stations where slight improvements were observed.

Using the mode-specific emissions led to better agreement with ozone observations. The coefficient of correlation of predictions with observations increased from 0.53 to 0.63 at the nearest site (i.e., Confederate Avenue Station) and the slope of the regression equation increased from 0.83 to 0.91 for the mode-specific case as compared to the characteristic value case. For $PM_{2.5}$, the mode-specific case led to a somewhat better agreement, an average of 10 percent improvement over the whole episode for daily PM values at a nearby observation site (i.e., Fort McPherson, Butler et al., 2003). Overall, using the mode-specific inventory led to better model performance.

4. Conclusions and future work

In this study, a comprehensive modeling approach was taken to assess the impact of the Hartsfield–Jackson International Airport. First, aircraft $PM_{2.5}$ emissions were estimated using a first-order approximation where emission rates are a function of SN and fuel flow rate for different engine types in different modes of operation.

Two different inventories were prepared one using the characteristic value of the SN for each engine type and another using mode-specific SN regressed from available data. The use of characteristic value SN resulted in 70 tons yr^{-1} $PM_{2.5}$ emissions, which were approximately 2.5 times larger compared to the mode-specific SN case. Emissions from the Hartsfield–Jackson airport constitute a small fraction of the total emissions in the Atlanta non-attainment area which is dominated by emissions from point, area and mobile sources. Then, the aircraft emissions were distributed in time and space. The temporal profile of aircraft activity at Hartsfield–Jackson airport is significantly different from the default profile used in a widely used emission model (SMOKE). A new technique was developed for distributing aircraft emissions in 3-D space by using aircraft location data. This results in significant dilution of aircraft emissions in comparison to the case where all aircraft emissions are injected into the first-layer (the model layer closer to the ground) grid cells containing the runways. Finally, the impact of aircraft and ground support equipment (GSE) were estimated by taking the difference between the results of air quality model simulations with and without those emissions.

Using the characteristic value inventory and injecting emissions into the first-layer cells, the maximum impact of aircraft emissions at Hartsfield–Jackson airport on ozone was estimated to be 56 ppb. The impact was more than 5 ppb over most of the Atlanta non-attainment region. The maximum impact on $PM_{2.5}$ levels was 25 $\mu g\,m^{-3}$ but the impacts were less than 4 $\mu g\,m^{-3}$ in most of the metropolitan area. The mode-specific inventory with 3-D spatial distribution of emissions led to a less intense ozone and $PM_{2.5}$ impact. The maximum impact on ozone was found to be 20 ppb with an average impact less than 1 ppb. The maximum impact on $PM_{2.5}$ was, 4.4 $\mu g\,m^{-3}$ in this case. Both sets of results, however, still have uncertainties and are limited by application to a 10-day period. Ozone and $PM_{2.5}$ impacts at other times may be more or less, depending on meteorology. Distribution of emissions spatially was found to impact ozone and $PM_{2.5}$ significantly. Our results suggest that emissions from ground support equipment impact ozone and $PM_{2.5}$, but to a lesser extent and more locally compared to aircraft emissions.

Acknowledgements

Our study could not be completed without the contribution from a large number of individuals. Risking omission, we thank Dr. Tom Nissalke, Director of Environment & Technology, Hartsfield–Jackson Atlanta International Airport, Mr. Doug Strachan of Hartsfield–Jackson Atlanta International Airport, Dr. Steven Baughcum of Boeing Company, Mr. Curtis Holsclaw, Manager of Emissions Division, Federal Aviation Administration, Ms. Debbie Calevich–Wilson of Kimley–Horn and Associates, Inc., Ms. Julie Draper of the Office of Environment and Energy, Federal Aviation Administration, Dr. Chowen Wey of NASA Glenn, and NASA Glenn for financial support.

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