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THE IMPACT OF ROAD STRUCTURES AND BUILDINGS ON URBAN AIR QUALITY

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Abstract: Urban air pollution is dominated by emissions from vehicles traveling on highways and streets between buildings. The highways can be at the same level, elevated, or depressed relative to the surrounding area, and often have sound barriers located next to them. Urban buildings also have a major effect on dispersion of emissions from vehicles. This paper describes models that extend those developed earlier to account for the effects of urban structures on near road concentrations. Here we address dispersion models that account for road depression and vegetative barriers. We also describe an internal boundary layer model that provides estimates of micrometeorological inputs to apply these models in urban areas.

Key words: *Urban dispersion, depressed roads, building effects, vegetative barriers*.

INTRODUCTION

Urban air pollution is dominated by emissions from vehicles traveling on highways and streets between buildings. The highways can be at the same level, elevated, or depressed relative to the surrounding area. They can also have sound barriers located next to them. These configurations affect dispersion of vehicle emissions, and thus have an impact on near road air quality. Urban buildings also have a major effect on dispersion of emissions from vehicles. Dispersion models to account for some of these urban structures have been described in earlier publications (Amini et al., 2016; Ahangar et al., 2017; Schulte et al., 2015; Venkatram et al., 2016). Here, we extend this suite of models by accounting for additional features of urban areas. The first model estimates concentrations associated with emissions from urban highways that are depressed relative to surrounding terrain. The second model accounts for the effect of adding vegetation to solid noise barriers. The third model facilitates the application of a model developed earlier to estimate the impact of buildings on street-level concentrations (Schulte et al., 2015): an internal boundary layer (IBL) model that estimates roof-level turbulence using meteorological data from a surface based instrument located at an upwind location, such as an airport. The results from the models have been evaluated with data from tracer, field, and wind tunnel studies.

DEPRESSED ROAD MODEL

There are relatively few studies on the effects of road configurations on near-road dispersion; configurations refer to roads that are elevated or depressed relative to the surrounding terrain or have other structures complicating the flow in the vicinity of the roadway. Heist et al. (2009) performed an experimental study in the U.S. EPA's Meteorological Wind Tunnel (Snyder, 1979) to explore the effects of different road configurations on the dispersion of traffic-related pollutants downwind of roads. The modelled freeway is a six lane divided highway at 1:150 scale. The configurations that were studied in this paper are: a 6 m deep depressed roadway with vertical sidewalls (D690), a 6 m deep depressed roadway with 30° angled sidewalls (D630), and a 9 m deep depressed roadway with vertical sidewalls (D990). The letter 'D' in a case name stands for 'Depressed', the first single digit number represents the depth of the road in meters, and the two-digit number at the end of a case name denotes the angle between the sidewalls and the roadbed, in degrees. Laser Doppler Velocimetry (LDV) was used for all velocity measurements in this study, details of which are described in Heist et al. (2009). The tracer gas used in the study was high-purity ethane (C_2H_6) which is only slightly heavier than air.

The impact of road configurations on the flow field is shown in Figure 1. Roadways with vertical sidewalls create recirculating flow in the depressed regions. This recirculation disappears in the depressed region with the angled side walls. All the configurations decrease surface concentrations relative to those of the flat terrain case.

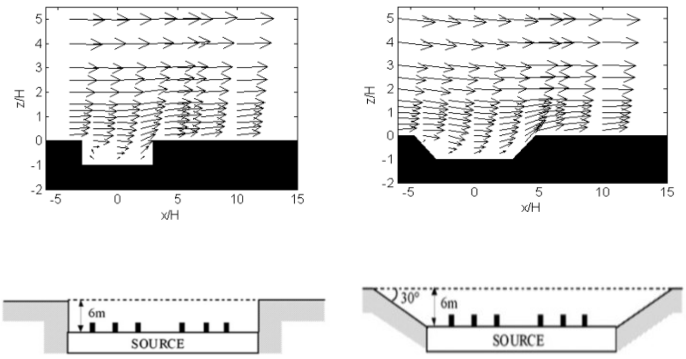

Figure 1. Two configurations studied in the wind tunnel. Left panels shows the flow field and source configuration for the 6 m depressed road. The right panels show the effects of angled side walls.

Our model is based on a modification of a model proposed by van Ulden (1978) which has been evaluated with observations from the Prairie Grass experiment (Barad, 1958). This model, which is the analytical solution of the eddy diffusivity-based mass conservation equation, expresses the concentration C associated with an infinitely long line source with strength q, as

$$\frac{C(x,z)}{q} = \frac{A}{\overline{U}\overline{z}} \exp\left(-\left(\frac{Bz}{\overline{z}} \right)^{s} \right) \tag{1}$$

where x and z are the distance from the source and height above ground level, $\overline{U}$ is the mean horizontal wind speed, and the value $s = 1.3$ provides an excellent description of the vertical concentration profiles observed at various downwind distances in the wind tunnel. For a neutral boundary layer, we can express the mean height of the plume $\overline{z}$ as (Venkatram, 2004)

$$\overline{z} = \left(a \frac{\beta u_*}{U_r} z_r^p x + b h_0^{p+1} \right)^{\frac{1}{p+1}}, \tag{2}$$

where a and b are constants, and β is a factor to account for the effect of the depression on turbulence. U_r is the velocity measured at z_r, a reference height, u_* is the surface friction velocity, p is the exponent of the power law fitted to the upwind wind profile, and h_0 is the initial plume spread induced by the depression.

We model the concentrations downwind of the freeway as the sum of concentrations due to six individual line sources. We account for the effects of the depressed road through two parameters: a factor β which multiplies the flat terrain friction velocity in equation 2 and an initial mixing height h_0. The values of these parameters are obtained by fitting results from the modified van Ulden model to the measurements made in the wind tunnel. The values of these parameters, listed in Table 1, suggest that it is necessary to use an initial mixing height of 1.2 m to describe the concentrations for the flat terrain case. One effect of the road depression is to increase the initial mixing height. A 6 m depressed roadway with straight edges adds 3.7 m to the initial mixing height of flat terrain, while a 9 m depressed roadway with straight edges adds 4.8 m. The D630 case adds 2.4 m to the flat terrain initial mixing height, indicating the smaller role of turbulent mixing in the presence of sloping walls.

The second effect of the road depression is an increase of β, which is interpreted as an increase in the rate of vertical plume spread. The D690, D630, and D990 cases result in increases of 12%, 37%, and 31% in this rate compared to those of flat terrain, respectively. One reason behind the highest rate for the D630 case could be the vertical velocities induced by the upward slope of the downwind edge of the depression. Figure 1 appears to support this hypothesis.

Table 1. Values of empirical parameters of different cases that form the σ_z expression ($s = 1.3$ and $p = 1/7$)

Case	$h_0(m)$	β
FLAT	1.2	1.00
D690	4.8	1.12
D630	3.6	1.37
D990	5.9	1.31

VEGETATIVE BARRIER MODEL

Vegetative barriers, which have been suggested as a potential method to decrease air pollution near roadways, affect downwind air quality through dry deposition and by altering the flow field. For most pollutants, including small particles, dry deposition plays a minor role compared to flow field effects in altering concentrations. It is useful to think of the flow as consisting of two components: one that is forced over the barrier, and the rest going through the barrier. The plume embedded in the over-barrier flow undergoes enhanced dispersion that results in lower concentrations relative to those without the barrier. The turbulence in the flow through the barrier is reduced relative to that upwind of the barrier, which results in higher concentrations relative to the those in the absence of the barrier.

A field study was conducted in Sacramento, California to understand the effects of tall vegetation behind a barrier on concentration levels in the vicinity of highway CA-99. The field study was conducted at two sites: a 5 m barrier extending over 500 m on the east side of the highway, and a barrier of the same height with a row of 15-18 m high pine trees planted next to it extending over 200 m along the highway. Ultrafine particle concentrations were measured with TSI Condensation Particle Counters (CPCs) at three locations, one upwind and two at a distance of 4 m downwind of each of these barriers. Meteorological variables were measured with Campbell Scientific CSAT3 3-D sonic anemometers at these locations, at a height of 2.5 m. The traffic flow in each lane of the freeway was obtained from the CalTrans Performance Measurements System. The measurements were conducted on 21st, 22nd, 25th, 26th, 27th, 28th, and 30th of June 2016 during 12:00–18:00 hours, during which time the wind blew primarily from the southwest.

The field data showed that the turbulence levels downwind of the vegetation-wall barrier were consistently lower than than those behind the barrier, indicating the role of vegetation in reducing turbulence. The concentrations behind the vegetation-wall barrier were lower than those downwind of the plain barrier 80% of the time suggesting the dominance of the blocking effect of the vegetation. However, the concentrations were higher 20% of the time, which indicated that greater impact of turbulence reduction on concentrations. We anlayzed the data using a dispersion model that was developed earlier (Venkatram et al., 2016) to account for the impact of a solid barrier on downwind concentrations. The model uses an entrainment factor, f_m, that is used to reduce the entrainment into the barrier wake during unstable conditions to ensure that the model describes the data from the Idaho Falls experiment (Finn et al., 2010). As the first step in modeling the complex effects of vegetation, we modified the solid barrier model to account for the effects of vegetation through: 1) the friction velocity was multiplied by the ratio of standard deviation of vertical velocity fluctuations behind the vegetation-wall to that behind the wall barrier, to model the reduction of turbulence by the vegetation, 2) the entrainment factor f_m was multiplied by the ratio of turbulent velocities, and 3) the effective height of the wall was increased to account for additional plume lofting induced by the vegetation. The results, shown in Figure 2, indicate that the model needs to be improved, although it provides a reasonable description of the distributions of observed concentrations.

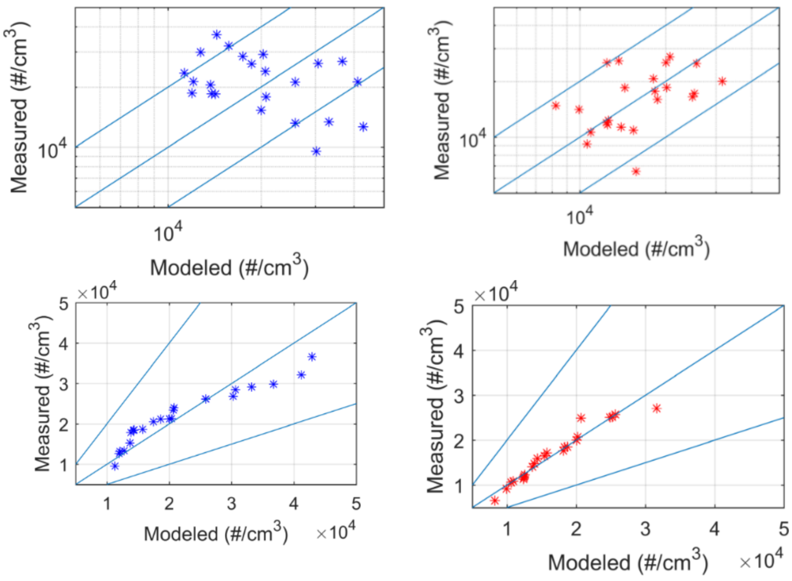

Figure 2. Scatter and quantile-quantile plot of measured and modeled values. Left panel: wall barrier. Right panel: wall-vegetation barrier.

INTERNAL BOUNDARY LAYER MODEL

Analysis of data collected in street canyons located in Hanover, Germany and Los Angeles, USA, suggests that street-level concentrations of vehicle-related pollutants can be estimated with the *v*ertical *d*ispersion *m*odel (VDM,Schulte et al., 2015), which assumes that vertical transport of emissions dominates the processes that governs these concentrations. One of the important inputs to VDM is the standard deviation of the vertical velocity fluctuatations at effective roof level, which in turn can be related to the friction velocity. In a follow-up study to evaluate VDM, we conducted a field study next to Market Street in Riverside, CA between July - September, 2015. One of the major objectives of this study was to evaluate a model to estimate the rooftop micrometeorological variables using measurements made upwind. A Campbell scientific CSAT3 sonic anemometer was used to measure the three components of the wind speed vector and the sonic temperature at 10 Hz at 3 m above the 25 m high roof of city hall. Another sonic anemometer was placed at Riverside airport, about 7.8 km southwest from city hall, at 2.7 m above ground level. The airport is usually upwind of the urban site.

The micrometeorological variables in Riverside were estimated using an internal boundary layer (IBL) model developed by Luhar et al. (2006) based on an approach suggested by Miyake (1965). The growth of the IBL height, *h*, is given by

$$\frac{dh}{dx} = A\left(\frac{\sigma_w}{U}\right)_u = f\left(\frac{h}{L_u},\frac{h}{z_{0u}}\right), \tag{3}$$

where the subscript '*u*' refers to urban values and *L* is the M-O length. This equation can be integrated to estimate *h* at the urban location, relating L_u to L_r. Then, the friction velocity at this location is computed by assuming that the mean velocities at *h* are the same at the urban and the rural upwind locations, $U_u(h) = U_r(h)$, giving

$$u_{*u} = u_{*r}\frac{\phi(h/L_r, h/z_{0r})}{\phi(h/L_u, h/z_{0u})} \tag{4}$$

where $\phi = \ln(h/z_0) + \psi_M(z_0/L) - \psi_M(h/L)$ and ψ_M is a function that accounts for stability.

The left panel of Figure 3 shows the comparison of the values of u_* estimated at the urban rooftop using the IBL model with corresponding observations. The comparison shows that the model underestimates the rooftop value by about 30% during unstable conditions. During stable atmospheric conditions the model significantly underestimates the urban surface friction velocity. The right panel of Figure 3 shows the comparison of the model with observations, where the model assumes neutral atmospheric conditions in the urban area. This improves the performance of the model for stable conditions, although the urban surface friction velocity is still underestimated.

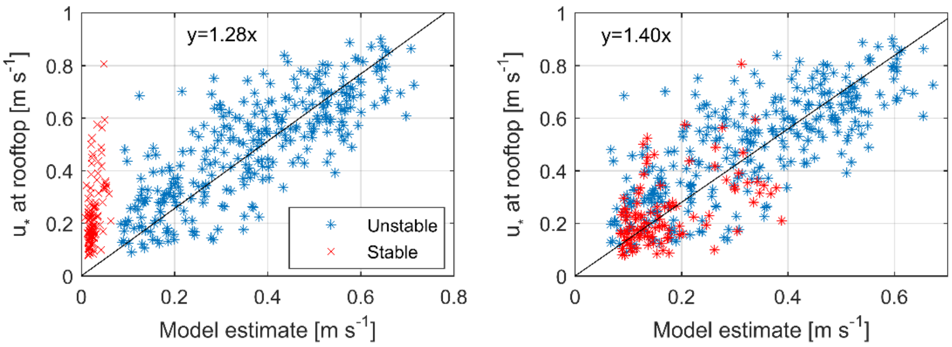

Figure 3. Model estimates of rooftop friction velocity compared with observations. Left panel: Accounts for stability. Right panel: Assumes neutral conditions in urban area.

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