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# Estimated health benefits of exhaust free transport in the city of Malmö, Southern Sweden

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A R T I C L E  I N F O

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A B S T R A C T

Air pollution is responsible for one in eight premature deaths worldwide, and thereby a major threat to human health. Health impact assessments of hypothetic changes in air pollution concentrations can be used as a mean of assessing the health impacts of policy, plans and projects, and support decision-makers in choices to prevent disease.

The aim of this study was to estimate health impacts attributable to a hypothetical decrease in air pollution concentrations in the city of Malmö in Southern Sweden corresponding to a policy on-road transportations without tail-pipe emissions in the municipality. We used air pollution data modelled for each of the 326,092 inhabitants in Malmö by a Gaussian dispersion model combined with an emission database with > 40,000 sources. The dispersion model calculates Nitrogen Oxides ($NO_x$) (later transformed into Nitrogen Dioxide ($NO_2$)) and particulate matter with an aerodynamic diameter < 2.5 μg/m$^3$ ($PM_{2.5}$) with high spatial and temporal resolution (85 m and 1 h, respectively).

The average individual reduction was 5.1 (ranging from 0.6 to 11.8) μg/m$^3$ in $NO_2$, which would prevent 55 (2% of all deaths) to 93 (4%) deaths annually, depending on dose-response function used. Furthermore, we estimate that the $NO_2$ reduction would result in 21 (6%) fewer cases of incident asthma in children, 95 (10%) fewer children with bronchitis every year, 30 (1%) fewer hospital admissions for respiratory disease, 87(4%) fewer dementia cases, and 11(11%) fewer cases of preeclampsia every year. The average reduction in $PM_{2.5}$ of 0.6 (ranging from 0.1 till 1.7) μg/m$^3$ would mean that 2729 (0.3%) work days would not be lost due to sick-days and that there would be 16,472 fewer restricted activity days (0.3%) that year had all on-road transportations been without tail-pipe emissions.

Even though the estimates are sensitive to the dose-response functions used and to exposure misclassification errors, even the most conservative estimate of the number of prevented deaths is 7 times larger than the annual traffic fatalities in Malmö, indicating a substantial possibility to reduce the health burden attributed to tail-pipe emissions in the study area.

## 1. Introduction

During the last decades, epidemiological and toxicological studies have provided enough evidence for the conduction of health impact assessments (HIA) of air pollution (WHO, 2013a, 2013b). Whilst epidemiologists often study the risk of a disease in the presence of exposure relative to the risk of a disease in the absence of exposure, a risk assessor, on the other hand, often asks how many excess cases of disease will occur in a population of a certain size due to exposure at a certain dose level (Hertz-Picciotto, 1995)? HIA generally applies a health impact function combining a risk estimate from the epidemiology literature that relate hypothesized air quality changes to a population at risk (Fann et al., 2011). HIA are used for example in large projects such as the Global Burden of Disease by WHO where air pollution is now valued as one of the largest health threats of our time, responsible for one in eight premature deaths worldwide (Cohen et al., 2004; WHO, 2011,

*Abbreviations:* HIA, health impact assessments; $PM_{10}$, particulate matter with an aerodynamic diameter < 10 μg/m$^3$; $PM_{2.5}$, particulate matter with an aerodynamic diameter < 2.5 μg/m$^3$; $NO_2$, nitrogen dioxide; $NO_x$, nitrogen oxides; ERFs, exposure response functions; RAD, restricted activity days

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2014). There were roughly equal contributions from household air pollution and ambient particulate matter pollution.

HIA can also be used as a systematic process to determine the potential health effect of i.e. air pollution from proposed policies, plans, programs or projects and can provide recommendations on monitoring and managing those effects (Chart-asa and Gibson, 2015). Results from HIA can thereby be a useful tool for policymakers and urban planners (Harris-Roxas and Harris, 2011; Shojaei et al., 2014). For example, Castro and colleagues estimated the health impacts attributable to a decrease in particulate matter with an aerodynamic diameter < 10 μg/$m^3$ ($PM_{10}$) and Nitrogen Dioxide ($NO_2$) concentrations due to certain policy measures to be about 1% to 2% of total all-cause annual mortality in the population of the Agglomeration Lausanne-Morges in Switzerland (Castro et al., 2017).

In the city of Malmö, Southern Sweden, questions are raised, politically and within different parts of the municipality, if it is worth the effort to work on improving air quality. The city of Malmö has had problems complying with EU Air Quality Guidelines for $NO_2$ in the past, and have had action plans since 2006 to reduce emissions, but the city now complies with EU Air Quality Guidelines for $NO_2$. Furthermore, the city of Malmö has agreed upon a policy to be fossil-free by the year 2020. The "Fossil-free initiative" is a transportation policy indicating no fossil fuels in on-road transportations used by the municipality. Decision-makers have raised the question of whether there are any health benefits from this policy. The Environmental Department of the city of Malmö therefore needs to quantify health effects of emissions from road traffic in Malmö, corresponding to a policy on on-road transportations without tail-pipe emissions, further referred to as exhaust-free transport, in the city of Malmö. More specifically, such a policy would imply a complete transition to electric vehicles, or replacing car transports with somatic energy transports such as cycling or walking. In line with that, our aim was to estimate health impacts of such a policy to assess the possible decreased health effects on citizens exposed to tail-pipe exhausts.

## 2. Materials and methods

### 2.1. Study area

The area of study is Malmö municipality. Malmö is Sweden's third largest city with a population of approximately 330,000 (326092) inhabitants. On-road transports and non-road mobile machinery are the largest sources of $NO_2$ emissions in Malmö, together they stand for 69% of the 2756 tons of $NO_2$ that were emitted in 2016 (Spanne et al., 2017). Around 5000 annual deaths per year can be attributed to air pollution in Sweden. Malmö is one of the cities with the highest levels of air pollution in Sweden (Gustafsson et al., 2014). For many years, the city of Malmö has exceeded the Swedish Air Quality Standards for daily concentrations of $NO_2$ (not > 60 μg/$m^3$ $NO_2$ for 7 days/year). The city has made efforts to tackle these problems with a mandatory air quality action plan adopted in 2007 and revised 2011 (Spanne et al., 2017). Compliance with the Swedish air quality standard for daily averages was achieved for the first time in 2014 and in 2016 $NO_2$ levels only exceeded 60 μg/$m^3$ for 4 days (Spanne et al., 2017). It should be emphasized that the air quality in Malmö is generally well within the present-day annual WHO air quality guideline value of 40 μg/$m^3$ (WHO, 2005) (Fig. 1). The annual mean $NO_2$ concentration was 14 μg/$m^3$ in urban background at the City Hall (Rådhuset) monitoring station and 30 μg/$m^3$ at the roadside location of Dalaplan in 2016 (Spanne et al., 2017). The $NO_2$ levels at the regional background site were 3 μg/$m^3$ in 2016. For particulate matter < 2.5 μm in aerodynamic diameter ($PM_{2.5}$) the concentration 2016 are also generally well within current air quality guidelines (annual mean 25 μg/$m^3$) with a concentration of 9 μg/$m^3$ at urban background site and of 12 μg/$m^3$ at a roadside site (Spanne et al., 2017).

### 2.2. Modelling hypothetical changes in air pollution concentrations

To calculate the hypothetical changes in air pollution concentrations of exhaust-free transport in Malmö municipality, we used a Gaussian dispersion model (AERMOD) combined with an emission database (EDB) with > 40,000 sources. Emissions from surrounding areas such as shipping emissions in the Oresund and emissions from Sealand, Denmark, are included in the EDB. To account for background levels from sources that are more distant, we added the regional background mean levels of 3 μg/$m^3$. Dispersion models are based on detailed knowledge of dynamical processes in the atmosphere by incorporating information on emissions and source characteristics, with meteorology to predict ground level concentrations (Holmes and Morawska, 2006). Gaussian dispersion models often rely on a Gaussian plume equation and uses data on emissions, meteorology and pollution concentrations to estimate spatial distribution of pollutant concentrations (Gilliland et al., 2005; de Hoogh et al., 2014). The Gaussian dispersion model was originally designed as an air quality management tool, but has been used widely for estimating long-term exposures (de Hoogh et al., 2014). The advantage of a dispersion model in this context is that we can amend emissions from certain sources to account for policy or planning processes. Dispersion models were used to calculate concentrations of Nitrogen oxides ($NO_x$), $PM_{2.5}$ with high spatial and temporal resolution (85 m and 1 h, respectively).

To convert $NO_x$ to $NO_2$ we used a formula based on empirical relationships between measured $NO_x$ and $NO_2$ levels in different environments and over different periods of time for South West Skåne, mainly in Malmö ((Naturvårdsverket), 2005). In this report, a basic formula was formulated. Since 2005, the formula has been refined and the parameters in Malmö have changed in order to obtain a better model. This formula was used here:

$$NO_2 = NO_x^{\left(0.72+\left(\frac{28}{NO_x}+142\right)\right)}$$

The EDB used and dispersion programs are owned by the city of Malmö. We have previously used this model in many epidemiological studies (Oudin et al., 2009; Malmqvist et al., 2013; Malmqvist et al., 2017). Modelled levels have also been compared to measured levels at residential facades with good correlations for $NO_x$ (N = 241, Spearman correlation of 0.8, $p < 0.001$) (Stroh et al., 2012). For $PM_{2.5}$, the $R^2$ for modelled vs measured level correlations was 0.86 (N = 96) in Malmö (Malmqvist et al., 2016). Individual air pollution concentrations were calculated with OPSIS EnviMan, which is a series of software modules for management of environmental information.

The dispersion model also enables modelling of specific pollutants by stratifying/excluding emissions from on-road traffic tail-pipe exhaust, wear and tear from tire and roads, marine shipping, wood smoke from residential heating, large-scale incinerators and long-range transported pollutants. Even if tail-pipe emissions were zero, vehicles produce particles from for example brake, wear and tear of tires and road surfaces. However, particles from wear and tear mainly do not affect the $NO_2$ or the $PM_{2.5}$ levels, but mostly the $PM_{10}$ levels. Since we based our calculations on $PM_{2.5}$ and $NO_2$, and not $PM_{10}$, particles from brake, wear and tear of tires and road surfaces were thus not relevant for our calculations.

### 2.3. Health impact assessment

We linked modelled air pollution concentrations to our study populations by residential geocodes of each individual. We used the whole population residing in the city of Malmö in 2016 (326,092 persons) for the analysis. Data on individual residential geocodes, age and sex were retrieved from Region Skåne (a self-governing administrative region responsible for health care). As descriptive statistics, we calculated the average individual hypothetical air quality change.