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Air quality impacts of using overnight electricity generation to charge plug-in hybrid electric vehicles for daytime use

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Abstract
The air quality impacts of replacing 20% of the gasoline powered light duty vehicle miles traveled with plug-in hybrid electric vehicles (PHEVs) in the region served by the Pennsylvania, New Jersey, Maryland classic grid are examined. Unutilized, base-load nighttime electricity generating capacity is assumed to charge PHEVs that would subsequently be used during urban commutes. The net impact of this scenario on the emissions of precursors to the formation of ozone is an increase in nitrogen oxide (NOx), volatile organic compound (VOC) and CO emissions from electricity generating units during nighttime hours, and a greater decrease in NOx, VOC and CO from mobile emissions in urban areas during daytime hours. The changes in maximum daily 8 h ozone concentrations, predicted using a regional photochemical model (CAMx), are decreases in ozone concentrations between 2 and 6 ppb that are widespread across the urban areas, and increases in ozone concentrations of up to 8 ppb in highly localized areas. Air quality indicators beyond maximum daily ozone concentration are also evaluated, and in general indicate air quality improvements associated with the use of PHEVs. However, a limited number of air quality indicators worsened with the use of PHEVs, suggesting that overall impacts of the use of PHEVs will be complex.

Keywords: plug-in hybrid electric vehicles (PHEVs), air quality monitoring, CAMx, ozone, emissions trading

1. Introduction

Despite more than 30 years of emission reductions, some of the most densely populated regions in the United States still fail to attain the National Ambient Air Quality Standards (NAAQS) for ozone. With the new, stricter standards recently proposed by the US Environmental Protection Agency (EPA), lowering the maximum 8 h average concentration limit from 84 to 75 ppb, meeting the NAAQS for ozone will continue to be a challenge [1].

Ozone is a secondary pollutant formed by the reactions of volatile organic compounds (VOCs) and nitrogen oxides (NOx) in the presence of sunlight. The chemistry of ozone formation is non-linear and introduces time lags between emissions and ozone formation. Both classes of ozone precursors (VOCs and NOx) have anthropogenic and biogenic sources, and emissions of each precursor have varying degrees of impact on ozone formation based on their relative concentration levels. The ozone production potential of VOC and NOx precursors vary between regions, and the temporal and spatial details of ozone precursor emissions can have a significant impact on whether or not the precursor emissions will lead to ozone formation. As meeting the ozone NAAQS becomes more difficult, management plans may look beyond total precursor emission reductions and also consider the time and location of emissions.

Some policies have consciously attempted to control the temporal pattern of ozone precursor emissions. For example, in Texas, regulations have been proposed limiting morning

construction and commercial lawn and garden activity during the ozone season [2], since morning emissions are more potent ozone precursors than emissions later in the day. Fees for driving at certain times in city centers, implemented in London and proposed for New York City [3], can also have the effect of shifting the time of emissions. This work will examine temporal shifting of emissions due to the possible use of plug-in hybrid electric vehicles (PHEVs).

PHEVs are capable of running on either electricity or gasoline. When operating on electricity, PHEVs have no tailpipe emissions. However, emissions are released when fuel is burned to generate electricity at power plants used to charge these vehicles. The scenario to be considered in this work is the use of PHEVs in the northeastern United States during the summer ozone season. It is assumed that the PHEVs will be charged at night with unused base-load electricity generating capacity. Specifically, the grid from which these PHEVs will be charged is the Pennsylvania, New Jersey, Maryland (PJM) classic grid. The distribution of electricity capacity (MW) in this grid, by type of fuel, is 38% coal, 26% natural gas, 21% nuclear and 14% petroleum [4].

For this analysis, it is assumed that the electricity used to charge PHEVs will come from the nighttime excess capacity of coal-fired power plants in the area. The PHEVs are assumed to be operated in urban areas within the PJM region, specifically the urban areas of Baltimore, Pittsburgh, Philadelphia, Newark and surrounding areas. This assumption is reasonable because PHEVs are targeted for short-distance commutes [5]. By using excess nighttime coal-fired electricity generating capacity to charge plug-in hybrid electric vehicles that would then be used for urban commuting the next day, nighttime emissions of NOx, VOC and carbon monoxide (CO) would increase from electricity generating units (EGUs), but daytime traffic-related emissions would decrease in urban areas. The resulting effects on ozone concentrations are not straightforward because of the complicated chemistry involved. Temporal and spatial details are important and so assessing the impacts of moving NOx, VOC and CO emissions from daytime in urban areas to nighttime in rural and urban areas requires regional photochemical modeling. The case study to be reported here uses photochemical modeling to examine the impacts on ozone formation of this shifting of emissions in time and location.

A number of previous studies have examined the feasibility and air quality effects of charging hybrid vehicles at night. A study by the Pacific Northwest National Laboratory (PNNL) found that the existing electrical infrastructure and capacity could support a switch to PHEV by 84% of US cars, pickup trucks and sports utility vehicles assuming nighttime charging [6, 7]. Another study, conducted for California, concluded that existing capacity has the capability of supporting PHEV nighttime charging [8]. Both of these studies only examined the ability of the electrical infrastructure to support the excess nighttime demand; air quality effects were not addressed beyond estimates of aggregate emissions.

The Electric Power Research Institute (EPRI) [9, 10] predicted the air quality effects of PHEV penetration into the market assuming PHEVs would become available in 2010, account for 15% of the new cars sales immediately, growing to 50% of all new automobile sales by 2030 when they would account for 40% of the total vehicle fleet. While the resulting higher electricity demand could be met using excess nighttime capacity, this shift may lead to higher emissions from EGUs if utility emission caps are altered reflecting their displacement of vehicular emissions. However, EPRI assumed that emission reductions due to NOx cap and trade programs and other emission control programs would be unaffected by increased demand due to shifting of emissions from the on-road sector. Thus, the EPRI scenario assumed a substantial emission reduction. The work presented here considers the impact of transferring emissions in one sector (on-road vehicles) to emissions in another sector (EGUs). Bradley [11] evaluated the effects of switching emissions between sectors and found that switching from conventional vehicles to PHEVs charged using coal-fired electricity would lead to emission reductions of NOx, VOC, CO and CO_2. As described later in this paper, this work confirms those findings but then examines the air quality impacts (i.e.: ton ozone formation) of shifting the emissions in time and location. This scenario has not been examined in previous work, and as will be demonstrated in this case study, these changes result in complex spatial patterns of air quality benefits.

In addition to examining the impact of shifts in emissions of ozone precursors between on-road vehicles and EGUs, the case study presented here will briefly examine the effect of the shifts on greenhouse gas (GHG) emissions. Many studies have reported that a switch to PHEVs from conventional vehicles will lead to reductions in GHG emissions [12–15]. EPRI examined 9 scenarios spanning 3 levels of PHEV penetration and 3 levels of electrical sector CO_2 intensity and found that even in the worst-case scenario, CO_2 emissions were reduced [9]. Stephan found PHEVs would reduce CO_2 emissions by 25% in the short term, and up to 50% in the long term using existing spare nighttime capacity [12]. Samaras looked at the entire life cycle of PHEVs and found that the worst-case scenario would lead to CO_2 emissions that are no greater than conventional vehicles [13].

2. Methods

The air quality impacts of shifting emissions from vehicles to EGUs are examined using a 3D Eulerian photochemical grid model. The model predicts the spatial and temporal movement, production and depletion of air pollutants using data on emissions, meteorology, chemistry and deposition. Several such models, approved for regulatory applications in the United States, are available. The model to be used in this work is the comprehensive air quality model, with extensions (CAMx, www.camx.com). CAMx was chosen for this work because of the availability of meteorological, land cover, boundary condition, initial condition and emission inputs for an air pollution episode from August 2002, which is used as a representative case study in this work. The modeling inputs for this episode were developed by the Central Regional Air Planning Association (CenRAP) for regional haze and visibility studies. A performance evaluation for the 2002 episode was conducted by ENVIRON [16].

Table 1. Electricity utilization factors and generation capacity by fuel type and state [4].

Electricity average yearly utilization for 2002	Coal	NG[a]	Nuclear	Petro
Pennsylvania	70.7%	12.3%	95.2%	9.3%
Maryland	66.9%	17.0%	82.2%	8.9%
New Jersey	51.6%	23.6%	90.9%	3.3%
Delaware	37.7%	12.7%	N/A	14.6%
Four state average	67.2%	18.4%	92.6%	8.0%
Electricity capacity 2002 (MW)				
Penn	18 384	6 223	9 127	3372
Maryland	4 897	1 490	1 685	2922
New Jersey	2 124	9 237	3 875	2533
Delaware	1 050	1 293	0	745
Total	26 455	18 243	14 687	9572
Daily total (MWh) potentially available from coal[b]	539 670			
Potential MWh available for PHEVs	107 900			

[a] NG = natural gas.

[b] Assumes plants are running 85% of the time (85% capacity factor).

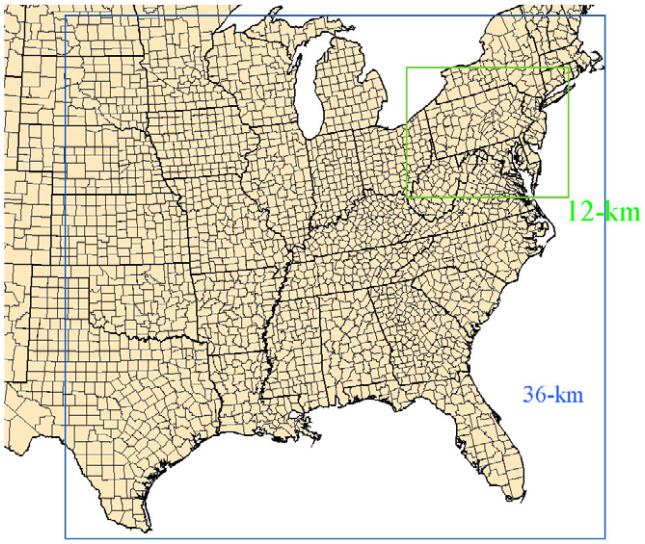

Figure 1. Air quality modeling domain considered for this study; blue box outlines grid modeled at 36 km resolution, green box outlines grid with 12 km resolution.

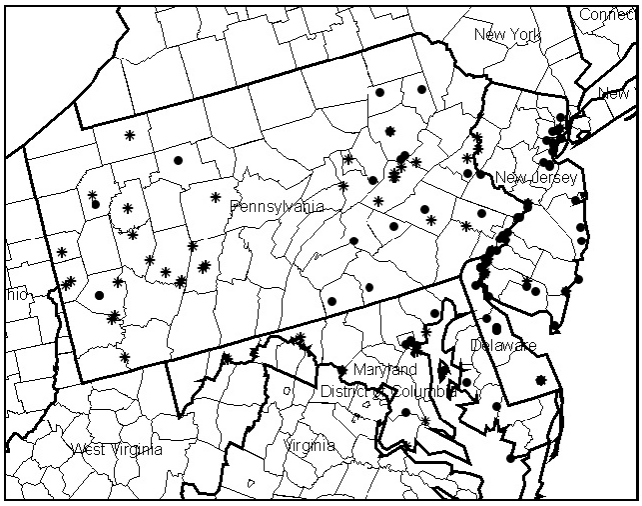

Figure 2. PJM region electric generating unit locations. Coal-fired power plants are represented with stars, all other EGU locations are represented by circles.

The air quality modeling domain is shown in figure 1. The modeling domain has a grid with 12 km horizontal resolution nested within a grid with 36 km horizontal resolution. The 12 km grid covers the Pennsylvania, New Jersey, Maryland (PJM) region that is the focus of this work. The classic PJM grid includes the three states for which it is named, as well as Delaware. The model inputs include meteorological data and emissions inventories. Meteorological data was developed by the Iowa Department of Natural Resources (IDNR) using the NCAR/PSU Mesoscale Model (MM5) and was then formatted for CAMx using an MM5CAMx processor. MM5 uses mathematics and physics to include topography, boundary conditions, and all meteorological inputs to develop a detailed layered meteorological input to air quality models [16]. Emissions inventory data, including point source, area source, mobile emissions on and off road, and biogenic emissions are based on the EPA's 2002 National Emissions Inventory with updates and corrections provided by individual states. The emissions inventories were processed using the SMOKE emissions processing system as preparations for EPS3, a CAMx preprocessing tool [16]. The biogenic emissions inventory was developed using the Global Biosphere Emissions and Interaction Systems (GlobeBeis) [16].

The EGU emissions are of particular interest in this work and therefore will be described in detail. Locations of power plants within the PJM region are shown in figure 2. The power plants for this analysis consist of a group of base-load plants, which are largely coal-fired, and peaking units, which are primarily gas-fired. Table 1 shows the electricity generating capacities of the EGUs in the region and the utilization factors. The coal-fired plants, which collectively have 26 GW of power generating capacity, are run primarily as base-load units. Nevertheless, they have a diurnal pattern of capacity utilization and emissions, shown in figure 3(a), which is due to decreased

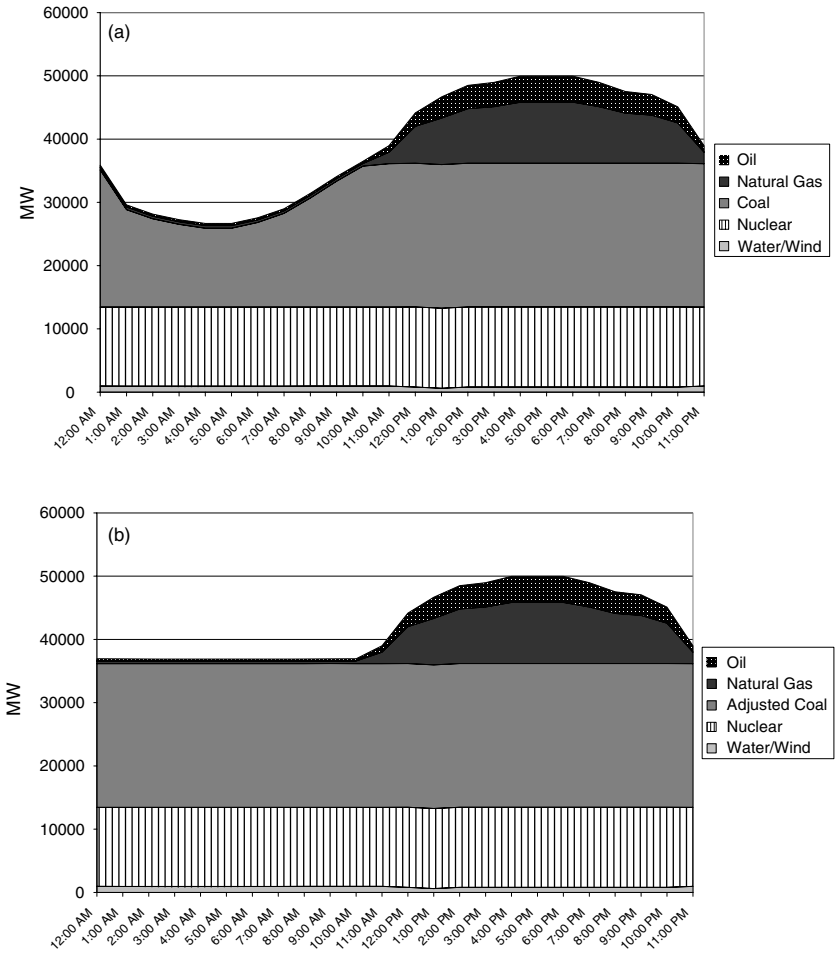

Figure 3. (a) Hourly electricity generation in the PJM classic grid region on August 12, 2002 (upper). (b) Hourly electricity generation, with nighttime PHEV charging using coal at a constant capacity factor to match the daytime maximum (lower) [17].

electricity demand at night [17]. In this work, it is assumed that these plants will be operated at constant capacity utilization, equal to their daytime maximum, and that the excess power generation at night will be used for PHEV charging. It was assumed that the generation from non-coal EGUs would not change. The modified diurnal profile is shown in figure 3(b). The excess generation available to PHEVs, if coal plants run at 85% capacity factor, is 107 GWh. It is assumed that this 107 GWh is 90% utilized, leaving 96 GWh for PHEV charging [17]. The entire 96 GWh additional electricity will be used to charge PHEVs, which will then displace emissions that occur during the day from on-road vehicles. The diurnal profile for weekday vehicular emissions is shown in figure 4.

Of the traditional fuel sources used for electricity, coal-fired power plants without controls release the most CO_2, NOx, and SO_2 throughout the life cycle [18] and so using coal generated electricity to charge PHEVs would represent a worst-case scenario for EGU emissions, assuming only existing capacity is utilized, and only at night. These additional emissions would be added to the grid, and once on the grid it is not possible to know that the additional capacity is used directly for PHEV charging. Nevertheless, it is assumed in this case study the additional generation is required because of the PHEV charging. The allocation patterns of additional nighttime demand would be decided using a traditional utility bidding system [17]. The assumptions that charging would occur only at night is dependent on the development of policy or technology that would limit charging to nighttime hours only.

The additional electricity available for nighttime charging of PHEVs is 96 GWh (approximately 20% of coal plant generation with an 85% capacity factor). The additional emissions associated with this increased capacity utilization are calculated in two ways. First, an EPA AP-42 emissions factor is used to calculate the approximate NOx emissions associated with the addition of 96 GWh nighttime electricity generation [19]. An average factor for bituminous coal on an uncontrolled boiler is used. Bituminous coal is the most prevalent type of coal in the US and is most commonly found in the northeast [4]. The EPA factor is 0.003 lb (1.4 g) of NOx emitted per kWh generated [19]. Using this factor, there would be approximately 144 English tons (131 metric tons) of NOx emissions associated with 96 GWh. The second method for calculating emissions is to assume that the 20% increase in capacity utilization, represented by the 96 GWh of additional generation, would increase emissions by 20%. The total NOx added to the point source emissions inventory in CAMx, assuming a 20% increase in NOx emissions due

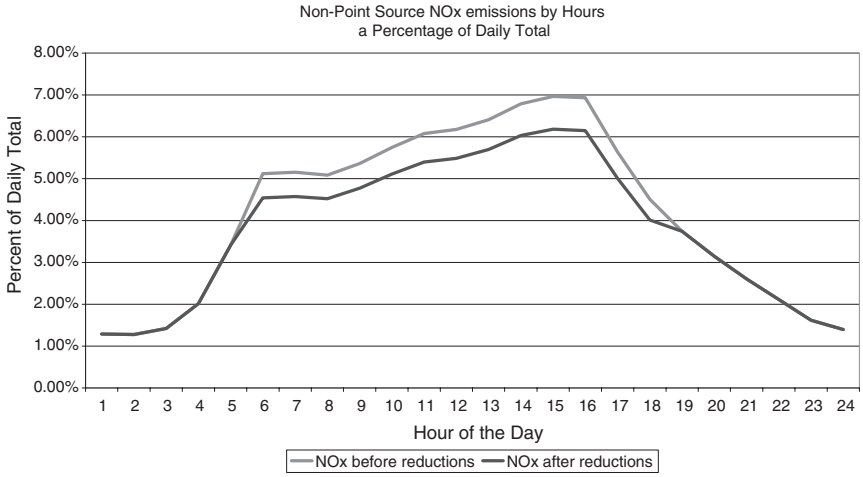

Figure 4. August 12th hourly non-point source NOx emissions from transportation as a percentage of total daily non-point source hourly NOx, shown before and after reductions due to PHEVs are realized.

Table 2. Emissions factors for light duty mobile vehicles by vehicle category [20–22].

Individual vehicle type	Gross vehicle weight (lb)	Percentage of light duty passenger fleet (%)	Grams NOx tailpipe emissions per mile	Grams CO tailpipe emissions per mile	Grams VOC tailpipe emissions per mile	MOBILE6 adjusted AC electricity consumption (Wh/mi)
Passenger cars	—	65.04	1.39	20.9	2.8	318.2
Gas truck (SUV)	0–6000	13.50	1.81	27.7	3.51	394.2
Gas truck	6001–8500	21.46	1.81	27.7	3.51	493.2

to additional nighttime generation at coal-fired plants, is 168 English tons (152 metric tons). Since these two approaches to estimating emissions lead to similar results, a 20% increase in emissions to the coal-fired power plants is applied because it is a slightly worse case. The temporal emissions profile is adjusted so that emissions assigned to coal-fired plants were constant throughout the day. This same 20% increase, with the same temporal allocation procedure, is applied to VOC and CO emissions from coal-fired EGUs.

To estimate the emissions reductions in the vehicle fleet associated with the use of PHEVs, the GWh available to PHEVs are converted to a total vehicles miles traveled (VMT) by the PHEVs. The resulting VMT available to PHEVs are distributed to three categories of light duty vehicles according to the percentages of those vehicles in the existing fleet, as listed in table 2. Table 2 also lists average energy economy factors for these three categories of light duty PHEVs. These estimates of energy use by PHEVs were obtained from an EPRI study [9, 16] and assume less efficient use of electrical energy by PHEVs, compared to other studies [6, 12]. For example, the PHEV economy factors used by the Pacific Northwest National Laboratory [6] include transmission and distribution losses, as well as battery charging and use losses. The PNNL economy factors for a mid-size sedan are 300 Wh/mile, as compared to the 318 Wh/mile reported by EPRI, and used in this study. NOx, CO and VOC emission factors for light duty gasoline vehicles were obtained from EPA average emissions and fuel consumption data as reported in MOBILE6. MOBILE6 is EPA's mobile vehicle emissions modeling software that is used to model the grams per mile of emissions from most types and ages of on-road mobile vehicles under various operating conditions [20]. These data assume an average, properly maintained vehicle on the road in July of 2000 and the average fuel economy for each vehicle class [21]. Percentages of each category of vehicle in service in 2002 are also listed and were obtained from Federal Highway Statistics data for 2002 [22]. Each of the energy economy factors (318–493 Wh/mile) are multiplied by the percentage of vehicles and the total excess nighttime MWh to calculate a daily total VMT available for PHEVs of approximately 245 million VMT.

Using the average light duty vehicle emission factors obtained from EPA data (table 2) and the percentage of each category of light duty vehicles on the road, the NOx emissions total from 245 million gasoline VMT is calculated to be 15% of the daily non-point source NOx emissions according to the EPA's National Emissions Inventory (NEI) emissions total for the 4 states. Non-point source emissions include on-road mobile, off-road mobile, and area emissions, but in the case of NOx, are dominated by on-road mobile emissions. For NOx, 15% of the non-point emissions inventory corresponded to 20% of the mobile emissions inventory. In the model, the mobile vehicle source NOx, VOC and CO emissions were reduced by 20% (300 t/d NOx, 180 t/d VOC and 2420 t/d CO) and these emission reductions were applied in the urban areas of the PJM region. Figure 5 shows a map of the PJM region, with the grid cells used in the modeling overlain. The urban areas in which PHEVs are assumed to operate are outlined in blue. The emission reductions are applied to daytime hours.

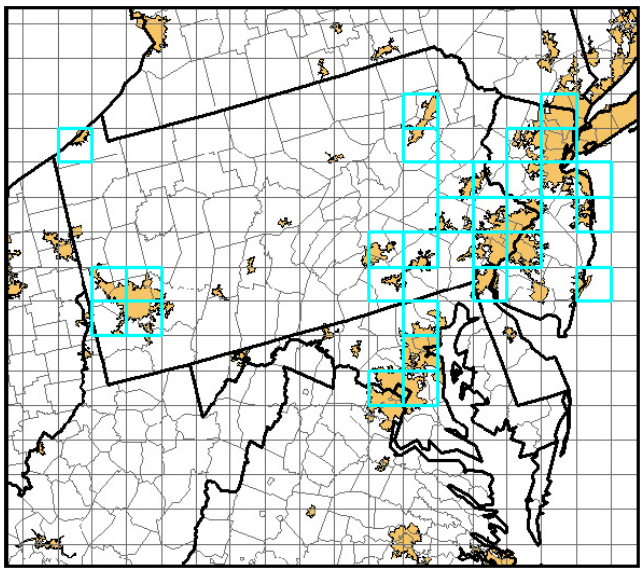

Figure 5. Map of the PJM area showing the urban areas in orange. Grid cells that received daytime mobile emissions reductions because of PHEV use are outlined in blue.

Figure 4 shows a time series of the non-point source NOx emissions on August 12, 2002 with and without the emission reductions due to the use of PHEVs.

To quantitatively evaluate the air quality impacts of PHEV use, four air quality metrics, in addition to total emissions and ozone concentrations, are used. The four metrics used are total population living in grid cells with eight hour averaged ozone concentrations above a threshold value of 75 ppb, daily maximum ozone above 75 ppb multiplied by population, time integrated ozone above 75 ppb, and time integrated ozone above 75 ppb multiplied by population. The following equations describe how the metrics are calculated.

(1) Total population above threshold (capita)

$$M_{\text{Total Population}} = \sum_{\text{g}} p_{\text{g}} \max\{\delta_{\text{g,h}}\}$$

$$\delta_{\text{g,h}} = \begin{cases} 0 & c_{\text{g,h}} \leqslant \text{threshold} \\ 1 & c_{\text{g,h}} > \text{threshold} \end{cases} \quad (1)$$

where p_{g} is the population in grid cell g, and $c_{\text{g,h}}$ is the ozone concentration in grid cell g in hour h. This metric is calculated by determining all ground level grid cell ozone concentrations in the PJM area for each day, calculating the maximum ozone concentration in each cell, and comparing the maximum concentration to the threshold. If the maximum concentration exceeded the threshold, then the population in that grid cell is added to the total.

(2) Maximum daily population exposure of concentrations above a threshold (ppb-capita)

$$M_{\text{Pop}} = \sum_{\text{g}} p_{\text{g}} \max\{e_{\text{g,h}}\}$$

$$e_{\text{g,h}} = \begin{cases} 0 & c_{\text{g,h}} \leqslant \text{threshold} \\ c_{\text{g,h}} - \text{threshold} & c_{\text{g,h}} > \text{threshold}. \end{cases} \quad (2)$$

This metric is calculated by determining the maximum ozone concentration in all ground level grid cells in the PJM area for each day. If that maximum is above the threshold, the excess is calculated by subtracting the threshold from the concentration in the grid cell. The excess is multiplied by the population in that grid cell and added to the total.

(3) Time integrated area $*$ excess (km^2 ppb)

$$M_{\text{TimeArea}} = \sum_{h} \sum_{\text{g}} a_{\text{g}} e_{\text{g,h}}$$

$$e_{\text{g,h}} = \begin{cases} 0 & c_{\text{g,h}} \leqslant \text{threshold} \\ c_{\text{g,h}} - \text{threshold} & c_{\text{g,h}} > \text{threshold} \end{cases} \quad (3)$$

where a_{g} is the area of grid cell g. This metric is calculated by determining the maximum ozone concentration in all ground level grid cells in the PJM area for each hour of each day. If that maximum is above the threshold, the area of the grid cell is added to the total for each hour that the threshold was exceeded.

(4) Time integrated population $*$ excess (ppb $*$ capita)

$$M_{\text{Time Pop}} = \sum_{h} \sum_{\text{g}} p_{\text{g}} e_{\text{g,h}}$$

$$e_{\text{g,h}} = \begin{cases} 0 & c_{\text{g,h}} \leqslant \text{threshold} \\ c_{\text{g,h}} - \text{threshold} & c_{\text{g,h}} > \text{threshold}. \end{cases} \quad (4)$$

This metric is calculated by determining the maximum ozone concentration in all ground level grid cells in the PJM area for each day. If that maximum is above the threshold, the excess is calculated by subtracting the threshold from the ozone concentration in the cell, and the excess is multiplied by the population density. The sum is taken over the area and over the hours of the day.

3. Results

Photochemical modeling simulations were performed for the period of August 5th through 16th, 2002. This episode was chosen because it contains one of the most severe photochemical events in the past decade in the northeast US, with PJM area monitoring stations on August 10th through the 14th measuring maximum 1 h average ozone concentrations of 117, 126, 143, 147 and 132 ppb, respectively. In addition to being severe, this episode presents a variety of meteorological conditions. Figure 6 shows 36 h back trajectories, calculated using the National Oceanic and Atmospheric Administration's HYbrid Single Particle Lagrangian Integrated Trajectory (HYSPLIT) model [23], for air parcels arriving at a point in NJ at 5pm on each day for the period August 10th–14th. The specific location in NJ represents the average latitude and longitude of the air quality monitoring stations located around the Philadelphia/Baltimore/Southern New Jersey non-attainment area. This map shows the path air travels during the 36 h prior to arriving in this area. During the period, the prevailing winds shift from southerly to westerly, leading to very different mixes of ozone precursors encountered by the air parcels arriving in the area as the episode develops.

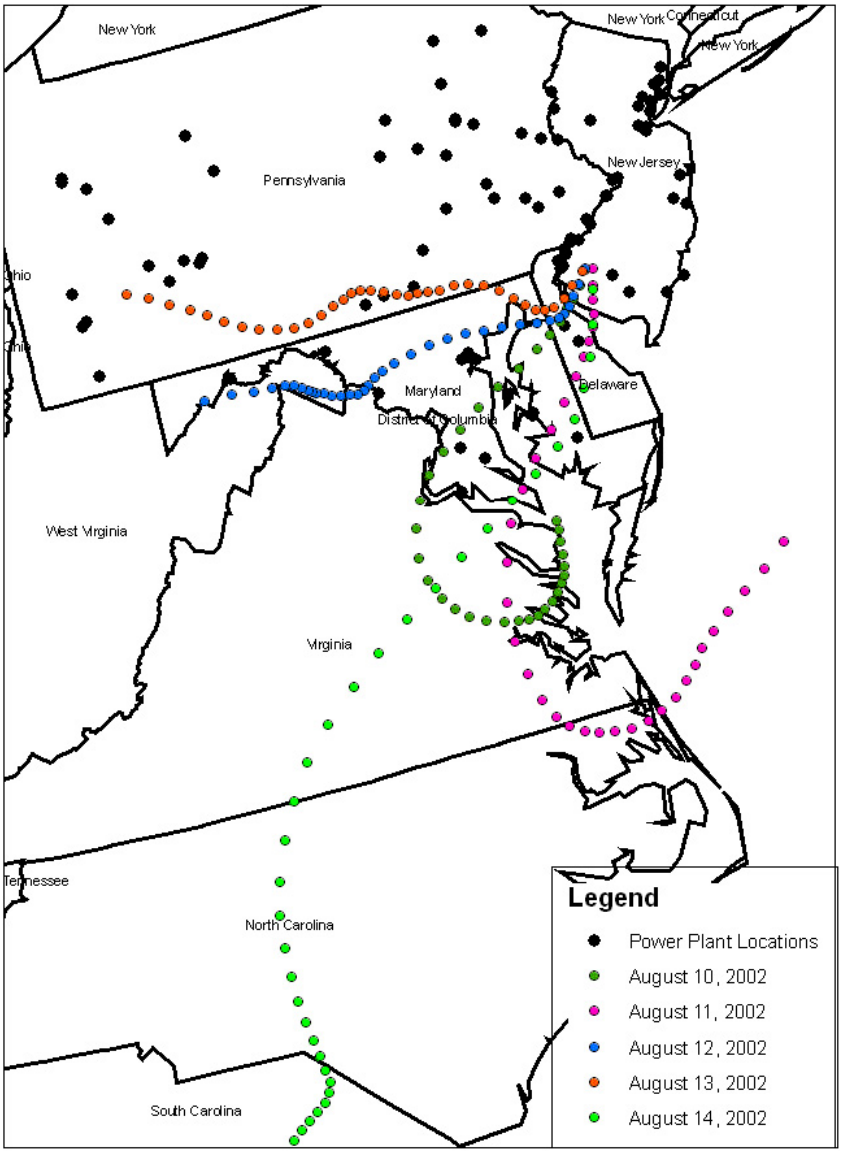

Figure 6. Map shows the 36 h back trajectories for air parcels arriving at a particular location in NJ at 5PM over the five days of an August 10th–14th ozone episode as modeled by HYSPLIT. The back trajectories show that prevailing winds rotated from southerly to westerly over the episode. The small black circles show locations of EGUs.

Four air quality modeling simulations were conducted using CAMx. The first is a basecase with unchanged emission inventories representing the emissions as they occurred during the modeling period. The second simulation represented the changes to the emissions inventory that would occur due to the full utilization of PHEVs, during the day, charged at night using electricity from coal-fired EGUs, as calculated above and is called PHEV-300. In this PHEV case, NOx emissions are increased by 168 tons during nighttime hours for EGUs in the four state PJM area because of increased nighttime electricity generation used to charge PHEVs. VOC and CO emissions are increased by 1.2 and 11.9 tons respectively. NOx emissions are decreased by 300 tons during daytime hours in urban areas (hence PHEV-300) from the substitution of gasoline vehicles by PHEVs. CO emissions are reduced by 2420 tons and VOC emissions were reduced by 180 tons.

For the third simulation, PHEV-150, emission reductions associated with PHEV use are halved. This notional scenario might arise due to lower charging and use efficiency, or the availability of a lower emitting basecase fleet of vehicles. The additional nighttime electricity generation and associated emissions remain the same. For this third case, the nighttime emissions are increased by 168 tons, 1.2 tons and 11.9 tons, for NOx, VOCs and CO respectively as with the second case, but the daytime NOx, VOC and CO emissions are decreased by only half the amount from the PHEV-300 simulation. NOx emissions are reduced by 150 tons, CO by 1210 tons and VOCs by 90 tons. For the fourth simulation, changes are made to the mobile emissions only. No increases are made to nighttime emissions from EGUs. NOx, VOC and CO are decreased during daytime hours, in urban areas by 300 tons, 180 tons, and 2420 tons respectively, as with the PHEV-300 run. This simulation serves as a sensitivity analysis as well as a best case scenario, since it assumes that electricity used to charge PHEVs resulted in no increases in EGU emissions (due to the

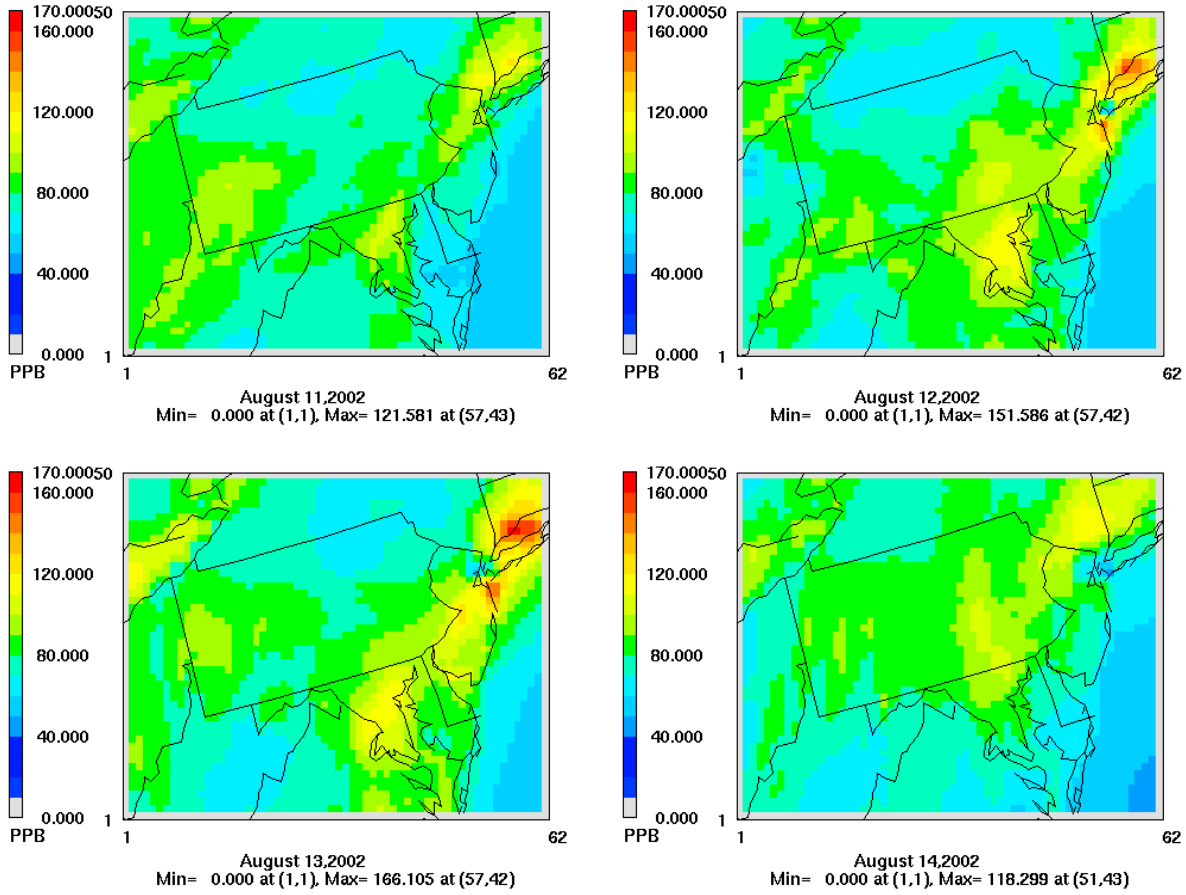

Figure 7. Maximum eight hour averaged ozone concentrations (maximum concentration achieved over the course of the day, regardless of time) for the basecase on August 11th–14th 2002 as modeled by CAMx.

existence of an emissions cap or the use of electricity from a non-emitting source like wind or solar).

Figure 7 shows the maximum ozone concentrations in the PJM area for the basecase simulations on August 11th through 14th, the days with the highest maximum ozone concentrations for the episode. These results are consistent with concentrations measured in the area on those days by monitor sites; data from two representative sites are shown in figure 8. The basecase CAMx simulation shows good agreement with concentrations measured by the monitor sites in both absolute values and hourly patterns.

In order to show the effects of PHEV utilization, the maximum daily 8 h average ozone concentrations are calculated for each case, in each grid cell for the modeling period. The differences between the maximum values for the basecase and for the PHEV-300 case are shown in figure 9. Negative values represent ozone reductions in the PHEV-300 case.

The air quality modeling results show air quality improvement over the urban areas in Pennsylvania as well as Baltimore and most of northern New Jersey. The changes in emissions due to the substitution of PHEVs, in the PHEV-300 case, lead to a 2–8 ppb decrease in maximum 8 h averaged ozone concentrations over these major urban areas on all days of the episode. The areas showing this ozone reduction have the highest basecase concentrations of ozone as well as the largest populations, meaning that PHEVs have the potential to significantly reduce ozone exposure in the northeastern US.

However, the August 12th–14th results also show a NOx disbenefit (ozone concentrations increasing as NOx emissions decrease) occurring in the cells above Newark, NJ. On the 14th, there is also a NOx disbenefit occurring over Philadelphia. Under basecase conditions, the NOx emissions in these areas are relatively high and the NOx reacts with ozone, reducing the ozone concentrations. These NOx disbenefit conditions also caused Newark and Philadelphia to have lower ozone concentrations, in the basecase, than the surrounding areas, between 50 and 70 ppb, while surrounding areas were as high as 120 ppb. When NOx was decreased across the area, the titration effect was reduced and the ozone concentrations in these high NOx areas increased to the mid-70s while the surrounding cells, not affected by the NOx disbenefit, decreased by a greater amount.

The results from the PHEV-150 case, as shown in figure 10, show a smaller air quality benefit. The maximum reduction in 8 h averaged ozone concentrations is only 3.9 ppb in this case as opposed to 8.7 in the PHEV-300 case. The PHEV-150 case shows the same NOx disbenefit, but to a lesser degree, above the cities of Newark and Philadelphia.

The data in table 3, which show the air quality metrics defined in equations (1)–(4), more completely summarize the results from the simulations. Metrics are calculated for

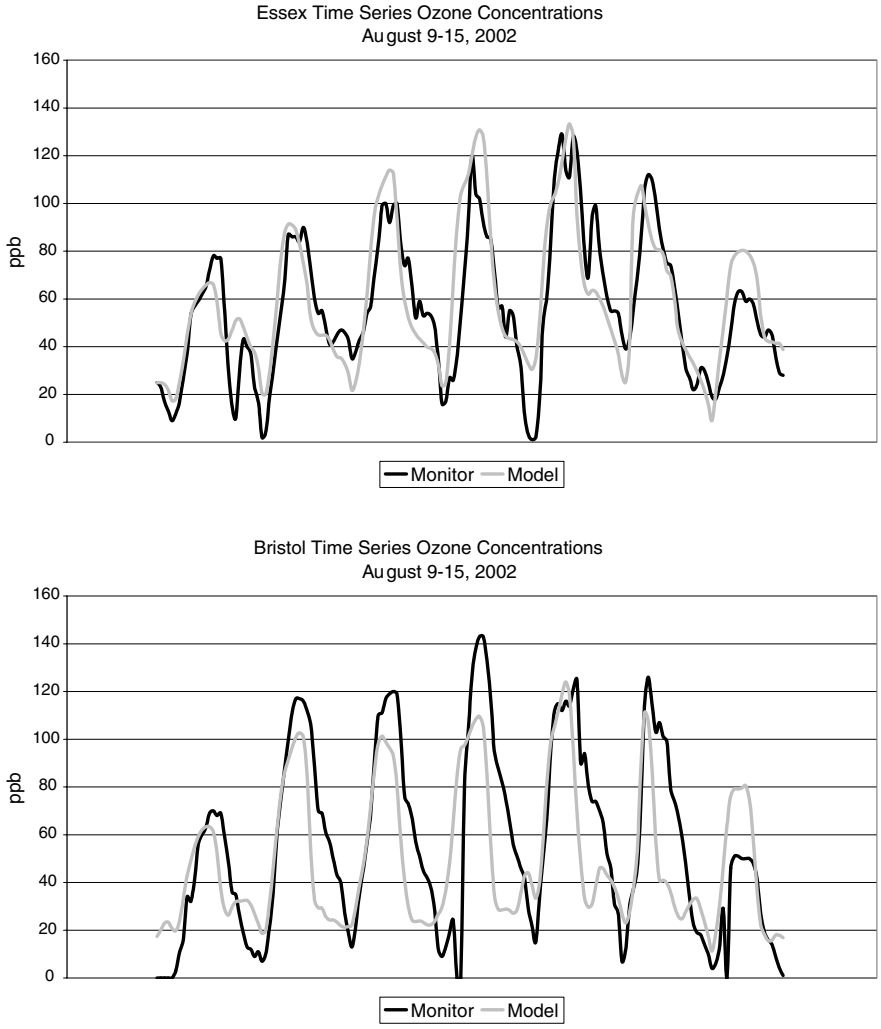

Figure 8. Comparison of modeled versus measured ozone values at two monitor locations in the PJM showing good agreement. Essex is located outside of Baltimore, and Bristol is located outside of Philadelphia.

two different ozone thresholds; the current ozone standard of 85 ppb (8 h average concentration), and the newly proposed standard of 75 ppb (8 h average). When the threshold is defined as 75 ppb, both the PHEV-300 and PHEV-150 cases show an increase in the total population above standard (metric 1), but the population exposure (a function of time and ozone concentrations) decreased, both at the time of maximum ozone and summed over all the hours of the day (metrics 2 and 4). Reduction of NOx emissions in urban areas is causing ozone concentrations in some very localized areas to increase above 75 ppb. The areas surrounding the urban centers have decreases in ozone concentration that are larger than the localized increases, but concentrations are still above 75 ppb. This effect can cause the total area and population exposed to concentrations above 75 ppb to increase, while decreasing both the time integrated population above 75 ppb, and time integrated area above 75 ppb.

When the ozone threshold is set at 85 ppb (8 h average) all metrics show air quality improvements, including the population above the standard. Population exposure, time integrated and at the time of maximum ozone, show decreases of 7% and 9% respectively. Time integrated area above the threshold showed a decrease greater than 10%. These metrics also show that, in areas where reduction of the NOx disbenefit is causing an increase in ozone concentrations, the resulting concentrations are below 85 ppb. At night, when EGU emissions increase, PHEV-300 nighttime 1 h ozone concentrations show increases of 2–4 ppb in rural areas with decreases of 3–5 ppb in urban areas.

Due to the complexity of ozone chemistry and its sensitivity to temporal and spatial changes in emissions of ozone precursors, the results are not straightforward. There are scenarios where PHEV use worsens air quality in localized areas, depending on how air quality is measured. However, these results highlight that PHEV scenarios, in general, reduce ozone concentrations and exposure. This reduction is especially true for the scenario where PHEVs are charged at night with lower or non-emitting sources (such as wind or controlled coal combustion).

Using the assumptions outlined in this paper concerning increased nighttime electricity generation at coal-fired power plants and decreased daytime utilization of gasoline vehicles in favor of PHEVs, the resulting change in total quantities of carbon dioxide (CO_2) released to the atmosphere was

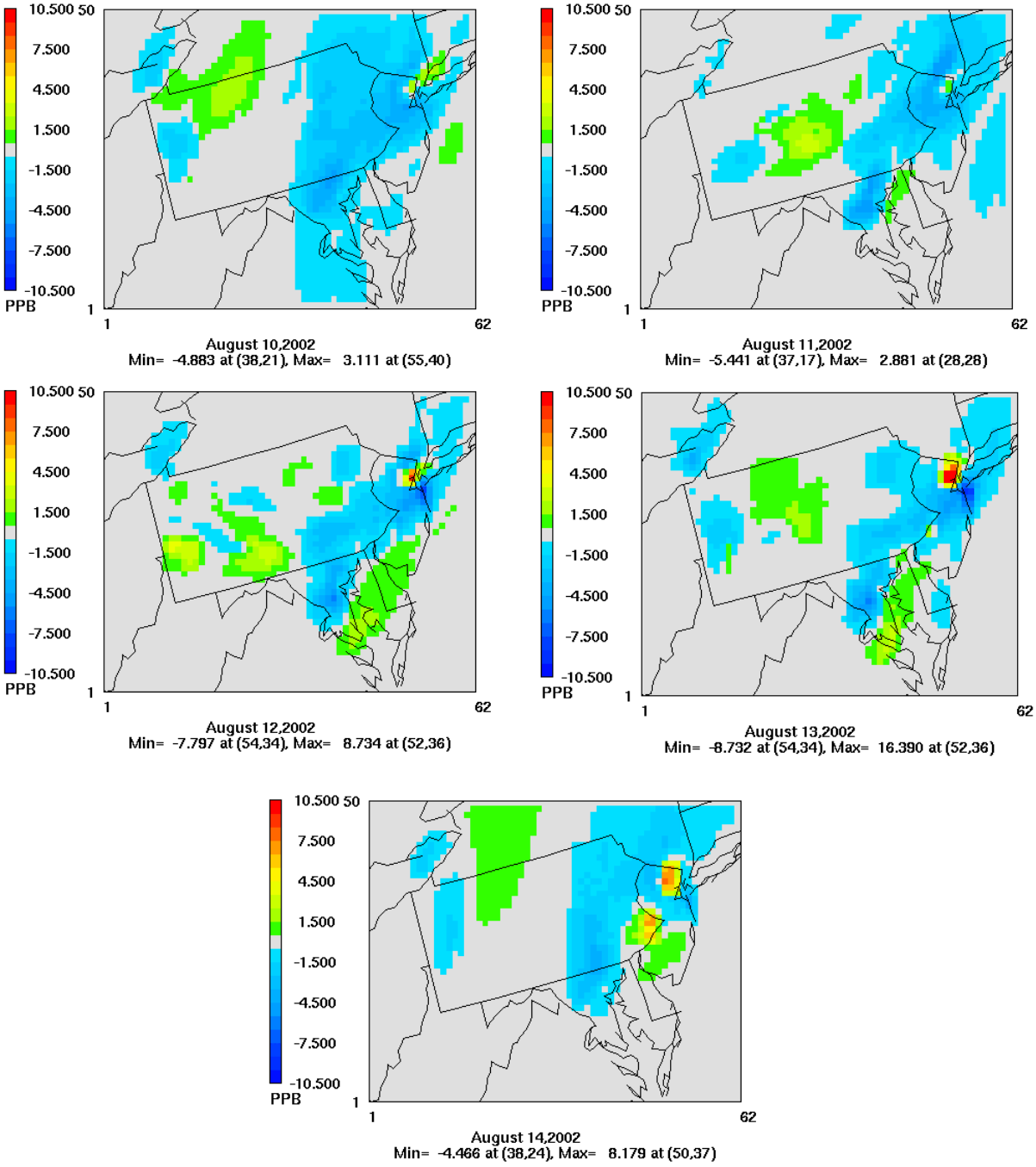

Figure 9. Eight hour maximum ozone, difference between the basecase and the PHEV-300 case for August 10th–14th. (PHEV-300 case—basecase).

calculated. The additional capacity utilization of the coal-fired EGUs would lead to an increase of approximately 99 610 tons of CO_2 emissions. This emissions estimate was calculated assuming a 35% efficiency of heat to electricity at coal plants and an average heat density for coal for the year 2002 obtained from the Energy Information Administration (EIA) [4]. The calculated decrease of CO_2 emissions from replacing gasoline vehicles with available PHEVs would be 122 670 tons. This value was calculated using average fuel economy standards (table 2) [21, 24]. Therefore, assuming that PHEVs are able to substitute for gasoline vehicles for at least 80% of the expected VMT calculated using an energy efficiency of 318 Wh/mi, the scenario considered in this work would not lead to increases in CO_2 emissions during fuel combustion. There are also likely differences between the greenhouse gas emissions due to producing, refining and delivering gasoline and mining and delivering coal. In general these emissions are substantially less that the greenhouse gas emissions associated with combustion, so a detailed estimate of these emissions was not prepared here.

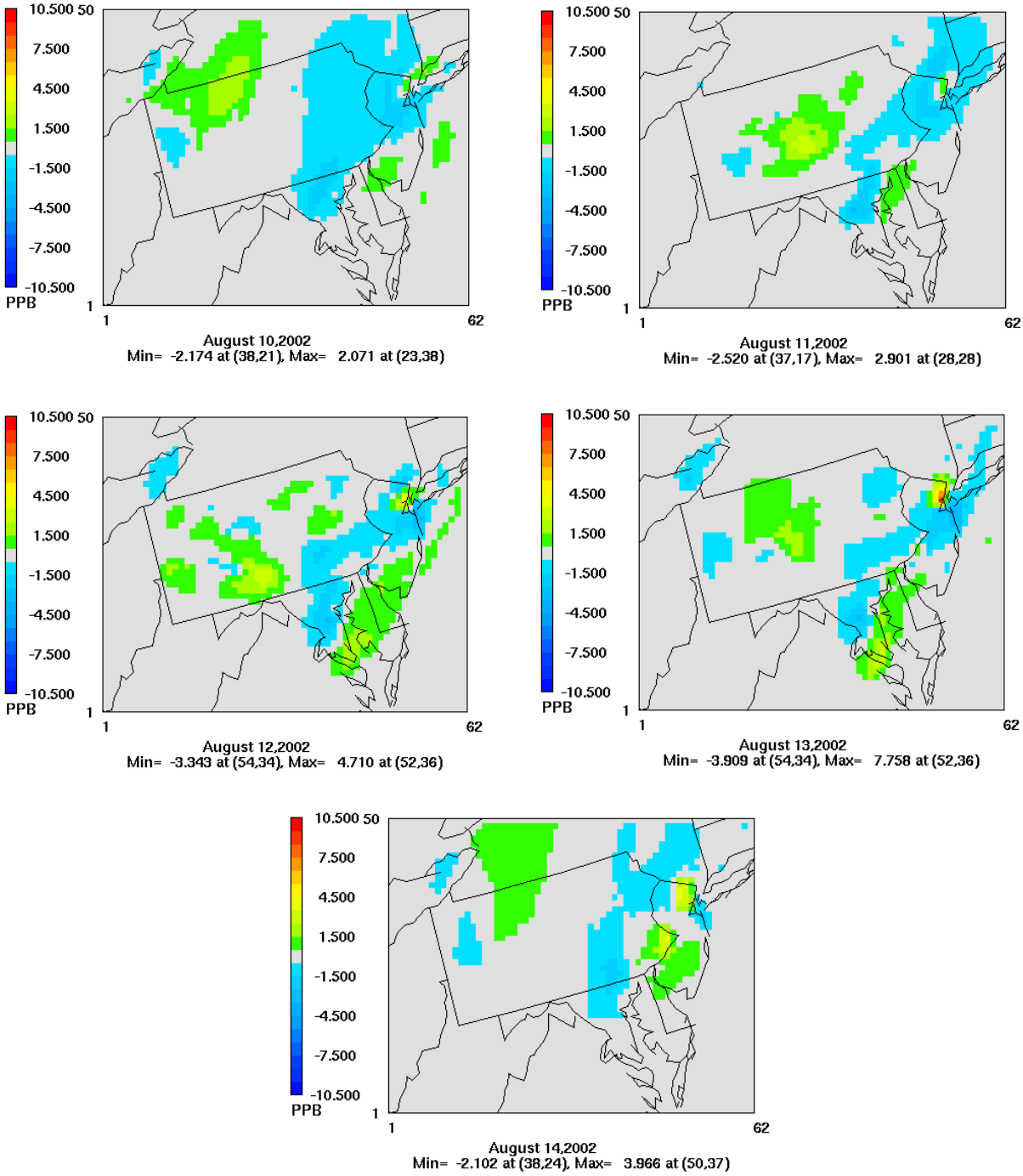

Figure 10. Eight hour maximum ozone, differences between the basecase and the PHEV-150 case for August 10th–14th. (PHEV-150 case—basecase.)

4. Conclusions

Air quality modeling of the four state classic PJM area show that substitution of PHEVs for just 20% of the mobile vehicle fleet VMT would reduce ozone by up to 8 ppb in the most densely populated areas in the PJM. The benefits would increase if cleaner sources are used to charge the PHEVs or if, subject to the availability of additional excess generation, PHEVs are substituted for a larger percentage of the mobile fleet. However, this work also indicates that while there is the potential for improvements in ozone concentrations, there is also the potential for localized worsening of ozone concentrations as the spatial and temporal patterns of emissions change. Further, the air quality impacts of PHEV use are not limited to ozone. If, for example, coal-fired power plants are used to generate electricity to power PHEVs, and local SO_2 emissions increased, then more particulate sulfate would be formed. On the other hand, reduced emissions from vehicle

Table 3. Changes in ozone concentrations (using multiple metrics, equations (1)–(4)) due to use of PHEVs.

Metric	PHEV-300 (%)	PHEV-150 (%)	PHEV_mobile_only (%)
Per cent change (from basecase) threshold = 75 ppb			
(1) Population above standard	1.11	1.68	−0.10
(2) Max population exposure	−5.46	−1.03	−8.51
(3) Daily area above standard	−5.76	−1.08	−9.22
(4) Daily population exposure	−3.87	−0.23	−6.96
Per cent change (from basecase) threshold = 85 ppb			
(1) Population above standard	−5.20	−4.50	−6.15
(2) Max population exposure	−9.31	−2.78	−12.64
(3) Daily area above standard	−10.64	−2.85	−15.41
(4) Daily population exposure	−7.18	−1.43	−10.98

exhaust would lower particulate matter concentrations. These and other air quality implications of the use of PHEVs will be examined in future work.

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