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The effects of congestions tax on air quality and health

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ABSTRACT

The "Stockholm Trial" involved a road pricing system to improve the air quality and reduce traffic congestion. The test period of the trial was January 3–July 31, 2006. Vehicles travelling into and out of the charge cordon were charged for every passage during weekdays. The amount due varied during the day and was highest during rush hours (20 SEK = 2.2 EUR, maximum 60 SEK per day). Based on measured and modelled changes in road traffic it was estimated that this system resulted in a 15% reduction in total road use within the charged cordon. Total traffic emissions in this area of NO_x and PM10 fell by 8.5% and 13%, respectively. Air quality dispersion modelling was applied to assess the effect of the emission reductions on ambient concentrations and population exposure. For the situations with and without the trial, meteorological conditions and other emissions than from road traffic were kept the same. The calculations show that, with a permanent congestion tax system like the Stockholm Trial, the annual average NO_x concentrations would be lower by up to 12% along the most densely trafficked streets. PM10 concentrations would be up to 7% lower. The limit values for both PM10 and NO_2 would still be exceeded along the most densely trafficked streets. The total population exposure of NO_x in Greater Stockholm (35 × 35 km with 1.44 million people) is estimated to decrease with a rather modest 0.23 $\mu g\ m^{-3}$. However, based on a long-term epidemiological study, that found an increased mortality risk of 8% per 10 $\mu g\ m^{-3}$ NO_x, it is estimated that 27 premature deaths would be avoided every year. According to life-table analysis this would correspond to 206 years of life gained over 10 years per 100 000 people following the trial if the effects on exposures would persist. The effect on mortality is attributed to road traffic emissions (likely vehicle exhaust particles); NO_x is merely regarded as an indicator of traffic exposure. This is only the tip of the ice-berg since reductions are expected in both respiratory and cardiovascular morbidity. This study demonstrates the importance of not only assessing the effects on air quality limit values, but also to make quantitative estimates of health impacts, in order to justify actions to reduce air pollution.

1. Introduction

Many cities have implemented congestion charging or low emission zones aiming at reducing traffic congestion and health impacts of traffic emissions. In Singapore traffic congestion was alleviated using first a manual Area licensing scheme starting in 1975 and subsequently an Electronic Road pricing system (ERS) from 1998 (Seik, 2000). London has a road charging zone around the city centre that recently was updated to cover a larger area. Several Norwegian cities charge drivers travelling with studded winter tires in order to reduce particle emissions due to road wear. In Rome, traffic is prohibited in the inner city on weekdays. Several cities in Europe have low emission zones. In several Swedish cities low emission zones

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apply to trucks and buses. Recently three German cities (Berlin, Hannover and Cologne) have applied a complete ban on all vehicles that have no catalysts or diesel particulate filters in zones of the city centres.

However, so far, there are very few papers on the quantitative effects of road pricing or low emission zones on air pollutant concentrations, population exposure and health. Beevers and Carslaw (2005) analyzed the air pollution impact of the London congestion charging. Road traffic data, combined with a traffic emission model, indicate that NO_x and PM10 emissions have been reduced by about 12% in the charging zone. But the emissions have increased on the inner ring road. The overall impact on air quality and health was not assessed. To our knowledge there is only one earlier study (Tonne et al., 2008) that has assessed the effects of a charging scheme not only on traffic and emissions, but also on exposure concentrations and health. They used a combination of dispersion modelling and regression calculations to analyse the air pollution and mortality benefits of the London congestion charge scheme (CCS). They concluded that the CCS lead to reductions in concentrations, although modest across Greater London, but greater in the charging zone wards. Predicted health benefits in the charging zone wards were 183 years of life per 100 000 people assuming conditions would persist over 10 years. This paper describes the effects of a road charge system in Stockholm on emissions, levels of air pollutants, and health of the population.

2. Description of the road charging system

On June 2, 2003, Stockholm City Council proposed testing congestion charging of traffic – called ‘‘The Stockholm Trial’’. On June 16, 2004 the Swedish Parliament adopted a law that made it possible to charge a congestion tax in Stockholm up to July 31, 2006. The Stockholm Trial consisted of three parts: extended public transport (16 new bus lines), congestion tax and more park-and-ride sites in the city and the county. The total public transport service was extended by 7% and the park-and-ride capacity was extended by 29%. The objectives of the trial included (i) reducing the number of vehicles in the congestion-charging zone during the morning and afternoon by 10–15%, (ii) improving traffic flows on the most heavily trafficked roads and (iii) reducing emissions of carbon dioxide, nitrogen oxides and particles in inner city.

Fig. 1 shows the extent of the congestion zone and the location of the toll stations. The inner city area is approximately 6 km $\times$ 6 km and has around 350 000 inhabitants. There are 23 000 workplaces, employing 318 000 persons of which two-thirds come from outside the zone. Drivers paid every time they passed a toll station (Table 1). Highest amount due was 20 SEK (corresponding to 2.2 EUR) during rush hours and lowest amount (10 SEK) early in the morning or in the evening. Maximum amount to pay for one day was 60 SEK. Night-time, holidays and weekends were free of charge. Taxis, buses, motorcycles, and cars classified as environmental vehicles (e.g. driven by electricity or bio fuels) were exempted. The Essingeleden bypass (Fig. 1) was free of charge for passage north–south through the toll zone.

3. Methodology

3.1. Measurements and modelling of road traffic

The effect of the Stockholm Trial on road traffic was quantified in terms of traffic flow by counting vehicles and by calculating road use, i.e. the number of vehicle kilometres travelled in the area (e.g. Baradaran et al., 2006; Forsman et al., 2006). Congestion was quantified in terms of journey times obtained from floating car measurements or from traffic cameras. Data on the composition of the vehicle fleet were acquired from manual recording of vehicle types over stretches of road where the control points already existed before the trial. For details evaluation reports in English are available on http://www.stockholmsforsoket.se/templates/page.aspx?id=12555 (accessed August 2008).

3.2. Calculation of emissions

3.2.1. Emission factors

The estimate of the change in road use with the Stockholm Trial was implemented in an existing traffic database (Airviro, SMHI, Norrköping, Sweden; http://airviro.smhi.se) (Johansson et al., 1999). Emissions from road traffic are described with emission factors for passenger cars (petrol and diesel), light commercial vehicles, heavy goods vehicles. Emission factors for NO_x were obtained from the EVA model of the Swedish Road Administration (Hammarström and Karlsson, 1994). Emission factors for PM10 were obtained from simultaneous measurements of PM10 and NO_x in the street canyon of Hornsgatan and at the urban background site (Torkel Knutssonsgatan), using NO_x as quantitative tracer as described in Ketzel et al. (2007). Emission factors for other roads were corrected for the vehicle speed dependence according to Bringfelt et al. (1997).

3.2.2. Uncertainties in emissions

There are several uncertainties in the estimation of the change in emissions due to congestion charging. Emission factors for NO_x depend on fleet composition and driving conditions (speed and congestion). With congestion charging, the relative contribution from commercial vehicles, especially heavy-duty vehicles, increased. This effect was not considered for every individual street. Instead the mean change in fleet composition was applied to all roads within the inner city.

Queuing time was reduced during the Stockholm Trial by one-third during the morning and by more than half during the afternoon/evening compared to without the trial (Söderholm, 2006). Estimates by Carlsson et al. (2006) using the Artemis emission factors indicate that reduced traffic congestion lead to a small reduction in the emissions of NO_x and exhaust PM, on the order of 1% for the whole day and 2–3% for rush hours. As shown below the reduction in emissions due to less traffic is much larger. Higher vehicle speeds increase vehicle-induced turbulence and may then increase dilution, especially during periods with low wind speed (Kastner-Klein et al., 2000), further decreasing the concentrations. The overall net effect on the emissions and concentrations for all streets of the area is

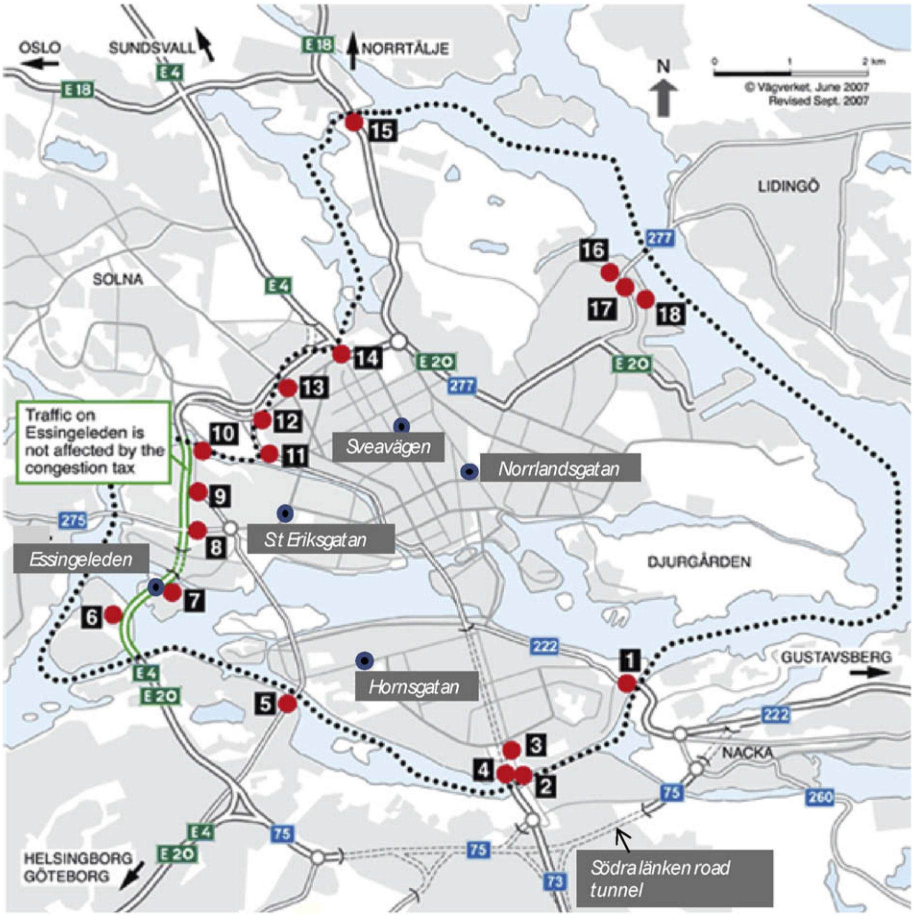

Fig. 1. Map showing the 18 toll stations of the congestion zone during the Stockholm trial (January–July, 2006). Dotted line indicates the inner city area which is reached only by passing a toll station. Black filled circles indicate sites mentioned in the text.

likely to be small and has not been accounted for due to lack of detailed road traffic measurements.

For PM10, only 10% is due to exhaust PM emissions; most of the road traffic emission (90%) is due to non-exhaust PM (Johansson et al., 2007). It has been shown that non-exhaust PM10 emissions depend on vehicle speed, especially when studded winter tires are used (e.g. Gustafsson et al., 2008). Also the suspension of road dust from dry roads depends on vehicle speed (Hussein et al., 2008). According to the field measurements by Hussein et al. (2008) the speed dependence for vehicle speeds below 50 km h^{-1} is quite weak, indicating that the change in average speed in the inner city should have a small effect on road dust emissions. Emission of brake wear generated particles is expected to decrease as congestion decrease, so that would tend to make non-exhaust PM emissions lower. So, for PM10, reduced congestion and increased vehicle speed may lead to both increased and decreased emissions of PM10 depending on the road considered. In this study we have not accounted for this due to lack of quantitative parameterisations of these processes and lack of traffic data on different roads.

Table 1
Congestion tax for every passage during different hours of weekdays. No tax was charged on Saturdays, Sundays, public holidays and the day before a public holiday. Maximum amount due was 60 SEK per day (10 SEK = 1.08 EUR, August 12, 2008).

Time	Amount
6.30 a.m.–6.59 a.m.	SEK 10
7.00 a.m.–7.29 a.m.	SEK 15
7.30 a.m.–8.29 a.m.	SEK 20
8.30 a.m.–8.59 a.m.	SEK 15
9.00 a.m.–3.29 p.m.	SEK 10
3.30 p.m.–3.59 p.m.	SEK 15
4.00 p.m.–5.29 p.m.	SEK 20
5.30 p.m.–5.59 p.m.	SEK 15
6.00 p.m.–6.29 p.m.	SEK 10

3.3. Air pollution measurements

Before and during the trial air quality was measured at 20 sites in the Stockholm area (SLB, 2006). In this paper we include data from four of these sites which are permanent measurements stations that have been in operation since several years. The locations of the stations are indicated on the map in Fig. 1. Three measurement stations were located along busy streets (10 000–35 000 vehicles day^{-1}) in street canyons in central Stockholm (Hornsgatan[1], Sveavägen[2] and Norrlandsgatan[3]). One station is located beside the Essingeleden bypass. The stations are equipped with automatic instruments for measurements of PM10 (Tapered Element Oscillating Microbalance, model 1400, Rupprecht and Pataschnik), NO, NO_2 (Environnement S.A., AC31M) and CO (Thermo Enviro Model 48). PM10 concentrations from the TEOM instrument were multiplied with 1.2 to account for losses of volatile compounds in this instrument (Johansson, 2003). Span and zero checks of the NO_x instruments using certified calibration gases with NO in N_2 were performed automatically every day. Conversion efficiency for reduction of NO_2 to NO is also checked every day by converting NO to NO_2 with O_3.

3.4. Dispersion modelling

3.4.1. Airviro

The annual mean PM10 and NO_x concentrations and exposures due to local road traffic emissions were calculated using a wind model and a Gaussian dispersion model (Airviro, SMHI, Norrköping, Sweden; http://airviro.smhi.se). Meteorological conditions were based on a climatology that was created from 10 years of meteorological measurements (15 min averages) in a 50 m high mast located in the southern part of Stockholm (Johansson et al., 2007). The dispersion calculations were performed on a 100 m resolution (122 500 receptor points). Individual buildings and street canyons were not resolved but treated using a roughness parameter (Gidhagen et al., 2005). In these calculations the meteorological conditions are the same with and without the Stockholm Trial. It is also assumed that the effect on the emissions during the 7-month trial would be representative for the whole year. That this is a reasonable assumption since mean daily traffic for January–July differs from the daily mean traffic for the whole year by less than 1%.

3.4.2. Uncertainties in 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 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 $\mu g\ m^{-3}$ and for the model calculations 21.4 ± 0.7 $\mu g\ m^{-3}$. These comparisons were made using time series of meteorological measurements for the same periods as the measurements were made (not with the climatology). The current paperfocuses on the change in annual concentrations and population exposure due to the change in emissions with congestion charging, i.e. they relate to a situation with compared to a situation without the Stockholm Trial. 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, which is discussed above.

3.5. Population exposure and health impact assessment

The population data (100 m resolution) include number of residents of different ages. People are linked to a specific location according to their registered residential address. The epidemiological studies providing exposure–response functions were also based on home addresses. If the exposure at place of work is considered, this has only a minor effect on the results. The reason is that the age category that contributes most to the premature mortality is 65+ with mostly retired persons without a work place.

The health impact assessment (HIA) presented is restricted to long-term effects on mortality, and uses the same approach as used by, for example, WHO (2004), Cohen et al. (2005) and in a Swedish national assessment by Forsberg et al. (2005). The excess number of cases per year, Δy, was calculated as:

$$\Delta y = (y_0 \cdot \mathrm{pop})(\mathrm{ERF} \cdot \Delta x - 1)$$

where y_0 is the baseline mortality rate, pop is the affected number of persons; ERF the exposure–response function (relative risk per change in concentration), and Δx is the estimated impact on exposure (in our case change in exposure calculated as the change in population weighted concentration). We used the change in the population weighted mean exposure (for all ages), assuming the ERF to be the same for all age groups. The rounded baseline used for this study was 1000 deaths per 100 000 person years (data from Swedish Cause of death register).

Several studies have used within-city gradients in NO_2 or NO_x as an indicator of traffic pollution exposure. Studies of this kind exist from Germany, New Zealand, France and Norway (Gehring et al., 2006; Scoggins et al., 2004; Filleul et al., 2005; Nafstad et al., 2004). These studies have arrived at very similar exposure–response factor for the importance of traffic emissions for mortality; 11% (95% CI = 1–21), 13% (95% CI = 10–15) and 14% (95% CI = 3–25) per 10 $\mu g\ m^{-3}$ increase in NO_2, respectively, in the studies from Germany, New Zealand and France. With the similarities in type of city, buildings, vehicle fleet and climate, the Norwegian study, carried out on adult men, was considered the most representative for the consequence analysis of the effects of the Stockholm Trial. In that study they arrived at an ERF of 8% (95% CI = 6–11) increase in mortality per 10 $\mu g\ m^{-3}$ increased NO_x concentration. Likely it is the

[1] Hornsgatan is 24 m wide street canyon with 24 m buildings, 35 000 vehicles day^{-1}.

[2] Sveavägen is a 33 m wide street canyon with 24 m buildings, 30 000 vehicles day^{-1}.

[3] Norrlandsgatan is 15 m wide with 24 m high buildings, 10 000 vehicles day^{-1}.

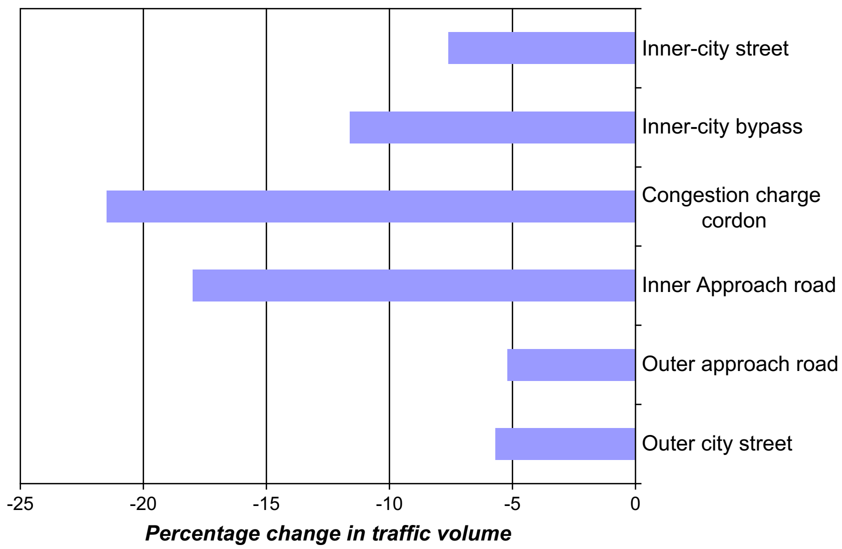

Fig. 2. Percentage reduction in traffic during weekdays on different types of streets in Stockholm after introduction of congestion charges.

fraction of exhaust particles, and not NO_x, that is the main candidate pollutant behind cardiovascular effects and premature mortality (Seaton and Dennekamp, 2003; Samet, 2007). NO_x is a good marker for vehicle exhaust particles 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). NO_x is a better marker for vehicle exhaust particles than NO_2, which depends on ozone levels. In addition, several studies (e.g. Carslaw, 2005; Carslaw et al., 2007) point out that the NO_2/NO_x ratio from road transport sources has increased in recent years making NO_2 a less suitable indicator of exhaust particles. There are almost no epidemiological studies of road dust, but studies of coarse PM in general support toxicological findings that dust particles, largely crustal, do not have the same influence on cardiovascular morbidity and mortality as widely demonstrated for finer combustion related particles (Brunekreef and Forsberg, 2005).

4. Results and discussion

4.1. Effects on traffic

Fig. 2 shows average traffic reductions for different types of roads. The reduction in total number of vehicle passages across the charge cordon over 24 h was 22%, corresponding to nearly 100 000 fewer passages to/from the inner city. The reduction was lower during the morning peak period (16%) and higher during the afternoon/evening peak (24%). For inner city streets the reduction in the number of vehicles was around 8% and for roads approaching the city around 5%. For the Essingeleden bypass, not subjected to congestion tax, vehicle numbers increased by 4%. The total road use within the zone with congestion tax decreased from 2 185 000 to 1 847 000 km (15%). Largest reduction in road use was estimated for passenger cars (16.5%). Road use by light commercial vehicles and heavy-duty vehicles decreased with 15.0% and 7.8%, respectively. The Stockholm Trial also involved extended bus traffic. The extra buses for the trial were mainly diesel fuelled (a few used biogas). According to Carlsson et al. (2006), the extended bus traffic corresponds to approximately 8000 km during a weekday within the zone. The fraction road use due to buses is thus very small, but increased from 1.9% to 2.7% of the total road use.

4.2. Effects on emissions

Compared with the situation in 2006 with no Stockholm Trial, NO_x emissions in the Greater Stockholm area (1.44 million inhabitants, 35 $\times$ 35 km) would decrease by 55 $\pm$ 16 tonnes as result of reduced traffic (Table 2). Most of this reduction occurs in Stockholm's inner city, 45 $\pm$ 13 tonnes.

For PM10, the corresponding reduction would be 30 $\pm$ 9 tonnes, of which about two-thirds would be the result of reductions in emissions in the inner city. Both exhaust and non-exhaust particles would have decreased. The increased NO_x emission from buses compensate for some of the reduction in emissions from passenger cars. This is the reason for NO_x emissions to be reduced by only 8.5% in the inner city, while total road use is reduced by 15%. For the other substances the emissions decrease approximately to the same extent as the road use. Reduced traffic congestion further lowers the emissions by 1% seen across the entire 24-h period, and by 2–3% at periods of peak traffic, but this is not taken into account (see discussion in Section 3.2.2).

4.3. Effects on air quality

4.3.1. Measurements

Fig. 3 shows running 7-day mean concentrations of NO_x, PM10 and the ratio of PM10/NO_x along Hornsgatan during January–July, 2003–2007. The mean NO_x levels

Table 2
Estimated reductions in emissions from road traffic in Stockholm for a situation for 2006 with the Stockholm Trial.

	Inner city:		City of Stockholm:		Greater Stockholm[a]:	
	tonnes year^{-1}	percent	tonnes year^{-1}	percent	tonnes year^{-1}	percent
Nitrogen oxides. NO_x	45 ± 13	8.5% ± 2.5%	47 ± 13	2.7% ± 0.8%	55 ± 16	1.3% ± 0.4%
Carbon monoxide. CO	670 ± 190	14% ± 4.0%	710 ± 200	5.1% ± 1.5%	770 ± 220	2.9% ± 0.8%
Particles. PM_{10} total	21 ± 6	13% ± 3.5%	23 ± 7	3.4% ± 1.0%	30 ± 9	1.5% ± 0.4%
"road wear particles"	19 ± 5.5	13% ± 3.5%	21 ± 6	3.3% ± 0.9%	28 ± 8	1.5% ± 0.4%
"exhaust particles"	1.8 ± 0.5	12% ± 3.6%	1.8 ± 0.5	4.4% ± 1.3%	2.1 ± 0.6	2.4% ± 0.7%
Carbon dioxide. CO_2	36 000 ± 10 300	13% ± 4%	38 000 ± 10 900	5.4% ± 1.5%	41 000 ± 11 700	2.7% ± 0.8%

[a] Defined as 35 km × 35 km across central Stockholm.

drop from 130 μg m^{-3} in 2003 to 102 μg m^{-3} in 2007, mainly due to lower emissions from the vehicles, indicated by the consistent steady decrease in both NO_x and CO concentrations (Fig. 4). Conditions are similar on Sveavägen, with lowest NO_x concentrations observed during the trial period of 2006. Considering the decreasing trend, mean NO_x concentrations for 2006 is expected to be higher than observed, indicating that the emissions were significantly lower due to the congestion charging. Fig. 4 also shows increased NO_x concentrations on Essingeleden during 2006 compared to 2005 and 2007, due to the increased traffic.

PM10 concentrations are substantially more variable than NO_x (Fig. 3). The pronounced peak in concentrations during spring is due to suspension of coarse PM in road dust as roads get dry after the (relatively wet and cold) winter period (Johansson et al., 2007; Omstedt et al., 2005). There are also large variations between the years in the development of the peak PM10 concentrations. In 2006 the peak was delayed by one month and there were also much lower concentrations during February–April compared to other years. The main factor influencing PM10 on Hornsgatan is road surface wetness and during periods with wet roads PM10 concentrations are much lower (Johansson et al., 2007). During March and beginning of April 2006 roads were wet due to a long winter with melting snow compared to the other years. But the reduced traffic in 2006 contributed to decreased road wear, and therefore also less road dust during the spring period. The mean PM10 values for the 7-month periods show no systematic trend, but the mean for 2006 is substantially lower than any of the other years (Fig. 4). But the reduction on Hornsgatan and Sveavägen is larger than can be explained by lower traffic emissions. Obviously it is very difficult to quantify the impact of reduced traffic during 2006 based on these measurements due to the strong impact of road conditions.

4.3.2. *Modelling*

In order to quantify the effect on the concentrations of the change in road use for the whole area and the whole population, air quality dispersion modelling was employed. Fig. 5 shows the geographic variations of the annual mean reduction in NO_x and PM10 concentrations due to the reduced traffic. Note that these concentrations represent roof top, not street level (the model cannot resolve individual streets and building effects), and that they are annual mean values assuming the effect of the 7-month trial would be the same for the whole year. Largest reductions are seen in the city centre, where concentrations are estimated to fall by up to 2 μg m^{-3} for both NO_x and PM10. The greatest improvements are found along the city centre and south of the city centre in connection to a road tunnel bypass (Södra länken). Concentrations of NO_x and PM10 increase in an area around the toll-free Essingeleden and near the portals of the road tunnel, due to the increase in traffic emissions.

Table 3 gives some examples of estimated effects on total annual mean NO_x, NO_2 and PM10 concentrations along some densely trafficked streets. Total PM10 concentrations with the Stockholm Trial are estimated to decrease by 4%–7% on four of the streets and remain unchanged on one street. For NO_x the corresponding decrease is 2%–12%, being

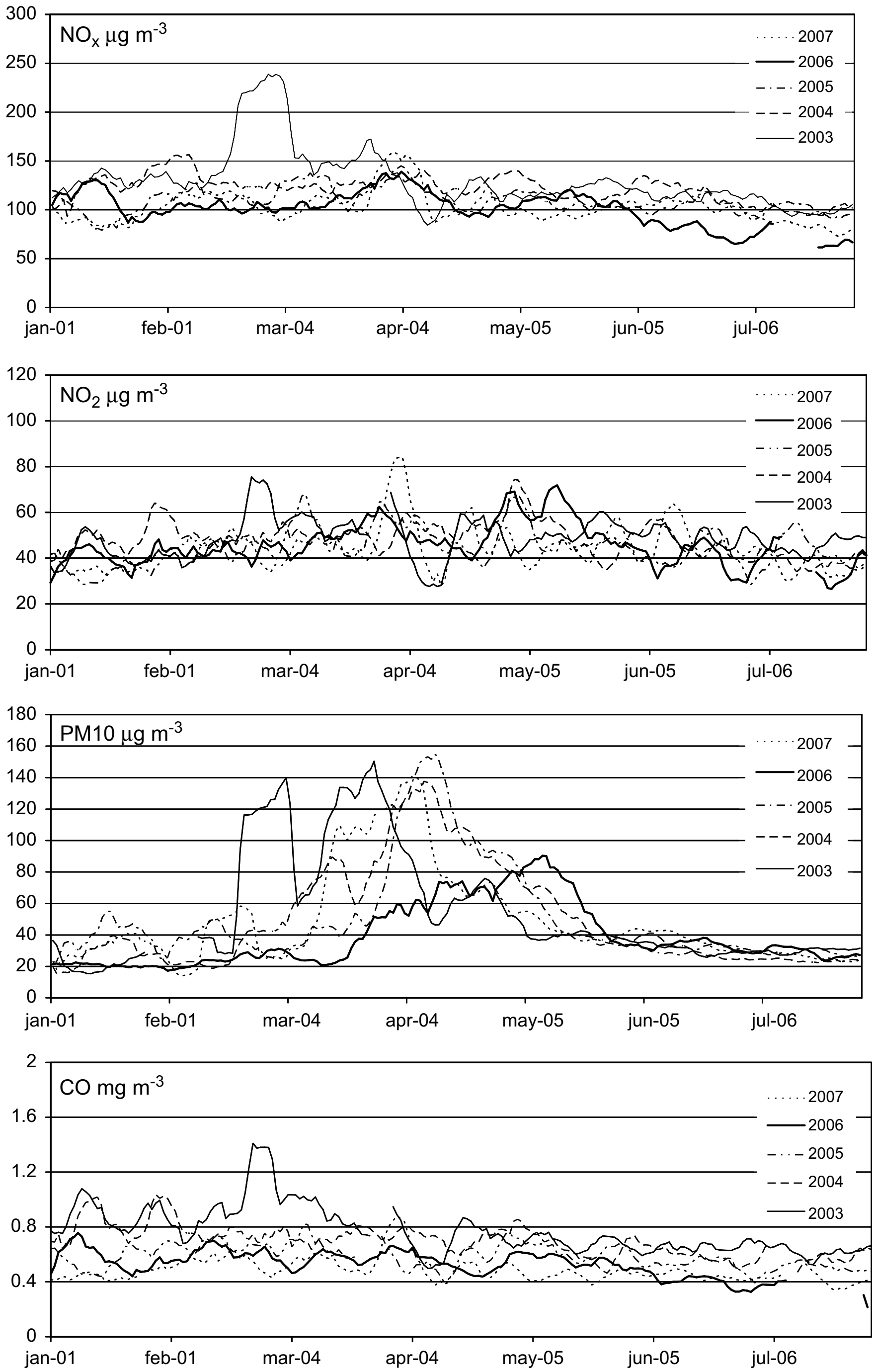

Fig. 3. Seven day running mean concentrations of NO_x, NO_2, PM10 and CO at street level on Hornsgatan in central Stockholm for the period January 1 to July 31 of 2003, 2004, 2005, 2006 and 2007. The Stockholm Trial was in effect during 2006 and is shown as a thick line.

unchanged on one street. It is interesting to note that the reductions are not large enough to achieve the limit values.

4.4. *Effects on exposure and health*

4.4.1. *Population exposure*

The mean population weighted changes in exposure of the whole population in Greater Stockholm area (1.44 million) and in the inner city (350 000 people) are presented in Table 4. The people in the city centre will have the largest reduction in exposure. Annual mean decrease in population weighted concentrations in the inner city is 10%, 7.6% and 10% for NO_x, PM10 and PM exhaust, respectively. The corresponding absolute values are 0.81, 0.21 and 0.022 $\mu g\ m^{-3}$, i.e. very small changes concentrations. PM exhaust constitutes only 10% of total PM10, due to the impact of road wear emissions.

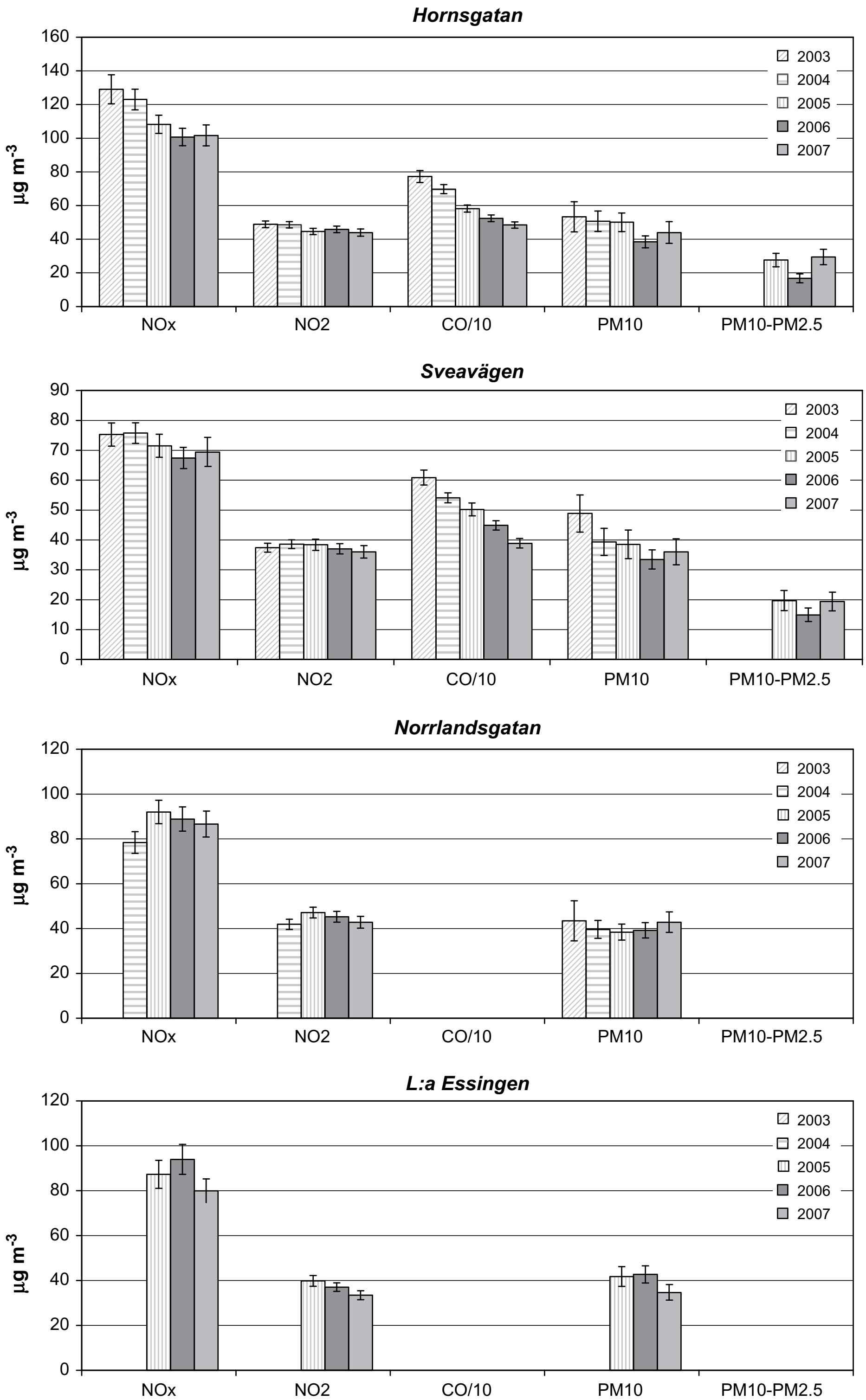

Fig. 4. Mean concentrations of NO_x, NO_2, CO and PM10, PM10–PM2.5 at kerb side sites in central Stockholm. Mean values are for the period January 1 to July 31 of 2003, 2004, 2005, 2006 and 2007. Bars indicate 95% confidence intervals.

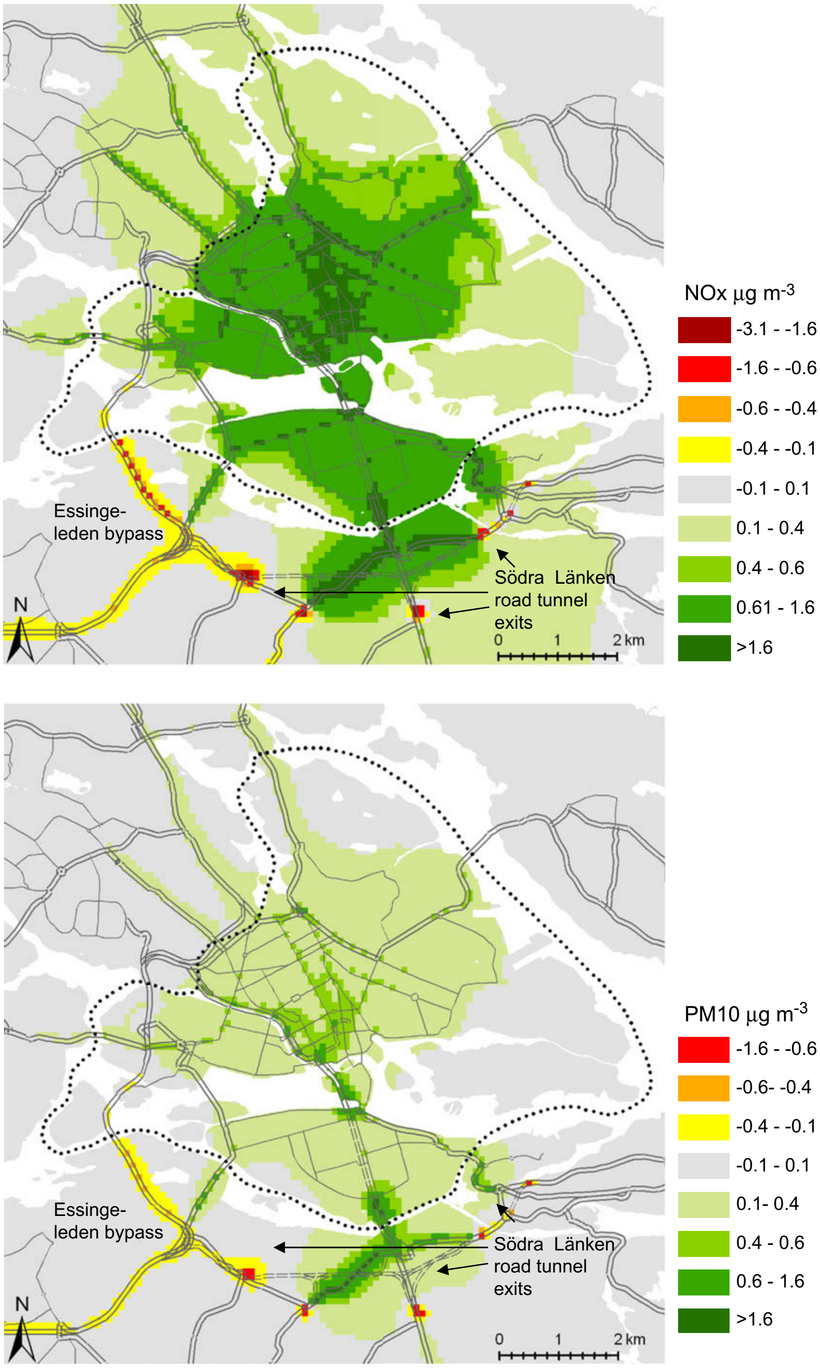

Fig. 5. Difference in annual mean concentrations of NO_x and PM10 with the Stockholm Trial compared to a situation without the Trial (with the same meteorology and other emissions than road traffic). Within the green areas the levels have fallen, within yellow to red areas there is an increase in levels. In the inner city changes refer to rooftop height (not street canyon). (For Interpretation of the references to colour in figure legends, the reader is refered to the web version of this article).

4.4.2. Mortality effects

The reduction in mortality is calculated using the exposure–response function for NO_x from Nafstad et al. (2004), 8% per 10 µg m^{-3}, and a rounded mortality frequency of 1000/100 000 inhabitants per year. As pointed out above NO_x is merely a marker for traffic emissions, not the actual cause of mortality. With the reduction in population weighted NO_x concentration for Greater Stockholm (0.23 µg m^{-3} in Table 4), 27 (20–37 lives, considering the 95% confidence interval of ER factor of Nafstad et al., 2004) premature deaths per year are gained. We have then assumed effects in mortality, not only in adults as included by Nafstad et al. but also among younger people, as studies have shown that air pollution

Table 3
Estimated total concentrations and percentage changes in levels along some streets in the inner city of Stockholm due to the Stockholm Trial (annual mean values for 2006). The locations of streets are shown in Fig. 1.

Street in city CENTRE	Number of vehicles per day (weekdays)	NO_x[a]	NO_2[b]	PM10[c]
Hornsgatan	35 000	103 $\mu g\ m^{-3}$	77 $\mu g\ m^{-3}$	78 $\mu g\ m^{-3}$
	−8%	- 8%	−3%	−5%
Sveavägen	30 000	76 $\mu g\ m^{-3}$	68 $\mu g\ m^{-3}$	62 $\mu g\ m^{-3}$
	−6%	−2%	−1%	−4%
Norrlandsgatan	10 000	77 $\mu g\ m^{-3}$	68 $\mu g\ m^{-3}$	62 $\mu g\ m^{-3}$
	−12%	−11%	−5%	−7%
Valhallavägen	38 000	35 $\mu g\ m^{-3}$	47 $\mu g\ m^{-3}$	58 $\mu g\ m^{-3}$
	−14%	−12%	−7%	−7%
S:t Eriksgatan	35 000	53 $\mu g\ m^{-3}$	58 $\mu g\ m^{-3}$	57 $\mu g\ m^{-3}$
	+5%	Unchanged	Unchanged	Unchanged
Essingeleden	140 000	54 $\mu g\ m^{-3}$	58 $\mu g\ m^{-3}$	67 $\mu g\ m^{-3}$
	+3%	+2%	+1%	+1%

[a] Annual mean value.
[b] 98th Percentile of daily mean values. The Swedish limit value is 60 $\mu g\ m^{-3}$. (There is no EU limit value for daily mean NO_2.)
[c] 90th Percentile of daily mean values. EU and Swedish limit value is 50 $\mu g\ m^{-3}$.

affect mortality in infants (Woodruff et al., 2008). The fact that younger people are included has however negligible significance on the result since the mortality is low for this group.

Life-table analysis indicates 11.2 years of life gained for each premature death. This means that 27 less deaths during one year would correspond to 20.6 years of life gained per 100 000 people with the full effects of the exposure reduction. This is similar to the YLG estimated for the London congestion-charging system, 183 years of life gained over 10 years per 100 000 persons (Tonne et al., 2008). The long-term effect depends on a number of factors, e.g. changes in future mortality rate, economical development, road planning and the development of the public transport system.

Emissions from road traffic also cause a variety of other adverse health effects. These include exacerbation of asthma, respiratory problems, allergies, lung cancer and cardiovascular effects (e.g. WHO, 2006). Morbidity and cost of illness should also be reduced if traffic emissions decrease. In Swedish studies, relevant for Stockholm, NO_2 has been used as marker for traffic exhaust and significant associations with acute asthma symptoms have been found (Forsberg et al., 1998). Also long-term exposure to traffic pollution, indicated by NO_2 levels, has been significantly associated with the prevalence of respiratory problems (Forsberg et al., 1997), lung cancer incidence (Nyberg et al., 2000), deaths in myocardial infarction (Rosenlund et al., 2006) and incidence of adult asthma (Modig et al., 2006).

Table 4
Annual mean contributions to total levels of nitrogen oxides and particles from emissions from road traffic with and without charges according to the Stockholm Trial. Mean levels have been estimated by weighting with regard to the number of residents in different parts of the area. Unit: $\mu g\ m^{-3}$.

	2006 without Stockholm Trial	2006 with Stockholm Trial	Difference
NO_x (Greater Stockholm)	4.42	4.19	0.23 (−5.3%)
PM_{10} (Greater Stockholm)	1.71	1.65	0.064 (−3.8%)
PM exhaust (Greater Stockholm)	0.102	0.0960	0.0062 (−6.1%)
NO_x (inner city)	8.41	7.60	0.81 (−10%)
PM_{10} (inner city)	2.76	2.55	0.21 (−7.6%)
PM exhaust (inner city)	0.21	0.19	0.022 (−10%)

5. Conclusions

The Stockholm Trial significantly reduced traffic emissions of NO_x and particles in central Stockholm. The reductions were mainly due to decreased traffic flow; lower congestion had small effect. For NO_x, emission reductions would have been larger without the extended bus traffic during the Stockholm Trial. Comparisons were made of NO_x, NO_2, CO and PM10 concentrations measured during the Stockholm Trial (the period January to July 2006), with the corresponding period in 2003, 2004, 2005 and 2007. Considering the trend in annual mean values due to cleaner vehicle fleet, NO_x concentrations were reduced at the most densely trafficked roads in the inner city with the trial in 2006 than they would have been without the trial, but the effect could not be quantitatively ascertained. For PM10, the temporal variations in the concentrations are very large, mainly due to different road conditions affecting road dust suspension. The concentrations during 2006 were significantly lower compared to the other years, but this was to a large degree due to an unusually wet spring period. Overall measured levels during the Stockholm Trial could not provide a quantitative answer of importance of the traffic reductions for the levels of air pollutants. The reduction in traffic emissions along the most densely trafficked streets was not sufficient for compliance of limit values.

The change in exposure due to the estimated change in traffic emissions was calculated keeping the meteorological conditions and other emissions the same. The annual average levels of NO_x and PM10 were estimated to fall by up to 2 $\mu g\ m^{-3}$. The greatest improvements in air quality are estimated to occur inside the charge cordon where most people live. Higher concentrations were calculated around the Essingeleden and Södra Länken by-passes, but considerably more people in Stockholm have reduced exposure of NO_x and PM10 compared to a situation without the congestion charge. On the basis of results from epidemiological studies that consistently show that there is a significant relationship between NO_x or NO_2 and mortality among the population, the total number of gained premature deaths was estimated. This indicates that in total, for the whole of the Stockholm area (1.44 million inhabitants, 35×35 km), it is estimated that 206 years of life would be gained per 100 000 people over a 10-year period provided the reduction in long-term exposure persist.

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