2.5.5 Mobility Scenarios in the Milan area: a modelling assessment of Air Quality

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Abstract

The purpose of this study is the estimation of possible environmental benefits deriving from the achievement of sustainability targets linked to mobility policies and consequently the human health impact. The focus is on the private road transport segment and the object of this study is the metropolitan city of Milan. At a first stage, the emissions of pollutants associated to road traffic, such as NO_X, and $PM_{2.5}$, are estimated in case of implementation of mobility policies and compared to a base case. This analysis is based on the variation of vehicle kilometres travelled (VKT) by different fuelled vehicles, assuming zero exhaust emissions associated to electric vehicles. Afterwards the impact on urban air quality due to road traffic is estimated, thanks to a WRF-CAMx modelling system. Following the impact pathway approach, a local scale health impact assessment is finally performed to quantify the actual benefits that might derive from such changes in air quality. Two mobility scenarios with different levels of incisiveness are analysed in this work: a *Fleet electrification only Scenario* and a *Fleet Electrification + Mobility Policy Scenario.* The former considers the introduction of 20% of pure electric vehicles (BEV) in the car fleet and the same amount of vehicles fuelled with LPG and methane, the remaining 60% being conventional. This mobility scenario leads to a slight reduction of $PM_{2.5}$ and an abatement of 17% of NO_2 annual mean concentration. However to get a more significant lowering of the atmospheric pollutants further actions are needed such as the enhancement of shared cars, more effectiveness of private vehicles, strengthening of public transport, including infrastructural interventions. In fact most promising results are associated with the second scenario. The decrease in $PM_{2.5}$ and NO_2 annual mean concentrations under this scenario entails a consistent reduction in the health burden associated to air pollution exposure: 118 premature deaths (1329 years of life lost) and 244 days of hospitalization due to cardiovascular and respiratory diseases would be avoided annually.

Introduction

This study aims to estimate the effects of a partial car fleet electrification on the mitigation of air pollution in a metropolitan context. The attempt here is also to assess the impact of such scenario in terms of human exposure to air pollution. The case test refers to the city of Milan and its surroundings where the harmful pollutants concentration limit values, imposed by the law, are often exceeded.

Air pollution is a major environmental risk to health (WHO, 2013) particularly in urbanized areas, where most part of European population live. It also has considerable economic impacts, cutting lives short, increasing medical costs and reducing productivity through working days lost across the economy. Europe's most serious pollutants in terms of harm to human health are fine particulate matter ($PM_{2.5}$), NO_2 and ground-level O_3 (EEA, 2018).

In urban areas on-road transport is one of the main sources of atmospheric pollutant emissions and passenger transport is the most critical segment. For a receptor placed in the city centre of Milan, Pepe et al., 2019 estimate that road traffic, residential heating by biomass, and long range transport are responsible for 73% of $PM_{2.5}$ annual average concentration. The transport sector yields the principal contribution (28%), with half contribution due to cars, 8% and 5% respectively due to heavy and light duty vehicles, and 1% due to mopeds and motorcycles. For NO_2 they find that road transport is, by far, the most impacting source, with a 72% share: 60% of NO_2

concentration derives from heavy duty (31%) and passengers cars (29%) emissions, 11% from light duty vehicles, and about 1% from mopeds and motorcycles (Pepe, et al., 2019). These estimates are obtained thanks to the source apportionment (SA) algorithm implemented in CAMx model (PSAT, Yarwood et al., 2004) and provide a comprehensive picture of the actual role of the different sources, thus representing a good starting point for further development of emission reduction strategies (Pepe et al, 2019). Among the strategies designed to reduce on road traffic emissions and their impact on air quality, fleet electrification is one of the most promising one. It comprises a wide spectrum of technological options that range from the early-stages hybrid vehicles to pure electric battery vehicles (BEV) that are entirely propelled by stored electricity with no direct exhaust emissions. Electric vehicles are particularly suitable to improve air quality in urban areas, where short distance trips at low speeds are prevalent (Soret, et al., 2014). Furthermore, higher potential benefits of reducing atmospheric emissions are found in highly populated areas (Ayalon, et al., 2013).

In this work the impact of a partial car fleet electrification on air pollution levels in a metropolitan context are estimated by means of air quality modelling. Additionally, the impact of the car fleet change is also assessed in terms of population exposure to air pollution and related adverse health effects. The modelling study is focused on Milan, the major metropolitan area of the Po Valley, a well-known European hot-spot for many air pollutants (Perrino et al., 2014). Surrounded by the Alps to the North and North-West and by the Apennines to the South, the Po Valley experiences poor circulation of air masses (Caserini et al., 2017). In addition, the occurrence of frequent and prolonged wind calm periods and atmospheric stability conditions, especially during the coldest months, favors the accumulation of locally-emitted pollutants as well as the development of high pollution episodes at regional scale, so that regulatory air quality limits are often exceeded.The work is focused on NO_2 and $PM_{2.5}$: besides being the most important pollutants associated to road transport and regulated by European air quality standards, they are suitable for a metropolitan scale study. On the contrary, the analysis of O_3 atmospheric concentration variation requires wider spatial scales.

In the following section 2 the set-up of WRF-CAMx modelling system, upon which the present work is based, is described together with the health impact assessment tool. The methodology developed to transpose mobility scenarios into passenger cars emission scenarios is then reported. This crucial step leads to the estimation of the effects of mobility policies on air quality, the main results being reported in section 3. Finally the expected impact on human health, resulting from the reduction of ambient $PM_{2.5}$ concentration for the two mobility scenarios considered, is discussed.

Methods

Modelling system

Air quality modelling simulations were performed through WRF (Weather Research and Forecasting meteorological model, Skamarock et al., 2008) and CAMx (Comprehensive Air Quality Model with Extensions, ENVIRON, 2016) for the entire 2010 calendar year.

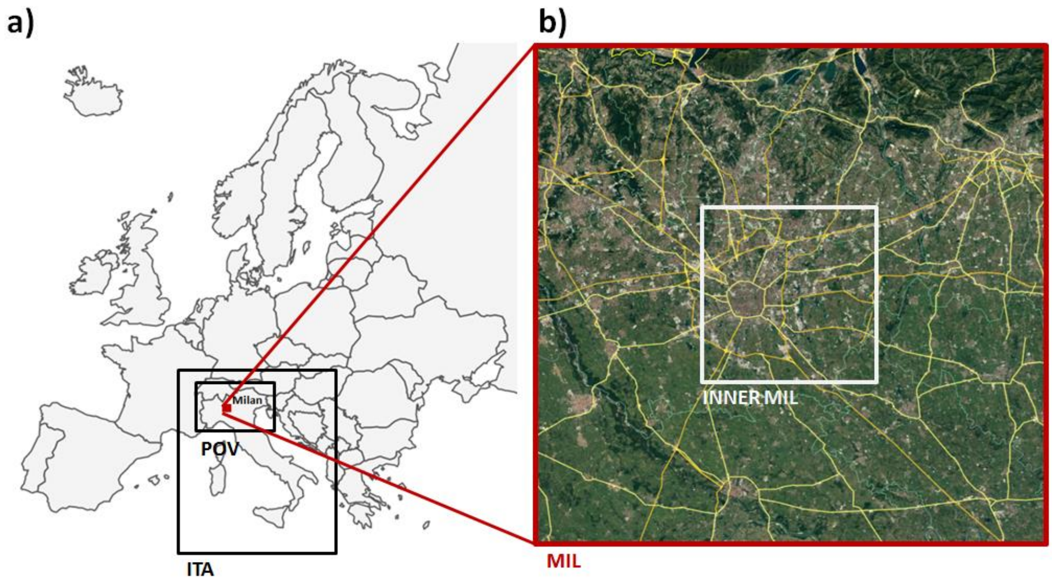

Figure 1: (a) WRF-CAMx domains of simulations: ITA and POV. (b) POV nested MIL domain (1.7 km of horizontal resolution) and the area of study (INNER MIL).

WRF model setup

WRF model (version 3.4.1) simulations were performed on four computational domains: a European domain and the three nested domains (ITA, POV and MIL) shown in Figure 1. The biggest domain has an European extent (3870 x 4140 km^2) and a horizontal resolution of 45 km. The first nested domain (ITA) covers the Italian peninsula (1350 x 1530 km^2) at 15 km of horizontal resolution; the second nested domain (POV) covers the Po Valley (600 x 420 km^2), and the last domain (MIL) covers Milan metropolitan area, spanning over the city of Milan and its surroundings (85 x 85 km^2). The horizontal resolution of POV and MIL domain are respectively 5 km and 1.7 km. All the domains are solved by 30 vertical levels, from about 25 m up to more than 15 km from ground level. This configuration of WRF model is forced on its open boundaries with ECMWF analysis fields at 0.5 x 0.5 degrees of resolution, both at the ground level and at different pressure levels. Further details on WRF set-up are given in Pepe et al., 2006.

CAMx model setup

In this work the simulation of dispersion phenomena and chemical processes is entrusted to CAMx model (version 6.3) that estimated concentration fields over the three innermost domains of WRF: ITA, POV and MIL (Figure 1). CAMx configuration shares the same horizontal resolution of WRF, with a slightly reduced grid dimensions in order to remove boundary effects. Conversely, the vertical grid is defined collapsing the first 21 WRF levels into 14 layers even though the layers up to 1 km above the ground level are kept at the same height.

Homogeneous gas phase reactions of nitrogen compounds and organic species are reproduced through CB05 mechanism (Yarwood et al., 2005). The aerosol scheme is based on two static modes (coarse and fine). Secondary inorganic compounds evolution is described by thermodynamic algorithm ISORROPIA (Nenes et al. 1998), while SOAP (ENVIRON, 2011) is used to describe secondary organic aerosol formation. Input emission fields were derived from emission inventory data at different spatial scale resolution: EMEP (European Monitoring and Evaluation Programme), available over a regular grid of 50 x 50 km2, ISPRA (Italian national inventory data), which provides province-scale emission data, INEMAR, which provides emission data at municipality resolution over the POV and MIL domain (INEMAR – ARPA Lombardia, 2015). Each emission inventory was processed using SMOKE model (version 3.5) (Sparse Matrix Operator for Kernel Emissions model, UNC, 2013) that allows the conversion of emission inventory data to the both spatial and temporal resolution needed by the air quality model. Additional details on CAMx configuration are available in Pepe et a., 2016.

Health impact assessment

The impact on human health of different air quality scenarios on human health can be calculated by combining gridded changes in $PM_{2.5}$ concentration fields, municipality-level population data and baseline incidence rates. Following the well-established European Environment Agency methodology (De Leuuw et al., 2016), a log-linear model has been used to derive the health impact function, which results in the following equation:

$$I_{ijk} = P_{ij} \cdot r_{ijk} \left(1 - \frac{1}{e^{\beta_k \Delta C_{ij}}}\right) \tag{1}$$

Where I_{ijk} is the impact on health endpoint k (e.g. avoided mortality) due to change in $PM_{2.5}$ concentration ΔC_{ij} in the grid cell (i,j), P_{ij} is the population in grid cell (i,j), β_k is a parameter which accounts for the sensitivity of impact on health endpoint k on a concentration change ΔC_{ij} and r_{ijk} is the baseline incidence rate of endpoint k in cell (i,j). In this work only two health endpoints have been taken into account (i.e. $k = \{1,2\}$), namely hospital admissions and premature deaths, as they are classified as reliable by the most up-to-date European literature on $PM_{2.5}$ health impact assessments (WHO, 2013). The value of parameter β_k is directly linked to the relative risk (RR_k), which is defined in the log-linear model as:

$$RR_k = \frac{r_{k,BC}}{r_{k,SC}} = e^{\beta_k \Delta C} \tag{2}$$

Where $r_{k,BC}$ stand for the baseline incidence rate whereas $r_{k,SC}$ is the incidence rate under a defined scenario. As RR_k values and their 95% confidence interval for different health endpoints k have been estimated during the European HRAPIE project (WHO, 2013) for a 10 μg m^{-3} change in $PM_{2.5}$, β_k can be simply computed as:

$$\beta_k = \frac{\ln(RR_k)}{10} \tag{3}$$

The total impact for the endpoint k is finally the sum of I_{ijk} over each grid cell $(i.j)$.

It should be noted that for premature deaths, ΔC_{ij} in Equation (1) is expressed in terms of the change in the annual average $PM_{2.5}$ concentration, as premature mortality is supposed to be an effect of a long-term exposure; on the other hand, for hospital admissions ΔC_{ij} is expressed as the change in daily mean $PM_{2.5}$ concentration (being the result of a short-term exposure), and Equation (1) is solved for each day of the year. We assume, coherently with WHO, 2013, that premature mortality affects only 30+ adults whereas hospital admissions are related to all ages. Following WHO 2013, in this work we use a value of RR per 10 μg/m^3 equal to 1.062 (1.040-1.083, 95% confidence interval) for premature mortality and equal to 1.0190 (1.0017–1.0166) and 1.0091 (0.9982–1.0402) for hospital admissions due to respiratory and cardiovascular diseases respectively.

Population and sanitary data have been collected from different sources. We used age-specific and municipality-level population and baseline incidence rates from the Italian National Institution of Statistics (ISTAT) for year 2010, which have been re-gridded onto the air quality modelling grid. The national average number of hospital admissions (2010), due to both respiratory and cardiovascular diseases, was instead recovered from the “European Hospital Morbidity Database”, which provides endpoint-specific data at a national level for each European country. Finally, RR_k values are taken from the results of HRAPIE project (WHO, 2013).

Base case and scenarios

Mobility scenarios

One base case and two mobility scenarios have been analysed. The base case refers to fleet composition and vehicle kilometres travelled (VKT) data for year 2010. The first scenario (*Scenario A - Fleet Electrification*) considers a car fleet composed by a share of 20% of Battery Electric Vehicles (BEV), a share of 20% of LPG and NGV cars, fuelled with both natural and liquefied petroleum gas, with the remainder 60% of the fleet being conventional cars. The second scenario (*Scenario B - Fleet Electrification + Mobility Policies*) considers the same car fleet composition of *Scenario A*, but assumes an overall transition toward a more sustainable mobility. More precisely, *Scenario B* supposes some political strategies designed in order to discourage the use of private cars in favour of bikes and the car pooling and by the extension of the restricted traffic zones. Moreover this scenario supposes the strengthening of public transport infrastructures. Basically the two scenarios differ from the base case (2010) by a different distribution of vehicle kilometres travelled (VKT) by three categories of cars. We assume that the changes imposed by the scenarios have immediate effects, so that the technological evolution of vehicles (e.g., Euro classes distribution) is neglected. More details on both scenarios can be found in Borgarello et al., 2016.

VKT gridded data are provided on a subdomain of the MIL domain (INNER-MIL in Figure 1b). These data have the same horizontal resolution of CAMx grid, i.e. 1.7 km. INNER MIL domain includes the city of Milan and 54 surrounding municipalities, 46 of them belonging to the area of Milan and the remaining 8 to the Monza and Brianza province. For the grid cells of INNER MIL domain, VKT data are provided with a hourly time step for a typical day traffic profile, with one peak during the morning and one peak during the afternoon and an overall traffic damping during the night.

VKT data refer to three macro classes of vehicles:

- Petrol and diesel cars, to whom we refer as *Conventional Vehicles (CV)*
- LPG and NGV cars, to whom we refer as *(LN)*
- BEV, to whom we refer as *Electric Vehicles (EV)*

According to car register data for the INNER-MIL domain in 2010 (ACI Automobile Club Italia 2010), CV class had a 65%-35% split between petrol and diesel cars, whilst LN class had a 84%-16% split between LPG and NGV cars. BEV were limited to very few vehicles in 2010 fleet.

A similar distribution of the car fleet characterizes the Monza and Brianza province. In this study we suppose that the share of electric vehicles are all BEV, neglecting plug-in electric cars.

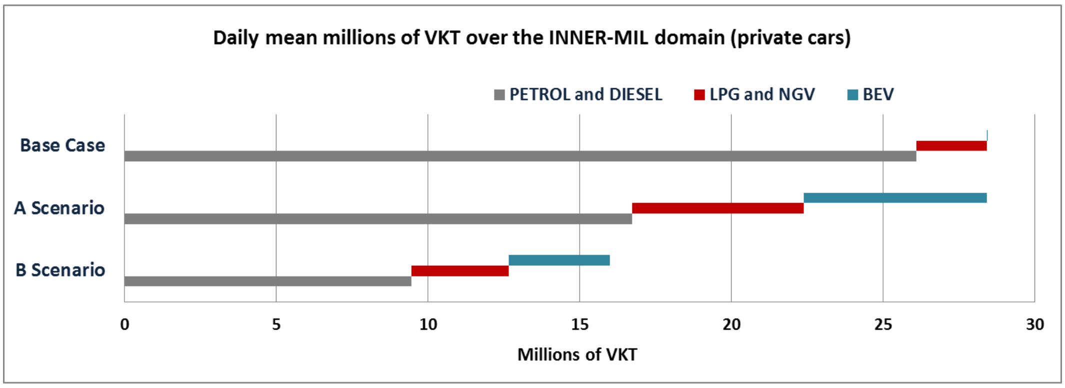

Figure 2: Daily mean millions of VKT averaged over the INNER-MIL domain for the classes of vehicles: CV, LN and EV, according to the Base case, *A* and *B Scenarios*.

Figure 2 provides the overview of the characteristics of the base case and of the two scenarios in terms of daily mean VKT by vehicle class. Almost all the vehicles are conventional in the base case (92%) where BEV vehicles are negligible. The two mobility scenarios introduce a drastic change to the distribution of the car fleet, with the total amount of VKT unchanged for *A Scenario* and reduced by 44% for *B scenario*, because of the replacement of private cars by other transport solutions.

Emission scenarios

Assessing the impact of mobility scenarios on air quality with a chemical and transport model, requires an intermediate and crucial step i.e., the careful rescaling of private road transport base case emission field in the area of study. For this purpose we use VKT data in order to calculate emission rescaling coefficients and emission factors (EF) as multiplicative weights. NO, NO_2, CO, NH_3, NMVOC, SO_2 rescaling coefficients are estimated according to equation 4, while for PM_{10} equation 5 is used. Both coefficients are estimated for each grid cell of the domain (i, j) - where VKT data are available - and for each hour of the day (t). We distinguish two coefficients C_gs and C_PM respectively for gaseous species and PM, in order to take account for PM_{10} non-exhaust emissions, such as brake, tire and road wear that characterize also EV cars.

$$C\_gs_{i,j,t} = \frac{\{\sum_n (VKT \times EF)_n\}_{SC}}{\{\sum_n (VKT \times EF)_n\}_{BC}} \tag{4}$$

$$C\_PM_{i,j,t} = \frac{\{\sum_n (VKT \times EF)_n + \sum_m VKT_{BEV} \times (EF)_m\}_{SC}}{\{\sum_n (VKT \times EF)_n\}_{BC}} \tag{5}$$

For gaseous species the coefficient C_gs is calculated as the ratio between the weighted sum of the VKTs in a scenario with respect to the base case. As weighting coefficient a mean EF, representative of each vehicle macro class n, is used (Figure 3). For C_gs we assume EV as zero exhaust-emissions. For what concerns C_PM we consider an additional term in equation 5, taking account also brake wear, tyre and road abrasion associated to EV (m index) (Ntziachristos et al., 2009). EF are provided by COPERT database (COPERT 2010, Gkatzoflias et al., 2010) and they are averaged for each class of vehicles, according to the car fleet composition in 2010 in the metropolitan area of Milan. Even though recent studies suggest higher road and tyre emissions for EV respect to CV, due to the greater weight of the vehicle, here we suppose the same order of magnitude. Conversely we suppose that non-exhaust emission fraction associated to brake wear of an EV corresponds to 20% of a CV one (Del Duce et al., 2016).

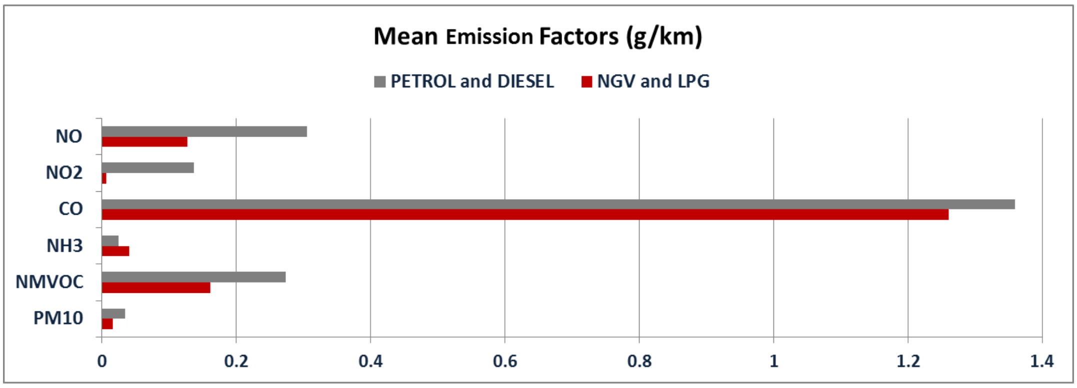

Figure 3: NO, NO_2, CO, NH_3, NMVOC and PM_{10} mean EF (g/km) (COPERT, 2010) employed in Equations 4 and 5. SO_2 coefficients are omitted because of the lower order of magnitude.

Daily mean rescaling coefficients, averaged over all cells of the INNER-MIL domain, are listed in Table 1. In general the overall effect of the *A scenario* is a reduction of 20% - 35% of private cars emission segment while the *B scenario* entails more than a halving of the base case passenger car emissions. This is mostly due to the shift of vehicles kilometres travelled by private cars toward the public transport.

Table 1: Daily mean NO, NO_2, CO, NH_3, SO_2, NMVOC and PM_{10} emission rescaling coefficients.

Emission Rescaling Coefficients		
	A scenario	*B scenario*
NO	0.72	0.46
NO_2	0.65	0.42
CO	0.77	0.49
NH_3	0.81	0.52
SO_2	0.64	0.41
NMVOC	0.71	0.45
PM_{10}	0.76	0.49

Finally, emission fields for *Scenarios A* and *B* are obtained, for each hour of the day, multiplying the base case emission field by a reduction matrix (composed by all the rescaling emission coefficients), over all the INNER-MIL domain. Other emission sources (e.g., the ones associated to industrial plants, residential and commercial heating with biomass, light duty and heavy duty vehicles) are unchanged respect to the base case, so the overall emission field suitable for CAMx model is obtained by the sum of all the emission macro-sectors.

It is worth mentioning that in this study the possible impact on air quality due to a possible increase in the emissions of the energy production sector has been neglected, due to their minor impact on air quality at regional scale (Balzarini et al., 2012).

Results

Mobility scenarios impact on air quality

The effects of sustainable mobility scenarios on air quality are estimated by the comparison of the scenario simulated concentrations with the base case ones. Figure 4a shows the yearly mean concentration of NO_2 simulated by WRF-CAMx modelling system for the base case in the MIL domain (year 2010). NO_2 concentration peaks are reached at the most urbanized areas of MIL domain, the yearly mean concentration ranging between 15 and 20 ppbV in the INNER-MIL portion of MIL domain. Figure 4b and 4c show the fields of the absolute concentration difference between the simulated scenarios and the base case. The effects of mobility policies are important in the INNER-MIL domain, where the partial electrification of the car fleet entails a reduction of about 17% of NO_2 yearly mean atmospheric concentration. This result is amplified in the case of *Scenario B,* which additionally implies an overall reduction of vehicle kilometres travelled by passenger cars, hence lading to a 30% abatement of the NO_2 concentration with respect to the base case.

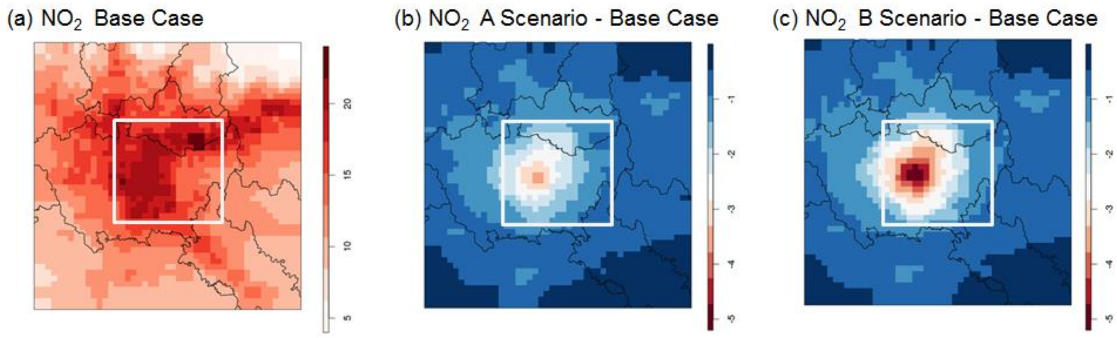

Figure 4: (a) Yearly mean NO_2 concentration (ppbV) for the base case in the MIL domain. (b) Absolute difference between *A Scenario* and the base case and (c) *B Scenario* and the base case. White box corresponds to INNER-MIL domain.

The exposure to fine PM reduces the life expectancy of the population, especially in polluted cities, so the estimation of mobility scenarios impacts on $PM_{2.5}$ atmospheric concentration is of primary importance. Figure 5a shows the base case concentration of $PM_{2.5}$ in the MIL domain, calculated with WRF-CAMx. Here the urban footprint of $PM_{2.5}$ is less evident than NO_2 since fine PM is more subject to transport and diffusion processes related to formation timescales of its secondary component. In the INNER-MIL part of the domain the yearly mean concentration of PM2.5 is about 17 µg m^{-3} while the reduction of $PM_{2.5}$ due to the mobility scenarios is about 4% and 8.5% respectively for the *A* and *B scenarios* (Figure 5b and 5c). Fleet electrification has a limited impact on $PM_{2.5}$ concentration. This is partially due to the fact that non-exhaust emissions (resuspension and brake, tyre and road abrasion) cannot be reduced by the variation of the composition of the car fleet.

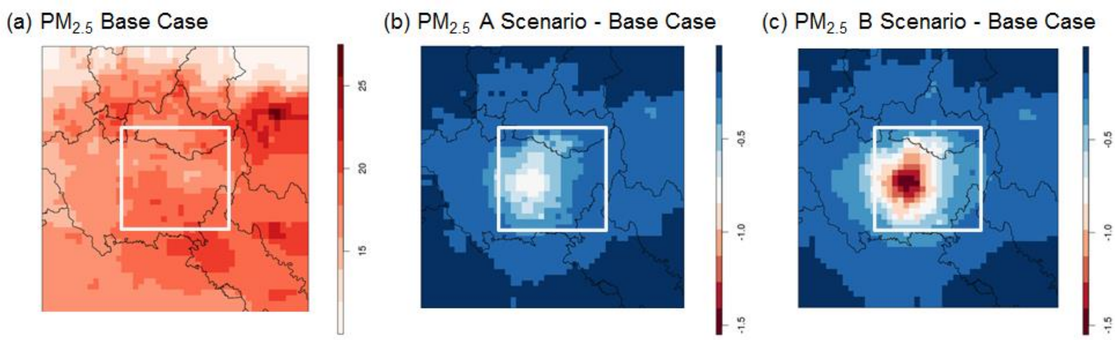

Figure 5: (a) Yearly mean $PM_{2.5}$ concentration (µg m^{-3}) for the Base case in the MIL domain. (b) Absolute difference between *A scenario* and the base case and (c) *B scenario* and the Base case. White box corresponds to INNER-MIL domain.

Mobility scenarios impact on human health

Differences in $PM_{2.5}$ concentration fields presented in the previous section entail a non-negligible impact on human health. On annual basis, the total amount of avoided premature deaths, achievable in the two different scenarios, is equal to 113±19 and 182±31 (mean ± standard deviation), respectively for the *A and B scenarios* (Table 2). The difference between the two scenarios can be ascribed to the impact of other mobility policies, which account for 69 avoided premature deaths (37.9% of the total). On top of that, 50±25 hospitalizations due to respiratory and cardiovascular diseases would be avoided annually in the *A Scenario* and 32 more would be avoided when including also the mobility policies (*B Scenario*), as summarized in Table 2. The uncertainty associated to our health impact calculations derives from the uncertainty in the relative risk estimation given by the HRAPIE project (WHO2013).

Table 2: Effects on human health due to the abatement of $PM_{2.5}$ (μg m^{-3}) respect to the base case, due to the *A* and *B Scenarios*. All the numbers refer to avoided effects.

	A Scenario	B Scenario
Avoided premature deaths	113 ± 19	182 ± 31
Avoided hospitalizations	50 ± 25	82 ± 42

The fine resolution modelling allowed assessing the spatial distribution of such avoided impacts, which is far from being uniform across the domain. As clear from Figure 6, positive impacts are mostly concentrated in the highly populated area of Milan, which is the middle of the MIL domain. This is related to the combined effect of the higher degree of $PM_{2.5}$ reduction in the city of Milan (Figure 5) and the higher population exposure (i.e., population density is greater in the city of Milan and it decreases in the city hinterland). In order to better quantify such spatial pattern, we computed the average avoided premature deaths per unit of area, both in the city of Milan and in the rest of the domain. We found that, under the *B scenario*, 0.609 deaths/km^2 are avoided in the city of Milan whereas only 0.013 deaths/km^2 could be avoided, on average, in the hinterland of Milan.

This finding implies that the most polluted areas (i.e. densely populated urban areas) would benefit the most from the two mobility scenarios, highlighting the importance of mobility policies in addressing current human health concerns in urban areas.

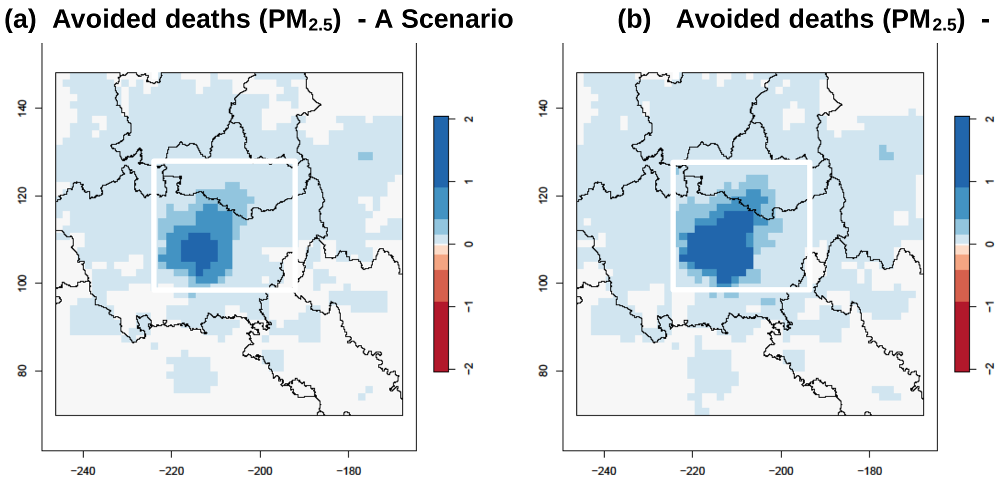

Figure 6: Estimated premature deaths avoided thanks to the abatement of the $PM_{2.5}$ (μg m^{-3}) resulting from the application of the *A scenario* (a) and the *B scenario* (b). The differences are calculated respect to the base case. White box corresponds to INNER-MIL domain.

Conclusions

The transition toward a more sustainable mobility is becoming increasingly more achievable thanks to the spread of electric and low carbon vehicles. Dedicated mobility policies can accelerate this trend. In this work the focus is on the private road transport sector, being amongst the main contributors to the poor air quality in urban areas. Particularly potential emission and air quality impacts of partial fleet electrification in the city of Milan and its surroundings are analysed. This area, which is located in the North-Western section of the Po Valley, is known for critical episodes of urban pollution. The air quality assessment has been performed with the WRF-CAMx modelling system and a novel tool has been developed and applied, over the INNER-MIL domain, to estimate the health impact associated with different policies.

Two fleet electrification scenarios (with 20% of vehicle kilometres travelled by BEV), have been defined and compared to a base case, where the *Scenario A* is an electrification-only scenario and the *Scenario B* takes account also of sustainable mobility policies. Hereinafter we provide average estimates of scenarios impacts, over all the INNER-MIL domain, even though both scenarios are most incisive in the city centre of Milan in terms of emissions, air quality and health impact. Both scenarios entail more than a halving of the base case passenger car emissions, the reduction being respectively of 58% for NO_2 and 51% for $PM_{2.5}$. The overall result in terms of pollutant atmospheric concentration abatement is less incisive, since other emission macro categories have an important weight. As expected the *B scenario* is more incisive than *A scenario* in terms of both emissions and air quality, since the effect of mobility policies is an overall decrease of vehicle kilometres travelled by passenger cars. In particular the more ambitious scenario (*B*) entails a 30% abatement of the NO_2 yearly mean concentration with respect to the base case in the INNER-MIL domain, while it has a limited impact on $PM_{2.5}$ (a lowering of 8.5%). This is partially due to the fact that non-exhaust cannot be reduced by the variation of the composition of the car fleet.

We conclude that fleet electrification presents potential air quality benefits even though the reductions achieved are insufficient to ensure proper air quality levels, especially for PM. Nevertheless we estimate non-negligible effects of *B scenario* on human health, due to the abatement of $PM_{2.5}$ concentration: 180 premature deaths are avoided in the area of Milan.

In order to reach greater benefits other management measures should involve all vehicle categories (two-wheelers, heavy-duty vehicles, buses and light-duty vehicles) not only passenger cars and further reductions should be considered also for other emission sources. Finally the WRF-CAMx modelling system, used to estimate the scenarios yearly mean concentration pollutant variations, is a suitable tool for the management and assessment of urban air quality, as well as the health impact assessment tool.

Acknowledgments

This work has been financed by the Research Fund for the Italian Electrical System in compliance with the Decree of Minister of Economic Development April 16, 2018.

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