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Impact of passenger car NO$_X$ emissions on urban NO$_2$ pollution – Scenario analysis for 8 European cities

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ABSTRACT

Residents of large European cities are exposed to NO$_2$ concentrations that often exceed the established air quality standards. Diesel cars have been identified as a major contributor to this situation; yet, it remains unclear to which levels the NO$_X$ emissions of diesel cars have to decrease to effectively mitigate urban NO$_2$ pollution across Europe. Here, we take a continental perspective and model urban NO$_2$ pollution in a generic street canyon of 8 major European cities for various NO$_X$ emission scenarios. We find that a reduction in the on-road NO$_X$ emissions of diesel cars to the Euro 6 level can in general decrease the regional and urban NO$_2$ concentrations and thereby the frequency of exceedances of the NO$_2$ air quality standard. High NO$_2$ fractions in the NO$_X$ emissions of diesel cars tend to increase the urban NO$_2$ concentrations only in proximity of intense road traffic typically found on artery roads in large cities like Paris and London. In cities with a low share of diesel cars in the vehicle fleet such as Athens or a high contribution from the NO$_2$ background to the urban NO$_2$ pollution such as Krakow, measures addressing heavy-duty vehicles, and the manufacturing, energy, and mining industry are necessary to decrease urban air pollution. We regard our model results as robust albeit subject to uncertainty resulting from the application of a generic street layout. With small modifications in the input parameters, our model could be used to assess the impact of NO$_X$ emissions from road transport on NO$_2$ air pollution in any European city.

1. Introduction

Residents of many larger European cities are still exposed to NO$_2$ concentrations that often exceed the established air quality standards (EEA, 2015). Most exceedances occur in city centres, mainly caused by traffic-related NO$_X$ (nitrogen oxides) emissions originating from diesel cars. Policy makers expected the NO$_X$ emissions of diesel cars to decrease considerably when introducing more stringent Euro 5 and Euro 6a/b emission limits in 2009 and 2014 (EC, 2012, 2008a,b). This expectation, however, has not materialized. Tests with Portable Emissions Measurement Systems (PEMS) suggest that Euro 5 and 6a/b diesels, just as Euro 3 and 4 diesels before, emit on the road several times more NO$_X$ than permitted by the applicable limit (Weiss et al., 2012; Franco et al., 2014; Kadijk et al., 2015). Also tests carried out by the national type approval authorities in the aftermath of the dieselgate scandal have shown that on the road Euro 5 and 6a/b cars emit on average five times more NO$_X$ than their respective limits of 180 and 80 mg/km (Degraeuwe and Weiss, 2017; Transport and Environment, 2016). In

view of the persistent air quality problems, the so-called Real-Driving Emissions (RDE) on-road test procedure with PEMS was introduced in the Euro 6 emission legislation (EC, 2017, 2016). Euro 6c introduces RDE for monitoring only without quantitative requirements for all new vehicles form 1/9/2018. Euro 6d-TEMP introduces binding RDE limits with a temporary conformity factor (CF) of 2.1. This means that NO$_X$ emissions measured on the road have to be below 80 × 2.1 = 168 mg/km. From 1/9/2017 this limit applies to new types and from 1/9/2019 to all new vehicles. Regulation EU2017/1151 (EC, 2017) specifies that the WLTC (Worldwide harmonized Light vehicles Test Cycle) replaces the NEDC for emission testing on the test bench from 1/9/2017 for new types and from 1/9/2018 for all new vehicles. With Euro 6d the CF will be lowered to 1 plus measurement uncertainty. Initially the uncertainty is set at 0.5 but will be revised annually. This means the NO$_x$ limit will be 120 g/km. Euro 6d will be introduced on 1/1/2020 for new types and in 1/1/2021 for new vehicles. For on road NO$_x$ emissions three subclasses of Euro 6 are important: Euro 6a-c, Euro 6d-TEMP and Euro 6d.

In spite of these provisions, it yet remains unclear whether the implemented conformity factors will effectively mitigate urban NO_2 (nitrogen dioxide) pollution (EEA, 2013). Degraeuwe et al. (2016) found in a case study for Antwerp (Belgium) that exceedances of the NO_2 air quality standard can be reduced substantially if diesel cars emitted on the road less than 80 mg NO_X/km. However, a more comprehensive assessment that takes a continental perspective and analyses the impact of the RDE conformity factors on the air quality in major European cities is still missing. Here, we address this knowledge gap by modelling urban NO_2 pollution for various NO_X emission scenarios. We pursue two objectives:

- analysing the reduction in urban NO_2 pollution resulting from a decrease in the distance-specific NO_X emissions of Euro 6 diesel cars;
- assessing whether a dedicated NO_2 emissions limit, applied next to the existing NO_X limit, could help reducing urban NO_2 pollution more effectively.

To this end, we generalise the air quality model developed by Degraeuwe et al. (2016) and apply it in a scenario analysis to 8 European cities. The outcome can assist the annual revision of RDE conformity factors and provide scientists with a simplified, yet accurate, tool to model NO_2 pollution for any city in Europe. The article continues in Section 2 with an explanation of methods and data sources. We present and discuss the results in Sections 3 and 4. The article finishes with conclusions for scientists and policy makers in Section 5.

2. Methods

2.1. General overview

Our modelling of urban NO_2 pollution is based on the method applied by Degraeuwe et al. (2016). We briefly explain this method in Section 2.2 and assess in a comparative case study for Antwerp (Belgium) and Milan (Italy) the sensitivity of the calculated NO_2 concentrations to key input parameters (Section 2.3). This assessment provides scope and justification for the introduction of model simplifications that allow us to apply the method to 8 European cities. Modelling of NO_2 pollution at the scale of a city requires large amounts of data; compiling and processing these within this research for a larger number of cities is unfeasible. We explain the simplified modelling approach applied here in Section 2.4.

2.2. Overview of the NO_2 pollution model applied to Antwerp

The NO_2 air pollution modelling of Degraeuwe et al. (2016) uses the NO_X emissions given for the year 2009 by the latest version of the TNO-MACC-II emissions inventory (Kuenen et al., 2014). These 2009 NO_X emissions constitute the baseline emissions scenario for subsequent pollution modelling. In a second step, nine emission scenarios for road vehicles are defined, assuming a situation where virtually the whole light-duty and heavy-duty vehicle fleet consists of Euro 6/VI-compliant vehicles as it may be the case around the year 2030. The vehicle emission scenarios include (i) two synthetic worst-case scenarios that assume both high NO_X emissions and high NO_2 fractions, (ii) a best-case scenario that assumes a complete fleet shift to gasoline cars and (iii) six scenarios with different combinations of NO_X emission factors and NO_2 fractions (see 2nd column of Table 3). The emission factors cover the whole spectrum of Euro 6, from a to d. The assumed emission factors are based on the following information:

(i) for light-duty diesel vehicles, on-road emission measurements as published by Franco et al., 2014); May et al., 2014; Vlachos et al., 2015; Weiss et al., 2012 and the RDE legislation (EC, 2017).

(ii) for heavy-duty vehicles, on-road compliance with the Euro VI limit was assumed, as suggested by the measurements of Vermeulen et al.

(2012).

The gradual implementation of Euro 6, from Euro 6a to d, is not modelled. The scenarios have to be considered as an asymptotical situation in which the major part of the fleet complies with Euro 6d. Hence also the year 2030 is indicative. Another study for Germany (Toenges-Schuller et al., 2016) takes the fleet evolution into account which allows to estimate when the NO_2 limit will be reached.

The future vehicle activity and the share of diesel cars in the vehicle fleet are retrieved from the TREMOVE model (Breemersch et al., 2010). By combining data on the distance-specific NO_X emissions of light- and heavy-duty vehicles, vehicle activity, the share of diesel cars in the vehicle fleet, the overall NO_X emissions from road transport for all European countries are modelled. The resulting emission data are then used as input for the chemical transport model LOTOS-EUROS (Schaap et al., 2008). A run at 28 × 28 km resolution for the entire European continent was done for the baseline and the nine NO_X emission scenarios. This run is then complemented by more detailed modelling at a 7 × 7 km resolution for the Benelux region that is used to establish the background NO_2 pollution level of Antwerp. To model the NO_2 pollution at high resolution in the city of Antwerp the IFDM (Immission Frequency Distribution Model) is used. IFDM is a bi-Gaussian plume model developed by Lefebvre et al. (2011). The model uses both the urban background concentration from LOTOS-EUROS (Schaap et al., 2008) and the NO_X emissions at street level to produce a high resolution concentration map for the city. For detailed street canyon calculations, IFDM incorporates the OSPM Model (Berkowicz et al., 1997; Ketzel et al., 2012). The latter uses the rooftop concentrations determined by IFDM, street canyon geometry, and the NO_X emissions in the canyon to calculate both hourly and annual average NO_2 concentrations in the specific street canyon under consideration.

2.3. Case study on the sensitivity of modelled NO_2 concentrations

The method described above requires detailed city-specific input on, e.g., the NO_2 background concentration, wind speed, temperature, building heights, street widths, fleet composition, and traffic intensity. To apply this model to other cities requires either collecting the necessary data for each individual case or applying generic assumptions. The latter approach reduces the demand for data and resources but inevitably introduces uncertainty into the NO_2 air pollution modelling. We consider it unfeasible, to collect within the scope of our research data, e.g., on building heights, street widths, and traffic intensities for a large number of European cities. We therefore assess via a case study for the cities of Antwerp and Milan the scope to simplify modelling parameters and based on this develop and apply generic assumptions to other cities. To this end, the following parameters are considered to influence the modelled NO_2 concentrations at the street canyon level: (i) hourly wind speed, (ii) hourly temperature, (iii) latitude dependent solar irradiation, (iv) the composition of the vehicle fleet and (v) background NO_2 concentration. We chose Milan for this case study because the conditions in this city differ considerably from those in Antwerp. In fact, the difference in the conditions between both cities appears to cover large parts of the wide range of climatic conditions and fleet compositions present in Europe:

- The average annual wind speeds in Antwerp and Milan are 6.5 km/h and 2.4 km/h, respectively, covering approximately the range of wind speeds observed in Europe. The North Sea coast is one of the windiest places in Europe (only Scotland is windier); the Po Valley has approximately the lowest wind speeds in Europe (DTU Wind Energy, 2015).
- The share of diesel cars in the passenger car fleet in Antwerp is very high (85%) while it is near the European average in Milan (64%).
- The urban NO_2 background concentrations in Antwerp are average (annual average NO_2 of 33 µg/m^3 in Schoten near Antwerp in 2009),

Table 1

Annual average NO$_2$ background concentrations and meteorological variables for 8 European cities; concentrations and meteorological data obtained from the LOTOS-EUROS model (Schaap et al., 2008) for the reference year 2009; percentage of kilometres driven with diesel cars from TREMOVE (Breemersch et al., 2010).

City	Annual average urban background concentrations of pollutants				Meteorological conditions		Percentage of passenger car kilometres driven by diesel cars
	NO$_2$ [µg/m^3]	O$_3$ [µg/m^3]	PM$_{2.5}$ [µg/m^3]	PM$_{10}$ [µg/m^3]	Wind speed [m/s]	Temperature [°C]	
Antwerp	28	38	10	15	6.5	11	85
Athens	19	59	11	20	4.9	17	9
Barcelona	43	38	12	22	3.6	16	77
Krakow	13	47	9	12	4.5	10	41
London	33	33	8	13	6.0	9	44
Milano	42	38	15	21	2.4	15	63
Paris	40	34	20	44	5.3	11	74
Stockholm	8	52	4	5	5.1	6	26

Table 2

Disaggregating the difference in air quality index for a generic street canyon in Antwerp and Milan into the effects of weather (i.e. combination of wind, temperature and latitude), vehicle fleet composition and background pollution. The first row for each air quality index represents the reference concentration in Antwerp, while the last rows represent the concentration under Milan conditions. The intermediate rows show the concentration corresponding to simulations performed with the Antwerp reference case but adapted for the conditions in Milan for the specific boundary condition.

	NO$_2$ (µg/m^3)		O$_3$ (µg/m^3)	
	ann. avg.	99.79%tile	ann. avg.	93.15%tile
Antwerp	**39**	**127**	**29**	**68**
Antwerp with parameter settings for Milan				
Weather	53 (+15)	195 (+68)	21 (−8)	54 (−13)
- Wind	54 (+15)	195 (+68)	22 (−8)	55 (−12)
- Latitude	39 (0)	130 (+2)	29 (+0)	67 (−0.5)
- Temperature	39 (−0.3)	127 (−0.6)	29 (−0.6)	67 (−0.5)
Fleet composition	37 (−1.4)	110 (−17)	29 (+0.2)	69 (+1.2)
Background NO$_2$	51 (+12)	145 (+18)	42 (+13)	123 (+55)
Milan	**62 (+23)**	**182 (+54)**	**34 (+5)**	**102 (+34)**

whereas those in Milan (annual average of 48 µg/m^3 in Milan Lambro in 2009) rank among the highest found in Europe.

We conduct a total of 7 model runs; 5 runs altering the 5 parameters individually, one run with the 3 weather parameters together (wind, temperature, latitude), and a final simulation run with the Antwerp street configuration and the climatic conditions, vehicle fleet composition, and NO$_2$ background concentration of Milan. As the results of our sensitivity study suggest, model simplifications are feasible (see Section 3.1). The generic assumptions made to estimate the NO$_2$ air pollution in 8 European cities are explained next.

2.4. Simplified modelling of NO$_2$ pollution in 8 European cities

We determine the effect of diesel car NO$_X$ emissions and their NO$_2$

fractions on the NO$_2$ air pollution in the following 8 cities: Antwerp, Athens, Barcelona, Krakow, London, Milano, Paris, and Stockholm. The chosen cities (i) comprise major metropolitan areas with a large population, (ii) resemble a wide range of geographic and climatic conditions across Europe, and (iii) are frequently exposed to elevated NO$_2$ pollution levels (EEA, 2013).

We base our modelling, on data for the local vehicle fleet composition as obtained from the TREMOVE model (Breemersch et al., 2010). The local weather conditions (i.e., wind and temperature) are obtained from the LOTOS-EUROS simulations (Schaap et al., 2008). Likewise, the NO$_2$ background concentration for each city and emission scenario is obtained from the LOTOS-EUROS simulations (see Section 2.2). Besides the Europe-wide simulation with a 28 × 28 km resolution, we conduct for three regions (i.e., Benelux, Po-Valley and South Poland) simulations at a 7 × 7 km resolution. These three regions are chosen because they constitute hot spots of high NO$_2$ air pollution. For the cities located in one of these three areas the 7 × 7 km runs are used, for the others the 28 × 28 km runs. We calculate the NO$_2$ concentration in a generic street canyon for each of the 8 cities, by applying the IFDM (Lefebvre et al., 2011) and OSPM (Berkowicz et al., 1997; Ketzel et al., 2012) models.

Following the outcome of our sensitivity study (see Section 3.1), we will use the street grid of Antwerp combined with the local boundary conditions (Table 1) for all 8 cities. This simplification is justified because a street grid represents a generic collection of street canyons with different orientations, building heights and traffic intensities rather than the precise street map of a city. The assumption of a generic street grid allows controlling for the effect of city geometry on the dispersion of NO$_2$ pollution for the range of assumed emission scenarios under a location-specific set of boundary conditions. Yet, the simplification also comes with a loss of accuracy which we discuss in Section 4.

For each city, we model at high-resolution the NO$_2$ pollution in a typical street canyon with a width of 50 m and a building height of 18 m. The generic canyon has a two-way single carriageway with 2 drive lanes per direction. The orientation is east-west. We assume a

Table 3

NO$_2$ concentration (µg/m^3) in a generic street canyon with boundary conditions of 8 European cities for the baseline and nine NO$_X$ emission scenarios; percentages in parentheses indicate the change in NO$_2$ concentrations relative to the baseline scenario.

Scenario	Diesel car EF/fNO$_2$ (mg/km, %)	vkm year	Antwerp	Athens	Barcelona	Krakow	London	Milan	Paris	Stockholm
1 (base)	EF (%NO$_2$)	2010	39	42	56	41	57	62	38	22
2	350 (35%)	2030	30 (−23%)	27 (−36%)	47 (−17%)	33 (−20%)	47 (−18%)	47 (−24%)	30 (−22%)	14 (−35%)
3	100 (50%)	2010	22 (−44%)	25 (−40%)	32 (−42%)	27 (−34%)	40 (−31%)	35 (−44%)	21 (−45%)	11 (−53%)
4	200 (70%)	2010	25 (−35%)	26 (−37%)	37 (−33%)	29 (−29%)	43 (−25%)	40 (−35%)	24 (−35%)	12 (−46%)
5	31 (2%)	2030	20 (−48%)	25 (−39%)	31 (−45%)	27 (−35%)	38 (−33%)	33 (−47%)	19 (−49%)	10 (−53%)
6	450 (5%)	2010	28 (−27%)	26 (−38%)	41 (−26%)	31 (−25%)	45 (−22%)	45 (−28%)	28 (−27%)	13 (−40%)
7	550 (20%)	2030	34 (−14%)	27 (−34%)	52 (−7%)	35 (−15%)	50 (−13%)	53 (−15%)	34 (−11%)	17 (−25%)
8	550 (50%)	2030	36 (−8%)	27 (−34%)	57 (3%)	37 (−11%)	53 (−8%)	57 (−9%)	37 (−3%)	17 (−24%)
9	550 (70%)	2010	33 (−14%)	26 (−37%)	49 (−11%)	34 (−18%)	50 (−13%)	53 (−15%)	33 (−12%)	15 (−34%)
10	550 (70%)	2030	37 (−4%)	27 (−34%)	60 (8%)	38 (−8%)	54 (−5%)	59 (−5%)	38 (1%)	17 (−23%)

peak-hour traffic of 870 vehicles per hour per lane, which is close to the maximum hourly capacity of a lane of around 1000 vehicles (UKHA, 1999). It is representative of a busy access road to a city centre. For this canyon, we likewise apply the ten emission scenarios described in paragraph 2.2. We compare the modelled background NO_2 and O_3 concentrations with the measured ones for all 8 cities. For the baseline year 2009, we establish a correlation between the modelled and measured daily average concentrations. Based on this correlation, we correct our modelled NO_2 concentrations of the various emission scenarios. We continue in the next section by presenting the results of our sensitivity study and the findings for the NO_2 background and street canyon concentrations in the 8 selected cities.

3. Results

3.1. Results – sensitivity study for Antwerp and Milan

By applying the set of boundary conditions (i.e., climate, NO_2 background concentration, and composition of the vehicle fleet) from Milan to the street grid of Antwerp, we find that the annual average NO_2 concentration in Milan exceeds the one in Antwerp by 23 μg/m3 (Table 2). The low wind speed and the high NO_2 background concentration each increase the NO_2 concentration by 15 μg/m^3 and 12 μg/m^3, respectively. In fact the frequently stagnant air in Milan causes both higher NO_2 background concentrations and higher NO_2 concentrations in street canyons compared to Antwerp. By contrast, the higher average temperature in Milan (14.7 °C) compared to Antwerp (10.8 °C) has virtually no effect on the average NO_2 concentrations. Likewise, the lower share of 63% versus 85% of passenger vehicle kilometres driven by Diesel cars in Milan compared to Antwerp only results in a 1 μg/m^3 reduction in ambient NO_2 concentrations. This seems a small effect but it is worth noting that the background is heavily influenced by the diesel share. Likewise, The 99.79[th] percentile of hourly NO_2 concentrations in both cities, which is a measure for the exceedance of the hourly limit of 200 μg/m^3, show the same behaviour as the annual average NO_2 concentrations in both cities (Table 2). The annual average ozone concentration in Milan is 5 μg/m^3 higher than in Antwerp. Two compensating effects cause this: The comparatively low wind speed in Milan causes higher NO concentrations that, in first instance, reduce the ozone concentrations by 8 μg/m^3. However, this effect is compensated by a background ozone concentration that is higher in Milan than in Antwerp, which leads to an increase in the ozone concentration of 13 μg/m^3. Based on these findings, we conclude the most influential parameters are the wind speed, the background concentration and the vehicle fleet.

3.2. Results – NO_2 pollution in 8 European cities

3.2.1. Impact of scenarios on the urban NO_2 background concentration

The background urban NO_2 concentration decreases with a reduction in NO_X emissions per-km from diesel cars. The example of the Po Valley suggests a decrease of up to 15 μg/m^3 if the NO_X emissions of diesel cars decreased from 550 to 100 mg NO_X/km (Fig. 1). This impact is not limited to big cities only but extends over large areas (see also Degraeuwe et al., 2016). The decrease in NO_X emissions also causes a decrease of the 93rd percentile of the O_3 maximum daily 8-h mean ozone concentration. The limit value of 120 μg/m^3 is exceeded mostly around the Mediterranean (EEA, 2015). E.g. scenario 5 (shift to gasoline) gives an improvement of about 20 μg/m^3 in the Po Valley where values above 140 μg/m^3 are common.

Likewise, an increase in the NO_2 fraction of the NO_X emissions of diesel cars can increase the regional NO_2 background concentrations. A comparison between Scenarios 7 and 10 (NO_X emissions of 550 mg/km but different NO_2 fractions of 20% and 70%, respectively) suggest the annual average NO_2 concentration can increase by up to 8 μg/m^3 when the NO_2 fraction in the NO_X emissions is increased from 20 to 70%
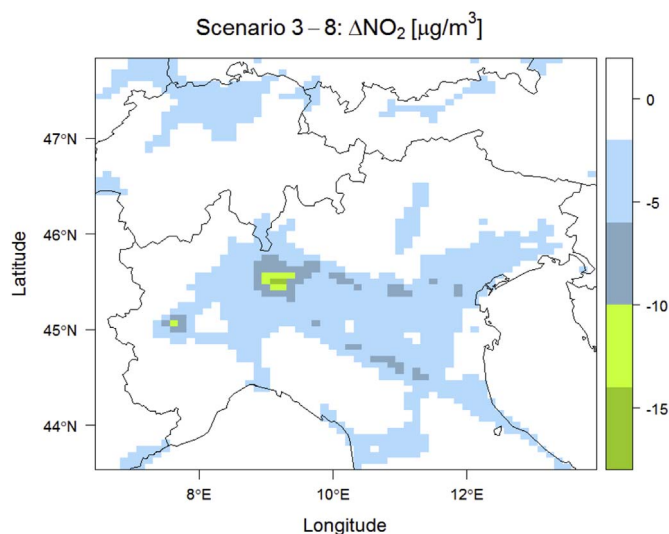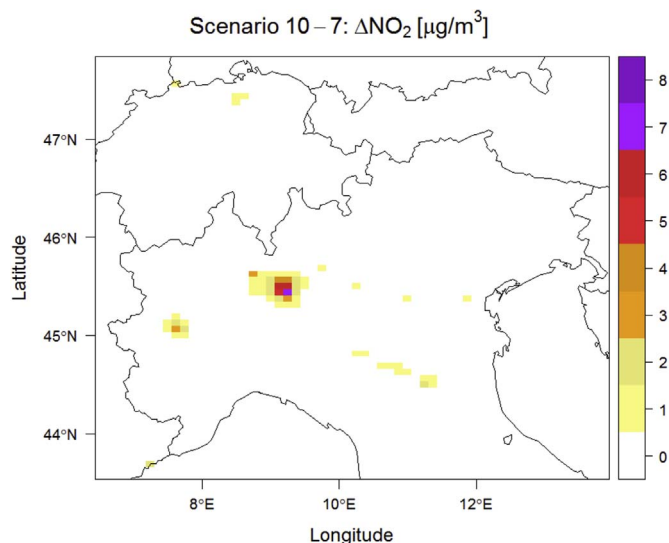

Fig. 1. Impact of a change in the NO_X emission reduction on annual NO_2 background concentrations in the Po Valley zoom area (Scenario 8: 550 mgNO$_X$/km, f-NO$_2$ = 50%, Scenario 3: 100 mgNO$_X$/km, f-NO$_2$ = 50%).

Fig. 2. Impact of a change in the f-NO$_2$ emission ratio on annual NO_2 background concentrations in the Po Valley zoom area (Scenario 10: 550 mgNO$_X$/km, f-NO$_2$ = 70%, Scenario 7: 550 mgNO$_X$/km, f-NO$_2$ = 20%).

(Fig. 2). Around big urban areas (Milan and Turin) a decrease in the NO_2 fraction also causes the ozone concentrations (annual average and 93rd percentile) to decrease. Contrary to the effect of the NO_X emissions, the effect of the NO_2 fraction is more localized around the big emission sources.

3.3. Urban NO_2 concentrations at street canyon level

The NO_2 concentrations in the generic street canyon vary substantially between the 8 cities for the various scenarios (Table 3). In the baseline scenario, Milan (62 μg/m^3), London (57 μg/m^3), and Barcelona (56 μg/m^3) show the highest annual average NO_2 concentrations. These cities also show the highest background NO_2 concentrations (Table 4). The behaviour of the NO_2 concentrations (annual average and the 99.79[th] percentile as indicator for compliance with the hourly limit of 200 μg/m^3) as function of the assumed scenarios on NO_X and NO_2 fractions is similar for all 8 cities. For cities where diesel cars contribute a large part to the urban NO_2 concentrations (e.g., Antwerp

Table 4
NO$_2$ concentration (µg/m^3) on an urban background location with boundary conditions of different European cities for the baseline and nine scenarios.

| Scenario	Antwerp	Athens	Barcelona	Krakow	London	Milan	Paris	Stockholm
1 (base)	29	29	34	27	50	34	31	9.5
2	23 (−20%)	24 (−18%)	30 (−11%)	24 (−11%)	44 (−12%)	29 (−15%)	27 (−13%)	9.1 (−4%)
3	19 (−33%)	24 (−19%)	28 (−17%)	23 (−14%)	39 (−21%)	25 (−26%)	23 (−26%)	8.7 (−8%)
4	21 (−27%)	24 (−17%)	29 (−14%)	24 (−12%)	41 (−17%)	27 (−21%)	25 (−20%)	8.8 (−7%)
5	19 (−36%)	24 (−18%)	28 (−19%)	23 (−14%)	39 (−22%)	25 (−28%)	22 (−29%)	8.7 (−8%)
6	23 (−22%)	24 (−18%)	30 (−12%)	24 (−12%)	42 (−15%)	28 (−18%)	26 (−18%)	9 (−5%)
7	25 (−13%)	24 (−17%)	32 (−7%)	24 (−10%)	45 (−9%)	31 (−10%)	29 (−8%)	9.3 (−2%)
8	26 (−11%)	24 (−17%)	32 (−6%)	24 (−10%)	47 (−6%)	31 (−8%)	31 (−2%)	9.3 (−2%)
9	24 (−16%)	24 (−17%)	31 (−8%)	24 (−11%)	45 (−10%)	29 (−14%)	29 (−7%)	9.1 (−4%)
10	26 (−10%)	24 (−17%)	32 (−5%)	24 (−10%)	47 (−5%)	31 (−8%)	32 (1%)	9.4 (−1%)
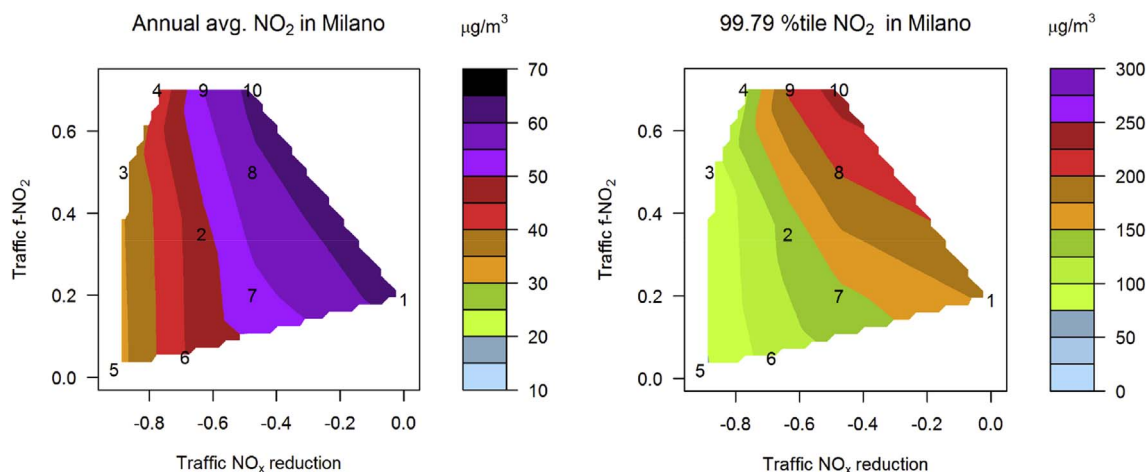
Fig. 3. Impact of NO$_2$ fraction and NO$_x$ emissions on annual average and 99.79[th] percentile NO$_2$ concentrations in a street canyon in Milan; the colour scale represents the annual average NO$_2$ concentration in the canyon. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

and Paris), reductions in the NO$_X$ emissions of diesel cars translate directly into substantial reductions in NO$_2$ concentrations. A reduction in the NO$_X$ emissions of diesel cars to Euro 6 levels (80 mg/km) would bring NO$_2$ concentrations in all 8 cities below the European air quality standard (Table 3). However, if the on-road NO$_X$ emissions of diesel cars remain at the current levels (compare baseline scenario with Scenarios 7 to 10), only small reductions of around 10% in the ambient NO$_2$ concentrations can be expected at best. Fig. 3 (left) depicting the specific situation in a street canyon in Milan suggests that a reduction in the NO$_X$ emissions yields a considerable decrease in the annual average NO$_2$ concentration while changes in the NO$_2$ fraction has little impact on the NO$_2$ concentration. However, the number of exceedances of the hourly limit of 200 µg/m^3, modelled here as the 99.79[th] percentile of the NO$_2$ concentrations, increase with an increasing NO$_2$ fraction in the NO$_X$ emissions Fig. 3 (right). It is even possible that increasing NO$_2$ fractions increase the NO$_2$ concentration in the street canyon, despite overall decreasing NO$_X$ emission levels (compare Scenario 1 and 10). This finding confirms the observation of Degraeuwe et al. (2016) for Antwerp and applies to some extent to all 8 cities covered by our analysis. The effects of a change in NO$_X$ emissions and their NO$_2$ fractions on the NO$_2$ concentration decreases with a decreasing share of diesel cars in the vehicle fleet (compare the results in Table 3 for Athens (9% share of diesel cars in the passenger car fleet) versus Antwerp (85% diesel cars in the passenger car fleet)). The observed reductions in NO$_2$ concentrations in the case of Athens mainly result from lower NO$_2$ background concentrations and decreased NO$_X$ emissions from heavy-duty vehicles. Likewise, a high contribution of the NO$_2$ background to the NO$_2$ concentration in the city street canyon (as is the case for Krakow) decreases the importance of the NO$_X$ emissions from diesel cars for urban air quality.

4. Discussion

4.1. Strengths and limitations of our research

This article assesses the effect of NO$_X$ emissions of Euro 6 Diesel cars on the NO$_2$ pollution in 8 European cities. Our modelling is based on the method developed by Degraeuwe et al. (2016) and applies city-specific conditions (such as wind speed, temperature, and vehicle fleet composition) to a generic street grid. Our modelling captures the wide range of scenarios for both NO$_X$ emissions and NO$_2$ fractions, thereby providing policy makers with clear indications about the impact of vehicle emissions on NO$_2$ air quality in the time horizon until 2030. The assumed model simplifications reduce the demand for data while still allowing us to account for relevant phenomena of pollution transport and atmospheric chemistry at European scale down to the level of the street canyon. With small modifications in the input parameters, our model can be used to assess the impact of NO$_X$ emissions from road transport on NO$_2$ air pollution in any European city.

Still, our research is also subject to limitations. The assumption of a generic street grid and NO$_2$ background concentrations improves the applicability of our model but also introduces uncertainty into our results. A comparison between modelled and measured NO$_2$ and O$_3$ background concentrations yields deviations between +13% and −63% for NO$_2$ and between −10% and +51% for O$_3$ (Table 5), suggesting that our modelling can indeed provide a robust first order approximation of the NO$_2$ air pollution for various NO$_X$ emission scenarios. These relatively limited deviations also suggest that the correlation function applied between the modelled and measured daily average NO$_2$ and O$_3$ concentrations is generally applicable. Depending on the cause of the differences between modelled and measured component concentrations, this generic correction may be too high or low.

Table 5
Urban background annual average NO₂ and O₃ concentrations [μg/m³], average of measurements in urban background stations and modelled by LOTOS-EUROS.

City	Annual average NO_2 ($\mu g/m^3$)			Annual average O_3 ($\mu g/m^3$)		
	Airbase measurement	LOTOS-EUROS modelling	Error	Airbase measurement	LOTOS-EUROS modelling	Error
Antwerp	32	25	−22%	37	41	10%
Athens	38	19	−51%	65	59	−10%
Barcelona	47	43	−8%	39	38	−3%
Krakow	33	13	−62%	31	47	51%
London	49	33	−34%	30	33	11%
Milan	48	45	−7%	49	39	−19%
Paris	36	40	13%	39	34	−13%
Stockholm	13	8	−40%	54	52	−4%

If the differences stem from factors that are unrelated to traffic emissions, no correction should be applied. If the differences stem from errors in the chemical transport model a proportional correction should be applied. The correction based on a correlation as implemented here constitutes a practical compromise between the two.

In cities like Paris and London, the dimensions of typical street canyons may be larger than the ones assumed here, i.e. streets may comprise 4 lanes in each direction. The NO₂ concentrations in such streets are likely to be higher than those modelled here, as the additional NOₓ vehicle emissions tend to over-compensate atmospheric dilution and mixing effects of a wider street canyon. Our modelling may thus systematically underestimate NO₂ pollution in the largest streets. The effects of the 10 NOₓ emission scenarios are, however, similar for larger and smaller streets; consequently the principal conclusions of our modelling should remain valid for large artery streets as well.

4.2. Effect of NOₓ emission levels and NO₂ fractions on urban NO₂ concentrations

In line with the findings of Degraeuwe et al. (2016), our analysis demonstrates that overall NO₂ concentrations in the 8 European cities addressed here depend to a larger extent on the NOₓ emission levels of diesel cars than on the NO₂ fractions (Table 3; compare Scenario 5 with Scenarios 7 and 10). The impact of NO₂ fractions on the ambient NO₂ concentrations is, however, relevant in bigger cities with a large contribution from traffic to total NO₂ pollution as it is the case, e.g., in Paris. As these observations suggest, reductions in the NOₓ emissions from diesel cars can substantially reduce the annual average NO₂ concentrations. This can be achieved with cleaner diesel cars, a shift to more gasoline cars in city centres or a reduction of motorised traffic. The European Commission has recently introduced the Real-Driving Emissions (RDE) test procedure to measure and control the NOₓ emissions of diesel cars on the road. Whether this measure will effectively limit the NOₓ emissions of diesel cars remains to be seen.

Moreover, a reduction in the NOₓ emissions of diesel cars may not be equally effective in addressing air quality problems in the various European cities. For cities like Athens in which diesel cars contribute little to air quality problems, complementary measures that also address the regional NO₂ background concentrations may need to be considered.

Our modelling results support the findings of Kiesewetter et al. (2014) who modelled the NO₂ pollution at 1950 air quality monitoring stations across Europe. Nevertheless, their method and the one we applied differ in key features. While we model the NO₂ concentrations in a street canyon by accounting for the NOₓ emissions in the canyon and the NO₂ background concentrations, Kiesewetter et al. (2014) derive the parameters (residence time in the canyon, and emissions) of the box model which computes the traffic increment from NO₂ measurements for the base year. In this way all European measurement stations can be considered, while our approach can only be applied to selected cities due to limited data availability and computational power.

Another difference is related to the impact of variations in NO₂ fraction on the background NO₂ concentrations. Kiesewetter et al. (2014) assume this ratio to be constant whereas we, on the other hand, account for variations of the ratio in the background with the chemical transport model (LOTOS-EUROS; Schaap et al., 2008). Finally, the method applied by Kiesewetter et al. (2014) can only model the annual average NO₂ concentrations and captures thus only the exceedance of annual mean NO₂ limits can be studied whereas we can address here also hourly NO₂ concentrations. This limitation of the approach chosen by Kiesewetter et al. (2014) is not much of a problem since almost all stations that exceed the hourly NO₂ limit on a regular basis, also exceed the annual NO₂ limit. However, if the NO₂ ratios of the NOₓ emissions continue to increase, exceedances of the hourly NO₂ limit could become more critical from the perspective of air quality.

As a consequence of the dieselgate scandal the French (UTAC, 2016), German (BMVI, 2016) and UK (Department for Transport, 2016) type approval authorities published a report on emissions of Euro 5 and 6 passenger cars. For a full discussion on these tests see Degraeuwe and Weiss (2017). They tested 78 Euro 6 diesel passenger cars, selecting the most common vehicles on the market. All vehicles comply with the NEDC on the test bench but when the same vehicle is driven on the road average emissions are 377 mg NOₓ/km (4.7 times the limit). Only 16% of the Euro 6 cars tested complied under on-road testing with the NEDC, by emitting less than 80 mg NOₓ/km, which demonstrates that it is technically possible to comply with the limit on the road. During the on-road test the ambient temperature was 10 °C (σ = 4 °C). Manufactures argue that this is because cars use legal defeat devices that reduce NOₓ abatement below 20 °C. They claim that this is necessary to protect the engine. However, the legislation also specifies that cars should not reduce after-treatment under normal conditions of use. The legal aspects are outside the scope of this paper. But the outcome of these reports has implications for urban air quality. The normal ambient temperature at which cars operate in Europe is 12 °C (σ = 4 °C; Malfettani et al., 2016). This means that the results are representative for normal conditions of use in Europe. Also the *Committee of Inquiry into Emission Measurements in the Automotive Sector* of the European Parliament investigated the discrepancy between the official and on-road NOₓ emission of diesel cars and published its final conclusions in April 2017 (Gieseke and Gerbrandy, 2017). They conclude that not the supposedly unrealistic NEDC but rather the use of defeat devices explains the biggest part of the difference between emissions on the road and on the NEDC (conclusions 1 and 5, Gieseke and Gerbrandy, 2017). Concerning the conformity factors, allowing diesel cars to exceed the limits by a factor 2.1 after 2019 and a factor 1 plus measurement uncertainty (currently set at 0.5) after 2021, the committee states that it allows the continuing use of less efficient technology. The technology to achieve the Euro 6 standard on the road is available but is not used for economic reasons. Hence, cities that want to tackle the exceedances of the NO₂ limits cannot count on clean diesels arriving soon. They will have to implement alternative measures such as a ban on diesel cars, a reduction of traffic, a shift to gasoline and electric vehicles or a modal shift to cycling,

walking and public transport.

5. Conclusions

We generalise in this article the NO_2 pollution model developed by Degraeuwe et al. (2016) and apply it to 8 cities in Europe. Benchmarking our results against measurements of actual NO_2 concentrations suggests that our modelling approach yields a reliable first-order approximation of the effect of various NO_X emission sources on urban NO_2 concentrations in European cities. We draw the following conclusions from our research:

- The absolute level of traffic NO_X emissions is more relevant for the NO_2 concentration in urban street canyons than the NO_2 fraction in the total NO_X emissions for most cities. A reduction in the NO_X emissions per km of diesel cars can decrease the ambient NO_2 concentrations in European cities.
- In cities with a low share of diesel cars in the vehicle fleet (e.g., Athens) or a high contribution to urban NO_2 pollution from background NO_2 concentrations (e.g., Krakow), additional measures addressing heavy-duty vehicles, and NO_2 sources in the manufacturing, energy, and mining industry are necessary to reduce urban air pollution.
- A sensitivity study shows that the NO_2 background concentration, the wind speed and the diesel share of the fleet have the biggest influence on the NO_2 concentration in a street canyon.
- The NO_2 fraction of NO_X emissions only has an effect close to the emission sources and has an impact on the exceedance of the hourly limit value of 200 $\mu g/m^3$. It could become a concern if cars continue to emit a lot of NO_x in combination with a higher NO_2 fraction.
- Compliance of diesel cars with the Euro 6 NO_x emission limits under real-world driving conditions yields substantial benefits for air quality and can effectively reduce the current exceedances of the NO_2 air quality standard. However, if Euro 6 diesel cars continue to emit on the road substantially more NO_X than permitted by the Euro 6 emissions limit, policy makers should consider alternative measures including forcing a shift to gasoline cars to achieve compliance with air quality standards.
- As a consequence of the dieselgate scandal a lot of data on real driving emissions of Euro 6 cars has become available. The on-road performance of the bulk of these cars is not good enough to solve the NO_2 problem in major cities. The allowed conformity factors will delay the widespread adoption of available emission control technologies. Cities with a severe NO_2 problem, may be obliged to consider alternative measures including reducing the share of diesel cars in the urban fleet.

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