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Using Backup Generators for Meeting Peak Electricity Demand: A Sensitivity Analysis on Emission Controls, Location, and Health Endpoints

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ABSTRACT

Generators installed for backup power during blackouts could help satisfy peak electricity demand; however, many are diesel generators with nonnegligible air emissions that may damage air quality and human health. The full (private and social) cost of using diesel generators with and without emission control retrofits for fine particulate matter ($PM_{2.5}$) and nitrogen oxides (NO_x) were compared with a new natural gas turbine peaking plant. Lower private costs were found for the backup generators because the capital costs are mostly ascribed to reliability. To estimate the social costs from air quality, the changes in ambient concentrations of ozone (O_3) and $PM_{2.5}$ were modeled using the Particulate Matter Comprehensive Air Quality Model with extensions ($PMCAM_x$) chemical transport model. These air quality changes were translated to their equivalent human health effects using concentration-response functions and then into dollars using estimates of "willingness-to-pay" to avoid ill health. As a case study, 1000 MW of backup generation operating for 12 hr/day for 6 days in each of four eastern U.S. cities (Atlanta, Chicago, Dallas, and New York) was modeled. In all cities, modeled $PM_{2.5}$ concentrations increased (up to 5 g/m^3) due mainly to primary emissions. Smaller increases and decreases were observed for secondary $PM_{2.5}$ with more variation between cities. Increases in NO_x emissions resulted in significant nitrate formation (up to 1 g/m^3) in Atlanta and Chicago. The NO_x emissions also caused O_3 decreases in the urban centers and increases in the surrounding areas. For $PM_{2.5}$, a social cost of approximately \$2/kWh was calculated for uncontrolled diesel generators in highly populated cities but was under 10 ¢/kWh with $PM_{2.5}$ and NO_x controls. On a full cost basis, it was found that properly controlled diesel generators are cost-effective for meeting peak electricity demand. The authors recommend NO_x and $PM_{2.5}$ controls.

IMPLICATIONS

Using installed backup generators to peak meet electricity demand for approximately 200 hr/yr is cost-effective compared with constructing a new dedicated peaking plant. Retrofitting these generators with emission controls for $PM_{2.5}$ and NO_x allows them to operate without severe air quality degradation and adverse human health effects. Modeling the air quality effects with a chemical transport model is necessary to evaluate this strategy.

INTRODUCTION

For approximately 200 hr/yr, electricity demand peaks. Because of congested transmission lines and expensive electricity generators, referred to as peaking plants, that operate only when electricity demand is high, the cost of a kilowatt-hour (kWh) can be more than 10 times greater than in off-peak periods. This effect is especially acute in urban centers, which are constrained by available transmission and distribution (T&D) and electricity generation capacity. In extreme cases, electricity supply and demand cannot be balanced, leading to brownouts and blackouts. To help improve the reliability of the electricity grid, end users with on-site generation capacity, specifically backup generators installed to operate during blackouts, could provide electricity during periods of peak demand.[1,2] These generators represent a significant source of underutilized capacity; for example, New York City is reported to have over 1000 MW of backup generation.[3] Several independent system operators (ISOs) have developed re- liability programs that harness these generators. The New York ISO (NYISO) allows backup generators to participate in emergency electricity and demand response programs as well as a special market for capacity.[4] Despite the po- tential benefits, many backup generators are excluded from these programs because they are diesel internal com- bustion engines (ICEs). Specifically, there is concern about adverse human health effects from exposure to the non-negligible air emissions from a diesel ICE used as a

backup generator without advanced emission controls for nitrogen oxides (NO_x) and fine particulate matter ($PM_{2.5}$).

In a previous paper, the authors evaluated the full costs (defined as the private (market) and social (air quality costs) costs) of using installed backup diesel and natural gas-fueled ICEs for meeting peak electricity demand in New York City compared with a conventional peaking plant (a simple cycle natural gas turbine).[5] To quantify the social costs from air quality, the emissions from these generators were converted to ambient concentrations using a "state of science" air quality chemical transport model (CTM), the Particulate Matter Comprehensive Air Quality Model with extensions ($PMCAM_x$).[6] It was found that a diesel ICE retrofitted with a catalyzed diesel particulate filter (DPF) could operate less expensively than a new peaking plant and without causing severe damage to human health. Since that work, the U.S. National Research Council concluded that there is sufficient evidence to support a causal relationship between exposure to ozone (O_3) and premature mortality.[7] Thus, there is an increased emphasis on reducing the emissions of O_3 pre- cursors from diesel ICEs, specifically NO_x. Retrofit emis- sions control options for NO_x that can be combined with a DPF on stationary diesel ICEs include low NO_x catalysts (LNCs) and exhaust gas recirculation (EGR).[8] There is also the possibility of retrofitting to a dual fuel operation, also known as bifuel. With this retrofit, the engine uses mostly natural gas with a smaller amount of diesel to ignite the mixture, reducing $PM_{2.5}$ and NO_x emissions.[9] The natural gas is supplied by the existing natural gas infrastructure.

Here, the authors extend their previous analysis of using backup generators for meeting peak electricity demand to include emission controls for NO_x. In addition to New York City, air quality modeling is also conducted for Atlanta, Chicago, and Dallas, which have different air pollution problems. Finally, the robustness of the full cost results is evaluated with respect to uncertainties in the mechanisms for the formation of secondary $PM_{2.5}$ in the CTM and health endpoints.

METHODS AND DATA

The full cost of using diesel backup generators with or without retrofitted emission controls for $PM_{2.5}$ and NO_x is calculated and compared to a simple cycle natural gas turbine on a per-kWh-generated basis. The private (market) and social (human health effects from air quality) costs are investigated and defined by eqs 1 and 2.

$$\text{Full Cost} \quad \text{Private Cost (PC)} \quad \text{Social Cost (SC)} \qquad (1)$$

$$\text{Full Cost} \quad [(CC \quad TD \quad RF)/HR]*CRF \quad FOM/HY \quad VOM \quad FC \quad SC \qquad (2)$$

where *CC* is the capital cost of the generator attributed to peak electricity generation (in $/kW), *CRF* is the capital recovery factor to convert one-time costs into equal annual payments at a specified discount rate (estimated at 15%), *FC* is the fuel cost (in $/kWh), *FOM* is the fixed operating and maintenance (O&M; in $/kW-yr), *HR* is the number of hours per year of operation (estimated at 200 hr), *RF* is the capital cost of the emission control retrofits (in $/kW), *SC* is the social cost (in $/kWh), *TD* is the capital cost of T&D or grid-generator interconnections (in $/kW), and *VOM* is the variable O&M (in $/kWh).

Table 1 shows the capital costs for retrofitting diesel backup generators with a catalyzed DPF, a DPF-LNC, a DPF with a low-pressure EGR, and dual fuel operation. Selective catalytic reduction (SCR) is not included as a NO_x emission control option for emergency generators. These costs are obtained from the Manufacturers of Emis- sions Control Association (MECA)[10] and the Western Re- gional Air Partnership (WRAP).[11] For the dual fuel retrofit, capital costs are estimated from $35/kW (the value at which the retrofitted dual fuel ICE would have approxi- mately the same levelized cost per kWh as an uncon- trolled diesel ICE at the mean fuel prices for diesel and natural gas) to $100/kW.[9,12] A cost of $125/kW is applied for interconnections required for the generators to operate in parallel with the electricity grid.[1] Finally, it is as- sumed that there is negligible fixed O&M associated with the DPF and the NO_x emission controls for the peak power application and a variable O&M cost of 0.01$/kWh for all backup generators. This is compared to a natural gas turbine with a capital cost of $500/kW, fixed O&M of $15/kW-yr, and variable O&M of 0.0055 $/kWh.[13] An additional $100/kW is estimated for T&D to transport the electricity from the turbine to the urban center. The fuel cost (FC) is calculated as the heat rate (in Btu/kWh) of the generator multiplied by the price of fuel for the relevant consumer class (in $/Btu). FCs are obtained from the Energy Information Agency (EIA) and are updated from the authors' previous work.[14] The mean, minimum, and maximum values from 2005 to 2008 are used. Diesel fuel is priced at $2.50/gal with a range of $1.50/gal to $4.00/gal and a 10¢ premium for ultralow sulfur diesel (ULSD) fuel. For natural gas, a price of $9.00/Mcf was used with a

Table 1. Heat rate in Btu/kWh and EFs in g/kWh for backup diesel generators without emission controls (costs in $/kW, heat rate in Btu/kWh).

Emission Control Retrofit	Incremental Capital Cost ($/kW)	Heat Rate (Btu/kWh) or Penalty (%)	NO_x (g/kWh or %)	$PM_{2.5}$ (g/kWh or %)	CO (g/kWh or %)	HC (g/kWh or %)	SO_2 (g/kWh or %)
Baseline	–	9750	18.8	1.40	6.40	2.0	1.25
DPF	25–40	0–4%	0%	85–99%	90%	90%	95%
DPF-LNC	40–60	0–7%	10–25%	85–99%	90%	90%	95%
DPF-EGR	40–55	0–5%	25–60%	85–99%	90%	90%	95%
Dual fuel (70–95% natural gas)	35–100	0–8%	50%	68–92%	–	–	95–99%

Notes: Fuel efficiency penalties and emission reductions from baseline in % for backup diesel generators with emission control technology retrofits.

range of $5.50/Mcf to $12.50/Mcf for electricity production and $14.50/Mcf with a range of $10.80/Mcf to $20.30/Mcf for commercial deliveries.

To calculate the social cost, a bottom-up impact pathway approach is used. First, an emission scenario for using the backup generators is developed, and these emissions are transformed into ambient concentrations using $PMCAM_x$. Second, these concentrations are translated into their equivalent human health effects and economic values.

$PMCAM_x$ is the Carnegie Mellon University research version of the Comprehensive Air Quality Model with Extensions (CAM_x). It is a "state of the science" CTM that simulates the emission, advection (convection), dispersion, gas- and aqueous-phase chemical reactions, and dry and wet deposition for 35 gaseous species, 12 radical species, and 13 aerosol species in 10 size bins on a three-dimensional Eulerian grid. Complete model details and evaluation can be found in Gaydos et al.[6] and Karydis et al.[15] This paper highlights the chemistry and $PM_{2.5}$ for- mation mechanisms in $PMCAM_x$. The gaseous chemistry is simulated using the Carbon Bond Mechanism (CBM) IV.[16] Inorganic species are transferred between the gas and aerosol phases using ISORROPIA, a bulk aerosol ther- modynamics model.[17] The secondary organic aerosol is determined using the Secondary Organic Aerosol Model (SOAM) II.[18,19] This model partitions condensable gases as a function of their volatility, the aerosol size distribution, and the composition assuming a pseudo-ideal solution. Condensable precursors are lumped into low- and high-yield products of the photooxidation of toluene and other aromatics; the oxidation products of paraffins, anthropogenic olefins, and cresol; and the oxidation of biogenic olefins. The temperature dependence of the saturation concentrations is described by the Clausius–Claperyon equation.

Table 1 shows the baseline composite emission factors (EF) and heat rate (efficiency) corresponding to an uncontrolled diesel ICE and the reductions from baseline for the retrofitted emission controls that are used to construct the emission profiles. The EF and fuel economy penalties for the generator/emission control retrofit combinations are derived from the California Air Resources Board (CARB),[20] the Environmental Defense Fund,[8] and the Natural Resources Defense Council.[21] For the baseline diesel generator, a diesel fuel with a sulfur content of 250 parts per million (ppm) is used. For all retrofit options, an ULSD fuel with a sulfur content of 15 ppm is used, which reduces the sulfur dioxide (SO_2) EF by approximately 95%. The authors note that as of 2010, all off-road diesel fuel will be ULSD. The emission reductions for the dual fuel option are highly dependent on the percent of diesel replaced by natural gas. A generator operating on 70 –95% natural gas is presented. For $PM_{2.5}$, it is assumed that the EF is a linear combination of the EFs for an uncontrolled diesel and natural gas ICE multiplied by the fraction of the two fuels. For NO_x, the emissions are a function of combustion characteristics with most retrofits reporting reductions of approximately 50%. There is also evidence that the carbon monoxide (CO) and hydrocarbons (HC) may be higher than the diesel ICE.[22] The EFs in Table 1 are further processed to be consistent with the representation of the species in $PMCAM_x$. The NO_x is divided into 85% nitrogen oxide (NO) and 15% nitrogen dioxide (NO_2). The HC are speciated with profiles from SPECIATE[23] and grouped into CBM-IV functional categories. $PM_{2.5}$ is divided into approximately 80% elemental carbon (PEC), approximately 20% organic carbon (OC) (POC), and less than 2% comprised of nitrate (PNO_3) and sulfate (PSO_4).[24] The $PM_{2.5}$ mass is divided equally into six size bins with aerodynamic diameters less than 2.5 m.

The peak demand scenario consists of 1000 MW of backup generators operating from 9:00 a.m. to 9:00 p.m. local time per day per city. This scenario is developed by isolating the top 200 hr with hourly load data from the Federal Energy Regulatory Commission (FERC) form 714.[25] For New York City and Chicago, electricity load data specific to the city control area are used. For Dallas and Atlanta, it is assumed that the city follows the same trend as the lower resolution state-level data. In New York City, 1000 MW is approximately the average peak load (defining the load as the difference in the electricity demand between any of the top 200 hr and the load in the 201st hour) and is consistent with estimates of available backup generation capacity in the area. Although 1000 MW exceeds the average peak electricity demand in Atlanta, Chicago, and Dallas, 1000 MW in each city is modeled to produce a conservative estimate of the costs from air quality. Using these loads, two peak periods of 3 days each from July 17–19 and 23–25, 2001, are also identified for the detailed air quality modeling. This time period was also evaluated extensively for the base-case $PMCAM_x$ simulations.[15] Although not all cities experience peak electricity demand on all days, these 6 days are run for all cities to capture additional variability in the meteorological conditions.

The CTM domain is discretized into a horizontal grid of 36 36 km with 14 vertical layers from the surface to 6 km. The lowest model layer is slightly less than 30 m thick vertically. The emissions from the diesel generators are modeled as an evenly distributed area source over the coarse grid cell that contains the majority of the urban area. The baseline concentrations for the modeling do- main are generated with emission files from the Lake Michigan Air Directors Consortium (LADCO).[26] Separate model runs are conducted for each city to capture all of the air quality effects from operating the generators in that city; these results are used to calculate the social costs. One model run is also conducted with generators operating in all cities at once and one run with no changes to the base-case emission fields; all figures in this manuscript show the difference between these two runs. It is assumed that the average of the six modeled days is representative of the change in ambient concentrations that would be observed on any peak electricity day and that there are 30 days of operation over the year. Because peak power demand is primarily a summertime phenom- enon, these assumptions are reasonable. If the generators were to operate in other seasons, additional air quality simulation should be performed to account for systematic seasonal differences in atmospheric chemistry.

The gridded ambient concentrations from $PMCAM_x$ are converted into human health effects with concentration-response (CR) functions derived from epidemiological studies. A range of CR functions are explored, focusing on premature mortality. The primary CR functions used in this work relate the long-term (annual) exposure to $PM_{2.5}$ and premature mortality. Specifically, a fixed pooling of the CR relationships from Laden et al.[27] and Pope et al.[28] is applied. Short-term (daily) mortality from $PM_{2.5}$ based on Klemm and Mason[29] and Schwartz[30] is also applied. In addition, a range of estimates for the carcino- genic effects from diesel particulate matter (DPM) is ex- amined. The U.S. Environmental Protection Agency (EPA) estimated a range of 1 10 ^{3} to 1 10 ^{5} cancer cases for each microgram per cubic meter (g/m^{3}) of DPM for continuous exposure over a 70-yr lifetime but concluded that the existing data are insufficient to derive quantita- tive risk factors.[31] The California Office of Environmental Health Hazard Assessment (OEHHA) and CARB estab- lished a "reasonable estimate" of 3.0 10 ^{4} g/m^{3} over a 70-yr lifetime.[32] To calculate the increase in cases, the change in the annual average concentrations is multi- plied by the cancer risk estimates and divided by 70 yr to estimate the number of cases on an annual basis. To evaluate the effect of O_3 on mortality, the estimates are compared using a 24-hr (daily mean) metric from Bell et al.[33–35] and a peak 1-hr metric from Levy et al.[36] The output of the CR function is multiplied by the exposed population, which is gridded to the same resolution as the output from $PMCAM_x$ using the population distribution from the Environmental Benefits Mapping and Analysis Program (BenMap), version 2.4.85.[37]

The total number of premature mortality events is multiplied by the value of a statistical life (VSL). The VSL is modeled as a Weibull distribution with a mean of $7.5 million (in 2005 dollars) (Weibell scale parameter: $8,300,000; Weibull shape parameter: 1.5096). Also reported are 5 and 95% confidence intervals. All cancer outcomes are cost at the VSL because most cases are lung cancer, which has a poor prognosis.

RESULTS AND DISCUSSION

Private Costs

Figure 1 shows the private costs of operating diesel ICEs with and without emission control retrofits compared with constructing a new natural gas simple cycle turbine. All backup options are less expensive for meeting peak electricity demand than the turbine because the capital costs of the backup generators do not need to be included when using the generator for the peak power application. The capital cost and the fixed O&M for the backup generators are already attributed to the increased reliability (e.g., protection from a blackout) provided by the generator to its owner. Because the additional number of hours of operation for peak power is small (e.g., 200 hr/yr), there is only minimal additional wear on the ICE. By contrast, the owner of a new peaking plant would need to recover the capital costs and the fixed O&M as well as the marginal cost of producing electricity. Using the costs in Table 1, the emissions control retrofits have a levelized cost of 2– 4 ¢/kWh. There is a small fuel efficiency penalty associated with the DPF, NO_x controls, and the dual fuel retrofit, which is included in Figure 1.

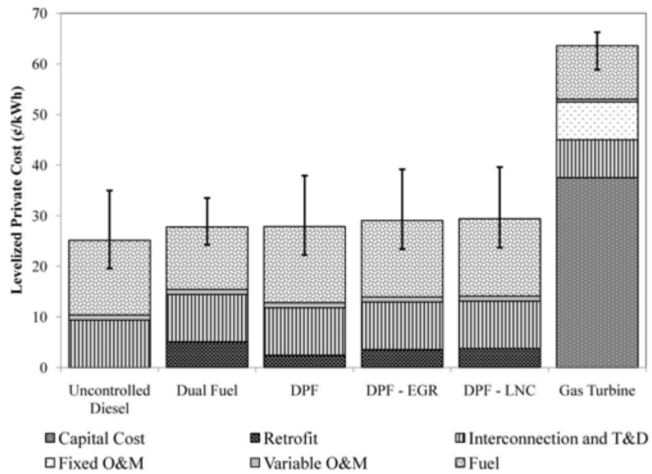

Figure 1. Private costs for installed backup generators with and without retrofit emission controls and a natural gas simple cycle turbine peaking plant in ¢/kWh. The costs shown are the average of the range presented in Table 1. The bars represent the low and high fuel prices. The fuel costs for the dual fuel retrofit is calculated as 80% natural gas and 20% diesel.

Because the full cost of all control options is similar, quantifying the effectiveness of the emission control options at reducing air quality and adverse human health effects is necessary to make a recommendation on the type of retrofit.

Air Quality Effects

Figure 2 shows the difference compared with base-case concentrations for 1-hr peak O_3 concentrations (in parts per billion [ppb]) as an average over the 6 days for an uncontrolled diesel ICE and a diesel ICE with DPF-EGR emission controls. The supplemental data (published at http://secure.awma.org/onlinelibrary/samples/10.3155-1047-3289.60.5.523_supplmaterial.pdf) show color versions of the 1-hr peak O_3 concentrations (Figure S1) and the average change in the daily mean O_3 concentrations (Figure S2) over the 6 days for an uncontrolled diesel ICE and a diesel ICE with DPF-EGR emission controls. The diesel ICE with a DPF has approximately the same changes in O_3 as an uncontrolled diesel ICE. The dual fuel option has approximately the same O_3 changes as a diesel ICE with DPF-EGR controls. For the DPF-LNC option (not shown), the changes in O_3 are between an uncontrolled diesel ICE and a diesel ICE with DPF-EGR controls. The high NO_x emissions from the uncontrolled diesel ICE leads to pronounced decreases in O_3 in the urban centers with smaller increases downwind. This effect is reduced as the NO_x emissions decrease with the emission control retrofits or by shifting to dual fuel. These O_3 concentra- tions are consistent with the representation of its precur- sors, volatile organic compounds (VOC) and NO_x, in $PMCAM_x$. When the initial ratio of NO_x to VOC is high (i.e., VOC-limited regime), adding more NO_x decreases the formation of O_3. At lower ratios (i.e., NO_x-limited regime), the additional NO_x increases the formation of O_3. Urban centers tend to have higher NO_x-to-VOC ratios; therefore, adding more NO_x results in the observed decreases.[38] In the NO_x-limited regimes outside of the urban centers, there are small increases in O_3. These increases are more pronounced around Atlanta and

Dallas, which are more

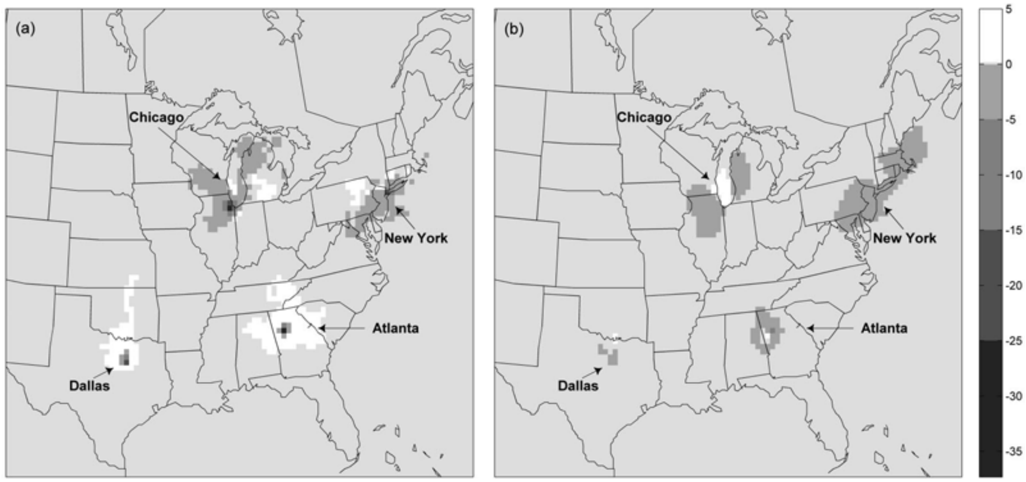

Figure 2. Change in maximum 1-hr O_3 concentrations in ppb averaged over 6 days of meeting 1000 MW of peak electricity demand for 12 hr/day using (a) an uncontrolled diesel backup generator and (b) a diesel backup generator controlled with a DPF and EGR (DPF-EGR). The EGR reduces the NO_x emissions by 50%. Color figures are available in the supplemental data as Figure S1.

strongly NO_x-limited in this model than around Chicago and New York City. These results are broadly consistent with the sensitivities of O_3 with respect to NO_x for these cities found in a modeling study by Liao et al.[39] The pattern of increases and decreases in O_3 can be observed for the difference in peak 1-hr and daily mean averaging periods, although the increases are more pronounced for the peak 1-hr O_3 concentrations. The choice of averaging period on health effects and social costs is addressed in the next section.

Figure 3 shows the change in daily mean $PM_{2.5}$ from base-case concentrations (in g/m^3) averaged over 6 days of operation for an uncontrolled diesel ICE and a diesel ICE retrofitted with a DPF-EGR. Color versions are available in the supplemental data (Figure S3). The increase in $PM_{2.5}$ is mainly due to increases in PEC and POC. Because much of the $PM_{2.5}$ is primary, adding a DPF reduces ambient concentrations. Also observed are small increases and decreases of approximately 0.5–1 g/m^3 in secondary $PM_{2.5}$, which is formed by chemical reactions of NO_x, SO_2, and VOCs. In VOC-limited urban centers, an increase in NO_x emissions results in a decrease of oxidant concentrations (e.g., O_3 and the hydroxyl radical, OH), which are necessary to oxidize the precursor gases to their aerosol form.[40] As a result, despite the increases in SO_2 and VOC emissions, PSO_4 and secondary organic aerosol (SOA) decrease slightly.[41] Further, although there is a decrease in oxidants, increases in PNO_3 are still observed because of the substantial increases in NO_x emissions. For PNO_3 formation, there must also be sufficient ammonia (NH_3) to neutralize the nitric acid (HNO_3) formed from the reaction of NO_x and the oxidants. The decrease in PSO_4 releases NH_3 to react with the HNO_3, making PNO_3 increases more likely. Predicting the amount of “free” NH_3 is challenging because it typically depends on small differences between NH_3 and PSO_4 concentrations and therefore tends to magnify errors in model predictions of both quantities.[42] Evaluating whether the modeled PNO_3

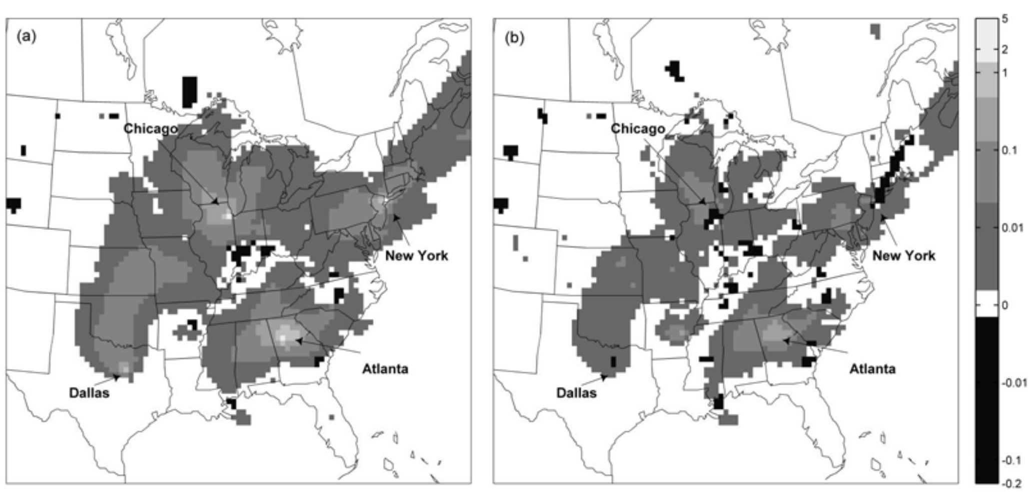

Figure 3. Change in daily mean $PM_{2.5}$ concentrations in g/m^3 averaged over 6 days of meeting 1000 MW of peak electricity demand for 12 hr/day using (a) an uncontrolled diesel backup generator and (b) a diesel backup generator with DPF and EGR (DPF-EGR). The DPF reduces the directly emitted $PM_{2.5}$ by 95%. A diesel fuel with 250 ppm of sulfur is used with the uncontrolled diesel backup generator. An ULSD fuel with 15 ppm of sulfur is used with the DPF. The EGR reduces the NO_x emissions by 50%. Color figures are available in the supplemental data as Figure S3.

increases are plausible is further compounded by uncertainties in NH_3 emission inventories and a lack of suitable measurements for gaseous or total NH_3.[43] However, the largest increase in PNO_3 is found in Atlanta, where previous measurements campaigns have found that the formation of PNO_3 is not limited by the available NH_3.[44,45] The magnitude of the increases and decreases also varies with the meteorology, especially for PNO_3. Days with higher PNO_3 formation have colder temperatures and higher relative humidity, which favors the partition of nitrate to the particulate phase.[46] $PMCAM_x$ also has known prob- lems representing PNO_3 because of difficulties simulat- ing the heterogeneous nighttime formation rate of HNO_3.[15] In addition, the chemical mechanisms that form SOA are poorly understood, and the current rep- resentation of SOAs in all CTMs is limited.[47] The next section discusses the implications and conducts a sen- sitivity analysis on the formation of secondary $PM_{2.5}$ with respect to health costs. See the supplemental data (Figures S4 –S11) for the speciation of $PM_{2.5}$ for each emission control combination.

Human Health Effects and Social Costs

Table 2 presents the social costs for operating the backup generators by city for long-term mortality from exposure to $PM_{2.5}$ for the following specifications: (1) sum of positive changes in $PM_{2.5}$ species only, (2) primary $PM_{2.5}$ species (directly emitted POC and PEC) only, and (3) sum of changes in all $PM_{2.5}$ species. These three measures of social cost are examined to explore the uncertainties in the representation of the formation of secondary $PM_{2.5}$ in $PMCAM_x$. It is found that changes in long-term mortality from $PM_{2.5}$ comprise the majority of the costs. Morbidity effects are approximately 10% of the long-term mortality cost. It is assumed that the CR functions derived for the total $PM_{2.5}$ concentrations hold for individual species.

Looking at individual $PM_{2.5}$ species, the primary components comprise most (70%) of the cost for the uncontrolled diesel. However, depending on the city, it is found that secondary $PM_{2.5}$ can have a significant influence on the total social cost once the primary $PM_{2.5}$ is controlled with a DPF. In Chicago, changes in secondary $PM_{2.5}$ result in an overall negative social cost for all $PM_{2.5}$ species approaches. Although this approach may be con- sidered a best estimate of the current model configura- tion, known problems and uncertainties in predicting different secondary $PM_{2.5}$ species limits the confidence in these results. Because the errors in predicting the different secondary $PM_{2.5}$ species are largely uncorrelated, it is pos- sible that operating the backup generators could lead to the increases predicted for some species but not the de- creases predicted for others. Therefore, only positive changes in $PM_{2.5}$ species are evaluated, which reflects this worst-case scenario.

It is also found that there is an order of magnitude difference in social costs by city, which is driven mainly by the population density in and around the urban center. The large populations in New York City and Chicago result in higher social costs than in Atlanta and Dallas. Local chemistry also plays an important role. This effect is pronounced in Atlanta and Chicago. In Atlanta, increases in secondary $PM_{2.5}$ species account for approximately 30% of the total health costs for an uncontrolled diesel ICE. This limits the effectiveness of the DPF in reducing

Table 2. Mean social cost in ¢/kWh with 5 and 95% confidence intervals from long-term (annual) mortality from $PM_{2.5}$ for the sum of positive changes in species concentrations, changes in primary species only, and the sum of all changes in species concentrations.

City		Uncontrolled Diesel	DPF	DPF-LNC	DPF-EGR	Dual Fuel
Atlanta	Positive changes	62.7 (14.0–150)	18.8 (4.20–45.0)	18.4 (4.10–44.0)	12.2 (2.72–29.2)	14.3 (3.20–34.2)
	Primary species	42.3 (9.45–101)	2.12 (0.47–5.07)	2.12 (0.47–5.07)	2.12 (0.47–5.07)	4.23 (0.94–10.1)
	All species	60.8 (13.6–146)	16.1 (3.58–38.4)	16.5 (3.70–39.6)	10.5 (2.35–25.2)	12.6 (2.82–30.2)
Chicago	Positive changes	101 (22.6–242)	27.1 (6.05–65.0)	22.8 (5.10–54.7)	16.6 (3.72–39.9)	20.5 (4.59–49.2)
	Primary species	78.2 (17.5–187)	3.91 (0.87–9.36)	3.91 (0.87–9.36)	3.91 (0.87–9.36)	7.82 (1.75–18.7)
	All species	80.2 (17.9–192)	2.01 (4.81 to 0.45)	3.32 (0.74–7.96)	0.67 (1.59 to 0.15)	3.24 (0.72–7.76)
Dallas	Positive changes	15.9 (3.55–38.1)	1.55 (0.35–3.71)	2.89 (0.64–6.91)	1.58 (0.35–3.79)	2.22 (0.50–5.32)
	Primary species	12.8 (2.86–30.7)	0.64 (0.14–1.53)	0.64 (0.14–1.53)	0.64 (0.14–1.53)	1.28 (0.29–3.07)
	All species	15.6 (3.47–37.3)	1.13 (0.25–2.71)	2.56 (0.57–6.13)	1.31 (0.29–3.14)	1.95 (0.44–4.67)
New York	Positive changes	186 (41.4–445)	28.1 (6.28–67.3)	27.0 (6.04–64.7)	19.3 (4.31–46.2)	26.5 (5.92–63.5)
	Primary species	161 (35.9–385)	8.04 (1.80–19.3)	8.04 (1.80–19.3)	8.04 (1.80–19.3)	16.1 (3.60–38.5)
	All species	173 (38.6–413)	10.7 (2.39–25.6)	15.3 (3.41–36.6)	8.46 (1.89–20.3)	16.5 (2.69–39.5)

Notes: The costs are shown by city for each generator/emission control technology retrofit option.

the social cost. When considering only increases in secondary $PM_{2.5}$, a similar effect is found in Chicago. In Dallas, the social costs are small because of a smaller population in the surrounding area and minimal formation of secondary $PM_{2.5}$.

By using the CR functions derived for all ambient $PM_{2.5}$, it is assumed that the $PM_{2.5}$ observed from operat- ing diesel and dual fuel ICEs is similar to the $PM_{2.5}$ ob- served in the epidemiological studies. This is not strictly true, because the composition of DPM differs from aver- age ambient compositions. Of specific concern is the re- lationship between DPM and lung cancer. Using the high- est estimate of the relationship from EPA, a social cost is calculated at approximately 10% of the cost calculated using the long-term $PM_{2.5}$ CR functions. The California OEHHA relationship yields a value of approximately 3% of the cost from long-term mortality from $PM_{2.5}$. How- ever, EPA also finds that a zero cancer risk cannot at present be dismissed. In addition, Laden et al.[27] and Pope et al.[28] include lung cancer as an end point in their studies linking ambient $PM_{2.5}$ to mortality. To the extent that ambient DPM contributed to this outcome, there is the potential for double counting if long-term mortality and carcinogenic effects are included. Nevertheless, at 3–10% of the total social cost, this does not represent a signifi- cant source of uncertainty in this assessment.

In this scenario, the generators would operate for only the top 200 hr of electricity demand per year causing changes in $PM_{2.5}$ concentrations on approximately 30 days/yr. However, the long-term CR function and the carcinogenic DPM relationship are based on constant changes in daily exposure to $PM_{2.5}$ (i.e., every day of the year). Thus, the social cost is evaluated from the short-term mortality CR function for $PM_{2.5}$. It is found that the costs from the short-term relationships are approximately 15% of the costs from using the long-term mortality relationships. However, short-term mortality from $PM_{2.5}$ is not included in the 2006 regulatory impact analysis for the revised National Ambient Air Quality Standard (NAAQS) for $PM_{2.5}$ because there is concern that including the long- and short-term effects may result in double counting.[48]

The U.S. National Research Council concluded that short-term exposure to O_3 is also associated with premature mortality on the basis of new evidence from recent studies.[7] These studies found a relationship between the daily mean and peak 1-hr O_3 metrics and premature mor- tality. Table 3 shows the social value of the health effects from these two metrics. With either metric, a social ben- efit due to the decreases in O_3 concentrations in the highly populated urban centers is calculated. Although the formation of $PM_{2.5}$ is subject to error and uncertain- ties, the chemistry that drives the formation of O_3 is well understood, and these results are robust in all modeled cities. By contrast, the magnitude of the relationship be- tween mortality and O_3 is subject to greater uncertainty. For example, using the peak 1-hr CR function from Levy et al.[36] generates benefits 2–3 times of those from using the daily mean CR functions from Bell et al.[33–35] It is not presently known which averaging period is a better pre- dictor of mortality. There is also a distribution issue with NO_x and O_3. The regions surrounding the urban centers experience small social costs from increases in O_3. In addition, although the benefits from the decreases in O_3 in the urban centers exceed the total cost from the in- creases in PNO_3, the suburban/rural areas experience up to a 10 times greater social cost from PNO_3 than the urban centers. Recognizing these uncertainties and possible eq- uity issues, these costs are not included in the total social cost. The authors suggest a conservative strategy of controlling NO_x emissions that decreases the benefit from O_3 reductions and the costs from increased secondary $PM_{2.5}$. For all $PM_{2.5}$ species and O_3, the distribution of the costs for each city between the urban center and the surrounding region is shown in the supplemental data (Tables S1–S10).

CONCLUSIONS

The decision to operate backup generators for peak electricity demand rather than construct a new peaking plant is based on the sum of the private costs and the social costs from changes in air quality. Figure 4 shows the full cost by city for each generator/emission control technology combination with the social costs for $PM_{2.5}$ as the sum of positive changes in species concentrations, the sum of primary species, and the sum of all changes in species concentrations. Only the costs for premature mor- tality from the long-term CR relationship for $PM_{2.5}$ are shown. The focus is on the long-term CR relationship because of the comparatively small costs from the short- term CR mortality relationship and the potential carcino- genic effects from DPM as well as the potential for double

Table 3. Comparison of the costs of mortality from O_3 in ¢/kWh with 5 and 95% confidence intervals for the daily 24-hr mean and peak 1-hr concentration–response (CR) functions.

City	Ozone Metric	Uncontrolled Diesel	DPF	DPF-LNC	DPF-EGR	Dual Fuel
Atlanta	Daily mean	17.7 (4.52 to	17.7 (4.52 to	15.0 (3.84 to	11.3 (2.88 to	11.3 (2.88 to
	Peak 1-hr	37.9 (12.1 to	37.9 (12.1 to	32.8 (10.5 to	24.2 (7.74 to	24.2 (7.74 to
Chicago	Daily mean	41.6 (10.6 to	41.6 (10.6 to	34.9 (8.90 to	25.4 (6.47 to	25.4 (6.47 to
	Peak 1-hr	138 (44.2 to 248)	138 (44.2 to 248)	116 (37.1 to 208)	84.1 (26.9 to	84.1 (26.9 to
Dallas	Daily mean	9.62 (2.45 to	9.62 (2.45 to	7.78 (1.98 to	5.25 (1.34 to	5.25 (1.34 to
	Peak 1-hr	25.6 (8.20 to	25.6 (8.20 to	21.1 (6.74 to	13.6 (4.36 to	13.6 (4.36 to
New York	Daily mean	55.4 (14.1 to	55.4 (14.1 to	44.7 (11.4 to	31.4 (8.01 to	31.4 (8.01 to
	Peak 1-hr	155 (49.7 to 278)	155 (49.7 to 278)	124 (39.6 to 222)	84.9 (27.2 to 152)	84.9 (27.2 to 152)

Notes: The costs are shown by city for each generator/emission control technology retrofit option. A negative value represents a social benefit from reduced mortality.

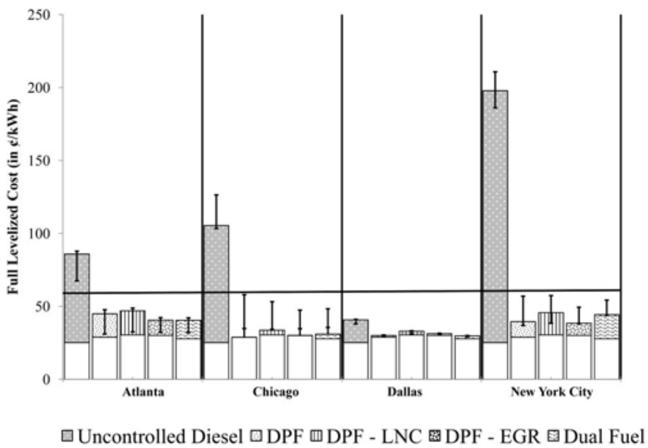

Figure 4. Full (private and social) cost by city and emission control technology in ¢/kWh. The white bars are the private costs. The patterned bars are the social costs from long-term mortality from changes in all species that comprise $PM_{2.5}$ as determined by PM- CAM_x. The high error bars represent the cost associated with posi- tive changes in $PM_{2.5}$ species. The low error bars represent the cost associated with changes in primary (directly emitted) $PM_{2.5}$ species (e.g., EC and OC). The black line is the levelized cost of constructing and operating a simple cycle natural gas turbine for the peak elec- tricity application (60 ¢/kWh).

counting mortality effects. A literature value of less than 1 ¢/kWh for the social cost associated with the natural gas peaking turbine is applied.[49]

Considering only the social cost from $PM_{2.5}$, the full costs for backup generators with any emission control technology are less than a new peaking plant for the three different estimates of social cost at their mean value. Although the differences are small, for changes in all $PM_{2.5}$ species, minimum full costs are found for a DPF without any NO_x controls and a DPF with 50% reductions in NO_x achieved with EGR controls. However, if only positive enhancements in $PM_{2.5}$ are considered, the lowest full cost is achieved with DPF-EGR controls. Given the low cost of NO_x controls, the authors recom- mend retrofitting the generators with these controls and a DPF. In the more densely populated cities of New York City and Chicago, the worst case of costing in- creases in $PM_{2.5}$ species only results in a full cost that is approaching the cost of a peaking plant. As a result, it is also important that the emission controls achieve their expected performance.

Adding the social benefit from the reductions in O_3 would decrease the backup generators' full costs to less than zero. However, the authors caution against a sim- plistic summing of the costs from $PM_{2.5}$ and O_3 because there is uncertainty in the strength of the relationship between O_3 and premature mortality. This effect is also not included because of the importance of decreasing NO_x emissions to reduce the formation of PNO_3. In areas where PNO_3 increases are projected to occur, the authors could have greater confidence in projecting the $PM_{2.5}$ impacts of using distributed backup power if a network of total NH_3 measurements were established.[43] To date, the National Atmospheric Deposition Program (NADP) has established a monitoring network, and Clean Air Status and Trends Network (CASTNET) is planning to measure NH_3.[50]

This sensitivity analysis confirms the results from the case study in Gilmore et al.[5] for New York City and shows that this strategy could also work in Atlanta, Chicago, and Dallas. On the basis of these results, the authors renew their recommendation that the relevant regulatory bodies reconsider their ban on using installed backup diesel ICEs for meeting peak electricity demand, taking care that in- dividual generators maintain appropriate emission stan- dards for $PM_{2.5}$ and NO_x and are properly sited so as not to cause a nuisance in the immediate area.

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