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Evaluation of the RIO-IFDM-street canyon model chain

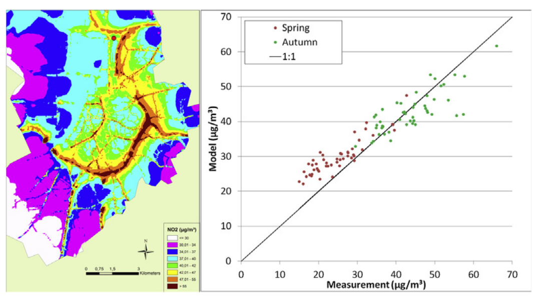
W. Lefebvre [a,*], M. Van Poppel [a,1], B. Maiheu [a,2], S. Janssen [a,3], E. Dons [a,b,4]

[a] VITO, Boeretang 200, 2400 Mol, Belgium
[b] Transportation Research Institute, Hasselt University, Wetenschapspark 5 Bus 6, 3590 Diepenbeek, Belgium

HIGHLIGHTS

- Setup of model chain ranging from regional to local scale.
- High resolution air quality maps for urban environments.
- Comparison to independent measurement data leads to very good validation results.
- Step-by-step analysis on the importance of the different model components.

GRAPHICAL ABSTRACT



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ABSTRACT

Integration of all relevant spatial scales in concentration modeling is important for assessing the European limit values for NO_2. The local NO_2-concentrations are influenced by the regional background, the local emissions and the street canyon effects. Therefore, it is important to consistently combine all these contributions in the model setup which is used for such an assessment. In this paper, we present the results of an integrated model chain, consisting of an advanced measurement interpolation model, a bi-Gaussian plume model and a canyon model to simulate the street-level concentrations over the city of Antwerp, Belgium. The results of this model chain are evaluated against independent weekly averaged NO_2 measurements at 49 locations in the city of Antwerp, during both a late autumn and a late spring week. It is shown that the model performed well, explaining between 62% and 87% of the spatial variance, with a RMSE between 5 and 6 µg m^{-1} and small biases. In addition to this overall validation, the performance of different components in the model chain is shown, in order to provide information on the importance of the different constituents.

1. Introduction

There is an old joke about a statistician who drowns in a river of on average half a meter deep. The same joke could apply to someone who wants to test the European NO_2 limits and uses a spatial average over a complete region, although it is questionable whether much laughter would be drawn from the public.

The European annual NO_2 limit of 40 µg m^{-1} has to be reached at every location. However, Eulerian models have limited spatial resolution and will provide an average concentration over a larger zone, typically about 1 km^2. A concentration in this zone which is lower than the limit is not instructive in assessing whether the limit is reached at every location within this zone (Thompson and Selin, 2012).

Measurements can solve this problem partially as they can measure at specific hotspots, although they are typically limited in

* Corresponding author. Tel.: +32 14 33 67 48.
E-mail addresses: wouterlefebvre@hotmail.com, wouter.lefebvre@vito.be (W. Lefebvre), martine.vanpoppel@vito.be (M. Van Poppel), bino.maiheu@vito.be (B. Maiheu), stijn.janssen@vito.be (S. Janssen), evi.dons@vito.be (E. Dons).
[1] Tel.: +32 14 33 53 65.
[2] Tel.: +32 14 33 67 45.
[3] Tel.: +32 14 33 67 02.
[4] Tel.: +32 14 33 51 90.

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space or time. As such, measurements do not provide concentrations averaged over a certain zone but result in point concentrations. While a well-distributed measurement network can thus give reliable information about exceedances of the European limits, it is in fact a series of point measurements. As a result, compliance to European Union limit values at all air quality monitoring stations does not necessary imply a compliance at every location in the area.

A number of experimental campaigns have quantified the spatial variability of urban pollutant concentration levels (da Silva et al., 2006; Thornburg et al., 2009; Vardoulakis et al., 2011). These papers have shown that for an assessment of the exposure of the population to air pollution, it is important to take into account both the heterogeneous spatial and temporal concentration distribution and the changing locations of the exposed population (e.g. Wilson et al., 2005; Beckx et al., 2009a; Setton et al., 2011; Dons et al., 2012; Dhondt et al., 2012). Furthermore, many questions can be raised about the representativeness of existing in-situ measurement sites (Buekers et al., 2011). There were recent attempts to characterize the spatial representativeness of air quality monitoring stations based on different approaches (Spangl et al., 2007; Joly and Peuch, 2011; Janssen et al., 2012). However, to our best knowledge, a unique robust methodology to assess the representativeness of in-situ measurements has not yet been achieved, especially on a street-level scale. It is therefore essential to create a reliable modeling framework which is able to capture both the spatial diverseness of the concentrations on a street-level scale, while still providing complete coverage over the studied region.

Several types of models have been applied in recent years to tackle this problem (e.g. Vardoulakis et al., 2003; Holmes and Morawska, 2006). The first group of models are the box models, such as OSPM (Berkowicz et al., 1997), which parameterize the effect of street canyons at the local scale. However, these models are unable to take into account the effect on the concentration by neighboring roads in a systematic way. Furthermore, these models need detailed inputs on the street-building configuration. As a result, these models are often applied only at one or some streets or at one or some locations (e.g. Hirtl and Baumann-Stanzer, 2007; Wang and Xie, 2009) or with reduced accuracy on the street-building configuration (e.g. Assael et al., 2008). Gaussian plume models, such as AERMOD, can easily take into account a complete city area. However, they lack the street canyon effect in their simulations, which can be significant for busy roads confined by continuous building-walls. An alternate approach is to apply CFD (Computational Fluid Dynamics) – based dispersion models, such as MISKAM. These models explicitly resolve the 3D geometry of the city and enable one to directly compute the dispersion in the street-canyon flow. However, due to computational restrictions typical for CFD models, it is currently practically unfeasible to apply them for a whole city and to simulate a complete year (Schatzmann and Leitl, 2011). For this reason there are only a very limited number of CFD applications used for operational policy support, although recent studies have been successful in demonstrating the potential of combining CFD calculations with a meteostatistics approach (Parra et al., 2010; Solazzo et al., 2011), albeit for a limited spatial domain.

Combinations between Gaussian models and box models, such as UBM-OSPM and ADMS-urban, have already been applied to some cities (Hirtl and Baumann-Stanzer, 2007; Berkowicz et al., 2008; Righi et al., 2009). When combined, these models can both perform yearlong simulations for a complete city and do this in a reasonable computing time, while taking into account both the street canyon effect and the effect of the neighboring roads. These model combinations have challenges in being consistent (e.g. eliminating double counting of emissions which are present both in the Gaussian model and in the box model, both in dispersion and in the chemistry, Lefebvre et al., 2011b), in devising a system in which detailed and accurate street-building geometry information is present for all street canyons (Jensen et al., 2001; Righi et al., 2009), in assimilating the known measurements in the city and thus correcting for eventual model biases or wrong estimations in the emissions, …

This paper evaluates such a consistent integrated modeling framework against independent measurements in the city of Antwerp. A comparison with measured concentration over two seasons is presented and the performance of different model components is discussed. This paper focuses on spatial validation. As such, we can be assured that major characteristics of the spatial concentration distribution are captured by the model. Nevertheless, this paper does not have the aim of validating the model framework in a complete fashion such as proposed in Jakeman et al. (2006) and Dennis et al. (2010). It will merely demonstrate its use for policy makers and thus will provide only part of what is needed to validate a model thoroughly in the framework of Jakeman et al. (2006) and Dennis et al. (2010).

2. Measurement campaign

Measurements reported in this paper are part of a larger multidisciplinary study (HAEPS; Health Effects of Air Pollution in Antwerp Schools; Van Poppel et al., 2012) dealing with health impact of traffic related air pollution on school children. To assess the exposure of the children at home, air quality measurements were performed at selected home locations.

NO_2 was measured over 7 days at selected locations in an urban area using diffusive sampling tubes (IVL, Sweden; Ferm and Svanberg (1998)) resulting in weekly average concentrations. The locations are characterized by differences in traffic exposure and street characteristics (e.g. street canyon locations, urban traffic locations and urban background locations), chosen to represent different ranges in concentration fields in an urban area. Diffusive samplers are placed in a dedicated rain shield attached to a rain-water pipe, a balcony or a streetlamp, near the front door, at a height of 2–3 m.

At each location, NO_2 was monitored during late spring (May–June 2011) and late autumn (November–December 2011). In both seasons, measurements were performed at 8 locations simultaneously during 5 consecutive weeks resulting in 40 locations sampled. In addition, all 40 locations were sampled simultaneously for one week in each season, including also 12 extra locations, resulting in 52 locations. During the entire sampling campaign, NO_2 was measured at an urban location of the AQ monitoring network.

To test reproducibility of the sampler a triple measurement was performed in spring at one location resulting in an average concentration of 25.9 ± 0.39 (SD) $\mu g\ m^{-3}$. Passive samplers were compared to reference monitors (Chemiluminescence) during the monitoring campaign at the urban location of the AQ monitoring station (5 weeks each season) and at a street location (1 week each season) resulting in 12 co-located measurements. Monitor data were averaged over the sampling period and plotted against the diffusive sampler values resulting in a regression line with intercept 2.95 (95% CI: −2.15–8.04) and slope 0.95 (95% CI: 0.84–1.06). The concentrations measured over different times are identified by season and week number. Measurements in late Spring or late Autumn are denoted by respectively S and A. The sample week is indicated by w1-w5 for week 1 up to week 5 respectively and by wAll for the week that sampling was performed over all locations simultaneously. Combining these time-related indicators, this results in e.g. S_w2 for the second week in the Spring campaign and A_wAll for the week in autumn in which all locations were measured simultaneously (Table 1).