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Impacts on air pollution and health by changing commuting from car to bicycle

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Highlights

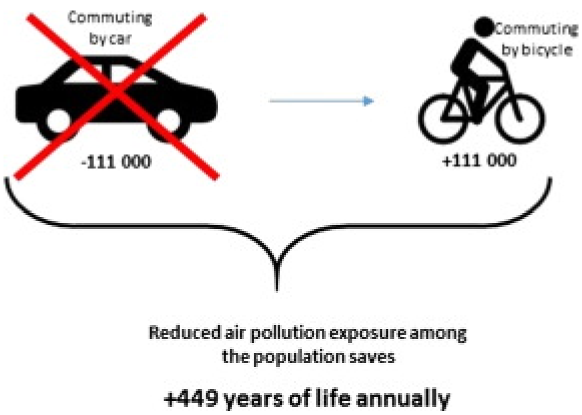
- A very large potential for transferring car commuters to cycling; more than 111 000 car commuters shifting.
- Reduced vehicle emission and thereby reduced population exposure, saves 449 years of life annually in Stockholm County.
- This is more than double the effect estimated in connection with the introduction of congestion tax in Stockholm.

Abstract

Our study is based on individual data on people's home and work addresses, as well as their age, sex and physical capacity, in order to establish realistic bicycle-travel distances. A transport model is used to single out data on commuting preferences in the County Stockholm. Our analysis shows there is a very large potential for reducing emissions and exposure if all car drivers living within a distance corresponding to a maximum of a 30 min bicycle ride to work would change to commuting by bicycle. It would result in > 111,000 new cyclists, corresponding to an increase of 209% compared to the current situation.

Mean population exposure would be reduced by about 7% for both NO_x and black carbon (BC) in the most densely populated area of the inner city of Stockholm. Applying a relative risk for NO_x of 8% decrease in all-cause mortality associated with a 10 μg m^{-3} decrease in NO_x, this corresponds to > 449 (95% CI: 340–558) years of life saved annually for the Stockholm county area with 2.1 million inhabitants. This is more than double the effect of the reduced mortality estimated for the introduction of congestion charge in Stockholm in 2006. Using NO_2 or BC as indicator of health impacts, we obtain 395 (95% CI: 172–617) and 185 (95% CI: 158–209) years of life saved for the population, respectively. The calculated exposure of BC and its corresponding impacts on mortality are likely underestimated. With this in mind the estimates using NO_x, NO_2 and BC show quite similar health impacts considering the 95% confidence intervals.

Graphical abstract



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Keywords

Air pollution; Vehicle emissions; Road traffic; Human health; Population exposure; Mortality; Cycling

1. Introduction

Road vehicle emissions are one of the most important sources of human exposure to air pollution. Depending on pollutant, mode of travel, travel distance etcetera, the exposure while commuting during rush hours along densely trafficked corridors may constitute a substantial fraction of the total daily exposure (e.g. Hänninen et al., 2004; Barrett et al., 2008; Dons et al., 2012). High exposures occur both inside vehicles due to the proximity of air intakes to exhaust emissions from neighboring vehicles as well as while walking or biking alongside the roads (Dons et al., 2012).

In the last years there have been attempts to develop estimates of the overall impact of transferring journeys from car to bicycle (de Hartog et al., 2010; Lindsay et al., 2011; Rojas-Rueda et al., 2011; Grabow et al., 2012). A Dutch study quantified the potential impact on all-cause mortality in 500,000 people that would make a transition from car to bicycle for a 7.5 or 15 km commute (de Hartog et al., 2010). In a similar study in Barcelona the change in cyclist exposure to exhaust was estimated (Rojas-Rueda et al., 2011). A study from New Zealand (Lindsay et al., 2011) shifting 5% of the vehicle kilometers to cycling, and an American study shifting 50% of car trips < 8 km to cycling (Grabow et al., 2012), both included estimates also of how the general population's health would benefit from reduced exhaust emissions.

These studies discuss specific cities, but all use very hypothetical scenarios and journeys. In an even more general European perspective, the benefits were estimated per individual driver who switches to active transport (5 km for bicycling and 2.5 km for walking) (Rabl and de Nazelle, 2012). Even if the published health impact assessments generally estimate very large potential benefits for commuters, the population wide benefits and interactions are not so well described (de Nazelle et al., 2011; Teschke et al., 2012).

Based on the results of a national travel survey in Sweden with thirty-nine thousand interviews conducted on a daily basis during 2011–2014 (Trafikanalys, 2015), we calculate that 51% of all car trips were shorter than 7.5 km. This means that the potential to shift car drivers to bicycles should be large. However, common objections relate to the Nordic climate, increased dose of traffic pollutants and injuries among cyclists, and especially, limited interest in the segments of the population who does not use a bicycle.

The main objective of this work is to assess the effect on emissions and population exposure of transferring car commuters to cyclists. Earlier studies on this matter have been based on hypothetical scenarios. Our scenario is based on detailed information on the individuals' home and work addresses, empirical data to establish which distances are reasonable to travel by bicycle and a transport model to single out data on commuting preferences in the County of Stockholm. This provides us with a possibility to demonstrate an integrated environment and health impact assessment built on realistic assumptions. This is useful for policy making and interventions.

2. Methods

Fig. 1 illustrates the different steps in the calculation of the population and commuter exposure and each step is described in the following sections. The basic data used for identifying individuals who can shift from car to bicycle are: 1) individual data on home and work addresses, age, sex and car ownership, 2) travel times and travel costs, and 3) traffic network data and vehicle fleet emission factors. The calculation steps are:

i) identify current volume of car commuting and the distance from home to workplace if these trips were made by bicycle based on the Astrid database and a travel demand model (LuTrans), both described later,

ii) calculate travel times by bicycle for current car commuters depending on sex and age considering their physical capacity based on physical capacity modelling,

iii) identify commuters with a travel time by bicycle of less than or equal to 30 min,

iv) calculate the new traffic flows, where remaining traffic may choose a different route based on the LuTrans travel demand model,

v) calculate spatially resolved reduction in air pollutant emissions and concentrations due to shifting car to bicycle commuting based on emission factors and the change in traffic due to less car commuters (here we take into account that the number of car commuters is slightly higher than the number of drivers (i.e. cars), why it is possible that more car commuters would shift to cycling than we now assume),

vi) calculate change in population weighted average exposure of the general population based on home address and spatially resolved concentrations,

vii) calculate the number of premature deaths avoided based on change in population exposure and exposure response functions for different pollutants, and

viii) calculate years of life gained for the population based on life table statistics for the population.

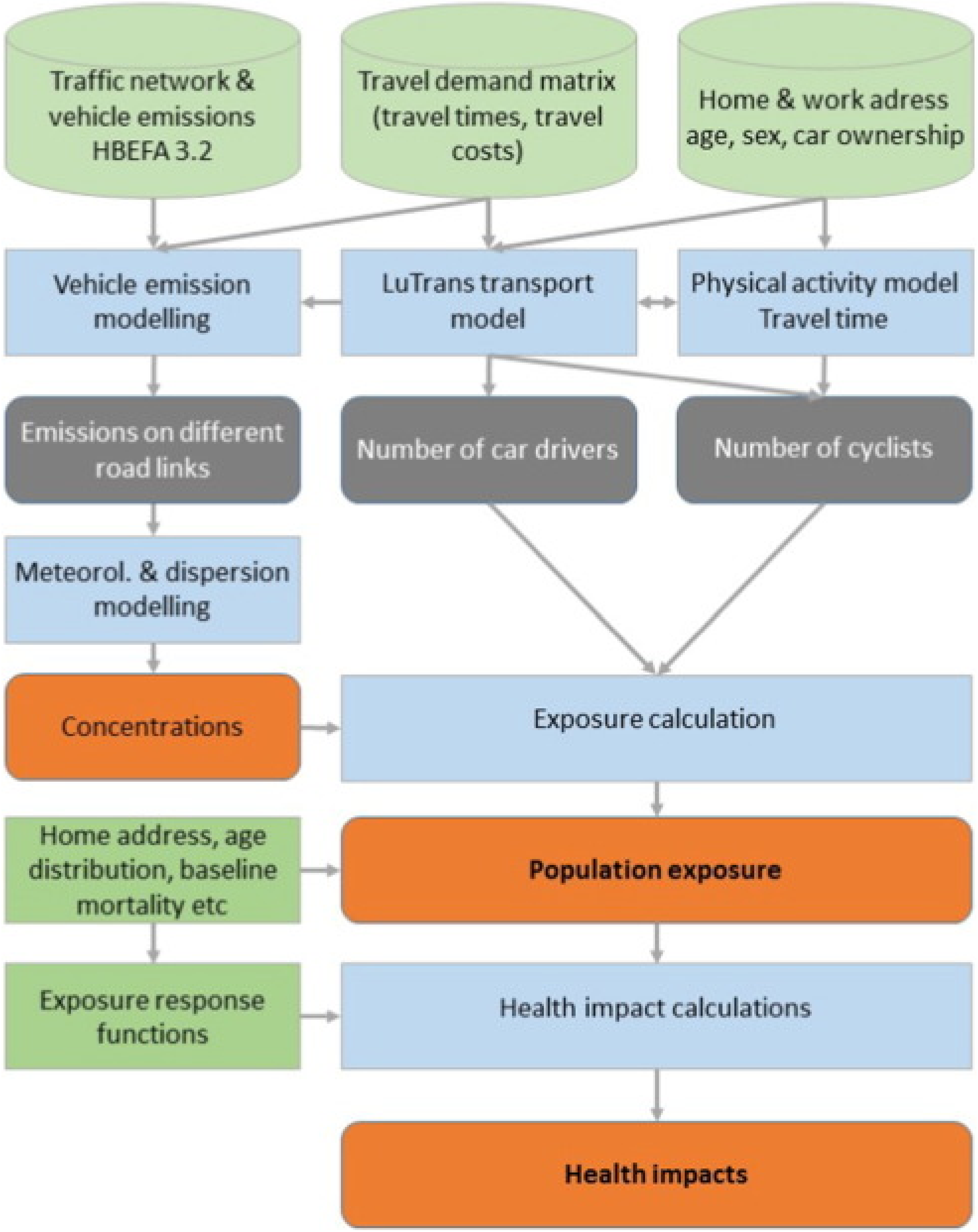

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Fig. 1. Illustration of calculation pathways, models and databases to obtain emissions, population and commuter exposures and health impacts. HBEFA 3.2 is the Handbook Emission Factors for Road Transport, version 3.2 (HBEFA, 2014).

Below we describe the models and data bases used in each step.

2.1. Scenario building through modelling of traffic flow and expected individual bicycle speed

2.1.1. Current modes of travel

Travel survey data was used to obtain an estimate of the proportion currently traveling to work with each mode of transport; walking, bicycling, public transport and car. These proportions were estimated on a fairly high resolution of combinations of living and work areas, where the size of each statistical area depend on the population density but also considering natural division of neighborhoods. Individual information on age, gender, home and work address and car ownership was obtained from the ASTRID database (Stjernström, 2011). This data was linked with the LuTrans model (Jonsson et al., 2011) together with data on traffic flows on roads. The LuTrans model is regularly calibrated based on traffic counts and the travel output is modelled as a logit model of: 1) travel survey data allocating individual trips to different modes of transport and 2) traffic counts to allocate car tips to specific car routes. The output is traffic flow on each link in the model, where a link is defined as the connection between two major intersections in the road network. In the present form there are auto links and public transport links included in the model. We allocated all study subjects a current mode of transport between home and work place.

2.1.2. Duration-distance relations considering physical capacity

As mentioned above, the methodology to obtain realistic duration-distance relations involves several steps, which are described in detail by Schantz et al. (2017). The first step was to establish the duration-distance relations in about 450 cycle commuters. For that purpose, the participants drew their own normal cycle commuting route to work on a map, and its distance was measured using a criterion method (Schantz and Stigell, 2009). These commuter cyclists were part of a larger group which has been described in detail by Stigell and Schantz (2015). Only cyclists with the last digit values of 1–4 and 6–9 in their duration reports (based on full number of minutes) were used for establishing the duration-distance relations. This is since such reports represent close to valid duration values (Kelly, 2013; Schantz, 2017). In this way linear duration-distance relations were established for each sex; for males: distance (*D*, *km*) = 0.347 km/min · time (*T*, *min*), and for females: distance (*D*, *km*) = 0.268 km/min · time (*T*, *min*).

Cycling speed is related to the maximal oxygen uptake, and therefore we measured the maximal oxygen uptake of 20 commuter cyclists, and compared their values with the age-matched values obtained within the normal population in the year of 1990 and 2000. The latter measurements were based on about 1700 individuals, aged 20–65 years. Since we wanted to evaluate the potential for commuter cycling in the present Swedish population, we added the secular increases in body weight up to the year of 2015 to the original individual data from 1990 and 2000. For that purpose, the secular body weight development in the population was established for the period 1988–2013. Those calculations were based on about 150,000 individuals.

It was noted that the commuter cyclists had significantly higher maximal oxygen uptake levels than the age matched values in the normal populations (Schantz et al., 2017). This difference between the sample of commuter cyclists and the general population led to a need of correction factors (a cycle commuter to population effect; for males: 0.717; and for females: 0.752) downscaling the duration-distance relation of the cycle commuters to population values with about 25–28%.

Due to that the maximal oxygen uptake decreases with age (Åstrand, 1960; Åstrand and Rodahl, 1970), we introduced age correction factors for the duration-distance relations. For that purpose we used the combined population data from 1990 and 2000, and created a relative age index.

The empirical formula predicting the distance (*D*, *km*) based on duration (*T*, *minutes*) and age (*A*, *years*) for males cycling in the age span of 20–65 years is:

$$D = T \cdot 0.347 \ \mathrm{km}/\mathrm{min} \cdot 0.717 \cdot (1.612 - 0.0142 \cdot A)$$

where the factor 0.717 reflects the cycle commuter to population effect (see above).

For females in the same age span, the formula is:

$$D = T \cdot 0.268 \ \mathrm{km}/\mathrm{min} \cdot 0.752 \cdot (1.532 - 0.0123 \cdot A)$$

where 0.752 reflects the cycle commuter to population effect.

2.1.3. Alternative scenario

Implementation of these duration-distance relations on individual data of home and work address, age and gender contained in the ASTRID database (Stjernström, 2011) identified individuals that have the potential to cycle to work within 30 min. ASTRID is a longitudinal, georeferenced database with individual-level demographic and socio-economic data for the entire Swedish population. The database serves as the basis for research on dynamic population development and contains detailed information about individuals' economic and social conditions, and geographic information about work and housing. The coordinates for the home and workplace address were extracted and the shortest path along a network of possible roads and bicycle paths were retrieved. If the individual was determined to have the potential to bicycle to work within 30 min based on age and gender, and the individual was previously allocated as traveling to work by car, the individual will in the alternative scenario shift to travel by bicycle.

The LuTrans traffic model (Jonsson et al., 2011) was used to model traffic flows in the alternative scenario where car trips has been transferred to bicycle. A demand matrix is used to estimate the route for each car trip, and since the demand within the road system decreases due to reduced number of cars, a new traffic flow is estimated where remaining traffic may choose a different route. The result of such as change will be at the road link level, which thereafter will be used to calculate vehicle emissions and traffic pollution exposure.

A more detailed description of the methodology for the considered alternative scenario and resulting changes in traffic has been described by Strömgren et al. (2017).

2.2. Vehicle emissions

The emission inventory for the county of Stockholm includes some 40,000 road links and an annual traffic volume of 12,000 million vehicle km's (LVF, 2015). NO_x, PM-exhaust and black carbon (BC) emissions from road traffic are described with emission factors (grams per km driven). Vehicles are grouped into passenger cars (petrol and diesel), light commercial vehicles, heavy goods vehicles and buses. Emission factors of NO_x for different vehicle types, speeds and driving conditions are calculated based on the Handbook Emission Factors for Road Transport, version 3.2 (HBEFA, 2014). BC emission factors are based on the Transphorm project (Transphorm, 2013). Vehicle emission factors for the fleet composition in the area are for the year 2013.

2.3. Dispersion and exposure modelling

The concentrations and exposures with and without the car-to-bicycle scenario are compared using the same meteorological conditions, i.e. only changing the emissions due to the change in commuter's transports. The concentrations due to local road traffic emissions are calculated using a wind model and a Gaussian air quality dispersion model, both part of the Airviro Air Quality Management System (SMHI, Norrköping, Sweden; http://airviro.smhi.se). The system has been used in Stockholm during > 20 years and it has provided exposure estimates for several epidemiological studies and health impact assessments (Nyberg et al., 2000; Bellander et al., 2001; Johansson et al., 2007; Johansson et al., 2009; Meister et al., 2012; Olsson et al., 2015; Orru et al., 2015).

Meteorological input for the dispersion model is based on a climatology created from 15 years of meteorological measurements (15 min averages) in a 50 m high mast located in the southern part of Stockholm. The climatology consists of a list of hourly events, each of them with a certain frequency of occurrence, which together will yield a distribution of different weather conditions that is similar to the distribution of the full scenario period (for further details, see Johansson et al., 2007). A diagnostic wind model (Danard, 1977) is used to obtain the wind field for the whole model domain considering variations in land-use and topography. This concept assumes that small scale winds can be seen as a local adaptation of large scale winds (free winds) due to local fluxes of heat and momentum from the sea or earth surface. Any non-linear interaction between the scales is neglected. It is also assumed that the adaptation process is very fast and that horizontal processes can be described by non-linear equations while the vertical processes can be parameterised as linear functions. The large scale winds as well as vertical fluxes of momentum and temperature are estimated from profile measurements in one or several meteorological masts (called principal masts).

The dispersion calculations are performed on a 100 m resolution over three different areas as shown in Fig. 2. The effects of buildings on the dispersion are considered using a street canyon model part of the Airviro system. The concentrations alongside streets are used to estimate the exposure dose for people biking instead of driving a car.

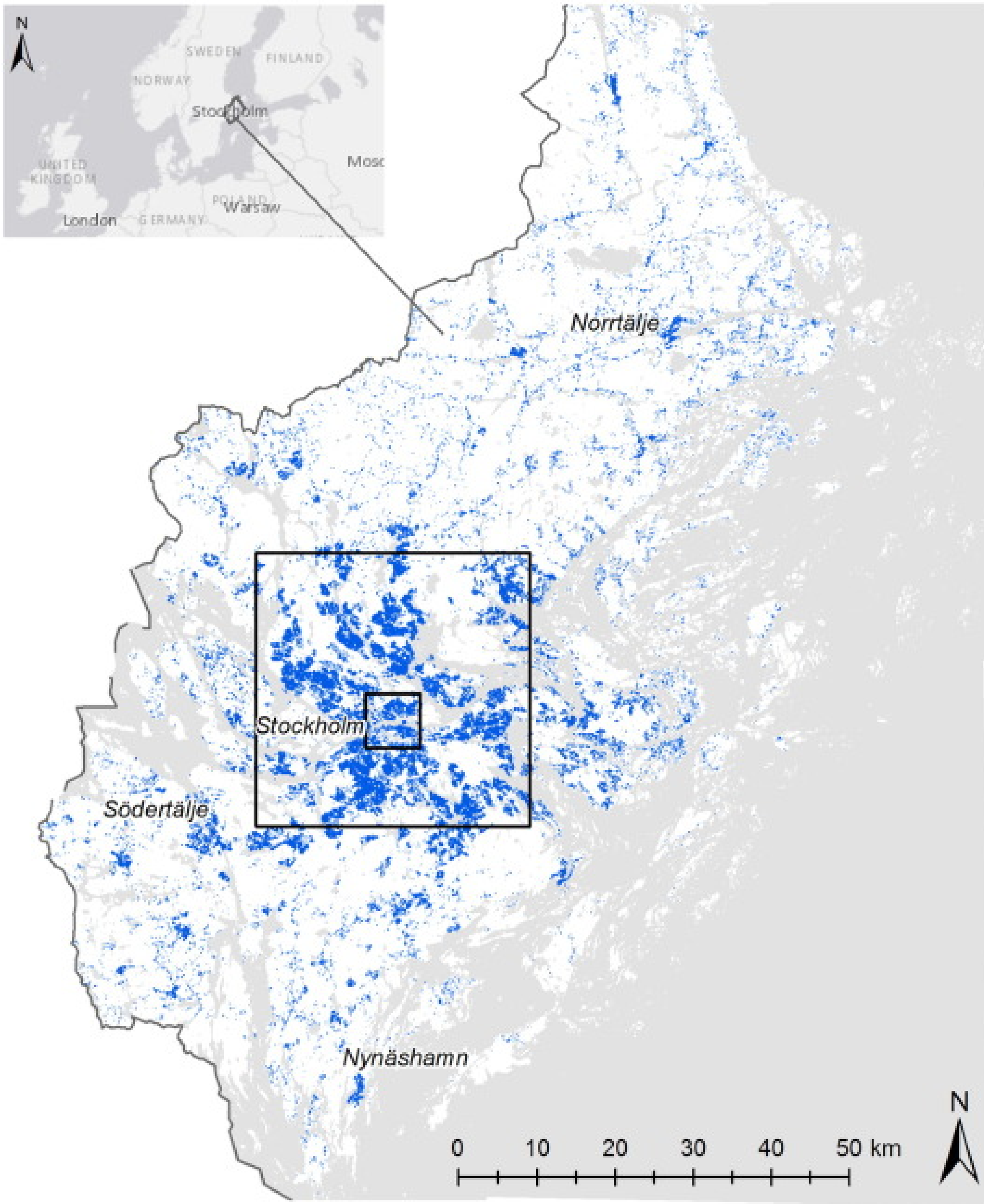

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Fig. 2. Calculation domains. The whole area is the County of Stockholm, the large square is the Greater Stockholm and the small square the inner City of Stockholm. Populated areas are indicated by blue colours. The resolution is the same for all areas, namely 100 m.

Population weighted exposure, C_{Pop}, is calculated based on home address as:

$$C_{Pop} = \frac{\sum C_i P_i}{\sum P_i}$$

where C_i and P_i are concentrations and population in each calculation grid cell (122,500 square cells, each cell is 100 m times 100 m) in the area studied. Age segregated (10 year classes) population data are for 2011 from Statistics Sweden (SCB).

2.3.1. Uncertainties in dispersion modelling

Uncertainties in calculated emissions and concentrations is part of the overall uncertainty in the dispersion modelling, and it is difficult to separate the error due to vehicle emissions and dispersion modelling. Dispersion model calculated concentrations have earlier been compared with measurements of NO_2 by Johansson et al. (1999) and Eneroth et al. (2006). Based on the data of Johansson et al. (1999), who reported measured and modelled annual mean NO_2 concentrations at 16 sites in the county of Stockholm, R^2 is 0.93 and the relative root-mean-square error (RMSE) of 23%. Eneroth et al. (2006) found R^2 to be 0.71 and relative RMSE 35%, when comparing model calculations with diffusion tube measurements (519 weekly samples) at fixed points within the Greater Stockholm area. The mean and standard deviation of measurements in Eneroth et al. (2006) were 25.0 ± 1.0 μg m^{-3} and for the model calculations 21.4 ± 0.7 μg m^{-3}.

This paper focuses on the change in annual concentrations and population exposure due to the change in emissions with less car commuting, i.e. they relate to a situation with compared to a situation without the car commuting. In this case, calculation errors are not due to different meteorological conditions, emissions other than road traffic and background concentrations (that represent contribution of sources outside the calculation domain) do not influence the conclusions. The main uncertainty in this methodology lies in the estimated change in traffic and its emissions.

2.4. Health impact calculations

To estimate health impacts we have used NO_x, NO_2 and BC exposure as indicators of adverse health due to exposure to vehicle exhaust. For NO_x we used a Norwegian study of 16,000 men from Oslo, of whom 25% died during the follow up, which used modelled NO_x in the residential area as the exposure indicator (Nafstad et al., 2004). This cohort, with people of between 40 and 49 years of age at the start of the study, was followed from 1972/73 through 1998. NO_x was estimated in a model with 1000 m grids, and a street contribution added for the largest streets. When the median concentration of NO_x for 1974–78 was used (10.7 μg m^{-3}), the relative risk for total mortality was 8% per 10 μg m^{-3} (95% CI 6%–11%). A Swedish study obtained similar results for men in Gothenburg aged 48–52 years of age at the start of follow up 1973, non-accidental mortality increased by 6% (95% CI 3%–9%) per 10 μg m^{-3} NO_x (Stockfelt et al., 2015).

There are more studies using NO_2 as indicator of mortality. Therefore we have also used Faustini et al. (2014) whom reviewed epidemiological studies on long-term associations between NO_2 and total mortality. The pooled effect based on 23 studies from 2004 to 2013 was 4% (95% CI 2%–6%) increase in mortality for an increase of the annual NO_2 concentration of 10 μg m^{-3}. For European studies the pooled estimate was 7% (95% CI 3%–10%).

For BC we used the pooled estimate for premature mortality associated with long-term exposure to elemental carbon of 6% per 1 μg m^{-3} (95% CI 5%–7%) as reported in a review by Hoek et al. (2013).

We have used life table analysis based the WHO AirQ + software tool for health risk assessment of air pollution (WHO, 2016) to calculate years of life gained due to reduced air pollution exposures among the general population. The baseline mortality for individuals older than 30 years of age in the county of Stockholm for 2013 is 1124 per 100,000 (NBHW, 2013). The same age specific mortality as for the county was applied for the other subareas (Greater Stockholm and Inner City of Stockholm).

3. Results

3.1. Effects on commuting by bicycle and car

The mean travel distance depends on age and sex. Since we have exact locations of home and workplace addresses we can calculate the exact travel distances for the scenario, i.e. for the travel by bicycle between home and workplace for individuals that can make the trip within a maximum of 30 min. The scenario resulted in 111,487 more cyclists, an increase of 210% compared to the current situation (Table 1). Of all new cyclists 52% were men and the mean age of all cyclists was 42 years. Average one-way cycle distances and durations were 3.7 km and 14 min among men and 3.1 km and 15 min among women.

Table 1. Number of commuters for different ravel modes and number of cars in the current situation and in the maximum 30 min bicycle scenario.

Mode of travel	Current situation (% of all commuters)	Scenario (% of all commuters)	Scenario minus current
Bicycle	53,206 (6%)	164,693 (18%)	+ 111,487 (+ 210%)
Car (drivers)	352,614 (38%)	241,127 (26%)	− 111,487 (− 32%)
Car (passengers)	35,297 (4%)	35,297 (4%)	0
Public transport	352,412 (38%)	352,412 (38%)	0
Walking	130,441 (14%)	130,441 (14%)	0

From Table 1 it can be noted that 18% of all commuters would be cyclists in the 30 min scenario compared to 6% in the current situation. Likewise the share of car drivers decreases from 38% to 26%. We have assumed that it is car drivers that will shift to cycling, car passengers are here not assumed to become cyclists (but to be passengers in another car, see further under Discussion).

The largest increase in number of cyclists occurs in the inner City of Stockholm, were distances between home and workplace in general are shorter. Streets in the inner city are estimated to have up to 2600 more cyclists per day. This corresponds to 4.5% of the total number of cyclists (58370) passing into and out of the inner City of Stockholm every day in 2015 (Stockholm Traffic Administration, 2015).

3.2. Effects concentrations and population exposure

Fig. 3 shows the geographical distribution of the change in annual mean NO_x concentrations for Greater Stockholm and the inner City of Stockholm if the 30 min scenario is realized. The maps show that the largest reductions in NO_x concentrations occur in the inner City of Stockholm. Here the concentration is reduced by up to 5 μg m^{-3}. This corresponds to a 20% reduction in the traffic contribution to NO_x concentrations in the inner City of Stockholm. There is a slightly different geographical distribution of the traffic in the 30 min scenario and this leads to increased emissions and thereby concentrations in some small areas (up 1.7 μg m^{-3}), but overall there is a reduction. The spatial variation is almost identical for NO_x and BC due to the same types of vehicles contributing to the emissions and concentrations.

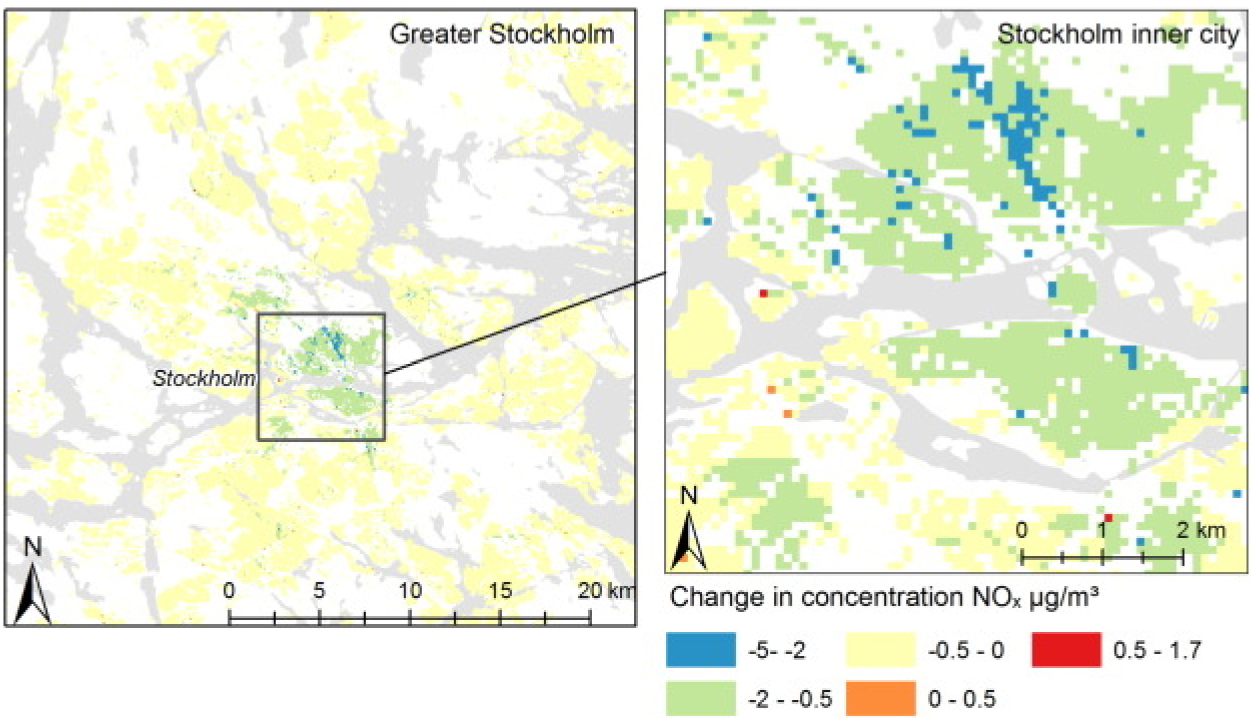

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Fig. 3. Calculated reductions in NO_x concentrations in the county and inner city of Stockholm with the 30 min scenario compared to the current situation.

Table 2 shows the annual mean population weighted concentrations of NO_x and BC for the current situation and the 30 min cycling. As can be seen, the weighted concentrations increase going from the whole county to the inner City of Stockholm. Comparing the county and the inner city, the increase in population weighted concentrations is around a factor two reflecting the higher density of emissions and population in the inner City of Stockholm compared to the whole county. The mean reduction in NO_x concentration for the County, Greater Stockholm and Inner city of Stockholm area is 6.5%, 6.6% and 8.4%, respectively (Table 3). Corresponding values for BC are similar.

Table 2. Annual mean population weighted concentrations in the different areas (μg m^{-3}).

Area	NO_x			BC		
	Current situation	30 min scenario	Difference	Current situation	30 min scenario	Difference
County of Stockholm	5.20	4.87	0.33 (6.5%)	0.350	0.327	0.023 (6.4%)
Greater Stockholm	6.02	5.62	0.40 (6.6%)	0.410	0.383	0.027 (6.5%)
Inner City of Stockholm	10.5	9.65	0.85 (8.4%)	0.742	0.682	0.060 (8.1%)

Table 3. Annual number of premature deaths and years of lives gained for the different calculation areas using NO_x, NO_2 and BC as indicators of health effects.

Area	Population	NO_x		NO_2		BC	
		Pre deaths	YLG	Pre deaths	YLG	Pre deaths	YLG
County of Stockholm	2,086,993	63 (CI: 47–87)	449 (CI: 340–558)	55 (CI: 24–79)	395 (CI: 172–617)	32 (CI: 26–37)	185 (CI: 158–209)
Greater Stockholm	1,628,528	58 (CI: 43–80)		51 (CI: 22–72)		29 (CI: 24–34)	
Inner City of Stockholm	398,742	31 (CI: 24–43)		27 (CI: 12–39)		16 (CI: 13–19)	

3.3. Impacts on premature mortality

Estimated number of avoided premature deaths associated with reduced exposures of NO_x, NO_2, and BC is presented in Table 3. The estimates are based on relative risks from the different epidemiological studies as described earlier. The estimate for NO_2 is based on the assumption that the exposure reduction will be equal to the reduction calculated for NO_x (see further below under Discussion).

Using NO_x as indicator of health impacts we obtain 63, 58 and 31 avoided premature deaths annually for the whole county of Stockholm, the Greater Stockholm area and inner City of Stockholm, respectively. The corresponding numbers when using NO_2 are slightly lower and when BC is used as indicator the numbers are about half of those using NO_x.

We have estimated the long-term effects on years of lives gained (YLG) because of reduced vehicle pollution exposure. This is done only for the county as we don't have death statics for the other areas. Using NO_x as indicator the 449 years of lives is gained annually. Using NO_2 and BC as indicator, we get 395 and 185 YLG, respectively.

4. Discussion

There are many studies assessing the impact on air pollution and health of increased biking. Generally, these studies use hypothetical scenarios and journeys, e.g. assuming some percentage or all of the commuters switching from car to bicycle and not knowing the exact pathways (Rojas-Rueda et al., 2011; Mueller et al., 2015). In our study we have used detailed information on individuals living addresses and places of work, travel distances, traffic modelling and physical capacity for cycling, etc. to obtain as accurate as possible information on the number of potential cyclists and taking into account the changes in geographic distribution of traffic emissions and population exposures. We also include only persons with a registered workplace (excluding e.g. unemployed and disability retirees). Individuals that are able to drive a car but too disabled to make a short bicycle trip, must represent a very minor part of the working population of Stockholm.

In their systematic review, Mueller et al. (2015) compared 17 studies which included health impacts due to changes in physical activity, traffic accidents, air pollution exposure to the general population and exposure to active travelers. Most of the studies (15) showed small health benefits to the general population due to reduced air pollution exposure, only a few percent of the benefits related to physical activity. Only two studies showed relatively large estimated health impacts associated with reduced air pollution exposures (Grabow et al., 2012; Dhondt et al., 2013). In the study by Grabow et al. (2012) they replaced 50% of car round-trips ≤ 8 km with bicycle and found that the health benefits due to reduced exposure for the general population was similar to the benefits of increased physical activity. In the case of Dhondt et al. (2013) they based their scenario on an increased fuel price of 20% and found the mortality benefits due to reduced population exposure to be > 2 times larger than physical activity.

Part of the reason why Dhondt et al. (2013) got a relatively large health impact of reduced exposure compared to other studies is likely that they used EC (elemental carbon) as proxy for health impacts. Most of the earlier studies estimate health impacts based on an exposure response function (ERF) for $PM_{2.5}$ (Mueller et al., 2015). The value used, 6% increase per 10 μg $PM_{2.5}$ m^{-3}, is mainly derived from large cohorts across regions reflecting differences in health effects based on urban background monitoring, and to a large degree influenced by secondary (non-local) particulate matter (e.g. Hoek et al., 2013). Dhondt et al. (2013) calculated 5 times less YLG when $PM_{2.5}$ was used instead of EC.

In our study we have used NOx, NO_2 and BC as indicators of health impacts. All three are indicators of adverse health effects associated with vehicle exhaust emissions. Especially NO_x and BC are highly correlated with other toxic constituents in vehicle exhaust and therefore it is difficult to judge independent effects of either indicator. Both NO_x and BC have been shown to be associated with increased premature mortality (Nafstad et al., 2004; Janssen et al., 2011; Grahame et al., 2014). Epidemiological studies with a finer spatial resolution which can capture the gradients in exposure to local traffic pollutants indicate an important effect of local traffic emissions, resulting in high relative risks (Roemer and van Wijnen, 2001, Hoek et al., 2002). NO_x is a good marker for vehicle exhaust particles in Stockholm as indicated by the high correlations between NO_x concentrations and total particle number concentrations at kerb-side sites (Johansson et al., 2007; Gidhagen et al., 2003) and close to a highway (Gidhagen et al., 2004).

We have used both NOx and NO_2 as indicators of health effects and we obtain slightly lower effects on mortality when NO_2 is used. For NO_x the relative risk estimate from Nafstad et al. (2004) is confirmed by a similar estimate for a Swedish cohort only slightly older at the start of follow up than participants in the study from Oslo (Stockfelt et al., 2015). But there is much more evidence of a long-term effect of NO_2 and the coefficient found by Nafstad et al. (2004) for NO_x, is in line with many other studies using NO_2 as indicator. Based on a meta-analysis of long term studies on mortality associated with exposure to NO_2, Faustini et al. (2014) for European studies obtained a pooled effect on mortality of 6.6% (95% CI 2.9%–10.4%) per 10 μg m^{-3} increase in annual mean NO_2 concentration. In a study in Stockholm of the incidence of lung cancer in men, the relative risk was found to be 12% for a 10 μg m^{-3} increase in modelled concentration of historic NO_2 levels at the home address (Nyberg et al., 2000). The model used in that study was very similar to the one used in this paper. A large fraction of NO_x (> 50% in the urban background) is in the form of NO_2 and they are highly correlated in urban areas. But the relation between NO_x and NO_2 is not linear. In fact NO_x is a better marker for vehicle exhaust particles than NO_2, which depends on ozone levels. At low NO_x concentrations ozone is in large excess and almost all NO is oxidized to NO_2, but when NO_x concentrations are very high, ozone is being depleted, and only a fraction of the NO is oxidized. This means that NO_2 concentrations are not linearly proportional to (potentially toxic) vehicle exhaust emissions. In addition, several studies (e.g. Carslaw, 2005; Carslaw et al., 2007) point out that the NO_2/NO_x ratio (as well as the NO_2 to exhaust particle ratio) from road transport sources has increased in recent years making NO_2 a less suitable indicator of exhaust particles. Therefore we believe that NO_x is a better indicator for health risks associated with vehicle exhaust exposure than NO_2.

We have also used BC, which is an additional air quality indicator to evaluate the health risks associated with air pollution exposure, especially for primary combustion particles (Janssen et al., 2011). Cohort studies provide sufficient evidence of associations of all-cause and cardiopulmonary mortality with long-term average BC exposure (Janssen et al., 2011; Grahame et al., 2014). Studies of short-term health effects show that the associations with BC are more robust than those with $PM_{2.5}$ or PM_{10}, suggesting that BC is a better indicator of harmful particulate substances from combustion sources (especially traffic) than undifferentiated PM mass (e.g. Olstrup et al., 2016).

Compared to NO_x we obtain much lower impacts on mortality and years of lives gained when BC is used. Partly this can be due to uncertainties in emission factors for BC. Recently we have compared the BC emission factors for diesel and gasoline vehicles in Stockholm based on Transphorm (see Methods) with real world measurements in a busy street canyon (Krecl et al., 2016). This analysis shows that the emission factors from Transphorm that we have used here seems to be too small indicating that the health impacts using BC would be larger and quite similar to using NO_x as indicator.

Our results using NO_x as indicator of the effect on mortality show that the car-to-bicycle scenario is even more beneficial than the effects of introducing the congestion charge in Stockholm (Johansson et al., 2009). One of the motives for the congestion charge in Stockholm was to improve air quality and its impact on health. Dispersion modelling of population exposure using the same methodology as in this study showed upon 27 fewer premature deaths per year using the same relative risk as in this study. For the same area and same indicator, our 30 min cycling scenario is estimated to save 63 premature deaths. This indicates that policies promoting car commuters to change to biking can potentially be very efficient for improving air quality and its impact on peoples' health.

In addition to the impacts on mortality, traffic pollution has been associated with several other adverse health effects, like for example respiratory and cardiovascular morbidity, cognitive impairment and pregnancy outcomes (Thurston et al., 2017). Reduced traffic will also reduce noise and emissions of climate related pollutants.

Our 30 min scenario resulted in rather modest biking distances for most commuters. Given that clearly longer cycling durations have been reported among existing cycle commuters in Greater Stockholm (Stigell and Schantz, 2015), it is likely that there is a greater potential than shown with the 30 min scenario. We assume that all car drivers with a maximum 30 min cycling distance to work will shift mode to cycling. However, the number of car passengers is not reduced. If these persons would not continue to be car passengers (in other cars), but use public transport, this would not change the results of these health impact calculations, since we are only looking at the effects on the general population. In coming papers we will assess the change in exposure and subsequent health impacts for the commuters (Sommar et al., 2017), i.e. the former car commuters becoming new cyclists as well as all commuters not shifting travel mode but being subject to lower exposures due to lower total vehicle emissions. We will also compare the overall benefits for the general population, commuters considering also the benefits of increased physical activity of new bikers and the effects on accidents (Sommar et al., 2017).

5. Conclusions

This study indicates that around 111,000 car commuters in the Stockholm region have the physical capacity and short enough travel distance to potentially switch to commuting by cycling within a duration of 30 min. The reduced number of car travels result in lower emissions of vehicle generated air pollutants and thereby lower exposures of the general population. The health benefits as indicated by premature mortality of reduced exposure are estimated to be twice as large as the benefits associated with reduced emissions when the congestion tax system around the Inner City of Stockholm was installed.

We have also shown that using different indicators (NO_x, NO_2 and BC) of health risks associated with reductions in vehicle exhaust emissions gives slightly different estimates of impacts on mortality. Even though there are many more studies on health risks using NO_2, it is a less good indicator of vehicle exhaust emissions compared to NO_x due to effects of photochemistry and varying share of NO_x in emissions. BC is likely the best health indicator as it is directly proportional to the exhaust particles, but it is associated with larger uncertainties for estimating vehicle emissions compared to NO_x.

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