TECHNICAL PAPER

ISSN:1047-3289 *J. Air & Waste Manage. Assoc.* **58**:451–461
DOI:10.3155/1047-3289.58.3.451

Resolving Local-Scale Emissions for Modeling Air Quality near Roadways

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ABSTRACT
A large body of literature published in recent years suggests increased health risk due to exposure of people to air pollution in close proximity to roadways. As a result, there is a need to more accurately represent the spatial concentration gradients near roadways to develop mitigation strategies. In this paper, we present a practical, readily adaptable methodology, using a "bottom-up" approach to develop a detailed highway vehicle emission inventory that includes emissions for individual road links. This methodology also takes advantage of geographic information system (GIS) software to improve the spatial accuracy of the activity information obtained from a Travel Demand Model. In addition, we present an air quality modeling application of this methodology in New Haven, CT. This application uses a hybrid modeling approach, in which a regional grid-based model is used to characterize average local ambient concentrations, and a Gaussian dispersion model is used to provide texture within the modeling domain because of spatial gradients associated with highway vehicle emissions and other local sources. Modeling results show substantial heterogeneity of pollutant concentrations within the modeling domain and strong spatial gradients associated with roadways, particularly for pollutants dominated by direct emissions.

IMPLICATIONS
This paper presents a generic, readily adaptable method for developing emission inventories and conducting air quality modeling in an urban area to improve the characterization of pollutant concentrations near roads. Application of this methodology will help assess mitigation strategies to address elevated pollutant concentrations near roads and resultant adverse health effects.

INTRODUCTION
A large body of literature, much of it published in recent years, suggests increased risks for exacerbation of asthma and other respiratory diseases, premature mortality, certain cancers, and adverse birth outcomes from air pollution exposures in populations residing in relatively close proximity to roadways.[1,2] Furthermore, several air quality monitoring studies have revealed the presence of elevated concentrations of pollutants emitted directly by motor vehicles near major roadways when compared with overall urban background levels.[3–8] These elevated concentrations generally occur within a few hundred meters of the road, although this distance may vary depending on traffic patterns, environmental conditions, and the presence of near-roadway urban structures. Pollutants with elevated concentrations measured near major roadways include coarse, fine, and ultrafine particulate matter (mass and particle number), black carbon (BC), polycyclic aromatic hydrocarbons, carbon monoxide (CO), oxides of nitrogen (NO_x), and benzene.

Air quality modeling can provide a linkage between emissions from activity on roadways and resultant air concentrations. Establishing such linkages is critical for transportation planning and developing mitigation strategies. Thus, environmental and transportation planning officials are increasingly interested in developing more accurate geographic representations of highway vehicle emissions in regional and urban-scale assessments, to better identify where air quality is most impacted by traffic as well as populations likely to experience elevated health risks from air pollutant exposure occurring along roadways. Current approaches for characterizing ambient air pollutant concentrations at an urban scale rely on developing a detailed emissions inventory and applying a dispersion model, such as the American Meteorological Society/U.S. Environmental Protection Agency (EPA) regulatory model (AERMOD), which can simulate large

gradients in modeled ambient concentrations of toxic pollutants in urban areas.[9]

In the past, air quality modelers relied on data from national inventories, such as EPA's National Emission Inventory (NEI) for emission inputs to be used with air quality models. These inventories typically rely on a "top-down" approach to estimate highway vehicle emissions. Under a top-down approach, emissions are estimated at the county level, typically starting from more aggregated information (e.g., state or national level). Spatial surrogates, thought a priori to correspond with activities that produce emissions, are then used to allocate emissions to grid cells or Census tracts for modeling. Air quality modeling using such inventories can be useful in elucidating patterns of ambient concentrations across broad geographic areas, and can help characterize air quality trends and potential impacts of controls at a broad geographic scale.[2,10,11] However, such inventories are often inadequate in identifying spatial gradients of pollutant concentrations resulting from highway vehicle activity, and locations where the highest risks are likely to occur. Thus, several recent studies have used "bottom-up" approaches, which rely on more local inputs to estimate better motor vehicle emission factors and vehicle counts or activity data from a metropolitan area's travel demand model (TDM), to develop a more accurate geographic representation of motor vehicle emissions.[12–14]

These bottom-up approaches provide greater power to predict the actual location of emissions. Such approaches use similar traffic data as studies that have predicted ambient concentrations of mobile source-related pollutants using land use regression. These studies report that ambient concentrations of mobile source-related pollutants such as benzene, formaldehyde, fine particulate matter ($PM_{2.5}$), and BC are associated with traffic within several hundred meters of an ambient monitor.[15,16] This suggests that using similar traffic-related variables, such as traffic, population, and industrial land use to predict the location of motor vehicle emissions may improve the predictive capacity of urban-scale air quality models.

In this paper, we present a readily adaptable methodology using a bottom-up approach to develop a detailed highway vehicle emission inventory that includes emissions for individual road links. This method also uses geographic information system (GIS) software to improve the spatial accuracy of the activity information obtained from a TDM model. In addition, we present an application of this methodology to New Haven, CT, a small urban area with typical emission sources. Figure 1 depicts the modeling domain, including locations of major roads. The red dots represent the location of major sources that emit greater than 10 t/yr of any pollutant, and the black dots show the location of the centroids for individual U.S. Census block groups (a Census block group is a combination of Census blocks, which are the smallest geographic areas for which the U.S. Census Bureau collects data.) As shown in Figure 1, there is a dense network of roadways in the New Haven modeling domain, and most of the Census block group centroids are located close to the roads. This illustrates the importance of characterizing the near-road impacts in modeling assessments.

Air quality modeling using this detailed highway inventory was done using a hybrid approach.[17,18] Although numerical photochemical grid models are the model platform of choice for simulation of atmospheric chemistry and fate of airborne pollutants on a larger scale, there are various transport and dispersion models that have been

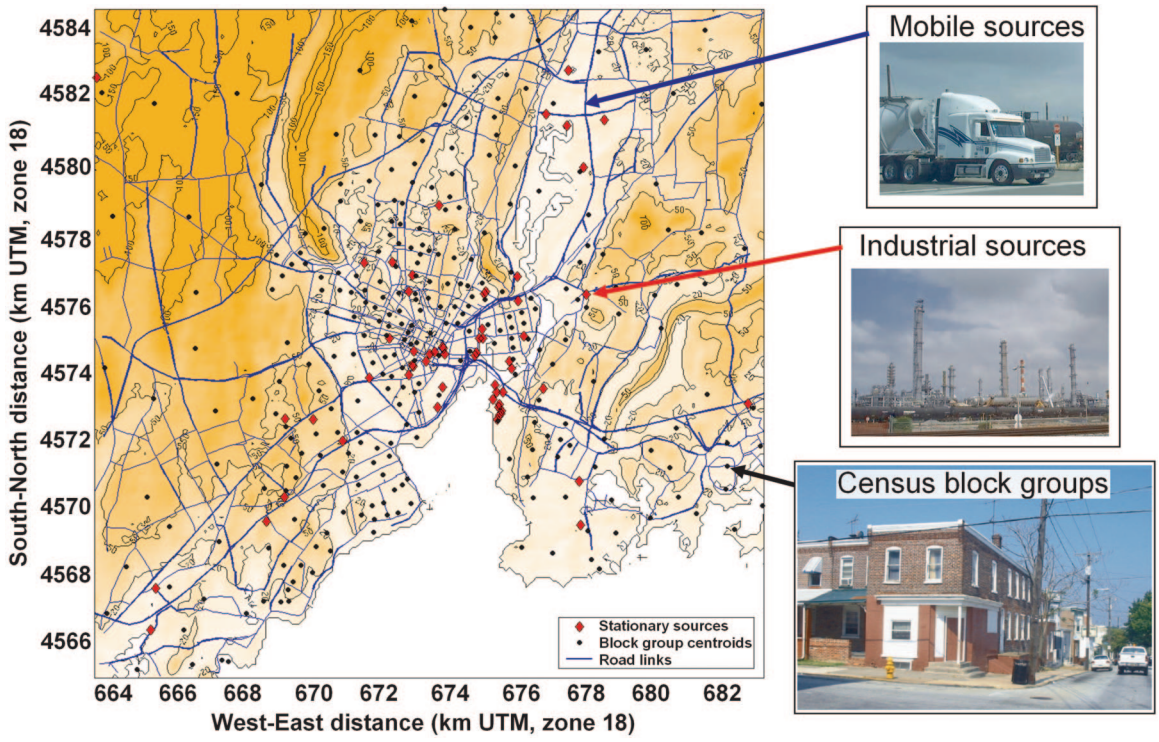

Figure 1. New Haven modeling domain, including major sources and road locations.

developed to simulate the fate of those airborne pollutants that are relatively chemically inert. Although lacking detailed atmospheric chemistry, dispersion models can provide detailed resolution of the spatial variations in hourly-average concentrations of airborne pollutants. To date, local-scale dispersion models have been used to provide the desired detailed description of the concentration pattern. However, local-scale dispersion models cannot properly treat photochemically generated pollutants. Also, an estimate of background concentrations must be provided to local-scale models; these can be provided directly by the grid model. Combining the capabilities of numerical photochemical models and dispersion models into one model is desirable, but this is a yet evolving area of research and development.

In this study, the hybrid approach used existing results from the Community Multiscale Air Quality (CMAQ) grid model and the AERMOD dispersion model to simulate concentration gradients in the modeling domain.[19,20] CMAQ provides volume-average concentration values for each 12- × 12-km grid cell in the modeling domain, given stated conditions that can change hourly. Emissions are assumed to be instantaneously well mixed within the cell where they are emitted. In contrast, AERMOD provides detailed resolution of the spatial variations in hourly-average concentrations of airborne pollutants because of the improved spatial emission allocations. Results of both model simulations can be combined to provide the total ambient concentrations. The study relied on detailed information from the bottom-up inventory for highway vehicle exhaust and evaporative running emissions on roads, as well as local point sources, marine port, and airport emissions data, to provide this resolution. Emissions of area sources and nonroad sources not associated with ports and airports were assumed to be uniformly distributed across grid cells. These emissions are allocated to grid cells using surrogates such as population and land use. However, data that could significantly improve allocations within grid cells were not readily available. Although this study is based in one city, the methodologies developed through this project will have broader application to other urban areas within the United States where vehicle activity data from TDMs are available.

METHODOLOGY TO CREATE LINK-BASED EMISSIONS INVENTORY

This methodology allows us to create a spatially and temporally resolved emissions inventory for mobile sources (i.e., hourly emissions for all pollutants modeled, by vehicle class, by road link).

The total emissions are calculated from emission factors multiplied by traffic activity for each link:

$$E_i(s) = EF_i(s) \times A(s) \quad (1)$$

where $E_i(s)$ is the emission rate (mass per unit of time) for pollutant i from a source (or road link) s, $EF_i(s)$ is the emission factor (mass per unit of activity) for pollutant i from source s, and $A(s)$ is the activity level for source s (e.g., vehicle miles traveled [VMT]) over a given time.

In this process, emission factors are representative for a vehicle class in the study area and are a function of speed, temperature, and road type (e.g., freeway, arterial). Traffic activity level is the VMT by vehicle classes on each link over a given time interval. Therefore, three major inputs are required to develop a bottom-up highway vehicle inventory for exhaust and evaporative running emissions that are primarily emitted on major roadways. These are:

(1) locations of individual road links;
(2) traffic activity by vehicle class on those links; and
(3) emission factors by vehicle class.

Location of Individual Road Links

In this study, the geographic database and associated attribute data used to create the link-level road network for the New Haven area were obtained from the Connecticut Department of Transportation (CDOT). The data were developed using the TRANPLAN (TRANSportation PLANning) four-step travel demand integrated model (www.citilabs.com/tranplan/). This database contains information on all major roads, and roadway activity data is output in the form of average annual daily traffic (AADT) for each road link. Other output from TRANPLAN are average speed for each road link during peak and offpeak periods, coordinates of the segment endpoints (nodes), roadway type, and number of lanes. Speeds from TRANPLAN were used to develop link-specific emission rates.

Although use of TDM data greatly improves geographic representation of emissions over top-down approaches that rely on spatial surrogates, a significant limitation for air quality modeling purposes in populated areas is that TDM data represent curvilinear roads as one or more straight line segments. A single model link may represent more than one road in the actual transportation system and links can intersect where none exist. As a result, misalignments with actual road locations of up to several hundred meters can occur.[21] Because recent research has found strong spatial gradients of pollutants near roads that are localized within a few hundred meters,[22–24] mislocation of roads can result in large errors in estimating ambient concentrations at receptors. Potential misalignment of road locations can be addressed by merging the TDM dataset with a geographic dataset with better spatial accuracy. Merging two geographic datasets to produce an improved target dataset is known as conflation and is used here to combine attributes from the TDM (AADT, speed) with the better spatial accuracy of the U.S. Census Topographically Integrated Geographic Encoding and Referencing (TIGER) 2000 road network. TIGER is readily available from the U.S. Census Bureau.[25] This approach was used in development of the Mobile Emission Assessment System for Urban and Regional Evaluation (MEASURE) under a cooperative agreement between the Georgia Institute of Technology and EPA.[26]

Overlaying the datasets in the GIS (ArcInfo) showed enough spatial similarities existed that a link-by-link assignment could be established. Individual travel model links were selected as source features and the corresponding accurate road segments were selected as target features that then assigned the link attributes to the TIGER road

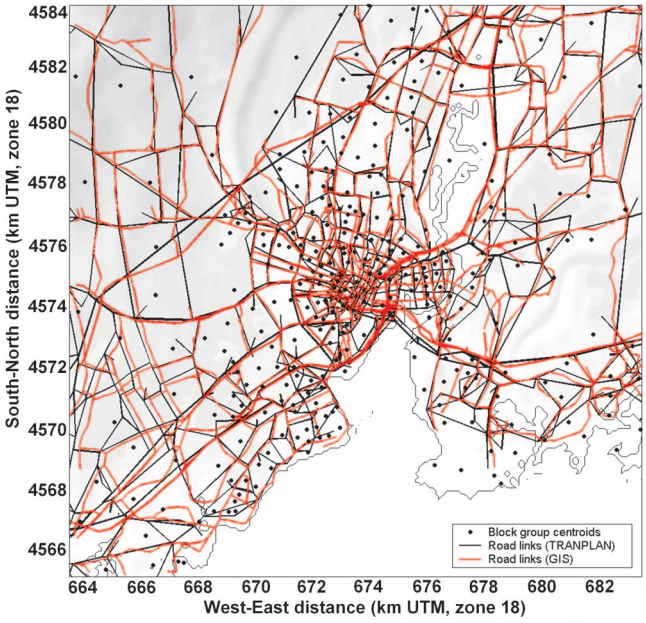

Figure 2. New Haven road links from TRANPLAN vs. TIGER road segments, along with centroids for individual U.S. Census block groups.

segments. Figure 2 compares the locations of the New Haven road links from TRANPLAN and TIGER, along with centroids for individual U.S. Census block groups. If sufficient resources are available, spatial accuracy of an urban scale assessment can be improved even more by transferring activity data to a global positioning system (GPS)-validated network. GPS-derived road network databases have greater spatial accuracy than TIGER. In the Southern California Children's Study, activity data were transferred to a GPS-validated network.[21]

Traffic Activity on Road Links

To determine activity level on each link, we used the TRANPLAN database for calendar year 2002, which consists of a roadway type indicator, directional AADT values, number of lanes, and average speed. Directional AADT values were summed and multiplied by the roadway segment length to obtain daily VMT values. Daily VMT was allocated to eight individual vehicle classes by roadway type using estimated fractions of VMT for each vehicle class obtained from CDOT. These eight vehicle classes are (1) HDDV, heavy-duty diesel vehicles; (2) HDGV, heavy-duty gasoline vehicles; (3) LDDT, light-duty diesel trucks; (4) LDDV, light-duty diesel vehicles; (5) LDGT1, light-duty gasoline trucks 1; (6) LDGT2, light-duty gasoline trucks 2; (7) LDGV, light-duty gasoline vehicles; and (8) MC, motorcycles. MOBILE6.2 estimates emissions for 28 vehicle classes, which can be aggregated into these eight broader classes in the output.

Because emission factors are estimated only on a monthly basis, and activity is estimated by TRANPLAN as annual average daily traffic, activity must be allocated to time of day and time of year. Seasonal temporal profiles were obtained from an ancillary file developed for the Emission Modeling System for Hazardous Air Pollutants (EMS-HAP) database, and were based on data developed for regional emission modeling studies under the National Acid Precipitation Assessment Program (NAPAP).[27] The temporal profiles were specific to vehicle type (across all road types), season, hour of day, and day of week (weekday, Saturday, or Sunday). Multiplying the annual average daily traffic count by the temporal profiles allocated the traffic counts to the appropriate season, hour of day, vehicle, and day of week.

Emission Factors

The two primary emission factor models used for highway vehicles in the United States are MOBILE6.2 and EMission FACtors (EMFAC). The latter model is used in California, and MOBILE6.2 is used throughout the rest of the United States.[28,29] However, EMFAC emission factors represent entire vehicle trips whereas MOBILE6 has more flexibility to estimate link-specific emission rates. Thus, the use of EMFAC presents challenges in modeling emissions at the link level.[30] In this study, we used MOBILE6.2 to generate an emission factor "look-up" table for calendar year 2002 that provides running exhaust and running evaporative emission factors for each of the vehicle classes as a function of speed and temperature. This was done using MOBILE6.2's spreadsheet output option, which produces output from batch runs in a tab-delimited ASCII file. Emission factors also vary by road type (freeway and arterial), so the average emission rate at a given speed and temperature was estimated as an average of emission rates for the individual road types. Nonrunning emissions are not associated with roadway links, and thus were not included in the table. Brake and tire wear particulate emissions for coarse particulate matter (PM_{10}) and $PM_{2.5}$ were included.

Both speed and temperature significantly impact emission rates from highway vehicles. Figures 3 and 4 provide examples for benzene, formaldehyde, CO, and NO_x in calendar year 2002. The impacts of speed and temperature shown in these examples are consistent with those observed in other analyses.[31,32] It should be noted that in MOBILE6.2, there are no speed or temperature effects for PM emissions.

Figure 3 illustrates the variability of emission factors by vehicle class as a function of vehicle speed for four pollutants in New Haven. These factors vary considerably by vehicle class and pollutant. In general, fleet average emission factors for gaseous air toxics are higher at lower speeds. Emissions factors for CO and NO_x increase at both low and high speeds. HDDVs experience a large increase in NO_x emissions at high speeds. Figure 4 shows the emission factors by vehicle class as a function of ambient temperature for various pollutants. Emission factors for gaseous air toxics are high at low and high ambient temperatures. It should be noted that recent research indicates that emissions of gaseous air toxics from gasoline vehicles with advanced emissions control technology are much higher at low temperatures than currently estimated by MOBILE6.[2] CO shows a very large dependence on temperature for gasoline vehicles; in contrast NO_x shows little effect. Temperature has no effect on diesel emissions in MOBILE6.

Input files for MOBILE6.2 for 2002 were obtained from the CDOT. In addition to these input files, modeling

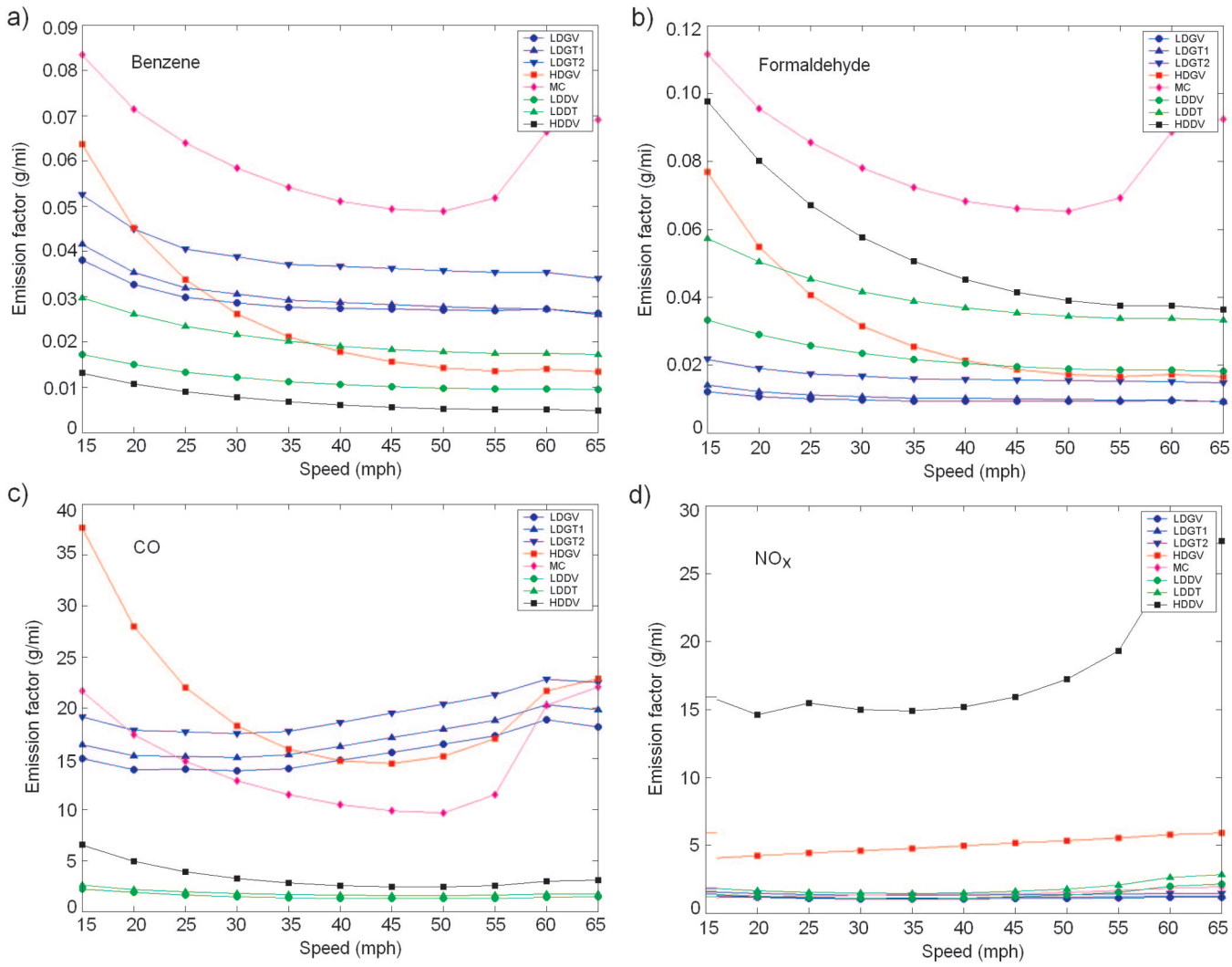

Figure 3. Emission factors by vehicle class as a function of speed for various pollutants—(a) benzene, (b) formaldehyde, (c) CO, and (d) NO_x—for New Haven, CT, 2002.

of air toxics using MOBILE6.2 requires additional fuel parameters not needed for modeling those pollutants for which EPA has the National Ambient Air Quality Standards (NAAQS) for pollutants such as PM_{10}, $PM_{2.5}$, NO_x, sulfur dioxide (SO_2), and CO. These fuel parameters include aromatics level, benzene level, olefin level, and percentage of gasoline evaporated at 200 and 300 °F (E200 and E300). Seasonal properties are available from surveys of gasoline stations. For New Haven, such data were collected as part of the Federal Reformulated Gasoline program.[33]

Calculation of Link-Level Emissions

Using hourly traffic activity and emission factors described in previous sections, the hourly emissions by vehicle class for each of the air toxics for individual links are calculated using eq 1. In this process, emission factors are matched by speed on any specific road link and by temperature for each hour. This procedure is repeated for each of the vehicle classes.

The approach described above can be used in any area of the United States with link-level vehicle activity data. In addition, a tool to obtain the emission rates at the link level has been developed for the Great Lakes Region. This tool, the Consolidated Community Emissions Processing Tool (CONCEPT),[34] is freely available and can significantly reduce the time and resources needed to develop a detailed highway inventory for states in that region as outlined in this paper. Spatial resolution of an inventory developed using CONCEPT could also be enhanced using the GIS approach outlined above.

MODELING APPLICATION IN NEW HAVEN, CT

To illustrate how this methodology can be used to create link-based emissions for near-road modeling applications, we conducted a modeling demonstration of these link-based emission techniques in New Haven, CT. The New Haven modeling effort is part of a broader feasibility study to evaluate impacts of regulatory and voluntary actions to reduce air toxic emissions. New Haven is one of EPA's nationally funded Community Air Toxics projects and has implemented several voluntary air pollution programs aimed at reducing both

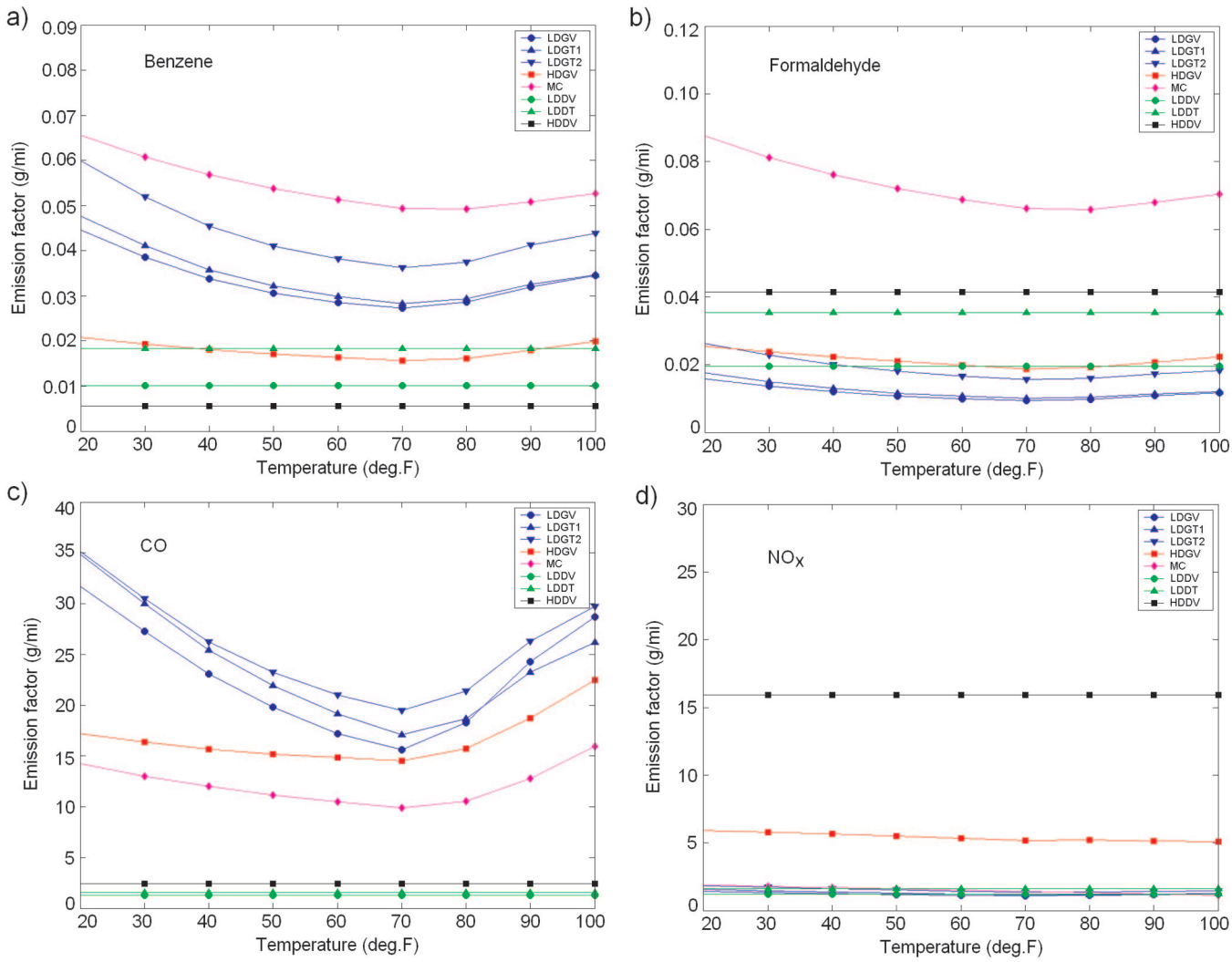

Figure 4. Emission factors by vehicle class as a function of ambient temperature for various pollutants—(a) benzene, (b) formaldehyde, (c) CO, and (d) NO_x—for New Haven, CT, 2002.

criteria and air toxic pollutants. In this study, the AERMOD model was used to estimate concentrations for several pollutants: PM_{10}, $PM_{2.5}$, NO_x, SO_2, CO, benzene, 1,3-butadiene, formaldehyde, acrolein, trivalent and hexavalent chromium (Cr^{3+} and Cr^{6+}), and diesel particulate matter. In this paper, we present results for benzene and CO because these are primarily mobile source pollutants and CMAQ simulation results were readily available.

The AERMOD dispersion model provides detailed resolution of the spatial variations in hourly-average concentrations of nonreactive pollutants associated with highway vehicle emissions and major sources. The AERMOD model provided hourly concentrations at Census block group centroids (~380 receptors in the New Haven modeling domain). The study area includes multiple stationary sources, mobile sources, a marine port, and an airport. Emission rates from sources with sufficient information on source location such as point, on-road mobile, and marine port and airport (nonroad) were directly input in AERMOD. The level of detail to allocate most nonroad mobile and area source emissions was not of sufficient detail for local modeling with AERMOD. CMAQ provides volume-average concentration values for each 12- × 12-km grid cell in the modeling domain, given stated conditions that can change hourly. CMAQ modeled concentrations were assumed to represent the contribution of all remaining emission sources (not directly input into AERMOD): nonroad mobile sources, area sources, and background due to long range transport. Results of both model simulations are combined to provide the total ambient air toxics concentrations.

All point source emissions were extracted from a version of the 1999 NEI inventory. This inventory was thoroughly reviewed when used in the 1999 National Air Toxics Assessment (NATA)[10] and was the most recently available air toxics inventory at the time of this study for illustrative purposes. Because of the desire for accuracy for the local-scale modeling, extensive location quality assurance was performed using Google Earth, EPA's online Facility Registry System (FRS), and in some cases, the 2002 draft NEI. All sources with latitude-longitude coordinates inside a 20- × 20-km modeling domain centered on the city of New Haven are included. These stationary source coordinates as well as stack parameters were input to AERMOD.

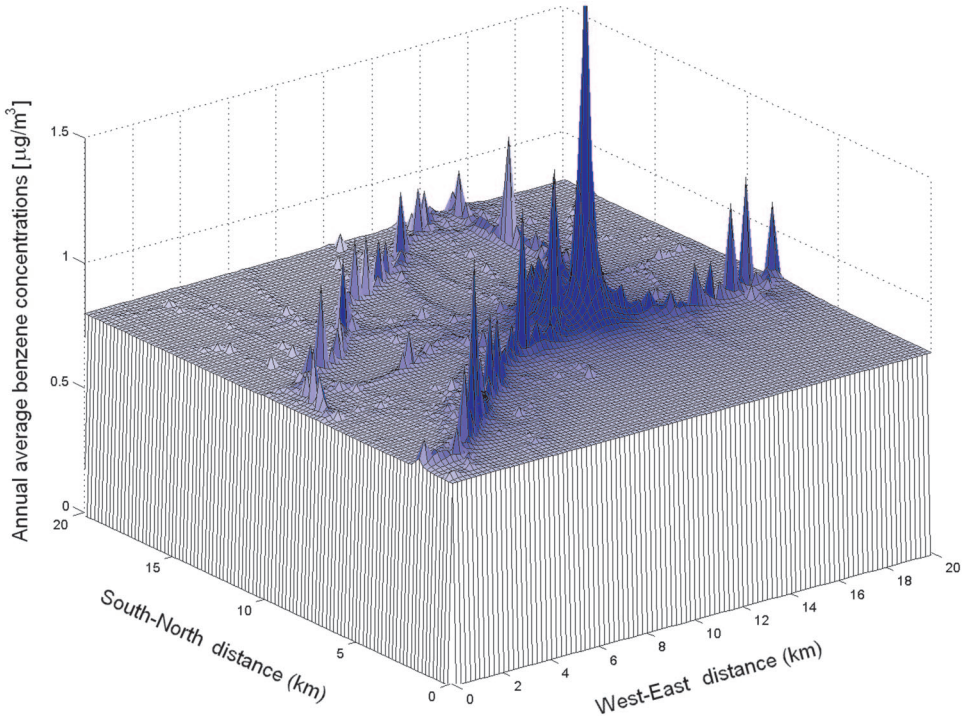

Figure 5. Spatial distribution of modeled annual average benzene concentrations in New Haven, CT.

For the near-road impact of mobile sources, the highway vehicle inventory was developed using the bottom-up versus top-down approach. Locations of individual road links along with link configuration and emission rates were input into AERMOD and modeled as area sources.[35] An extensive discussion of differences between the top-down and bottom-up approaches can be found elsewhere.[12] Also, whereas the highway vehicle inventory used in AERMOD modeling included only the running emissions associated with road links, CMAQ modeling included other types of vehicle emissions such as vehicle start exhaust, evaporative emissions caused by daily temperature changes while a vehicle is parked (diurnal emissions), evaporative emissions produced after a vehicle is

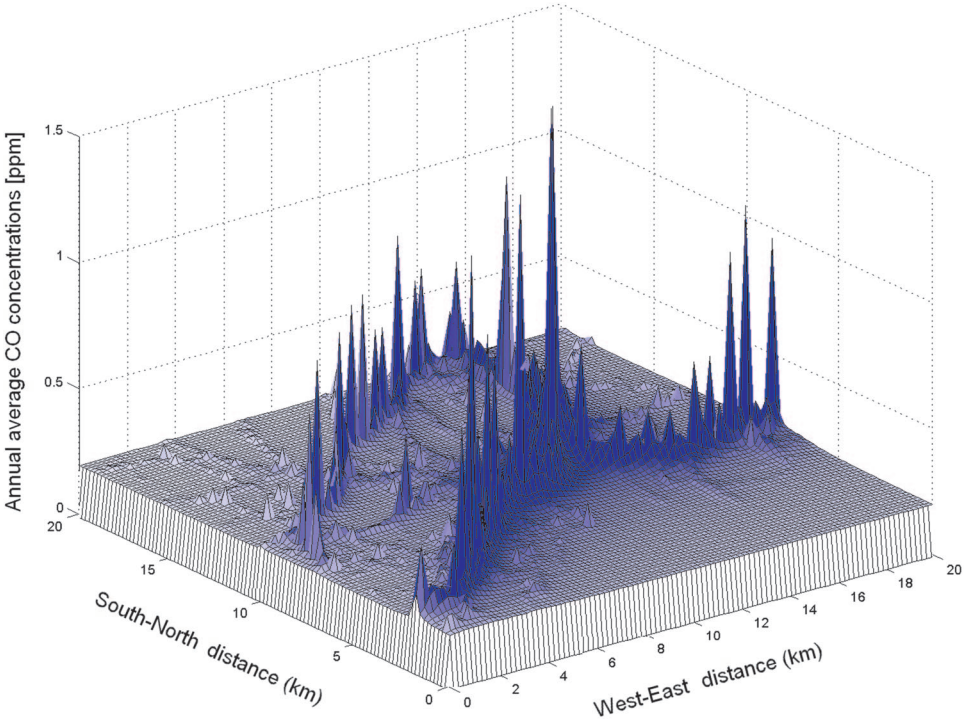

Figure 6. Spatial distribution of modeled annual average CO concentrations in New Haven, CT.

stopped and turned off (hot soak emissions), and emission from leaks or permeation of gasoline from the fuel system while the vehicle is off (resting loss emissions). Whereas the exhaust and evaporative running emissions used in AERMOD were for calendar year 2002, these emissions were obtained from the 1999 NEI.

Figures 5 and 6 show the spatial distribution of hybrid-modeled annual average concentrations in New Haven for two pollutants: benzene, representative of air toxics, and CO, representative of criteria pollutants. For visualization purposes, the model was run with an equally spaced 200-m receptor grid. Both pollutants are mobile-source driven. As shown in Figure 5, benzene concentrations are highest along major roadways and also downwind of the port area

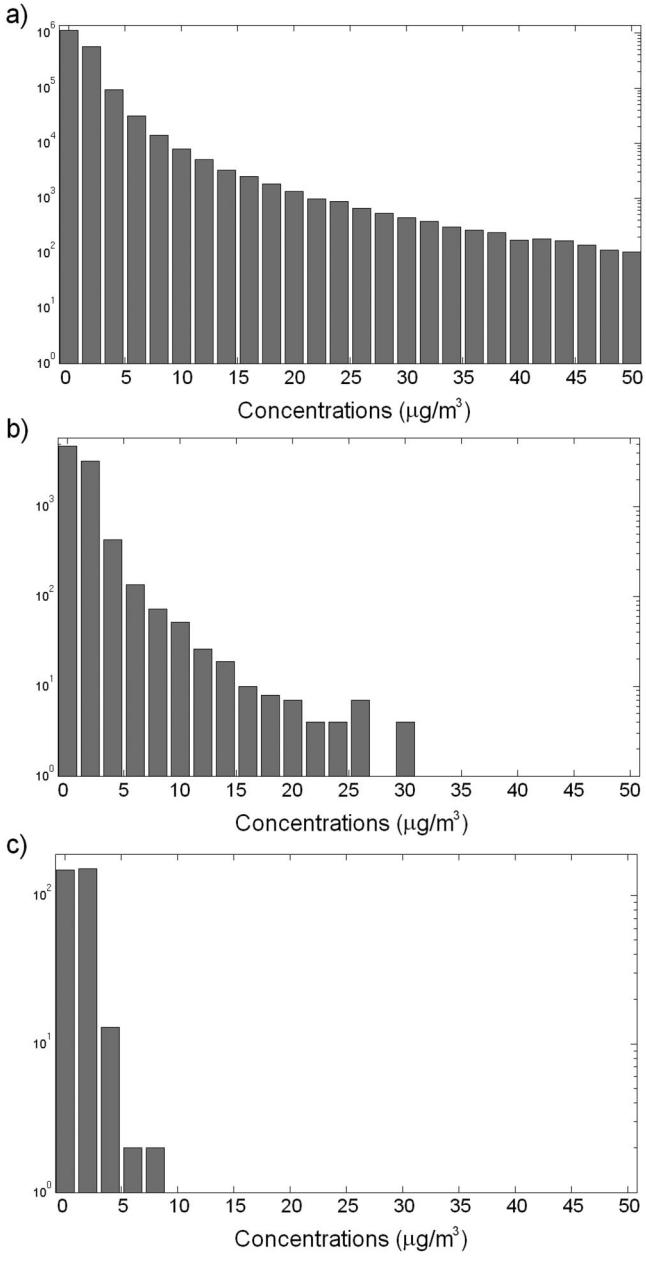

Figure 7. Distributions of modeled benzene concentrations in New Haven, CT: (a) hourly, at each of the 318 block group centroids; (b) hourly, averaged over the entire modeling domain; and (c) annual averages for each of the 318 block group centroids.

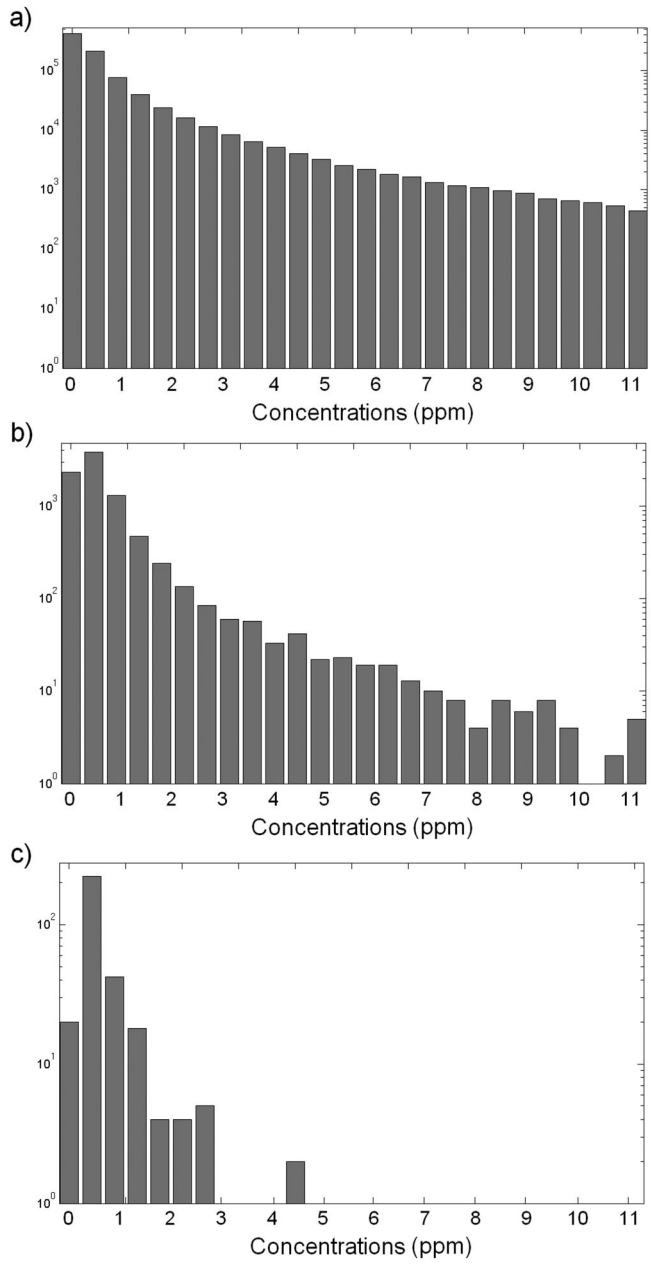

Figure 8. Distributions of modeled CO concentrations in New Haven, CT: (a) hourly, at each of the 318 block group centroids; (b) hourly, averaged over the entire modeling domain; and (c) annual averages for each of the 318 block group centroids.

(highest peak in the figure) where there are large emission sources. Figure 6 shows the spatial distribution of modeled annual average CO concentrations. These figures are similar in that they show high concentrations near major highways because these two pollutants are primarily emitted from mobile sources.

Figure 7 shows the spatial and temporal variations of benzene concentrations in the study area. The top panel (a) shows the frequency distribution of benzene concentrations for all hours in the study period, that is, hourly concentrations at each of the 318 block group centroids in calendar year 2002. As expected, the frequency of occurrence decreases as concentrations increase. Panel b shows the distribution of these hourly concentrations as they are spatially

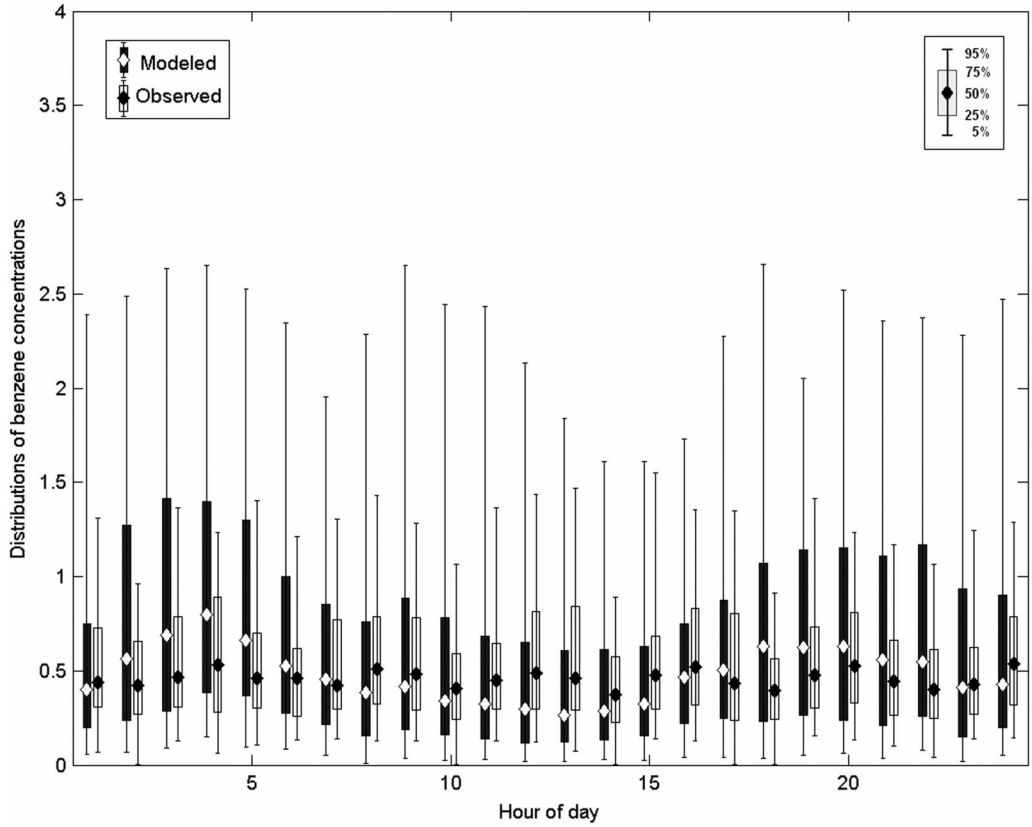

Figure 9. Comparison of modeled and observed distributions of benzene concentrations ($\mu g/m^3$) in New Haven, CT, at monitor no. 9005 (residential, suburban), by hour of day. Dots represent median values of the distribution; boxes represent 25–75% range; and whiskers represent 5–95% range.

averaged over the entire domain. The range of concentrations across the domain is less than in panel a, but is also high nevertheless. When the hourly concentrations are averaged at each Census block group over the entire year (c), the range in concentrations is much smaller, thus indicating that overall variability in ambient benzene is dominated by temporal variability. This is not unexpected because these pollutants are primarily emitted by mobile sources, which have a strong diurnal emission pattern. A similar pattern is observed for CO (Figure 8).

To evaluate model results, we compared hourly benzene and CO model predictions with existing monitor data in the New Haven area. Figure 9 shows a comparison of modeled and observed distribution of benzene concentrations by hour of day at monitor 9005 (latitude 41.34111 N, longitude 72.921389 W) in the New Haven area. This monitor is at a residential, suburban location. Generally, modeled results agree with observed values, and are within a factor of two. However, the modeled concentrations show a wider range of variability than the monitored values. Both modeled and observed concentrations vary according to time of day. This is expected because benzene emissions from automobiles are higher in the morning and afternoon rush-hour periods. In addition, meteorological conditions are more favorable to low dispersion during the morning.

Figure 10 shows a comparison of modeled and observed distribution of CO concentrations by hour of day at monitor 0025 (latitude 41.309167 N, longitude 72.923333 W). This monitor is located at the commercial center of the urban area, and does not show significantly higher concentrations during rush-hour periods. Here also, modeled results agree with observed values and are within a factor of two.

Although the approach presented in this paper represents a substantial advance over more traditional modeling approaches, there are still a significant number of limitations and uncertainties. Among these limitations and uncertainties are:

- Speeds obtained from travel demand models can be inaccurate.[36] In addition, the study used daily average speed and did not account for differences in speed during peak and offpeak hours.
- Roads can be inaccurately located in TIGER.
- Distributions of total VMT among vehicle types for road links can be mischaracterized.
- Differences in diurnal temporal distributions of activity for weekdays versus weekends were not accounted for in modeling.
- MOBILE6.2 does not account for speed or temperature effects on PM emissions, or temperature effects on diesel vehicle emissions.

SUMMARY

With more accurate geographic representation of spatial gradients associated with roadways and point sources, the results presented here appear to be better suited for developing mitigation strategies to address elevated pollutant concentrations near roads and resultant adverse health

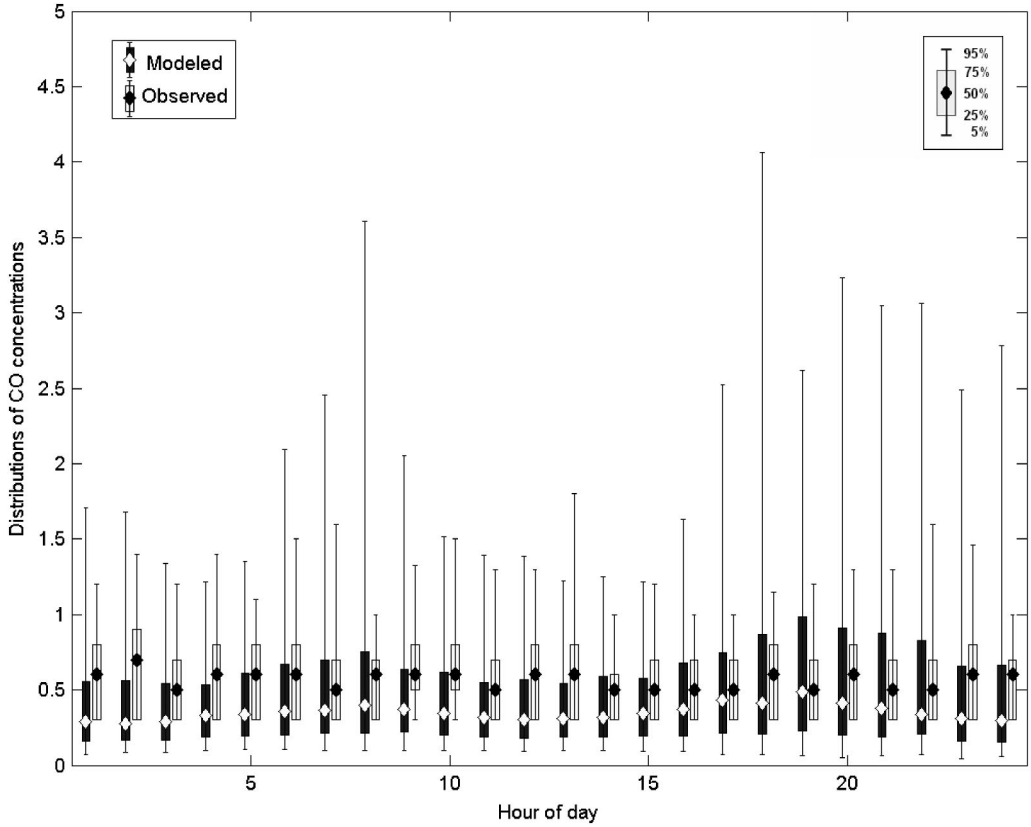

Figure 10. Comparison of modeled and observed distributions of CO concentrations (ppm) in New Haven, CT, at monitor no. 0025 (urban and center city, commercial), by hour of day. Dots represent median values of the distribution, boxes represent 25–75% range, and whiskers represent 5–95% range.

effects than more traditional air quality modeling approaches. In addition, the hybrid modeling approach combines the advantages of both models to provide better spatial resolution than either model alone. More refined approaches are especially important given the growing body of literature on near-road health effects. Furthermore, these data can be aligned with socioeconomic indicators and other population data in environmental justice analyses. Although the approach presented in this paper is more resource intensive than traditional top-down approaches in which emissions are allocated to grid cells for air quality modeling, it is practical and readily adaptable. Such approaches are likely to become more widespread as more tools, such as CONCEPT, are developed to link activity data from travel demand models with emission rates from emission factor models, and as hybrid air quality modeling tools are refined. In addition, EPA's new emissions model currently under development, the Motor Vehicle Emission Simulator (MOVES), will use a modal emission-rate approach that includes acceleration, rather than estimating emission rates based on average speeds.[37] This will improve capabilities to do more refined modeling of emissions at the local scale.

In this illustrative study, many assumptions were made that should be re-examined in real-world applications. For example, emission inventory must be consistent when the results from the AERMOD model are added to the CMAQ model. Also, meteorological data periods must be consistent between AERMOD, CMAQ, and the monitoring data.

ACKNOWLEDGMENTS

The authors thank Rob McConnell and Paul Buckley from CDOT for assistance in executing this effort. The authors also thank Dr. Haluk Ozkaynak and Dr. Danelle Lobdell for leading the research program and management for this study, and Chad Bailey, Dr. Madeleine Strum, and Dr. Richard Baldauf for comments on the manuscript. The research presented here was performed under the Memorandum of Understanding between EPA and the U.S. Department of Commerce's National Oceanic and Atmospheric Administration (NOAA) and under agreement number DW13921548. This work constitutes a contribution to the NOAA Air Quality Program. It does not necessarily reflect Agency policies or views.

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