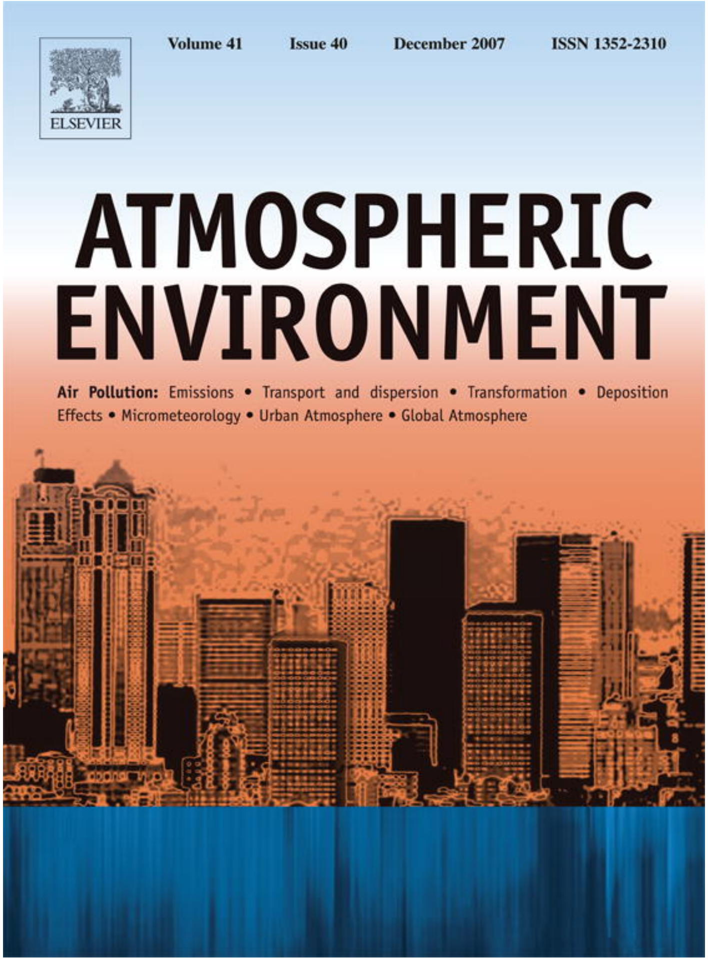
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# Analysis of air quality data near roadways using a dispersion model

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**Abstract**

We used a dispersion model to analyze measurements made during a field study conducted by the U.S. EPA in July–August 2006, to estimate the impact of traffic emissions on air quality at distances of tens of meters from an eight-lane highway located in Raleigh, NC. The air quality measurements consisted of long path optical measurements of NO at distances of 7 and 17 m from the edge of the highway. Sonic anemometers were used to measure wind speed and turbulent velocities at 6 and 20 m from the highway. Traffic flow rates were monitored using traffic surveillance cameras. The dispersion model [Venkatram, A., 2004. On estimating emissions through horizontal fluxes. Atmospheric Environment 38, 2439–2446] explained over 60% of the variance of the observed path averaged NO concentrations, and over 90% of the observed concentrations were within a factor of two of the model estimates.

Sensitivity tests conducted with the model indicated that the traffic flow rate made the largest contribution to the variance of the observed NO concentrations. The meteorological variable that had the largest impact on the near road NO concentrations was the standard deviation of the vertical velocity fluctuations, $\sigma_w$. Wind speed had a relatively minor effect on concentrations. Furthermore, as long as the wind direction was within $\pm 45^\circ$ from the normal to the road, wind direction had little impact on near road concentrations. The measurements did not allow us to draw conclusions on the impact of traffic-induced turbulence on dispersion. The analysis of air quality and meteorological observations resulted in plausible estimates of on-road emission factors for NO.



*Keywords:* Air quality; Line source; Dispersion modeling; Traffic emissions; Mobile sources

## 1. Introduction

Emissions from motor vehicles influence the temporal and spatial patterns of regulated gases, particulate matter (PM), and toxic air pollutant concentrations within urban areas. Air quality monitoring studies conducted near major roadways

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have detected elevated concentrations, compared to overall urban background levels, of motor-vehicle-emitted compounds, including carbon monoxide (CO), nitrogen oxides ($NO_x$), coarse ($PM_{10-2.5}$), fine ($PM_{2.5}$), and ultrafine ($PM_{0.1}$) particle mass, particle number, black carbon (BC), polycyclic aromatic hydrocarbons (PAHs), and benzene (Kim et al., 2002; Hutchins et al., 2000; Zhu et al., 2002a, b; Kittelson et al., 2004). A number of epidemiological studies have reported associations between living close to high traffic roadways and adverse health effects such as asthma and other respiratory impacts, birth and developmental effects, premature mortality, cardiovascular effects, and cancer (e.g. Harrison et al., 1999; Brauer et al., 2002; Hoek et al., 2002; Finkelstein et al., 2004).

Results from these air quality and health effects studies motivated a comprehensive field study, conducted by the U.S. EPA, to characterize the influence of traffic-generated emissions on the temporal and spatial variability in air pollutant concentrations in the near road environment. One of the objectives of this study was the evaluation of existing emissions and dispersion models for near road applications.

The objective of this paper is to identify the variables that govern air quality in the immediate vicinity of a highway. This is achieved by interpreting the variation of long-path NO concentrations measured near the road with a dispersion model that incorporates concurrent meteorological measurements.

Studies similar to ours have been conducted in the past. Noll et al. (1978) report on a study conducted in the vicinity of a major road in Nashville, Tennessee, during July and August 1973. Carbon monoxide concentrations were measured continuously using five probes located at several distances from the road. The meteorological measurements consisted of wind speeds and directions measured at 4 and 9 m above ground. Traffic counts were measured using pneumatic counters and recorders. The data from this field study were used to evaluate three highway line-source dispersion models formulated by the U.S. EPA and the California Air Resources Board.

Chock (1980) describes a comprehensive dispersion study, referred to as the General Motors Sulfate Dispersion Experiment, in Milford, MI, in which $SF_6$ was released from eight pickup trucks and 352 automobiles that were driven on a 10 km, four lane test track for 17 days. Winds were measured using three-dimensional $u$-, $v$-, $w$-wind component Gill anemometers located on five towers at distances ranging from 3.8 to 100 m from the downwind edge of the roadway, and two towers on the dominant upwind side of the road. These towers also sampled $SF_6$ at 0.5 and 3.5 m from ground level. This field study resulted in a comprehensive dataset that has been used by several investigators (Rao, 2002; U.S. EPA, 1979; Chock, 1978) to develop and evaluate models for dispersion of emissions from roadways.

The study described in this paper differs from previous studies in several ways. It is not confined to using observational data to evaluate existing dispersion models, as in Noll et al. (1978). This paper proposes a semi-empirical dispersion model to interpret air quality and meteorological measurements made next to a major highway. The model is simple compared to numerical dispersion models (Rao et al., 2002) to facilitate the identification of controlling variables. At the same time, the model incorporates sufficient physics to represent the first step toward developing a highway dispersion component of a regulatory model such as AERMOD (Cimorelli et al., 2005). The model has been evaluated with unique measurements of NO using long-path optical measurements that result in effective crosswind averaging of plumes originating from motor vehicles on the highway. Such measurements also permit the estimation of on-road emission factors. Because nitric oxide is the dominant component of $NO_x$ emitted from most on-road motor vehicles (Heywood, 1988) and urban background levels are low, NO can be used as a tracer of primary motor vehicle emissions on the short temporal and spatial scales experienced near the road to understand dispersion of motor vehicle emissions.

The next section describes the dispersion study conducted by the U.S. EPA.

## 2. Field study

The field measurements were collected from 27 July to 10 August 2006, adjacent to U.S. Interstate 440 (I-440), in Raleigh, NC. The study was designed to obtain highly time-resolved measurements of traffic activity, meteorology, and air quality at varying distances from the road. Selected air quality parameters represented the complex mixture of pollutants emitted by motor vehicles. In addition to real-time air quality monitoring, selected time-integrated measurements allowed for detailed chemical speciation and the evaluation of particle toxicity (Baldauf et al., 2007).

Fig. 1 shows the project location, adjacent to I-440 in Raleigh, NC. I-440 is a limited-access highway supporting approximately 125,000 vehicles per day. An open field, at-grade with the highway, extends for approximately 120 m to the north of I-440, with only a guardrail and shrubbery approximately 1 m in height and width between the field and I-440 travel lanes. An access road supporting fewer than 200 vehicles per day runs parallel to the highway approximately 10 m from the northern edge of the nearest I-440 travel lane. South of the highway, there is an approximately 5 m drop in elevation at a 45° angle. One- and two-storey office buildings are located at the bottom of the hill; thus, the rooftops of these buildings are essentially at-grade with the highway. With the exception of the highway, no other major air pollution sources were identified within a 5 km radius of the study site.

Historical meteorological monitoring data, collected at Raleigh–Durham International Airport's (RDU) National Weather Service (NWS) station, approximately 13 km from the site, indicated predominant wind directions were from the south and southwest for this area. Based on this data, downwind monitoring sites were established to the north of I-440, as shown in Fig. 1.

Traffic surveillance cameras mounted on a 12 m utility pole located approximately 5 m from the edge of I-440 provided video data of traffic activity on I-440 and the access road. TigerEye$^{TM}$ software (DTS Inc., Albuquerque, NM, USA) remotely calculated vehicle frequency, speed, and class (motorcycles, light-duty cars, light-duty trucks, and heavy-duty truck) as a function of time during daylight hours of the study. Computer hard drives on-site stored all traffic video information to allow for visual confirmation of the software outputs.

Meteorological measurements included wind speed, wind direction, temperature, and humidity. Wind speeds and directions were measured with two cup-and-vane anemometer stations and four sonic anemometers (Model 81000 Ultrasonic Anemometer, R.M. Young Company, Traverse City,

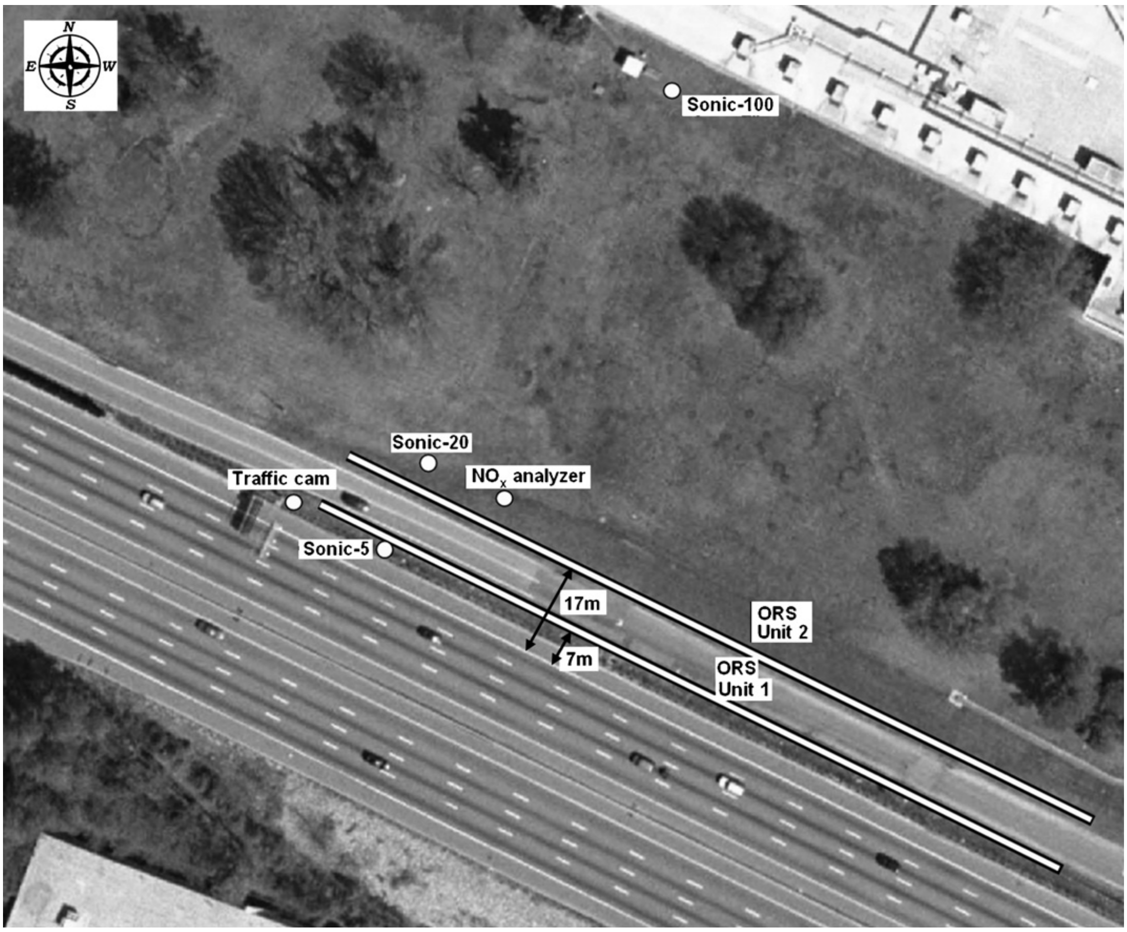


Fig. 1. Map of study location showing the location of the monitoring equipment. The solid lines indicate the ORS path lengths.

MI, USA). Downwind sites at 5, 20, and 100 m contained sonic anemometers. At the 5 m site, one sonic anemometer was located at a height of 2 m above ground. At the 20 m site, two sonic anemometers measured wind speed and direction at heights of 4 and 8 m above ground. Comparison of the data at the 5 and 20 m sites provided information on the horizontal and vertical extent of the turbulent mixing zone from the highway.

Air quality monitors measured pollutant concentrations at multiple distances from the road. Measurements of regulated gases, PM, and air toxics provided information on the concentration of these pollutants during changing traffic and environmental conditions. An on-site master clock provided time synchronized measurements for all of the monitoring equipment. Real-time gas analyzers, meeting criteria in the EPA federal reference method (FRM) or federal equivalent method (FEM), measured regulated pollutant concentrations, including oxides of nitrogen ($NO/NO_2/NO_x$), at 20 and 275 m from the road.

A unique feature of this field study was the application of optical remote sensing (ORS) to measure NO and other pollutant concentrations along multiple paths near the highway. Ground-based ORS instruments utilized infrared, visible and/or ultraviolet light beams projected over open paths to measure spatially averaged gaseous pollutant concentrations in the intersected air column using optical absorption spectroscopy. ORS instruments systems using a novel path-integrated optical measurement technique called Deep UltraViolet Differential Optical Absorption Spectroscopy (DUV-DOAS) were configured in close proximity to I-440. The bistatic DUV-DOASs systems (ORS Units 1 and 2) were set up parallel to the road at distances of 7 and 17 m from the travel lane. The open-path sampling distance was 149 m and the optical paths were 2 m above the ground. The systems were time-synchronized and produced a concentration reading every 5 s (Thoma et al., 2007).

## 3. Model for dispersion of pollutants from highway

The model used to interpret the observations assumes that the highway can be treated as a set of line sources, each of which corresponds to an individual traffic lane. At the receptor distances considered here, we will assume that the line sources are infinitely long. This assumption is not always necessary because there are simple approximations for modeling dispersion from finite line sources (see, e.g. Venkatram and Horst, 2006).

The model used here has been applied by Venkatram (2004) to estimate $PM_{10}$ emission factors for unpaved roads. It is based on the model proposed by van Ulden (1978) and evaluated with observations from the Prairie Grass experiment (Barad, 1958). The concentration associated with a line source with strength $q$, can be written as

$$\frac{C(x,z)}{Q} = \frac{A}{\bar{U}\bar{z}\cos\theta}\exp\left[-\left(\frac{Bz}{\bar{z}}\right)^s\right], \tag{1}$$

where $\theta$ is the angle between the wind and the normal to the line sources, $\bar{z}$ is the mean plume height, and $\bar{U}$ is the wind speed averaged over the plume depth. The value of the shape parameter, $s$, depends on stability. The emission rate $Q$ is given by $T_r e_f$ where $T_r$ (vehicle $s^{-1}$) is the traffic flow rate per lane and $e_f$ is the emission factor (g $m^{-1}$) per vehicle.

If we assume that the horizontal velocity $U(z)$ is described by a power law

$$U(z) = U_r\left(\frac{z}{z_r}\right)^p, \tag{2}$$

where $U_r$ is a reference velocity at height $z_r$, we can show (Venkatram, 2004) that the mean wind speed, $\bar{U}$, is given by

$$\bar{U} = f_u U_r\left(\frac{\bar{z}}{z_r}\right)^p, \qquad A = \frac{sB}{\Gamma(1/s)},$$

$$\text{where } B = \frac{\Gamma(2/s)}{\Gamma(1/s)} \quad \text{and} \quad f_u = \frac{\Gamma((p+1)/s)}{[\Gamma(1/s)B^p]}, \tag{3}$$

where $\Gamma(p)$ is the gamma function given by $\int_0^\infty x^{p-1}\exp(-x)\,dx$.

To account for oblique winds, we use Calder's (1973) approximation and evaluate $\bar{z}$ at a distance, $x/\cos\theta$, from the line source, where $x$ is measured normal to the road. If we assume that the meteorological conditions are close to neutral, the mean plume height can be expressed as

$$\bar{z} = \frac{1}{t}\left[(p+1)\kappa t\frac{\sigma_w}{\alpha U_r}\frac{x}{\cos\theta}z_r^p\right]^{1/(p+1)}, \tag{4}$$

where

$$t = \left[s\left(\frac{\Gamma(2/s)}{\Gamma(1/s)}\right)^s\right]^{1/(1-s)}, \tag{5}$$

where $\alpha = 1.25$. We account for the height of release by using a virtual distance $x_0$ obtained by equating the release height, $h_0$, to $\bar{z}$ at $x_0$. This distance is then added to all downwind distances.

Combining Eqs. (3) and (4) yields the following expression for the denominator of Eq. (1):

$$\bar{U}\bar{z}\cos\theta = a\sigma_w x + b,$$

$$\text{where} \quad a = \frac{kf_u(p+1)}{\alpha t^p} \quad \text{and}$$

$$b = f_u U_r \left(\frac{h_0}{z_r}\right)^p h_0 \cos\theta. \tag{6}$$

This result suggests that the concentration is relatively insensitive to the mean wind speed and wind direction except through the second term, $b$, associated with the release height, $h_0$.

In using Eq. (1) to interpret the concentration data, we will treat the emission factor, $e_f$, as a parameter whose value is determined by fitting model estimates of NO to corresponding observations. Before applying the model to interpret observations of NO, we examine the role of meteorological variables in the variation of near road concentrations.

## 4. Variations of meteorological variables and concentrations

The following analysis uses measurements from two sonic anemometers, whose locations relative to the highway are shown in Fig. 1: Sonic-5 (anemometer 1), located 5 m from the edge of the road at a height of 2 m, and Sonic-20 (anemometer 2), located 20 m from the road edge, at a height of 8 m. It is reasonable to assume that meteorological measurements at Sonic-5 are affected by turbulence generated by traffic activity. On the other hand, traffic generated turbulence is less likely to influence measurements at Sonic-20. Comparing the relative performance of the model using the two sets of data as model inputs provides information on the importance of including traffic-induced turbulence in modeling dispersion.

Fig. 2 shows the relationships between the variables measured by the two sonics. The measurements correspond to 10 min averages. The top left panel shows that the wind speeds at Sonic-20 are generally higher than those measured by Sonic-5. This is to be expected in view of the height difference. However, the correlation between the wind speeds is poor, suggesting the possible influence of traffic-induced turbulence on Sonic-5. The correlation between the vertical velocity fluctuations is better, with $\sigma_w$ at Sonic-5 being about 0.6 of that at Sonic-20. The horizontal velocity fluctuations, $\sigma_v$, measured by the two sonics are similar in magnitude and are correlated; values at Sonic-5 tend to be slightly higher. The wind directions at the two sonics are also highly correlated, although there are substantial differences during any 10 min period.

Fig. 3 provides more insight into the relationship between the two variables that govern dispersion, $\sigma_w$ and $U$. The plots show averages over all values corresponding to a particular hour. The left panels of the figure show that the winds are relatively light: $<1.5\,\mathrm{m\,s^{-1}}$ at Sonic-5 and below $2\,\mathrm{m\,s^{-1}}$ at Sonic-20. Minimum winds are less than or close to $0.5\,\mathrm{m\,s^{-1}}$ at both sites. The vertical turbulent velocities reach their maxima of about $0.5\,\mathrm{m\,s^{-1}}$ at 15:00 h, when the wind speeds also attain their maximum values at the two sonics.

The right panels of the figure indicate that $\sigma_w$ is correlated with $U$ when the wind speeds exceed $1\,\mathrm{m\,s^{-1}}$. Below this threshold, $\sigma_w$ varies from 0.1 to $0.2\,\mathrm{m\,s^{-1}}$. The turbulent intensities ($\sigma_w/U$) vary from 0.2 to 0.3; the turbulent intensities are higher at Sonic-5, suggesting the possible influence of vehicle-induced turbulence at 5 m from the road.

The relationships between the governing meteorological variables measured by Sonic-5 and concentrations observed at Unit 1 are shown in Fig. 4. The NO concentration decreases with wind speed, but the scatter is large. The concentration increases with $1/\sigma_w$ and traffic flow rate. The NO concentration is correlated best with the combination (traffic flow rate/$\sigma_w$), a result that is consistent with the model, as we will see later. The next section provides results on using the dispersion model to explain the variation of observed NO concentrations.

## 5. Application of dispersion model

The model is applied by assuming that the emission factor $e_f$ is unknown, and the predicted concentration is written as

$$C_p(x) = e_f D(\beta), \tag{7}$$

where $D(\beta)$ is a function of $\beta$ which represents the known inputs in Eq. (1): the source–receptor geometry and the meteorological inputs. The emission factor, $e_f$, expressed as grams of NO per kilometer traveled by a typical vehicle on the road, represents an average over the vehicles traveling on the road over the time period of the experiment.

The unknown emission factor is determined by assuming that the observed NO concentrations are log-normally distributed about the model estimate

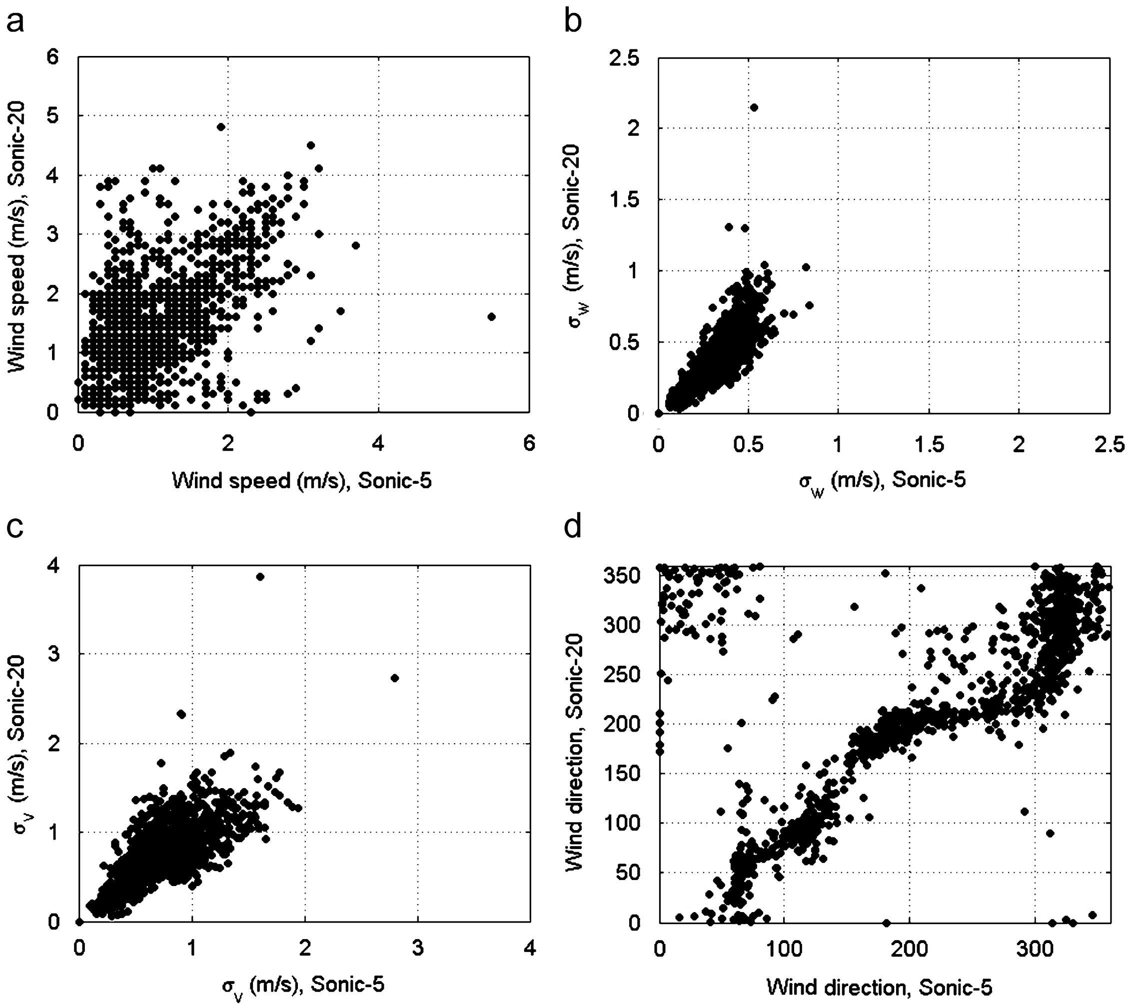


Fig. 2. Relationships between meteorological variables measured by sonic anemometers.

so that

$$\ln C_0 = \ln e_f + \ln D + \varepsilon, \tag{8}$$

where $\varepsilon$ is a random variable with zero mean. Then, an estimate of $e_f$ is given by

$$e_f = \exp\left(\text{mean}\left(\ln\left(\frac{C_0}{D}\right)\right)\right). \tag{9}$$

The configuration of the road relative to two receptors is an important model input. The road has eight lanes, the total width of the paved section is 30 m, the median is 6 m, and each shoulder is 4 m wide. The road is represented by eight line sources, each of which is placed in the middle of the individual lane. ORS Unit 1 is 7.6 m from the edge of the paved road and Unit 2 is 10 m from Unit 1.

Fig. 5 shows the results obtained by applying Eqs. (1)–(4) to interpret the NO concentrations observed at Units 1 and 2. The exponent $s$ in Eq. (1) was taken to 2, based on results obtained by Britter and Hanna (2003). The wind profile exponent, $p$, was estimated by assuming that the shear stress is constant in the surface layer so that

$$K(z)\frac{\mathrm{d}U}{\mathrm{d}z} = u_*^2. \tag{10}$$

Substituting Eq. (2) for the wind profile, and using the neutral expression for $K(z)$ in Eq. (10), we find

$$p = \frac{u_*}{kU_r} \approx \frac{\sigma_w}{\alpha k U_r}, \quad \text{at } z = z_r. \tag{11}$$

This estimate of $p$ is tentative because it is not consistent with the assumption that it is constant with height. The sensitivity of model results to this parameter will be examined later. Most of the results presented here include only wind angles that are within $\pm 45^\circ$ from the normal to the road to ensure the applicability of Eq. (1), which assumes an infinitely long road. However, we do examine the

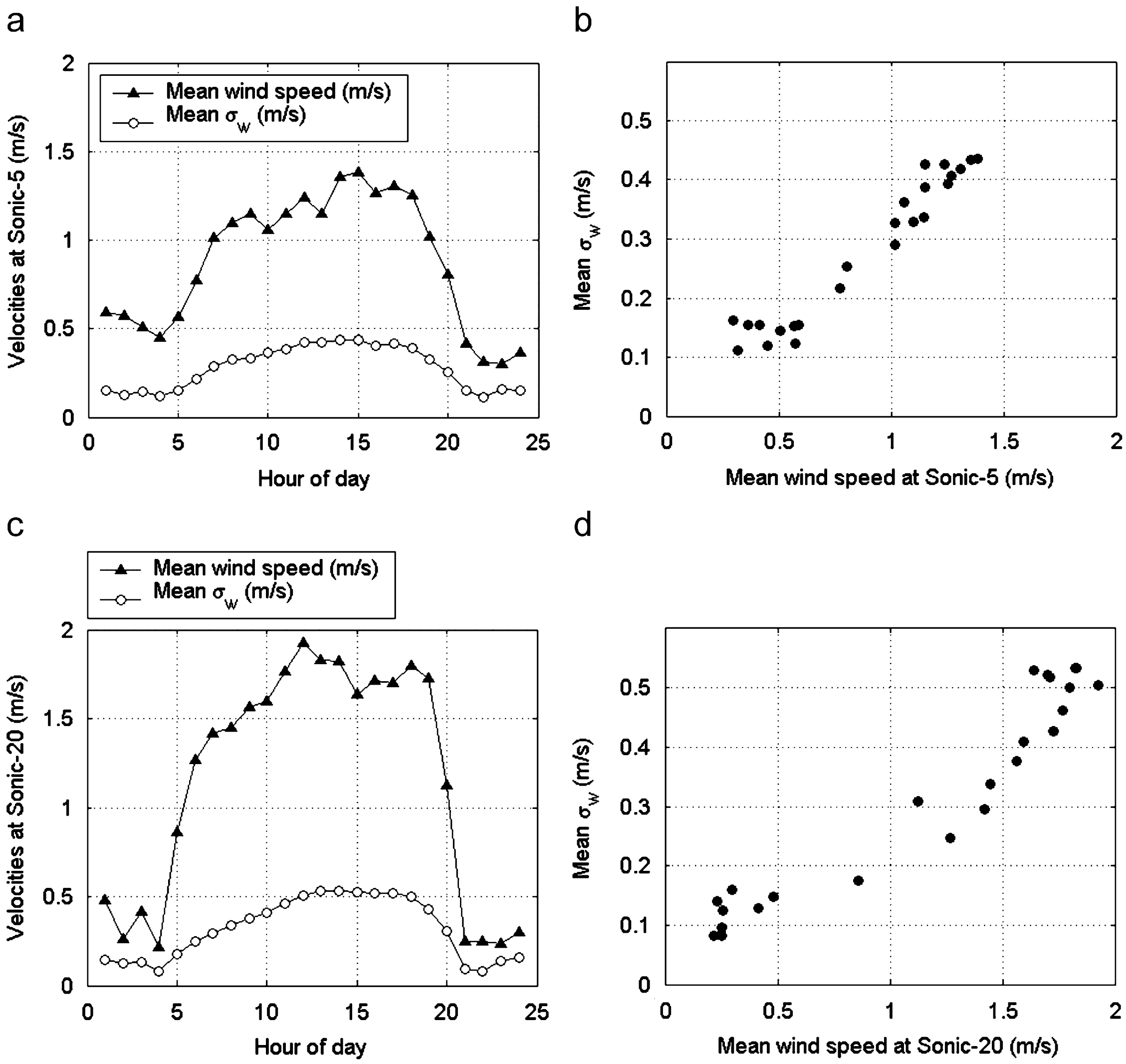


Fig. 3. Relationship between wind speed and vertical velocity fluctuations.

performance of the model at wind directions that are almost parallel to the road.

The correlation coefficient, $r^2$, refers to the fraction of the variance of the observed concentrations—actually log(observed) to be consistent with Eq. (7)—that is explained by the model. The fraction of observed concentrations within a factor of two of the model estimates is called 'fac2', and is expressed as a percentage.

The top panels of the figure refer to meteorological inputs from two sonics—Sonic-5 and Sonic-20, respectively. We see that measurements from Sonic-20 located 20 m from the road at 8 m above ground lead to similar performance measures between observed and estimated NO concentrations as that obtained with measurements from Sonic-5 located 5 m from the highway at a height of 2 m from the ground. The NO emission factor averaged over the period of the field study and over the vehicles on the road during that time varied between 0.51 and 0.57 g km$^{-1}$, which convert to 0.78–0.87 g km$^{-1}$ expressed as emissions of $NO_2$. This range of emission factors is consistent with tunnel derived measurements summarized in McGaughey et al. (2004): the values range from 0.46 to 1.19 g km$^{-1}$.

The bottom panels indicate that the estimated relationship between the NO concentrations at Units 1 and 2 is close to the observed relationship, although the model predicts a higher range of concentrations. Note that the high degree of correlation between the observed NO concentrations reflects the fact that averaging the concentration over the 149 m optical path reduces the variance associated with point measurements. This reduction of variance also contributes to the relatively high correlation coefficient between model estimates and observations.

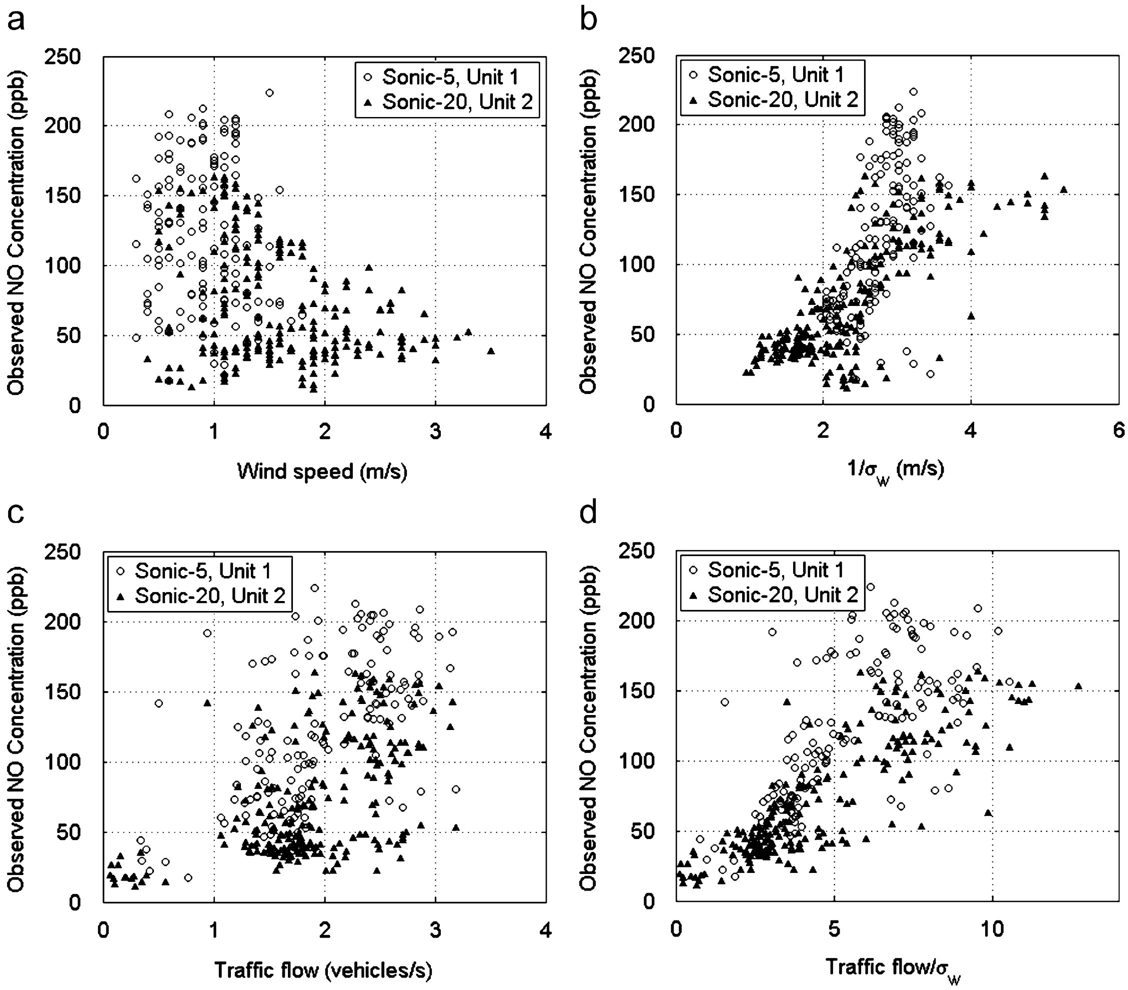


Fig. 4. Relationship between NO concentrations measured by Unit 1 and meteorological variables measured by Sonic-5.

The results presented in Fig. 5 suggest that the dispersion model captures the essential processes that govern transport of emissions from the roadway to the near road receptors. This is seen in the residual plot of Fig. 6. If the model inputs explain most of the variance of the observed concentrations, the residual, which is the ratio of the observed to the estimated concentration, should be uncorrelated with the model inputs. A visual examination of Fig. 6 indicates that the residuals are uncorrelated with the model inputs, a result which supports the formulation of the dispersion model. The next section discusses the role of different inputs on model performance.

## 6. Role of model inputs

Fig. 7 illustrates the role of wind direction in determining model performance. When the wind angle is allowed to vary between $+85^\circ$ and $-85^\circ$, model performance deteriorates compared to that for the smaller wind angles considered in Fig. 5. However, model performance is adequate even when the wind direction is almost parallel to the road. The model tends to overestimate concentrations by about 20% compared to the earlier result, which reduces the emission factor by about 20%. The deterioration in model performance occurs at wind angles over $60^\circ$; the model overestimates concentrations in this range suggesting the need to account for edge effects associated with large wind angles relative to the road (Venkatram and Horst, 2006).

Fig. 8a and b shows the effect of doubling the initial release height, $h_0$, associated with traffic-induced vertical dispersion, from 1.5 to 3 m. Model performance is essentially unchanged from the base case, except that the emission factor increases by about 15%, which is small compared to the 100% change in $h_0$. The role of a particular meteorological

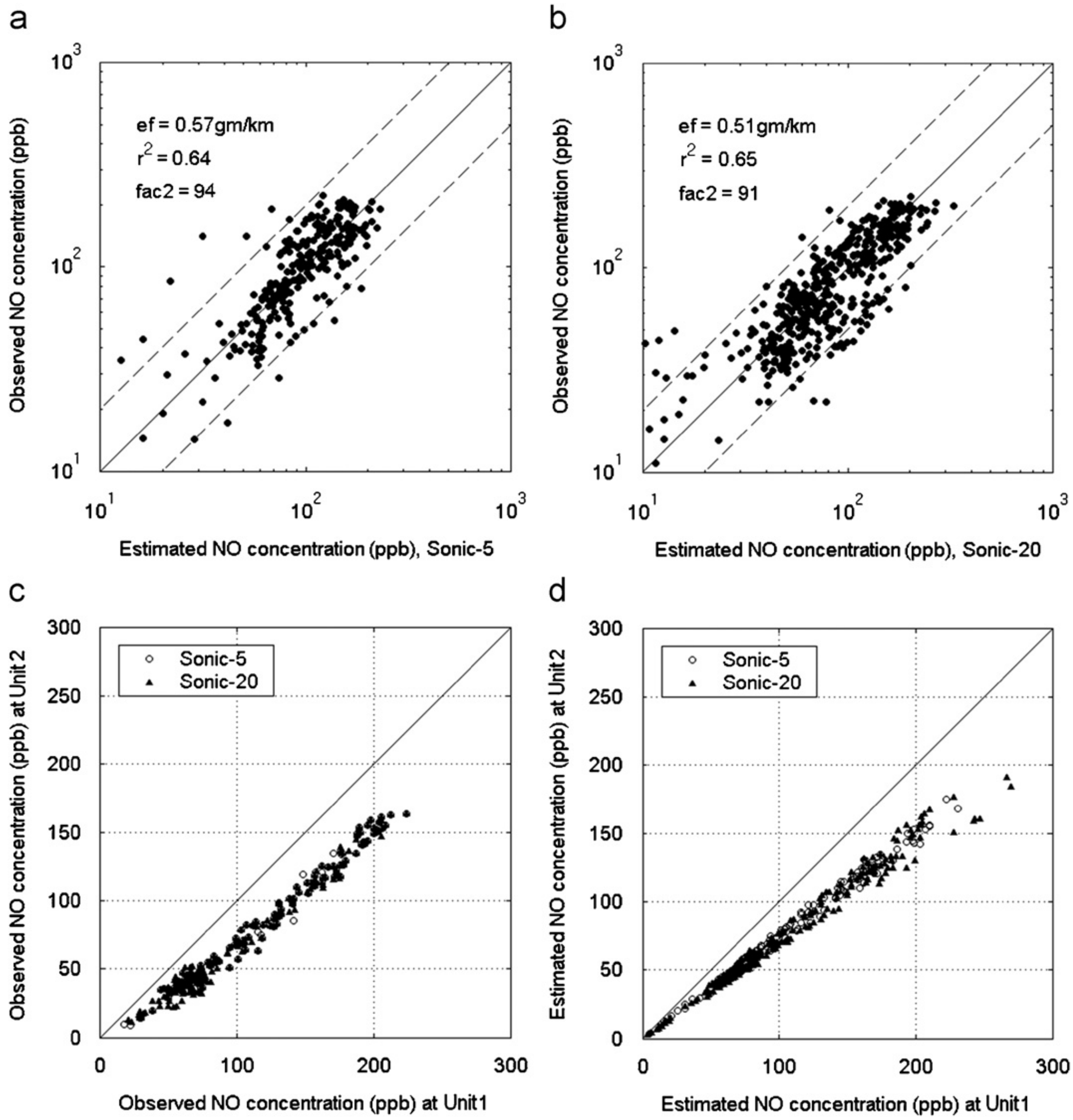


Fig. 5. Comparison of estimated and observed NO concentrations at Units 1 and 2. Wind angles restricted between $-45°$ and $+45°$ of the normal to the road. Initial dispersion height $h_0 = 1.5$ m.

variable in determining model performance is investigated by keeping the variable constant and allowing the others to vary. Fig. 8c and d shows model performance when wind speed is kept constant at the mean of the observed values. Model performance changes little from the base case, confirming the insensitivity of modeled concentrations to mean wind speed discussed earlier through Eq. (6). On the other hand, the standard deviation of vertical velocity fluctuations, $\sigma_w$, plays a more significant role as seen in Fig. 8e and f. Model performance deteriorates markedly especially when measurements from Sonic-20 are used. The maximum $r^2$ goes down from 0.62 to 0.4 and fac2 decreasing from 94% to 83%. However, the estimated emission factors are similar to those from the base case. Fig. 9a and b shows that the effect of using a constant traffic flow rate: the correlations go down significantly and the upper and the lower limits of the observed concentration range are not predicted. Fig. 9c and d shows results when the wind direction is assumed to be always normal to the road. We see that this assumption makes little difference to model performance, suggesting the minor role of wind direction in determining concentrations when the wind angle is between $+45°$ and $-45°$ to the normal to the road. Fig. 9e and f shows the sensitivity of model results to the

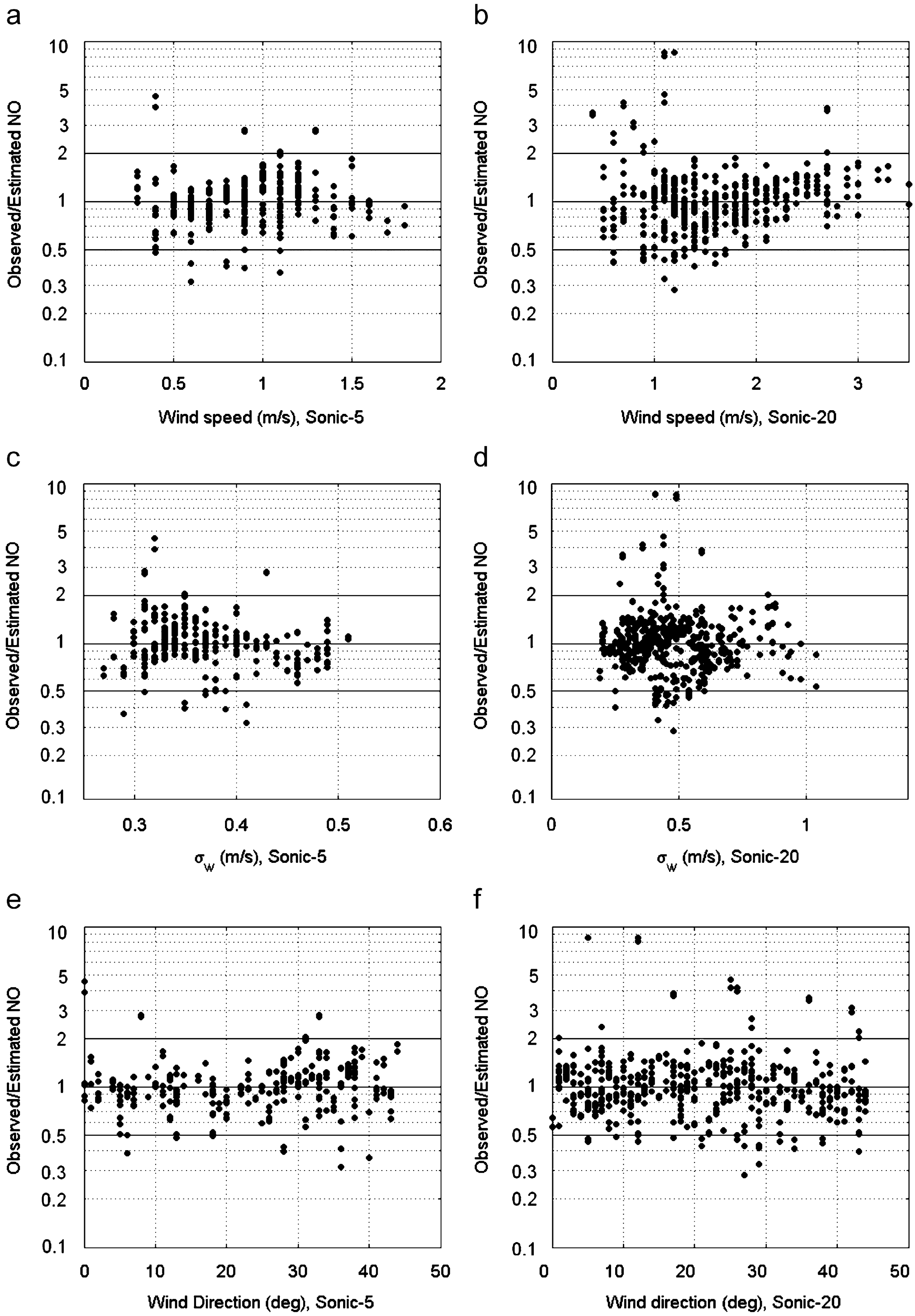


Fig. 6. Variation of the ratio of the observed to model estimated NO concentrations with model inputs. Wind angles restricted between −45° and +45° of the normal to the road. The horizontal lines represent factor of two interval.

parameter, $p$, used to characterize the wind profile. Setting $p$ to zero does lead to reduction of $r^2$ but the estimated emission factors are similar to the base case.

It is clear that the dispersion model presented here provides an adequate description of the observed concentrations. However, its computational

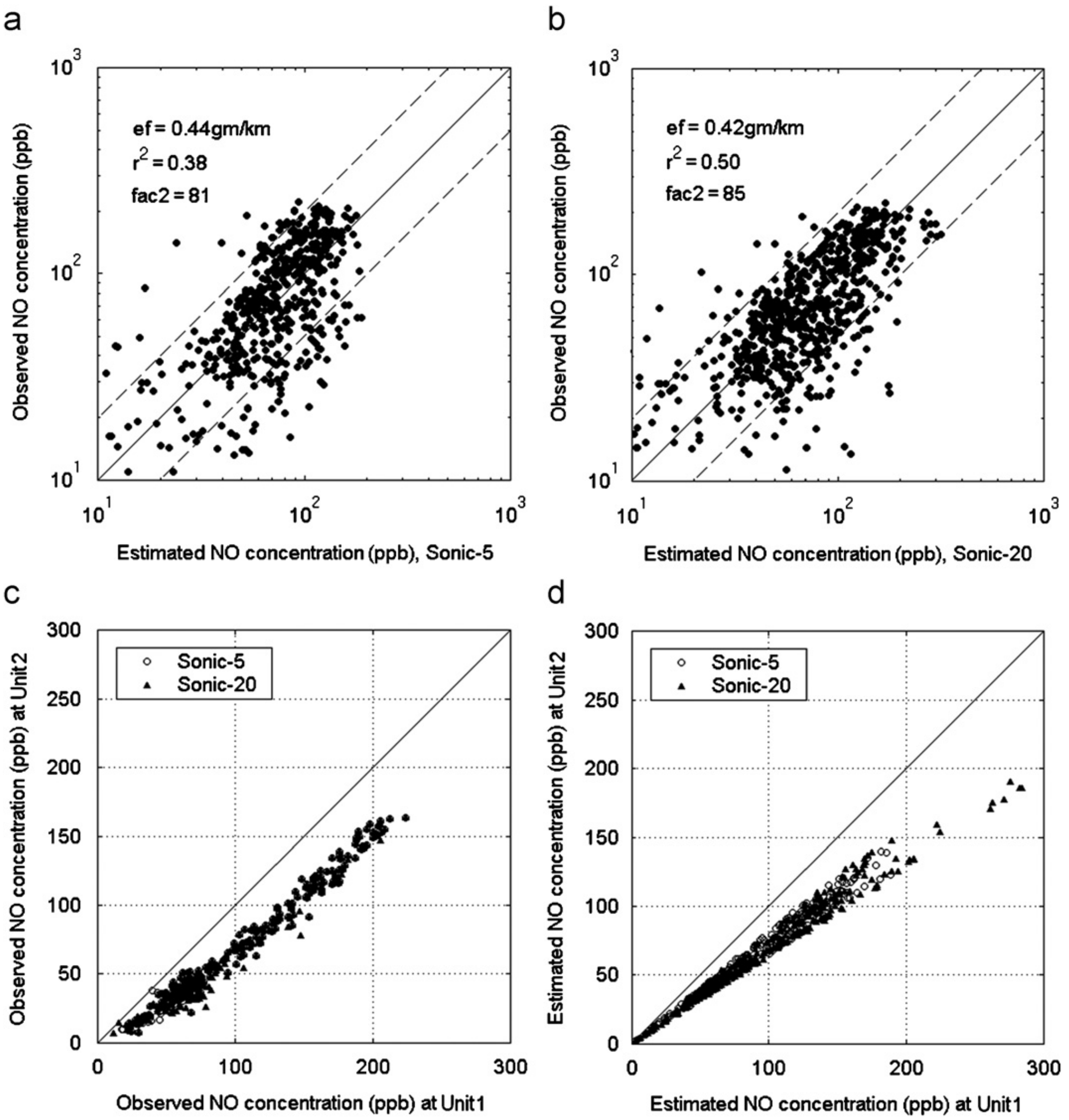


Fig. 7. Comparison of estimated and observed NO concentrations at Units 1 and 2. Wind angles restricted between $-85°$ and $+85°$ of the normal to the road.

requirements might become burdensome in applications that require accounting for a large number of roads. It turns out that a much simpler model can be constructed by adapting an urban dispersion model proposed by Venkatram and Cimorelli (2007).

## 7. A simple model

The simple model assumes that the highway is an area source with a width, $W$, measured along the wind direction blowing normal to the highway, and has infinite length along the highway. Then, the contribution of emissions from a strip of the highway, $\mathrm{d}x$, to the concentration at a receptor at a distance $x$ from the strip is given by

$$\mathrm{d}C = q\,\mathrm{d}x\frac{A}{\bar{U}\bar{z}\cos\theta}, \tag{12}$$

where $q$ is the emission density at a distance $x$, $\bar{U}$ is the mean wind speed, and $\bar{z}$ is the mean plume height defined earlier, and the $\theta$ is the angle between the wind vector and the normal to the road.

As a first approximation, assume that the emission density is uniform across the highway. Then, $q(\mathrm{g}\,(\mathrm{m}^2\,\mathrm{s})^{-1})$ is given by

$$q = \frac{T_\mathrm{r}e_\mathrm{f}}{W}, \tag{13}$$

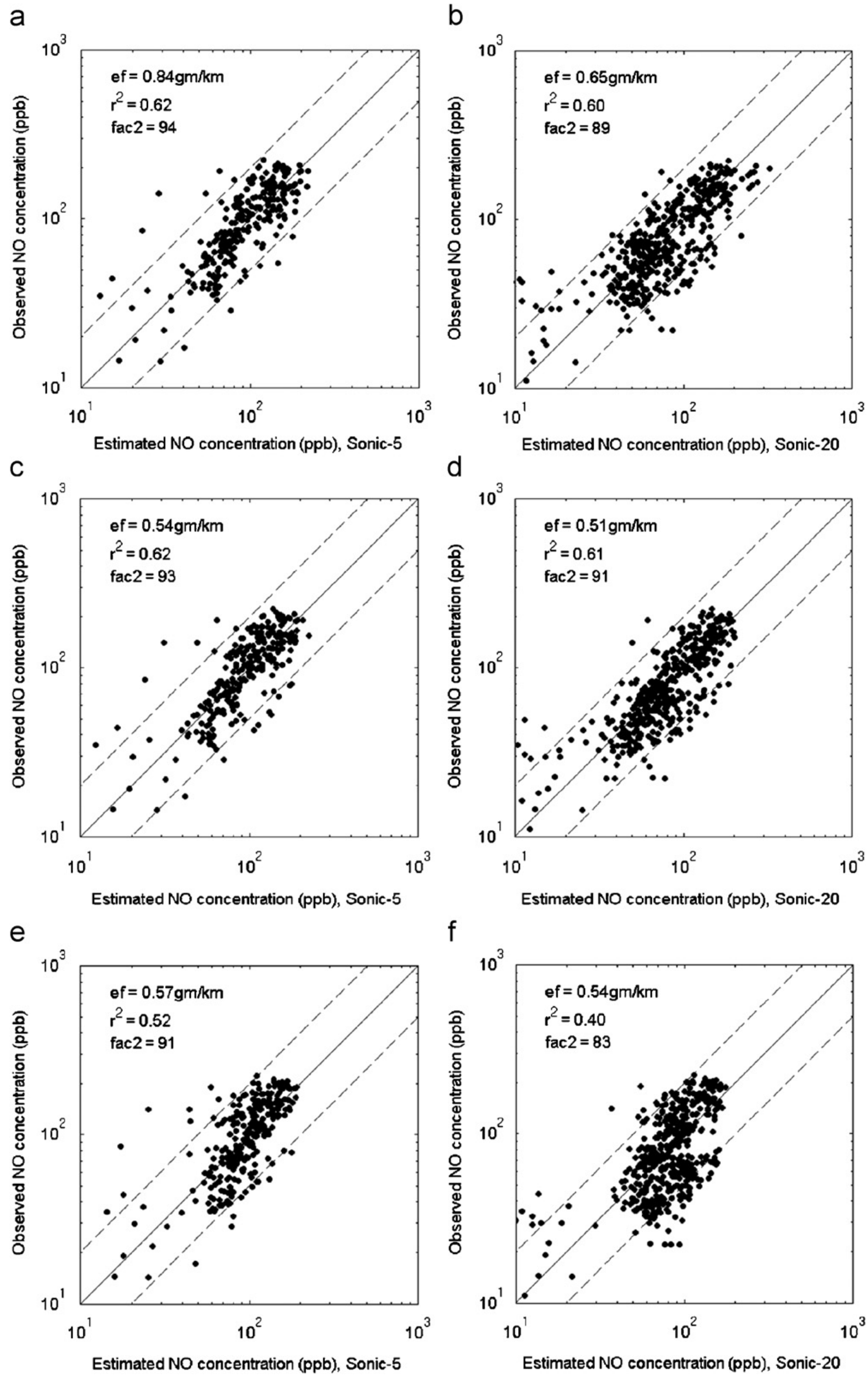


Fig. 8. Same as Fig. 5(a,b) except that: dispersion height $h_0 = 3$ m (a, b); wind speed is constant at mean of observations (c, d); and except that $\sigma_w$ is constant at mean of observations (e, f).

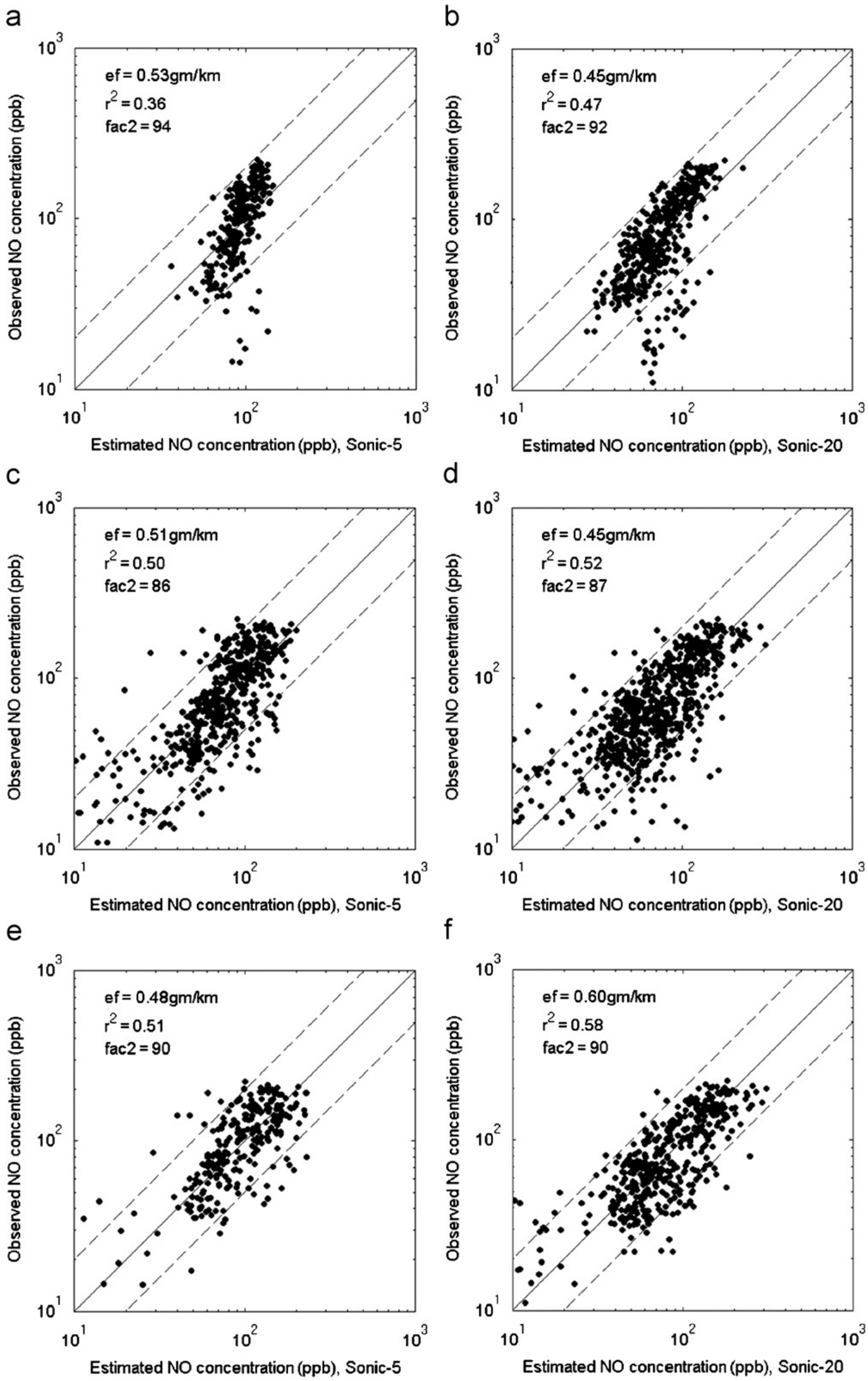


Fig. 9. Same as Fig. 5(a, b) except that: flow rate is constant at mean of observations (a, b); wind angle is taken to be zero (c, d); and except that $p = 0$ (e, f).

where $T_r$ is the traffic flow rate (vehicles $s^{-1}$), and $e_f$ is the emission factor (g (m vehicle)$^{-1}$). Note that the ratio, $T_r/W$, is a measure of the traffic flow rate per unit width of the highway.

If we integrate Eq. (12) from $x$ at the downwind edge of the highway to $x+W$ at the upwind edge of the highway, we get the concentration at the receptor at a distance, $x$, from the edge of the highway,

$$C = q\int_{x}^{x+W} \frac{A}{\bar{z}\bar{U}\cos\theta}\,\mathrm{d}x. \tag{14}$$

Substituting Eq. (6) in Eq. (14) and integrating results in the following expression for the concentration:

$$C(x) = \frac{AT_r e_f}{aW\sigma_w}\ln\left(\frac{b + a\sigma_w(x+W)}{b + a\sigma_w x}\right). \tag{15}$$

The concentration $C(x)$ is inversely proportional to standard deviation of the vertical fluctuations, $\sigma_w$, and not the wind speed $U_r$.

Note that the wind speed affects the concentration indirectly through the wind profile parameter, $p$, embedded in $a$ (see Eq. (6)), but as seen earlier, the sensitivity to this parameter is small compared to that for $\sigma_w$. The wind speed plays a more direct role at distances much smaller than the width of the road. At small $x \ll W$, the concentration becomes

$$C(x) = \frac{AT_r e_f}{aW\sigma_w}\ln\left(1 + \frac{a\sigma_w W}{b}\right), \tag{16}$$

and when the width of the road, $W$, is small enough for $(a\sigma_w W/b) \ll 1$, the concentration is governed by the initial vertical plume spread, $h_0$,

$$C(x) \sim \frac{T_r e_f}{U(h_0)h_0\cos\theta}. \tag{17}$$

The performance of the model described by Eq. (15) is shown in Fig. 10. We see the results are similar to that of the model based on Eq. (1), although the complete model performs slightly better. Sensitivity studies indicate that wind direction has little effect on concentrations when the wind angle is between $+45^\circ$ and $-45^\circ$ to the normal to the road. These results suggest that Eq. (15) is a useful alternative to Eq. (1) when computational constraints become important.

## 8. Summary

We have used a dispersion model to analyze measurements made during a field study conducted by the U.S. EPA in July and August 2006, to estimate the impact of traffic emissions on air quality at distances of tens of meters from an eight-lane highway located in Raleigh, NC. The air quality measurements consisted of long-path optical measurements of NO at distances at 7 and 17 m from the edge of the highway. Sonic anemometers were used to measure wind speed and turbulent velocities at 5 and 20 m from the highway. Traffic flow rates were monitored using traffic surveillance cameras.

The dispersion model used to analyze these measurements was presented in an earlier study (Venkatram, 2004) to estimate $PM_{10}$ emission factors from paved roads. In this study, the model explained over 60% of the variance of the observed path-averaged NO concentrations, and over 90% of the observed concentrations were within a factor of two of the model estimates.

Sensitivity tests conducted with the model allowed us to identify the observed variables that govern dispersion of emissions from the highway to receptors within 20 m. We found that

1. NO concentrations near the road were governed by the emission rate, as represented by the traffic flow rate, and the standard deviation of the vertical velocity fluctuations, $\sigma_w$.
2. The concentrations were relatively insensitive to the mean wind speed except at distances from the roads that are comparable to the width of the road.
3. As long as the wind direction was within $\pm 45^\circ$ from the normal to the road, the wind direction had little effect on near road concentrations.

The emission factor for NO, estimated from the model was about 0.50 g km$^{-1}$ (0.77 g km$^{-1}$ as $NO_2$), a value which compares reasonably with tunnel derived measurements reported by McGaughey et al. (2004). The performance of the dispersion model in explaining NO observations suggests that the analysis of air quality measurements near a road using a model can be the basis of a technique to estimate on-road emission factors of other species emitted by vehicles. However, the technique requires additional evaluation before it can be recommended for such applications.

The dispersion model for road emissions proposed here can be readily incorporated into the current generation of dispersion models typified by AERMOD (Cimorelli et al., 2005). Unlike CALINE

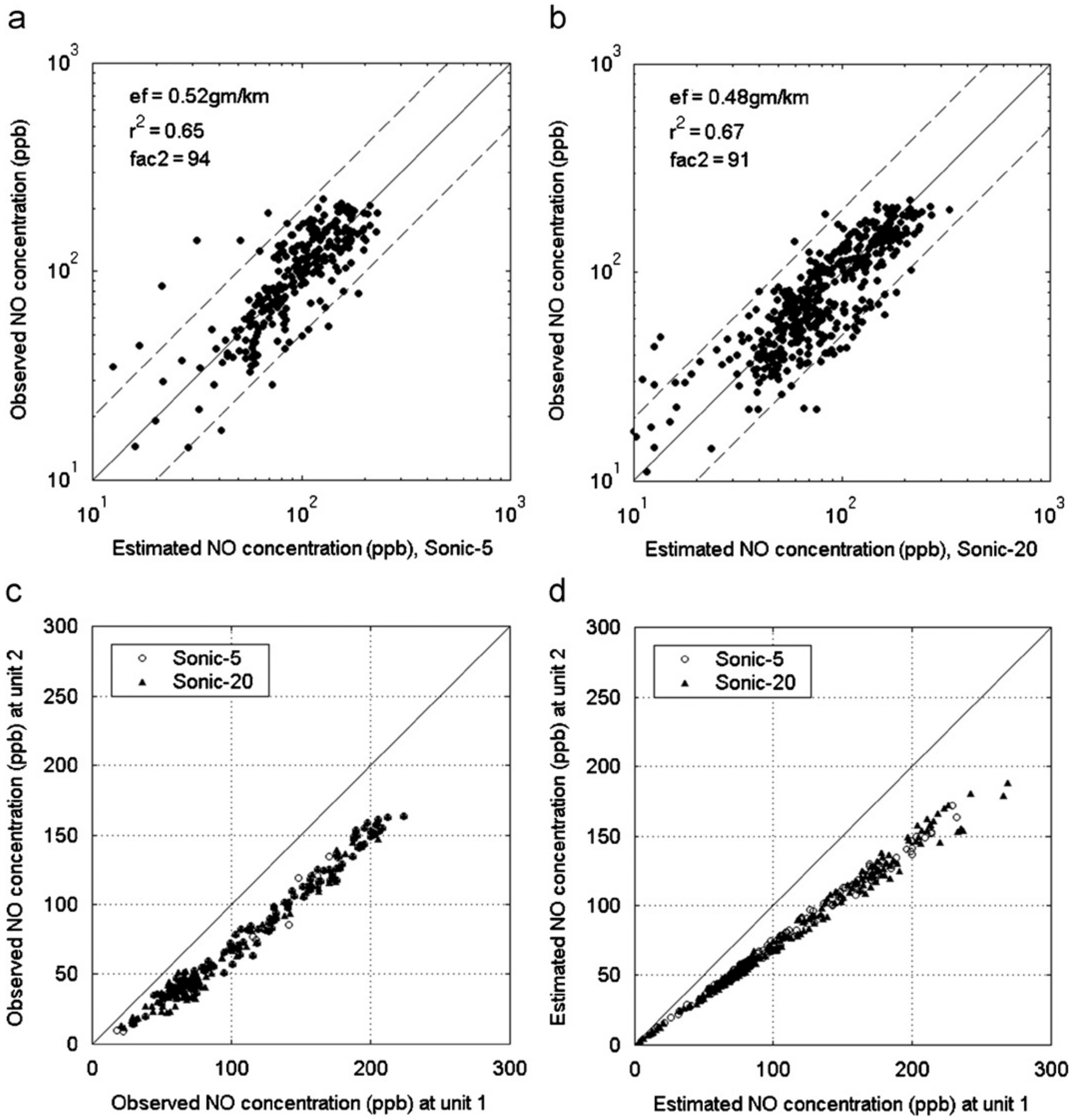


Fig. 10. Comparison of estimated and observed NO concentrations at Units 1 and 2. Wind angles restricted between −45° and +45° of the normal to the road. Initial dispersion height $h_0 = 1.5$ m and $W = 36$ m. Unit 1 is at 7 m and Unit 2 is at 17 m from paved edge of highway.

(Benson, 1992), which uses stability based Pasquill–Gifford dispersion curves, this model requires micrometeorological inputs compatible with those of AERMOD. The most important meteorological input is the standard deviation of the vertical velocity fluctuations. In principle, it can be estimated from the type of measurements customarily required by current models. Although this study does not draw conclusions on the impact of traffic-induced turbulence, it might be necessary to account for this effect, using methods such as those proposed by Kalthoff et al. (2005) and Bäumer et al. (2005).

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## Disclaimer

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