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EXTENDED ABSTRACT

Validation of Lagrangian dispersion models with wind tunnel experiments for an idealised industrial site.

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Introduction

Accidental releases that may happen in industrial environments represent a hazard for human populations and the environment. Examples such as leaks of heavy gases, fires and the spread of toxic compounds generate a threat for human health and provoke pollutions that must be monitored and removed. To understand which zones of the site and of the surrounding environment are impacted, it is necessary to model the plume spread, which is rendered complex by the hardly predictable effects of turbulence and the impact of built elements on the wind field. This is why the operational dispersion models used to assess the risks associated to pollutant spread must be able to consider the impact of the site geometry on pollutant dispersion within reasonably short simulation times.

This can be achieved running Lagrangian particle dispersion models, whose formulation can rely on different descriptions of the velocity field, balancing the need for computational speed with the accuracy of the results (Wilson and Sawford, 1996).

Method

We focus on two Lagrangian dispersion models to compute the mean concentration field but that differ in the approaches used to compute the wind field.

On one side, PMSS (Parallelised Micro Swift Spray) is a suite of tools based on the SWIFT wind computation model and the Spray pollutant dispersion model (Tinarelli et al, 2012). The vertical profile of incoming wind velocity is first interpolated before creating recirculating regions based on geometrical considerations nearby obstacles. The impact of built elements on the wind field is hence captured through a semi-empirical approach, which enables fast computations even for dense and complex geometries (Oldrini et al, 2017).

On the other side, SLAM (Safety Lagrangian Atmospheric Model, Vendel et al, 2011) is a Lagrangian model that we couple to a database of RANS simulations (Reynolds-Averaged Navier Stokes). These are performed beforehand and interpolated once the meteorological conditions (wind direction and atmospheric stability) of the specific simulated case are provided. The velocity field obtained hence captures most of the effects of obstacles with a high spatial precision.

The formulation of the pollutant dispersion is essentially the same in both models, relying on the solution of the Langevin equation and adopting the well-established well-mixing condition (Thomson, 1987) in order to determine the coefficients of the drift and diffusion terms.

Both models are parallelised in order to shorten simulation times. We validate these two modelling approaches by comparing the simulated time-averaged concentration fields to a dataset provided by wind tunnel measurements performed on the small-scale model of a realistic industrial site, designed in order to reproduce typical features of real production plants. The dispersion of a passive tracer on this site is studied in wind tunnel at scale 1:200 inside the neutrally stratified boundary layer previously investigated in detail by Nironi et al (2015). A key feature of the site is the presence of a building, placed upwind, including elements with a complex geometry (assembly of pipes, tanks…), reproduced in the reduced scale model as a bulk of columns forming a porous obstacle having a solid volume fraction of 0.24. Velocity is measured with a Laser Doppler anemometer (LDA) while concentration measurements are performed with a fast flame ionisation detector (FFID) providing time series of the one-point concentration signal, from which statistical parameters (mean, variance) are retrieved (Marro et al 2020).

Both velocity and time-averaged concentration fields are compared to the fields predicted by both models. Distances are divided by the upstream building height, and dimensionless concentrations are obtained following the method of Nironi et al (2015).

Results

Figure 1 shows the comparison of the dimensionless concentration field obtained with PMSS (background color) to the experimental concentrations (colored dots) over several horizontal planes. For a better readability of the figures, the minimum and the maximum of the color scale were chosen as the respective minimum and maximum of experimental concentration values, since concentration maxima of both models are higher than the highest concentration value measured in wind tunnel.

The simulated plume is narrower than the experimental plume, particularly close to the source. Notably, the model do not capture well the widening of the plume close to the source, due to the recirculation generated by the porous building. This is associated to higher concentration values close to the source in the PMSS simulation than in the measurements.

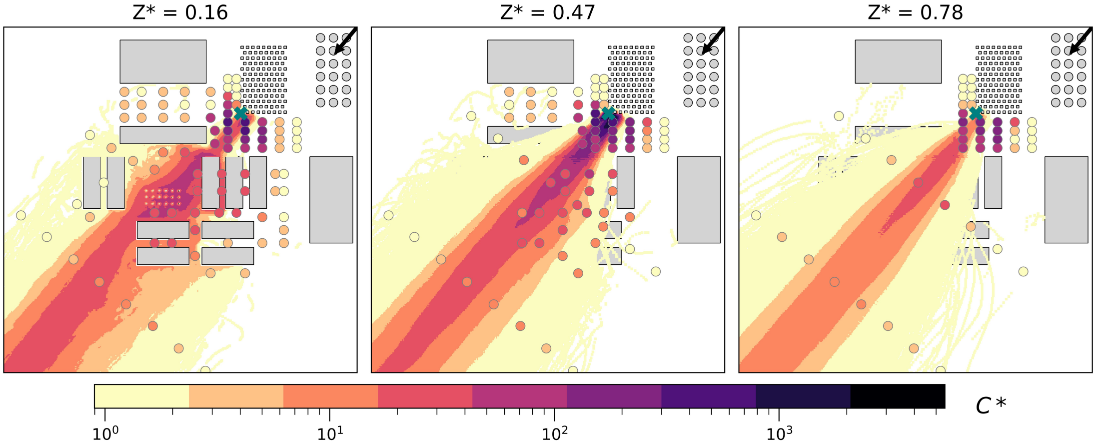

Figure 1: Average concentration on 3 horizontal layers obtained with PMSS (background colors) and experimentally (colored points).

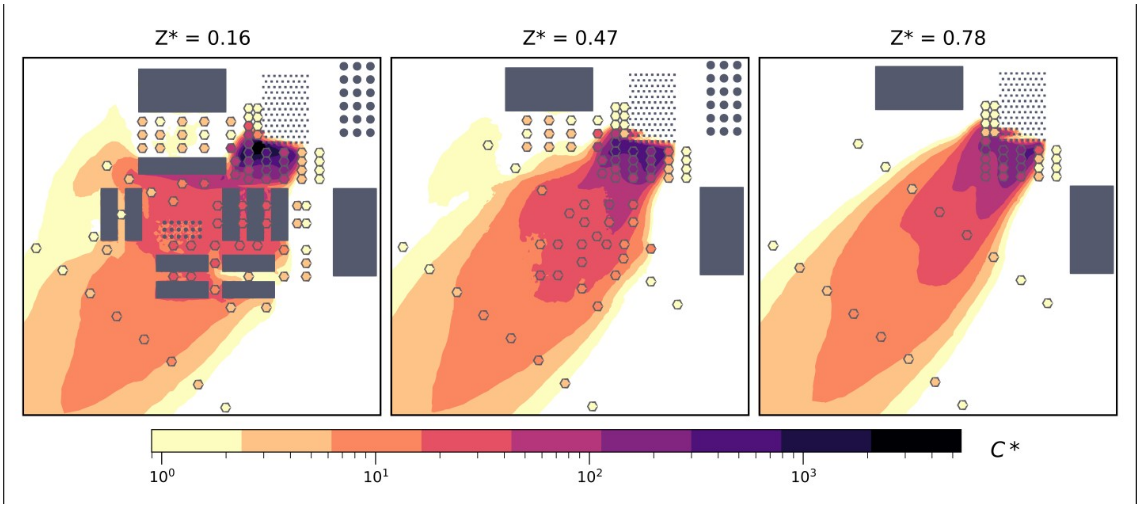

Figure 2: Average concentration obtained with SLAM and wind tunnel experiments. Same structure as Fig1.

A better agreement between PMSS and experimental data is observed in the mid- and the far field due to the weaker impact of the upwind obstacle. The global plume shape is captured by the model at all heights. Discrepancies appear mainly at ground level and at plume borders, since the experimental plume reaches a larger extent than the simulated plume, inducing higher concentrations in the simulation than in the measurements. Hence, the upwind building has a strong impact on the plume mainly in the near field, and capturing this effect is crucial to determine a reliable plume shape over the whole site in this configuration.

Figure 2 shows the concentration field simulated by SLAM at the same heights. The simulated plume is as wide as the experimental plume close to the source since the pollutant recirculates inside the wake of the porous building. The plume shape is very well captured at all distances from the source. However, higher concentration values are reached in the simulation close to the source compared to the experiment, which indicates an underestimation of dilution of the pollutant in the recirculation zone of the porous obstacle.

Conclusion

The site studied in this work has very specific geometrical features which harden the modelling of pollutant dispersion, notably due to the upwind position of the porous obstacle, which is composed of a group of columns. This induces a strong dependence of the simulated concentration field on the ability of the model to reproduce the global recirculation generated by this obstacle. The global wake is much larger than the sum of individual column wakes and can even reach a bigger extent than the wake of a solid obstacle of same dimensions (Taddei et al 2016). This is why the SLAM methodology, which relies on RANS simulations, produces more accurate plume shapes and concentration mean values than the PMSS method.

The present work focuses on the mean concentration field and does not consider peak concentration values, which can reach several orders of magnitude over the mean value. To estimate accurately the risks linked to concentration peaks, it is necessary to access at least to the mean and the variance of concentration (Bertagni et al, 2019). Simulating the concentration variance is thus a crucial feature that should be implemented in dispersion models for adequate risk estimates of accidental releases inside real built environments.

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