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The ratio of effective building height to street width governs dispersion of local vehicle emissions

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HIGHLIGHTS

- Vertical turbulent transport governs dispersion of emissions in urban streets.
- Dispersion model depends on street aspect ratio and vertical turbulent velocity.
- Surface concentrations are most sensitive to street aspect ratio.
- These results inform the design of transit oriented developments.

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ABSTRACT

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 a model that assumes that vertical turbulent transport of emissions dominates the governing processes. The dispersion model relates surface concentrations to traffic flow rate, the effective aspect ratio of the street, and roof level turbulence. The dispersion model indicates that magnification of concentrations relative to those in the absence of buildings is most sensitive to the aspect ratio of the street, which is the ratio of the effective height of the buildings on the street to the width of the street. This result can be useful in the design of transit oriented developments that increase building density to reduce emissions from transportation.

1. Introduction

A transit oriented development (TOD) is defined as a high density community of homes, offices, and shops built within walking distance of a transit station, such as a light rail or a bus station. Expansion of TODs is being pursued to reduce greenhouse gas and air pollutant emissions associated with transportation. This will reduce concentrations of air pollutants averaged over city scales (~10 km). However, there is concern that high building density can reduce dispersion of pollutants relative to that in terrain without buildings. This paper describes results from a research program, motivated by this concern, to estimate the effect of building morphology on dispersion within a TOD. The program involved 1) analysis of data from a five year measurement program conducted in a street canyon in Hanover, Germany 2) field studies conducted in Los Angeles and subsequent analysis of the data, and 3) development of a semi-empirical dispersion model to describe the data from Hanover and Los Angeles. The long term Hanover data allowed us to formulate the relationship between concentrations and micrometeorology, while the variation of the building morphology in the Los Angeles field studies revealed the impact of street aspect ratio on dispersion.

Results from field studies (Hanna et al., 2014), laboratory experiments (Barlow and Belcher, 2002), and numerical simulations (Ketzel et al., 2000; Hang et al., 2012) have provided valuable insight into the mechanisms that govern dispersion of pollutants in the urban canopy. This information is the basis for semi-empirical dispersion models such as the Canyon Plume Box Model (CPBM) (Yamartino and Wiegand, 1986), and the Operational Street Pollution Model (OSPM) (Berkowicz et al., 1997). These models apply primarily to street canyons between relatively uniform buildings,

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which are common in Europe where these models originate. They may not be applicable to the inhomogeneous building structures that characterize urban area cores in the United States, because the inhomogeneous environments produce complex flow structures inconsistent with the street canyon model formulation (Nelson et al., 2007). A field study (Karra et al., 2011) conducted in Nicosia, Cyprus at a street canyon with different building heights on the two sides of the street showed that concentrations of CO were usually higher on the windward side than those on the leeward side of the street canyon. This observation contradicts the predictions of street canyon models with uniform building heights.

Inhomogeneous built environments with tall buildings can induce flows that are significantly different from the idealized flows assumed in street canyon models. A wind tunnel study of an urban neighbourhood with a single tall tower found enhanced vertical dispersion in the wake of the tower (Brixey et al., 2009). Another wind tunnel model of Manhattan found strong transport of contaminants up the lee sides of several of the tallest buildings (Heist et al., 2004). Results of the Joint Urban 2003 field study showed that the flow within an inhomogeneous street canyon was complex, with different flow structures resulting from slightly differing wind directions (Nelson et al., 2007). Large downdrafts and updrafts that could transport pollutants vertically were also observed.

Because of the complexity of the flows in such areas, our objective is limited to capturing the essential features of dispersion in the presence of buildings through a semi-empirical dispersion model. This model cannot describe the variation of concentration across the street as OSPM does. Its output, which corresponds to concentrations averaged over the area of a city block, is designed to provide guidance on the design of TODs.

2. Development of the semi-empirical dispersion model

The relationship between vehicle related concentrations in a street and associated micrometeorology was formulated through an analysis of data collected by the Lower Saxony Ministry for Environment, Energy, and Climate, in Göttinger Straβe, Hanover, Germany, during 2003–2007. Göttinger Str. (Fig. 1) is 25 m wide with 20 m tall buildings on either side. Measurements of NO and NO_2 concentrations were made at two locations: one on the southwest side of the road 1.5 m above ground level (AGL), and the other on the southwest building rooftop above the surface monitor. Wind speed and turbulence measurements were made using a sonic anemometer near the surface concentration monitor at 10 m AGL, and mean winds were measured near the rooftop monitor at 42 m AGL. Traffic flow measurements were made with automatic counters, and were converted into emission rates using emission factors of 0.465 g/km and 6.18 g/km for passenger cars and trucks, respectively, determined using EMFAC 2007 (California Air Resources Board, 2007).

We evaluated four dispersion models with the data from Göttinger Str. to determine the micrometeorological variables that best describe the concentrations. Thus, the models have different dependencies on the surface wind speed, U_s, and the standard deviation of the vertical velocity fluctuations, σ_w at 10 m AGL. The first model for the surface concentration, C_s, is the direct contribution term of OSPM (Berkowicz et al., 1997):

$$C_s = \sqrt{\frac{2}{\pi}} \frac{Q}{W\sigma_w} \ln\left(1 + \frac{W\sigma_w}{h_0 U_s}\right) + C_r \quad (1)$$

where C_r is the rooftop concentration, h_0 is the initial vertical plume spread, Q is the emission rate per unit length of the road, and W is the width of the road. The second model assumes that dispersion is dominated by the initial plume spread (Venkatram et al., 2007):

$$C_s \sim \frac{Q}{h_0 U_s} + C_r \quad (2)$$

The third model is derived by assuming concentrations are mixed over the height h_0, and follow a Gaussian profile above h_0, resulting in the following expression:

$$C_s \sim \frac{Q}{\sigma_w W}\left(1 + \sqrt{\frac{2}{\pi}} \frac{h_0 U_s}{L\sigma_w}\right)^{-1} + C_r \quad (3)$$

where L is the length of the road. The final model is given by Equation (4):

$$C_s = \alpha \frac{Q}{\sigma_w W} + C_r \quad (4)$$

where α is a parameter whose value is determined empirically. Equation (4), which is the leading term of Equation (3), is similar to the OSPM recirculating contribution model (Berkowicz et al., 1997), although in OSPM, the rooftop value of σ_w is prescribed, and is determined from the rooftop wind speed through a constant turbulent intensity. This equation is consistent with the scaling suggested by Kastner-Klein et al. (2003), who found that σ_w is a better scaling velocity than U_s for the concentration.

Scatterplots comparing the models with observed NO_x in Göttinger Str. are shown in Fig. 2. In Equation (4), the parameter α is taken to be 0.9, and σ_w in all the models corresponds to the measured value at 10 m AGL. The performances of Equations (3) and (4) are similar with both explaining about 54% of the variance of the observations. This suggests that the term, Q/σ_w, is the dominant contributor to the variation of observations. So we adopted Equation (4) to keep the model as simple as possible.

Fig. 3 shows that Equation (4) describes the observed concentration as a function of the deviation of the wind direction from the normal to the road even though it does not explicitly depend on wind direction. The dependence is implicit through the variation of σ_w with wind direction: the values of σ_w are smallest when the surface wind direction is normal to the road. Note that the modelled value is proportional to $1/\sigma_w$; so its variation with wind direction is equivalent to the variation of $1/\sigma_w$ with wind direction. These results appear to suggest that in this particular street canyon, the concentration at the surface is dominated by vertical transport.

We can provide a tentative basis for this model by assuming that the horizontally averaged concentration in the street canyon is governed primarily by vertical transport, so that the flux of pollutants at the surface is matched by vertical turbulent diffusion:

$$Q \sim KW\left(\frac{C_s - C_r}{H}\right) \quad (5)$$

where Q is the emission rate per unit length of the street canyon, W is the width of the canyon, and H is the equivalent height of the canyon. If we assume that the background makes similar contributions to C_s and C_r, we can evaluate Equation (5) with observations by using the difference between the observed C_s and C_r.

The eddy diffusivity K is expressed as

$$K = \sigma_w l \quad (6)$$

where σ_w is the vertical average, between the surface and roof, of the standard deviation of vertical velocity fluctuations, and l is a mixing length. The vertically averaged σ_w is estimated from the measured surface, σ_{ws}, and roof, σ_{wr}, values from