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► **To cite this version:**

Anaïs Pasquier, Michel Andre. Considering criteria related to spatial variabilities for the assessment of air pollution from traffic. *Transportation Research Procedia*, 2017, 25, 10.1016/j.trpro.2017.05.210 . hal-01757121

HAL Id: hal-01757121

<https://hal.science/hal-01757121v1>

Submitted on 3 Apr 2018

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World Conference on Transport Research - WCTR 2016 Shanghai. 10-15 July 2016

Considering criteria related to spatial variabilities for the assessment of air pollution from traffic

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Abstract

Understanding the impacts of traffic on air pollution is essential to improve methodology of assessment of population exposure which is depending on concentrations of pollutants in the atmosphere. However, concentrations are variable in space, and several parameters influence pollutant concentrations: traffic characteristics, meteorological conditions, pollutant characteristics, road structures, topography etc. In order to improve air pollution assessment methodology, we conducted a review that focuses on parameters that affect spatial variations in studying the link between concentrations and distance to the road for traffic pollutants as PM_{2.5}, PM₁₀, NO₂ etc. This step enabled the identification of pollutions classification. Then, we identified external factors as meteorology or road structures, barriers etc. that affect concentrations variabilities on the road and on urban area.

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Peer-review under responsibility of WORLD CONFERENCE ON TRANSPORT RESEARCH SOCIETY.

Keywords: Traffic; Roadside concentrations; Spatial variability; Meteorology; Barriers; Population exposure

1. Introduction

Road traffic in urban areas is identified as a major source of several air pollutants emissions, exhaust emissions or non-exhaust emissions as road abrasion or tire and brake wear which become more important in proportion with reduction of exhaust emissions. Those pollutants, as particulate matter (PM), nitrogen dioxide (NO₂), or benzene

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(C₆H₆) are responsible for adverse effects on human health. Indeed, the Health Effects Institute (HEI, 2010) reports that concentrations of traffic related air pollutants as ultra-fine particles, black carbon (BC), NO₂, benzene, are higher in the vicinity of roads than on regional backgrounds. Besides, according to the World Health Organization Regional Office for Europe (2013), adverse health effects to PM_{2.5}, PM₁₀, ultra-fine particles (UFP) occur both on short or long-term exposure. So, population exposure is depending on concentrations of air pollutants and their activities during the day, month or year. Thus, local concentrations of pollutants have to be taken into account to better know population exposure on-road, on sidewalk, at work or at home.

However, concentrations are variable in space, and several parameters and factors influence pollutant concentrations. Thus we aim at identifying these parameters to better understand spatialization of pollution at a urban and local scale and the population exposure. As a first step, to improve comprehension of population exposure to levels of traffic related air pollution in the vicinity of roads, the review focuses on local scale in studying the link between concentrations and distance to the road for pollutant as PM_{2.5}, PM₁₀, NO₂ etc. This step should enable the identification of pollution that is specifically associated to “near-road pollutants”, and a pollution of background pollutants. Then, in a second time, as dispersion depends on several external factors as meteorology (wind speed and direction, precipitations, temperature etc.) or road structures, barriers etc., we will try to identify effects of those parameters on concentrations variabilities whether at the road scale or at the city scale. Finally, we consider how population exposure is affected by the previous parameters and how we could use it to improve traffic air pollution assessment.

Nomenclature

BS	Black Smoke
BC	Black carbon
C ₆ H ₆	Benzene
CO	Carbon monoxide
EC	Elemental Carbon
HGV	Heavy Goods Vehicles
NO	Nitrogen monoxide
NO ₂	Nitrogen dioxide
O ₃	Ozone
PM	Particulate matter
PM _{2.5}	Particulate matter with diameter of 2.5µm or less
PM ₁₀	Particulate matter with diameter of 10µm or less
p-PAH	Particle-bound Polycyclic aromatic hydrocarbons
RH	Relative humidity
SF ₆	Sulfur hexa-fluoride
UFP	Ultra-fine particle
VOCs	Volatile organic compounds

2. Methodology

We conducted a review of studies about four main topics. Firstly, we focused on the relation between the concentrations of major pollutants from road traffic and distance to the road. So we searched studies about “spatial variabilities” of “traffic related air pollutants” concentrations in the “near road” environment in “urban” areas. During this work, several factors as the presence of road-side barriers or the infrastructure geometries appeared to be of interest in order to understand spatial variabilities of air pollution in road vicinity. Thus, we looked for studies concerning the effects of “road-side barriers”, or “roadway configurations” effects on “dispersion”. Thirdly, as the meteorology plays an important role in the dispersion of pollutants in the atmosphere and was often cited in the previous studies, we sought studies that are about the influence of meteorological parameters on pollutant dispersion. Finally, we looked for few cases that dealt with external sources and background concentrations which

affect spatial variabilities, and impact area of air pollutants emitted by road traffic. However, even if we classified those studies in four parts, some studies are dealing with more than one of these subjects, but are presented in the part for which the most relevant results or conclusions were found. In this work we concentrated our attention on recent studies and on urban areas. In order to establish an exhaustive view of studies reviewed, we summarize them with their main outcomes. In the review, some concepts will be used: “upwind” is for a wind blowing from the point of interest to the road, while “downwind” is for wind blowing from the road to the point of interest (sampling sites for instance), the “atmospheric boundary layer” is “the layer of air directly above the Earth’s surface in which the effects of the surface (friction, heating and cooling) are felt directly in time scales less than a day, and in which significant fluxes of momentum, heat or matter are carried by turbulent motions on a scale of the order of the depths of the boundary layer or less.” (Garratt 1994).

3. Summary of studies

This summary will enable a discussion about effects of several phenomena and factors on air pollution, and particularly pollutant concentrations with distance to the road, to identify pollutants having specific exposure issues in a near-road environment.

3.1. Relation between concentrations and distance to the road

Firstly, we focused on studies that are about local concentrations and gradients of traffic related air pollutants with distance to a road axis.

Roorda-Knape et al. (1998) reported measurements of traffic related air pollutant in six districts near major motorways in Netherland. PM_{10} , $PM_{2.5}$, black smoke (BS) and benzene outdoor concentrations were measured at four distances from the roadside in two districts, and NO_2 outdoor concentrations in all districts. The study showed that concentrations of BS strongly decreased with distance to the road, NO_2 concentrations also declined with the distance to the road, but gradients were not found for PM_{10} , $PM_{2.5}$ and benzene. However, significant declines of concentrations were found only between 15m (first distance) and 115m (second distance) for PM_{10} and benzene. Gradients found for BS and NO_2 were curvilinear and influenced by wind exposure: in high downwind situation gradients were more obvious.

Hitchins et al. (2000) measured the number concentrations of particles (UFP between 0.015 to $0.697\mu m$ and 0.5 to $20\mu m$ particles) at two sites near major roads near Brisbane, Australia, at increasing distances from 15 to 375m to the road. Results were highly dependent with wind conditions. In downwind conditions, concentrations of particle number smaller than $0.7\mu m$ decreased significantly with the distance to the road. At 150m, concentrations were about half of the concentrations at 15m. A similar trend was found for $PM_{2.5}$. Concentrations levels close to the road were higher for lower wind speeds and lower for higher wind speeds. When the wind was parallel to the road, the total particle number concentration also decreased with distance to the road, but half of the maximum concentrations were reached at about 50 to 100m for particle sized from 0.015 to $0.697\mu m$. When the wind was blowing from the sampling points to the road, no effect was found for particle at a distance larger than 15m: concentrations were similar to urban levels measured. According to this study an exponential decay curve would apply to UFP dispersion with distance to the road in downwind conditions.

In Tiitta et al. (2002), the mass concentrations of $PM_{2.5}$ were measured in a suburban environment in Finland near a major road at 12, 25, 52 and 87m from the center of the major road. Concentrations were found higher in vicinity of the road (12 and 25m) but the difference between concentrations measured at 52 and 87 were not statistically significant.

Zhu et al. (2002) compared measurements of concentrations and size distribution of UFP, in the range from 6 to 220nm, CO and BC, on an Interstate (710) in Los Angeles with previous measurements near a roadway (405) dominated by gasoline vehicles. The wind came from the freeway towards the sampling site 80% of the time, with a speed of 3m/s. CO, BC and particle number concentrations decreased about 60-80% in 100m and leveled off after 150m. The decrease of concentrations was exponential. The 710 Interstate had more than 25% heavy diesel trucks while the 405 freeway had less than 5%. Thus, comparison between the studies emphasized differences between concentrations of BC and CO in the vicinity of the roadways: concentration of CO near the 405 freeway was two

times more important than near the interstate, while BC concentrations were three times higher near the Interstate. They also studied the size distribution of UFP with increasing distance to the road and concluded that coagulation plays an important role for the smallest ultra-fine particles.

Gilbert et al. (2003) conducted a study to measure concentrations of NO₂ in function of distance to a major highway in Montreal, Canada. Passive samplers of NO₂ were installed at a height of 2.5m and at distances ranging from 0 to 1310m from the highway. They found that concentrations decreased with distance, and the major decrease occurred within 200m from the highway. Besides, concentrations were always higher downwind than upwind the highway. They concluded that distance to a major roadway may be a good surrogate for assessing exposure to some traffic related air pollutant.

Zhou and Levy (2007) conducted an analysis of the published studies between 1998 and 2005 about the spatial extent of air pollution from mobile sources. The concept of spatial extent is defined as the distance from a source of emissions at which individuals and population groups are exposed to elevate risks of adverse health effects. They categorized selected studies in function of pollutant type and background concentrations: inert pollutant and high background (PM mass without background removed in the analysis), or inert pollutant and low or removed background (carbon monoxide (CO), benzene, elemental carbon (EC), PM mass with background removed), or reactive pollutant and near-source removal (NO, UFP), or reactive pollutant and near-source formation (NO₂). They concluded on first-order rules for the spatial extent of impact from mobile sources: an order of 100 to 400 m for elemental carbon or PM mass concentration; 200 to 500m for NO₂; and 100 to 300m for UFP count.

Beckerman et al. (2008) studied NO₂, NO_x, ozone (O₃) and volatile organic compounds (VOCs) concentrations near a highway (401) in Toronto, Canada, to assess correlations between NO₂ and other pollutants. Sampling sites were along two transects across the highway. The wind predominant direction was north-east. Thus, points that were on the south of the way were downwind and sites on the north were upwind. The results showed that NO and NO₂ concentrations decreased as the distance to the expressway increased, but faster for NO concentrations. Concentrations dropped off rapidly in the first 50m and continued to decrease until background levels at 400m in the downwind conditions (200m in upwind conditions). The inverse pattern was observed for O₃ concentrations as levels increased as distances became larger. Besides, decay gradients with distance of benzene, n-hexane and methyl tert-butyl ether (MTBE) were similar to gradients of NO₂.

Karner et al. (2010) synthesized data about air pollutant concentrations collected in 37 studies at various distances to the road. They studied normalization of data: usual normalization of concentrations (according to wind speed or traffic volume) is problematic for an analysis of data from numerous studies (lack of information). Thus, the authors focused on two types of normalization: normalizing to background concentrations and normalizing to edge-of-road (avoiding issues when background concentrations are not available, and avoiding comparison problems as background concentrations assessment is not similar from one study to another). They classified pollutants concentrations with distance to the road into three groups: the first was defined by a rapid decrease (at least 50% decrease in the first 150m) and then a more gradual decline, the second had a significant reduction along the distance, and the third group do not show trend with distance to the road. According to the edge-normalization, CO, NO, NO_x and UFP decreased sharply in the first 150m and thus are in the first group, while benzene, EC, NO₂ and PM_{2.5} are in the second group as their concentrations declined less rapidly, and finally PM₁₀ and particle larger than 300nm are in the third group. Most of pollutant concentrations leveled off by 80 to 600m from the road.

Padró-Martínez et al. (2012) measured particle number concentrations (PNC), particle size distribution, PM_{2.5}, particle-bound polycyclic aromatic hydrocarbons (p-PAH), BC, CO, NO and NO_x concentrations in an urban area near a highway (Interstate 93), in Somerville, USA. They studied their dependence with meteorological and traffic conditions. Monitoring was made on 58 days during the four seasons, at weekdays or weekend, several hours of the day, and at numerous distances to the I-93. The results showed that concentrations were the highest in the 50m from the road. PNC were impacted by wind speed: when the wind speed was inferior to 0.3m.s⁻¹, PNC in all sampling sites were highest and when wind speed was greater than 1.6m.s⁻¹, PNC were lowest. Concerning NO_x concentrations, the highest levels were measured in the 50m from the road, and the concentrations decreased to background levels within 200m. The decay gradients with distance for NO, CO, p-PAH, and BC were similar to NO_x. But, the PM_{2.5} observed distance-decay gradients were flat as PM_{2.5} had regional source intake, and their concentrations were higher during spring and summer than in winter, which could be explained by the setting up of secondary aerosol.

Barros et al. (2013) analyzed vertical and horizontal variation of NO_2 and C_6H_6 concentrations along a major city ring motorway (VCI) in Oporto, Portugal. They choose to focus on three representative sub-domains along the ring: a street canyon type road, a flat open air type road and flat open air type mixed with a perpendicular intersection where VCI is elevated by a viaduct. They collected meteorological and atmospheric conditions to understand transport of pollutants emitted at the VCI for all sub-domains, to investigate their effects on urban canopy concentrations (about 50m of height), or concentrations variability at roadside and 100m to the road. They found that concentrations at 100m from the VCI were reduced in average of $32.7 \pm 15.9\%$ for NO_2 and about 25.7 ± 12.1 for C_6H_6 .

Patton et al. (2014) studied the decrease of traffic related air pollutants with distance in three near-highway (I-93) areas: Somerville, Dorchester/south Boston, Chinatown and in a background area, Malden, USA. They aimed at determine whether differences of concentrations between neighborhoods could be explained by traffic and meteorology. Particulate number concentration, p-PAH, NO, NO_x , BC, CO, $\text{PM}_{2.5}$ concentrations were monitored during 35 to 47 days in each neighborhood. Results showed that pollutants concentrations decreased as the distance to the road increased, and leveled off at 200m from the road. Thus the I-93 was identified as a major source for those pollutants, however, although experiments in three areas were conducted over similar conditions of meteorological and traffic conditions, concentrations gradients were varying between neighborhoods. They assumed that those differences could be explained by the infrastructure (elevated road in Somerville, and below-grade in Dorchester and Chinatown), noise-barriers, street-canyons etc. Their results suggested that one can not generalize gradients of concentrations with distance and contrasts between near-road areas and background. And spatial variabilities might be more important than could suggest measures of distance gradients at different points of a highway.

3.2. Infrastructure geometries effects on concentrations

As Patton et al. (2014) mentioned it, the infrastructures could induce some spatial variation of pollutants concentrations. Thus, among other factors, we assumed that geometries of road or the presence of walls at roadsides would affect dispersion and transport of pollutants, thus we looked for studies that investigate effects of infrastructure geometries on concentrations of pollutants.

Ning et al. (2010) conducted measurements of particle size distributions and co-pollutants concentrations at two freeways (I-710 and I-5) in Southern California, with two sampling sites in each freeway: one with noise barrier, and one without. Distributions and concentrations were investigated in the immediate vicinity and at different distances from roads. The results found that particle number concentrations reached background concentrations at 200m for the I-710 and at 120m for I-5. With noise barrier, PNC at I-710 and I-5 were, respectively, 43% and 45% lower at 15m than at 20m downwind without barrier. They also found that PNC increased as downwind distance increased, they reached their maxima at 100m for I-710 and 80m for I-5. Besides, maxima are 2.4 and 2.2 higher than those observed for corresponding sites without barrier for I-710 and I-5. PNC reached background concentrations at 400m. Similarly, with a noise barrier, CO, NO_2 and BC concentrations reached background concentrations around 400m and 250m for I-710 and I-5 respectively, after a peak at 80-100m whereas all pollutants had reached background concentrations at about 150m for sites without barriers. Thus this study highlighted a deficit zone in concentrations immediately behind the noise barrier, but concentrations were increasing until 80 to 100m, and more than two times higher than without barrier.

Finn et al. (2010) performed a roadway dispersion study, at the Idaho National Laboratory, to investigate the effects of a roadside barrier on the concentrations of pollutants due to the roadway emissions in various atmospheric conditions. Roadway emissions were simulated by the release of an atmospheric tracer, the sulfur hexafluoride (SF_6), by two line sources of 54m long, positioned 1m above the ground. The first line source was used for a 90m long noise barrier and the second for a control experiment without barrier. Results showed that the magnitude of the concentrations increased as atmospheric stability increased. They also noticed that lateral dispersion of concentrations and the horizontal plume spread were greater with the barrier. Besides, a deficit of concentrations around 50% or higher was observed behind the barrier compared to concentrations at same locations without the barrier for any atmospheric stability conditions. But upwind of the barrier, in the location that is supposed to be the roadway, concentrations were higher in stable conditions.

Hagler et al. (2011) designed a computational fluid dynamics roadside barrier simulation to observe the effects of barrier height and wind direction on concentrations. A wall of 3 to 18m high was located along a 6 lanes highway. Results suggested that barrier significantly reduce maximum and ground concentrations downwind of a major roadway, compared to a situation without barrier. Besides, a roadside barrier induced a vertical shifting of concentrations, mixing pollutant with cleaner air above the road and therefore reducing the plume concentrations. However the vertical shifting of concentrations is responsible for higher concentrations at the top of the barrier. Concerning the upwind side, the presence of a barrier increases on-road concentrations, and higher the barrier is, higher are the concentrations.

Hagler et al. (2012) monitored UFP concentrations at three locations in North Carolina, USA. Two sites had thin trees along the roadway. The third site had a 6m noise-reducing wall. UFP concentrations immediately behind the noise barrier were reduced by approximately 50% for several wind conditions. But, the results for tree barriers were not as obvious, levels observed were variable: whether higher or lower than a configuration without barrier. Indeed, those tree barriers appeared to be porous, they concluded that a less porous vegetation barrier could possibly improve near-road air pollution but additional researches must be done to assess the potential of vegetation on near-road air quality.

Schulte et al. (2014) reviewed results from field studies, laboratory experiments and numerical simulations to propose two semi-empirical dispersion models. Then, they illustrate their application by assessing impacts on near road concentrations of a barrier under different atmosphere conditions. Results highlighted that barriers affect dispersion by increasing vertical dispersion, inducing vertical mixing behind the barrier, and shifting concentrations above the barriers.

Baldauf et al. (2013) explored variabilities of traffic pollutants concentrations with roadway configurations in Las Vegas, Nevada, USA. They measured and simulated CO and NO₂ concentrations along the interstate 15, an urban highway. Pollutants were monitored by four stations located along an east-west transect (I-15 is a north-south axis), at 100 m upwind, and at 20, 100 and 300m downwind. This section of I-15 is depressed and surrounded by a sloped wall (5m and a slope of 20°). Besides, additional monitoring was performed at 500m from the south of the transect, at 20m downwind of I-15, the section was here at-grade with surrounding. Moreover, wind tunnel simulations were conducted. A 1:200 scale model of the domain was installed in the tunnel. Results of field measurements showed that maximum concentrations for NO₂ and CO were measured at the site at-grade with I-15 during low wind speed from west-southwest conditions. The monitor located at the top of the section more often measured higher concentrations than that at-grade site. However, at-grade site knew higher impact of primary pollutants as CO and NO. The tunnel simulations provided information about wind and building effects. When the wind was perpendicular to the road, the top of the depressed section experienced decreased pollutant concentrations (about 15 to 25%). And buildings impacted airflow: when downwind they increased turbulence, when upwind, they caused less turbulent mixing and therefore increased concentrations in the downwind side of the road.

Steffens et al. (2014) conducted experiments in a wind tunnel, where twelve road geometries were studied (elevated, at-grade, depressed roads, roadside barriers). The results were compared with simulations performed with a large eddy simulation model. Results showed that geometries or presence of roadside barriers affected pollutants dispersion in road vicinity. All these configurations, except one, decreased concentrations on roadsides at ground level, while concentrations on the road and at the height of the structure increased. The exception was an elevated road that allowed a decrease of pollutants concentrations on the road, and on the surrounding ground, but they noticed an increase of concentrations at the height of the elevation above the surrounding.

Rakowska et al. (2014) investigated the traffic related air pollutants in and near major streets in Hong Kong. The streets were chosen in a heavy traffic district, with high pedestrian usage. One street was wide and open and had 10 times the traffic of the two others that had high-rise buildings along them. They observed that peak concentrations occurred in the street with the lowest traffic level, confirming that street canyon restrict dispersion of traffic emissions. Moreover, meteorological measurements of road surface wind speeds in the two street canyons were reduced compared to the open street, making dispersion difficult.

Zhou et Levy (2008) studied the influence of street canyons on population exposure per unit of emission of motorized vehicles. They considered the street configuration (i.e. the street canyon versus other linear sources on open area) and individual exposure rather than residential concentrations. They conducted their calculations on mid-town Manhattan (New York City), which is an area with street canyons, high buildings, high traffic volumes and a

high population density. As they aimed at studying the population exposure and the relative importance of different subpopulations, they also choose this area because of the presence of diverse population groups such as pedestrians, residents, office workers, and drivers. Besides, they could use data from a campaign conducted in this site in August 2006 for the New York Metropolitan Exposure to Traffic Study (NYMETS). They modeled $PM_{2.5}$, PM_{10} , CO, NO, NO_2 and UFP concentrations for a base case and a sensitivity analysis, using. In the base scenario, they used a median height of 60m to represent buildings on both sides of streets, and a width of 30m. The vertical profile found for CO showed that 30% of the ground concentration remains at the top of the street canyon. The same type of profile was found for the other pollutants except for the NO_2 , for which 50% of concentration remains at the top of the street canyon. The sharpest decrease occurs within the first 15m above the ground.

3.3. Meteorological and topography conditions affecting concentrations

In the first paragraph, we saw number of cases that took into account wind conditions, thus we analyzed recent studies that included meteorological, seasonal elements to explain local concentrations variabilities, or regional intake of pollutants. Moreover, as meteorological conditions as wind speed and direction or atmospheric stability are affected by local topography, we considered several cases that considered topography affecting air pollution.

Padró-Martínez et al. (2012) found out that concentrations of pollutants related to traffic were higher in winter than during the three other seasons. Particle number concentrations were the highest during the winter, then during the spring, and were the lowest during summer and fall. They attributed these trends to higher levels of exhaust emissions, and more stabilized atmosphere during cold month.

Schleicher et al. (2013) investigated BC concentrations in Beijing over a two-year period on 5 sites. Results showed that concentrations are higher in winter than in summer. Meteorological conditions and other sources could explain that: stabilized atmospheric conditions and a lower height of the boundary layer and also lower temperatures in winter, and thus a higher intake due to heating with combustion of coal. Besides, they observed higher concentrations of BC during nights which could be explained by a lower boundary layer, and higher activity of heavy duty vehicles during nights.

Barros et al. (2013) also compared concentrations of NO_2 and C_6H_6 along the VCI regarding the seasonal variations. They found that the atmosphere was more stable during winter than spring or fall in the domain, and concentrations were higher in winter. The maximum of NO_2 concentrations was in April and in January for C_6H_6 , and minima for both NO_2 and C_6H_6 were in September. Besides, photochemical reactions of NO_2 and C_6H_6 oxidation are enhanced by the increase of sunlight in spring or fall. Thus, the variation of transport and dispersion conditions between seasons is an important factor. Concentrations of NO_2 and C_6H_6 decreased respectively of 15% and 33% in fall compared to winter.

Fruin et al. (2014) performed measurements of PM, EC and OC in size ranges of less than $0.2\mu m$, 0.2 to $2.5\mu m$ and 2.5 to $10\mu m$, and NO_x and NO_2 concentrations in eight Southern California communities. Results pointed out that primary pollutants as $EC_{2.5}$ and NO_x had higher concentrations during cool season (October to March) than during the warm season (April to September) due to weaker sea breezes, lower mixing heights and stronger stabilized atmospheric conditions during nights in cool season. For pollutants with significant contributions from secondary conditions, differences between seasons were less important and concentrations were as high or higher in warm season for $PM_{2.5}$ or water soluble OC. Besides, variabilities of $PM_{2.5}$ and $OC_{0.2}$ between each community were lower, that demonstrates regional and uniform concentrations.

In Eeftens et al. (2015) measurements of NO_2 , $PM_{2.5}$, PM_{10} , UFP number concentrations and $PM_{2.5}$ absorbance were performed in eight areas in Switzerland. NO_2 concentrations were measured in the eight areas and on 40 sites per area, and other pollutants concentrations were monitored on four areas and 20 sites per area. Samplings were performed three times during different seasons. They found that concentrations were higher in larger cities than in smaller ones, and were higher in winter than in summer for all pollutants except NO_2 .

Zhang et al. (2015) investigated relationships of six air pollutants ($PM_{2.5}$, PM_{10} , CO, SO_2 , NO_2 and O_3) with meteorological parameters (winds speed and direction, temperature and relative humidity (RH)) in three megacities in China (Beijing, Shanghai and Guangzhou) during a 12 months campaign in 2013-2014. They aim at improving the understanding of mechanisms that produce air pollution. They observed clear seasonal trends for $PM_{2.5}$, PM_{10} , CO, SO_2 and NO_2 : maxima in winter and minima in winter. Seasonal variation is partially due to the variation of the

boundary layer (lower in winter, higher in summer). The study of wind speed correlation with concentrations indicated that during summer, when the atmospheric boundary layer can grow up, vertical dispersion may play a more important role and horizontal dispersion plays a less important role. In Beijing, easterly wind led to the highest concentrations of $PM_{2.5}$, and then followed by southerly wind, because they transport pollutants from polluted areas as Tianjin and Central China. Whereas northerly wind led to the lowest $PM_{2.5}$ concentrations, as it brings cleaner air from mountains. In Shanghai, the highest $PM_{2.5}$ concentrations were related to westerly wind followed by north wind, indicating the transport of pollutants from north and west to Shanghai. While in Guangzhou, northerly winds were associated with the highest $PM_{2.5}$ concentrations. They also identified conditions for which highest 10% and lowest 10% of concentrations occurred. To sum up, the highest 10% $PM_{2.5}$, PM_{10} , CO , SO_2 , and NO_2 concentrations were related to lower wind speeds and lower temperature. In Beijing, the highest 10% of concentrations occurred during higher relative humidity (RH) but with lower values in Guangzhou, while in Shanghai between the highest and the lowest 10% concentrations, there were less fluctuations of RH.

Miao et al. (2015) investigated the boundary layer processes in the Beijing-Tianjin-Hebei (BTH) region. Indeed, this region experiences frequent heavy pollution during fall and winter. The pollution was often increased by unfavorable atmospheric boundary layer (ABL) conditions. The topography in this region impacts the ABL processes. But, these impacts on air quality were not yet clearly understood. Four numerical experiments with the Weather and Research Forecasting with Chemistry model were conducted, one for each season. Results suggested that seasonal variation significantly modulates ABL processes. In fall, thermal contrast between the mountains and the plain leads to an important breeze circulation from mountain to plain. In summer, southeasterly winds allow the sea-breeze front to penetrate farther inland (~150 km from the coast), and the breeze from mountain to plain is less marked. In spring and winter, with strong northwesterly synoptic winds, the sea breeze circulation is confined in the coastal area, and the mountain to plain breeze is suppressed. The ABL height is low in winter due to a strong surface stability, while ABL heights are high in spring due to strong mechanical forcing. The relatively low BL height in fall and winter may exacerbate the air pollution, thus contributing to the frequent severe polluted events in the Beijing-Tianjin-Hebei region.

3.4. External sources, background concentrations and a few other contextual conditions

Since the impact area of concentrations of pollutant related to traffic depends on the background concentrations, or external intakes of pollutants, we sought instances to know to what extent they affected the assessment of air pollution due to traffic.

Wu et al. (2015) analyzed distributions of BC, UFP and $PM_{0.5-2.5}$ concentrations in the urban environment in the south of the city of Edinburgh, UK. BC and UFP showed a high spatial variability, three times larger than that of $PM_{0.5-2.5}$. BC and UFP concentrations were affected by geographical locations and background concentrations variations, while $PM_{0.5-2.5}$ concentrations were influenced by regional sources. Concentrations measured during non-working days, where Heavy Goods Vehicles (HGV) flows are reduced, resulted on a lower BC/UFP ratio than that during working days. They concluded that HGV may contribute more to BC concentrations.

Beekmann et al. (2015) characterized air quality of Paris, France, during two campaigns of one-month and from one-year-observations of daily chemical compositions of $PM_{2.5}$. They found out that about 70% of the urban background of fine particulate matter is transported to Paris from upwind regions of France and continental Europe. They compared Paris case to other megacities around the world. In Paris, primary fossil fuel combustion emissions constituted little of carbonaceous fine PM: less than 20 % during winter and less than 40 % during summer. But, cooking, and during winter, residential wood burning were the major primary organic PM sources. Besides, low BC and EC levels (compared to other megacities) are consistent with relatively low emissions in a post-industrial megacity such as Paris. They concluded that on other post-industrial, mid-latitude, flat terrain megacities like New York City, London, and Tokyo there is a similar behavior. Whereas in megacities as Mexico City or Los Angeles, which are affected by factors such as larger local emissions, dispersion limited by topography and larger radiation, local sources are predominant.

Dimitriou and Kassomenos (2013) aimed at identifying local and regional sources of PM_{10} and $PM_{2.5}$ affecting their levels in Lisbon, Madrid, Marseille and Rome, four large cities in southern Europe. They used data from seven air pollution monitoring stations. Traffic was found to be the primary source of PM at all sites. However, seasonal

variations of $PM_{2.5}/PM_{10}$ ratio occurred, with a minimum during warm season. This is explained by an increase of dust resuspension, and biogenic coarse PM (pollens, seeds, or forest surroundings Marseille etc.). They also identified distance sources: long range transport of dust from Sahara desert and transportation of Mediterranean Sea spray or Atlantic Ocean spray were identified as the primary regional sources of exogenous PM_{10} in the four cities.

In Dimitriou and Kassomenos (2014), similar analysis were conducted to identify local and exogenous sources affecting PM levels in five cities of Northern Europe: London, Paris, Hamburg, Copenhagen, Stockholm. In each city, one background and one traffic air monitoring station were selected. They found that background sites were affected by important natural sources of particles. Moreover, the daily $PM_{2.5}/PM_{10}$ ratios were minima during warm period at stations with natural aerosol sources (dust resuspension or biogenic coarse particulate matter). In Stockholm, minima were found in a period from February to April and were attributed to studded tyres, road abrasion emissions and resuspension of particles. $PM_{2.5}/PM_{10}$ ratios were not variable at traffic hot spots. Finally the atmospheric trajectories analysis revealed that background site in Hamburg was affected by air masses coming from Poland and the Czech Republic, as background sites in Stockholm and Copenhagen. Stockholm and Copenhagen were also impacted by air flows from Scandinavian and Jutland Peninsulas respectively, and from Germany and France. Whereas London and Paris are influenced by air masses from France, Germany, UK and North Atlantic, carrying particles related to combustion or sea spray.

4. Discussion and identification of groups of pollutants, specificities and sub-groups of population

4.1. Classification of pollutions

The review of studies reporting the evolution of traffic air pollutant concentrations with distance to the road enabled the identification of works suggesting classification of pollutions. Particularly, pollutant dispersion and transformations are driven by their own chemical and physical properties. Thus, according to Zhou and Levy (2007), we can have a first classification of pollutions: pollution of inert pollutants or reactive pollutants. Inert pollutants are those for which concentration profiles are governed by dilution and diffusion. Reactive pollutants can be separate into three groups: pollutants which react in the atmosphere with other components to form other chemical species, pollutants that are formed by chemical transformations in the atmosphere and finally ultrafine particulate matter that coagulate. For instance, NO oxidizes in the atmosphere to form NO_2 , thus oxidization and dilution result in a sharp decrease of NO concentrations with distance. But NO_2 dilution is slowed down by the formation process, and the concentration decrease is more gradual. UFP coagulation combined with dilution result in a fast decrease of concentrations. Thus, for Zhou et Levy (2007), Benzene, CO, BS/ BC and particulate matter in mass are considered as relatively inert pollutants, NO and UFP are reactive pollutants that could react to form other species or particles, and NO_2 as a reactive pollutants that could be formed. To this classification, we can add O_3 which is not directly emitted by road traffic, but which is correlated to this as its formation in the troposphere depends on NO_x and COVs concentrations. However, this classification might be incomplete, as UFP can coagulate to form bigger particles, we could add particulate matter to the group of reactive pollutants that could be formed. Moreover, the qualifying “inert” does not mean that pollutants never react, but are inert at this local scale: their time of residence is higher than the time of transport to go outside the zones studied at the road scale. This classification is summed up in Table 1.

Table 1. First classification of pollutions

1 st classification				
Pollution of inert pollutants	Benzene	CO	BS/BC	PM in mass
Pollution of reactive pollutants that form other pollutants	NO	UFP		
Pollution of reactive pollutants and are formed by other pollutants	NO_2	PM		
Pollution of not traffic emitted pollutant, formed by other pollutants	O_3			

Although Patton et al. (2014) concluded that decrease gradients with distance can not be generalized, as concentrations might be highly variable in space, the review of studies that focused on concentrations gradients with distance to the road enables some useful insights, and a second classification: “near-road pollutants” and persistent pollutants with distance (local scale).

On one hand, according to Karner et al. (2009), we can identify that some pollutants as CO, NO, NO_x, and UFP are pollutants for which concentrations gradients with distance to the road are clear and reach 50% of the road concentrations within the first 150m and keep decreasing afterward. Thus, considering a major road in a urban environment without industrial sources, those pollutants can be associated to a pollution of “near-road pollutants” for which concentrations depend on the distance to the road, and reach background within 100m to 400m from a major road in downwind conditions. However, beyond NO_x, NO₂ concentrations declines are not as fast, because of its formation in the atmosphere with oxidation of NO, which then suffers a sharper decrease (Beckerman et al. 2008). Thus, NO₂ can not be considered only as a near-road pollutant.

On the other hand, although concentrations of benzene (Barros et al. 2013), PM₁₀ (Roorda-Knape et al. 1998), PM_{2.5} (Tiitta et al. 2002) and EC (Padró-Martínez et al. 2012) are observed higher in the vicinity of roadways, their concentrations decrease less rapidly than preceding pollutants, and no gradients are identifiable for PM₁₀. PM₁₀ and PM_{2.5} are more affected by regional sources, besides UFP particles can coagulate to form bigger particles. Thus, PM₁₀ and PM_{2.5} can not be identified as “near-road” pollutants, and are considered as a pollution of urban background. Moreover, although the main sources for benzene in urban areas are petrol cars, previous studies did not show an obvious trend for its decrease. Concerning EC, Karner et al. (2009) observed a decline in the first 100m but the concentrations level off at more than 60% of the road concentrations. This second classification is summarized in Table 2. Each pollutant considered above is associated with a gradient observed or not according to previous studies, for instance no gradient was found for PM₁₀ concentrations, but all other pollutants have gradients of decrease. Then, pollutants are associated to the distance at which they level off at background concentrations, and finally the distance at which they reach 50% of on-road concentrations. That means that no associated gradient or 50% of on-road concentrations were observed. This classification relies on mentioned studies about distance decrease concentrations to major roads, and shows approximate distances, that have to be taken as indications to compare pollutant generated by road traffic.

Table 2. Second classification of pollutions

2nd classification	Pollutant	Gradient	Distance at which background concentrations are reached	Distance at which 50% of on-road concentrations are reached
Pollution of “near-road” pollutants	CO	Yes	<400m	~50m
	NO	Yes	<400m	~60m
	NO _x	Yes	<400m	~100m
	UFP	Yes	<400m	~150m
Pollution of non “near-road”/urban background pollutants	NO ₂	Yes	>400m	-
	PM ₁₀	No	-	-
	PM _{2.5}	Yes	>400m	-
	EC	Yes	>400m	-

This review permits two classifications of pollutions at a road scale. We define a first classification according to chemical specificities: inert or reactive pollutants and a second classification of near-road and persistent pollutants with distance. Next, the review enables the identification of important factors influencing pattern of pollutant concentrations: concentrations in vicinity of major roadways are directly influenced by the road traffic, but their background concentrations, meteorological conditions and road geometries appears to be essential.

4.2. Road structures affecting dispersion of air pollutants from traffic

The impacts of road geometry and road side barriers are important on dispersion of pollutants emitted by road traffic. According to Ning et al. (2010) and Finn et al. (2010), the presence of roadside barrier greatly changes particle and other pollutants dispersion. Concentrations are lower just behind the barrier, and then grow with distances to maxima, and background concentrations are reached farther than without barrier (about 400m away from the road, against 150 to 200m without barrier according to Ning et al. (2010) experiments). However, just behind the barrier, concentrations appear to be lower than without, indeed (Finn et al. 2010) found that levels are 50% lower or more just behind the barrier. But, in contrast, with computational fluid dynamics simulations, Hagler et al. (2011) found that barrier decreases pollutant levels behind it whatever the distance is, compared to the situation without barrier. But concentrations on-road increased, and maximum concentrations were reached in the top of the barrier. The difference for concentrations behind the barrier might be explained by the model which does not take into account buildings or trees affecting air flow, or the coagulation of particles.

In conclusion, road-side barriers affect dramatically pollutant dispersion. And although, concentrations just behind the barrier are lower than without one, concentrations and exposure on the road and farther to the barrier are affected.

Concerning road geometries, according to Baldauf et al. (2013) and Steffens et al. (2014), geometries affect dispersion of pollutants and concentrations on-road, and near-road areas. In the same way, Rakowska et al. (2014), Van Dingenen et al. (2004) or Liu et al. (2005) showed that street canyons are responsible for high level of pollutant concentrations due to a limited dispersion by buildings.

Thus, the assessment of concentrations and population exposure on-road and near-road must consider the presence of roadside barriers and how the road is in function of the surrounding field. In Table 3 we summarized diverse configurations of road discussed above: at-grade road, up-grade, street canyon, and the presence of a side barrier. The at-grade road is taken as the reference case, then the qualitative effects are function of the open at-grade road, with no side-barrier. We tried to indicate pollution that is affected by each case, however, in several cases, we coped with uncertainties: in street canyons, are reactive pollutants producing secondary pollutants? Our review did not focus on that question, and further review should bring elements to answer that question.

Table 3. Road configurations, effects and pollutants considered

Road configuration	Effects	Pollutions affected
At-grade road	On road:	
	General case: high concentrations	All traffic pollutants (except PM ₁₀)
	Downwind:	
	Concentrations decrease with distance	
Up-grade road	Sharp gradients	Inert pollutants/reactive pollutants that produce other pollutants
	Gradual gradients	Urban background pollutants/reactive pollutants that are produced by other pollutants
	On road:	
Street canyon	Concentrations lower than general case	All traffic pollutants (except PM ₁₀)
	Downwind:	
	On the ground: lower concentrations	Inert pollutants
Side barrier	At road level: higher concentrations	Inert pollutants
	On road:	
Street canyon	Concentrations higher	All traffic pollutants (except PM ₁₀) (reactive pollutants?)
	On road:	
Side barrier	Concentrations are higher	All traffic pollutants

Downwind:	
At the height of the barrier: higher concentrations	All traffic pollutants
Just behind the barrier: low concentrations, cavity	Near-road pollutants
Distance increasing: concentrations levels up to a maximum	Near-road pollutants

Some configurations as the presence of tunnels do not have been studied here, but one could add to this table some configurations in order to improve their assessment. Moreover, an improvement of this classification will be an addition of parameters, as height of building and road width for street canyons or the height of side barriers etc. to better characterize effects of each configuration.

4.3. Meteorology affecting air pollution from road traffic

In an assessment study about concentrations of pollutants related to the traffic of a road as major source, among other meteorological factors we should define the wind conditions: direction of wind with regard to the road and its speed.

In general, gradients of concentrations with distance are different on upwind and downwind sides of the road. Gradients of concentrations are less obvious (Roorda-Knape et al. 1998) or inexistent (Hitchins et al. 2000) in upwind conditions. In upwind conditions, concentrations reach background concentrations nearer to the road than in downwind conditions. Thus, areas that are most of the time upwind to the road should be less exposed to concentrations of pollutants specifically due to the traffic on this road. Hence the identification of the wind profiles of direction would provide information on the location of the area with the most important concentrations of air pollutant from traffic of a major road.

However, this must be tempered according to atmospheric conditions and wind speed. During low wind speed condition and highly stabilized atmosphere, horizontal dispersion and vertical mixing are less important and concentrations will be high either the roadside (Padró-Martínez et al. 2012). Seasons are responsible for seasonal variations of concentrations, indeed Zhang et al. (2015) and Miao et al. (2015), showed that seasons are affecting boundary layer height and conditions of stabilized atmosphere often occur during winter (cold months) and pre-sunrise hours (Hu et al. 2009).

In Table 4, we identify important meteorological parameters that influence pollution concentrations.

Table 4. Meteorological parameter and effects on pollution

Meteorology	Effects
Wind context: direction and intensity	Horizontal dispersion
Precipitations: frequency	Leaching of pollutants
Temperature/sunlight	Physico-chemical reactions
Boundary layer: height	Vertical mixing
Seasons	Cover previous elements

This table enables the identification of several meteorological parameters, and we will lead to further research to identify more specific aspects, and qualitative data for these parameters to improve our understanding of the influence of meteorology on dispersion, and thus to understand which parameters are most influencing the dispersion, taking into account the local context as the presence of a mountain range or sea etc.

4.4. Background and external sources of air pollution

Next, the area of impact of pollutant concentrations is highly dependent on background concentrations. Background concentrations are depending on the time of atmospheric transport and the time of atmospheric residence of the species. In addition, background concentrations are affected by regional intakes. For instance Beekmann et al. (2015) estimated that fine particle matter concentrations in Paris might be due to about 70% from regional transport of pollutants. Moreover, as Dimitriou and Kassomenos (2013), Dimitriou and Kassomenos (2014) and Zhang et al. (2015) suggested, natural or anthropogenic particles and pollutants are transported by regional flows. Thus the presence of sea, or mountains, influences the circulation of flows, and natural sources as sea breeze, deserts have regional impacts on urban areas air quality (Sahara desert impacts southern Europe cities), as industries or agriculture.

This highlights that each cities have a context which depends on its background, potential external sources and regional intakes of air pollution. Thus we need to dissociate city or areas which are highly influenced by background concentrations and external sources, from those which have a low background context.

4.5. How these previous considerations should affect population exposure to traffic related air pollutants and its assessment?

Concentrations around roadways are influenced by road traffic, background concentrations, and number of parameters discussed above. And population exposure to these concentrations is depending on the time spent in each microenvironment: transport, street, home, office... The assessment of on-road and near-road concentrations contributes to understand exposure concerning vehicle drivers, users of public transport, pedestrians and cyclists and also residents and workers in the immediate surroundings. So the density of population is an important factor. However, we must remember that a low density will probably generate more car use, and then a higher number of kilometers traveled, and potentially to higher emissions. We start with the general case for which, the higher the distance to the road is, the lower the concentrations are until they reach background concentrations. In the most exposed side of the road (downwind), concentrations reach maxima. Thus, according to the review it is obvious that exposure to CO, NO_x, UFP is high on-road, and also within the first 150m of a major way. Thus, pedestrians, cyclists and drivers are directly exposed to the sources, and the only ways to avoid high level of exposure is either reduce the time spent on-road (by pulling pedestrians or cyclists away), or reduce traffic emissions, or for drivers to install advanced filtration systems. In order to reduce traffic emissions, many measures might be set up, however they must be assessed to know if they are really efficient, if they could have reverse effects on other areas. Concerning the workers and resident exposure, the first option is to avoid the construction of buildings in road vicinity, and the second option, as for drivers, is the setting up of filtration systems. However this solution can not be efficient for populations which do not spend all time indoor, as for schools for instance.

But, results from Zhou et Levy (2008) show that the vertical dispersion of pollutants can follow an exponential decay, thus the populations in high floor level will be less exposed to concentrations of pollutants from outdoor. Thus a repartition according to activities (office workers, residents) in function of the average time spent could minimize total exposure.

Another teaching from the review is that specific mitigation measures should be set up in cities in function of local characteristics, as the topography that affects dispersion. Moreover these measures could depend on the season, but this would imply a specific assessment according to the season to show if expected effects of a measure would be as significant in winter and summer for instance. Thus, the assessment of urban areas air pollution should firstly take into account a local context which includes topography, meteorology, background concentrations, and external intake of pollution. In a second time, the different pollutions identified must be analyzed separately. And finally, in several cases, important local pollutions must be assessed, and then street configurations, the presence of obstacle that limits dispersion etc.

5. Conclusion

This review focused on the identification of parameters affecting the variability of traffic related air pollutant concentrations. It enables the identification of a classification of pollutions as pollutions of “inert” and “reactive” pollutants (Ying Zhou et Levy 2007). Then, by reviewing studies focusing on decreasing gradients of concentrations with distance, we suggested another classification taking into account their pattern with distance to the road. Hence we identified CO, NO, NO_x and UFP as “near-road” pollutants, and PM₁₀, PM_{2.5}, and EC as persistent pollutants with distance to the road at the local scale.

Then we identified that road structures and side-barriers are directly affecting dispersion, thus on-road and near-road concentrations of pollutants. We did not focus on the effects of vegetation on concentrations and dispersion, however, according to Gromke et Blocken (2015), Hagler et al. (2012) and Steffens et al. (2012), vegetation will change the flow, and can limit dispersion of concentrations. Thus, further efforts to better understand the interactions of built environment and natural surroundings with pollutants concentrations are needed and could lead to the creation of indicators for roads including parameters as width, building heights, presence of side barriers, road level (at-grade, up-grade...), and the addition of some road configurations as the presence of tunnels would lead to better understanding their impacts on near-road air quality. The creation of indicators, and the classification of road configurations could improve the assessment of local air pollution and the impact on population exposure, by identifying infrastructure factors that impact them most.

Besides, as Beekmann et al. (2015) concluded, megacities are only partially responsible for their own concentration levels and this conclusion implies that regulation policies might not be as effective that wanted. Indeed, topography, meteorological and seasonal variations have important impacts on transport and dispersion. Thus, the air pollution assessment of cities with identifiable meteorological parameters that are highly depending on seasons and with topographic characteristics should include those data when considering mitigation measures. Then, build up classification characterizing cities and meteorology appears to be necessary. Those classifications would take into account topography, potential regional intakes, industries, residential wood burning, winter sanding or salting of roads, and beyond local accurate car fleet characteristics, the heavy good vehicles or diesel shares (Wu et al 2015, and Zhu et al 2002). Further works should bring some qualitative elements about meteorological parameters impacting the temporal variability of air pollution. The classifications of geographical contexts and meteorological parameters could allow us to study a city, and by knowing its context, taking into account the elements that are necessary to assess the impacts of road traffic on air pollution.

However this review has several limits. Firstly, we did not analyzed findings about O₃, because even if it represents a major issue on air pollution, it is a secondary pollutant, not directly emitted by exhaust, and its chemistry is quite complex, but produced in the troposphere from the oxidation of volatile organic compounds by OH in the presence of NO_x. Thus it follows a reverse pattern: its concentrations increase with distance to roads. But, its concentrations are also highly dependent with season (Zhang et al. 2015). Secondly, concerning population exposure in buildings, more specific studies about indoor/outdoor exchanges were not considered, but would provide useful clues to better understand inside exposure. And finally, as Zhu et al (2002) mentioned it, fleet composition can induce differences in emissions and thus on near-road air pollution. Indeed, Carter et al (2014) fleet composition can be highly variable locally, and then the local characterization of the fleet composition can be necessary to correctly evaluate the impacts of road traffic on local air quality.

To conclude, air pollutants from road traffic impact air quality at several scales of space and time, because of the physical and chemical processes that are affecting them in the atmosphere. A future direction is that the knowledge of those scales could improve the assessment of the impacts of policies set up to reduce air pollution from traffic in urban areas, because these mitigation measures are themselves enforced at several scales. Thus, their potential of effects (as the emissions, or the air quality and population health) will depend on those scales.

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