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Dieselization in Sweden - blessing or curse?

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Abstract

In this paper we discuss, based on research on the external cost of air pollution, if diesel as a fuel in the transport sector should be encouraged or discouraged in Swedish environmental policy. There are two main reasons for posing this question. The first is the international context where the use of diesel is generally considered to be a bad, due to its negative health effects. The second is the Swedish context with an ambitious vision for a fossil free vehicle fleet in 2030 where the use of diesel produced from forestry residues could be part of the solution. In recent years the use of diesel cars has been encouraged by various policy measures, for example a subsidy based on assessments of emissions for CO2 per kilometer. Is this a policy that should be continued or abandoned? In this paper we focus on the health impacts and our conclusion is that dieselization is more a blessing than a curse. The reason is that Sweden is a sparsely populated country and therefore the health costs of emissions from road transport are low by international standards.

Keywords: Environmental policy, fossil free vehicle fleet, fuel use, external health cost, impact pathway approach, cost-efficient emission reductions

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1. Introduction

In Sweden environmental concern, and climate change in particular, has been high on the policy agenda for over two decades. In 2008, a vision for a fossil free vehicle fleet was adopted by the parliament (Government bill 2008/09:162) and recently a parliamentary committee established the goal of a 70% reduction of greenhouse gas emissions from inland transport to 2030 compared to the emissions in 2010. How to reach this vision, and now this goal, has been the focus of several inquiries commissioned by the government. Most recently, in 2016, the Swedish Energy Agency was given the task to present, jointly with five other national agencies, a strategic plan for the conversion to a fossil-free transport sector (Swedish Energy Agency, 2017). One of the feasible options is to use forest residues to produce bio-diesel, but how should politicians assess this option given the ongoing discussions about "Dieselgate" and diesels contribution to air pollution?

Until ten years ago, diesel was mainly used in heavy trucks and other machinery in Sweden[1]. Since 2006 however the share of diesel cars has increased from about 6 percent up to closer to 30 in 2015. There are several reasons for this as described in Kågeson (2013). One reason is the subsidies that was introduced in 2004 for "environmental cars". The design of this subsidy has changed somewhat over time, but its consequence was that diesel cars are more subsidized than equal gasoline cars since the assessment is based on CO_2 emissions per km. "Environmental cars" has also benefited from other advantages such as free parking in many cities, and where also exempt from the Stockholm congestion tax until 2012. Another factor that is likely to have contributed to the change is that one litre of gasoline has been priced 3 percent above one litre of diesel for several years up to 2013. Taken together, these factors seem to have provided incentives for consumers to buy diesel cars to a larger extent than before. This change has most likely been beneficial for the climate, but the question is if it has come at an important cost for public health.

The impact on health of diesel cars has been discussed in research for a long time (Ecotraffic, 2002; Mayeres and Proost, 2001; 2013; Michiels et al., 2012). The question of diesel versus gasoline has also been discussed in Sweden before (Ecotraffic, 2002; Vägverket, 2004, Ministry of Finance, 2009; Swedish Environmental Protection Agency, 2017). Vägverket (2001) concluded that the difference in environmental and health impacts between gasoline and diesel cars were small and that it was more important, from an environmental point of view, to choose fuel efficient cars. Ecotraffic (2002) on the other hand found that diesel cars have higher emissions of NOx and PM (unless they have a diesel particle filter). In the paper by Mayeres and Proost (2013), the calculation and comparison of external costs results in higher conventional air pollution cost of 24 euro per year for the diesel car with respect to a gasoline version. For a fuel-efficient diesel, emitting less than 105 g CO_2/km, the additional cost is 9 euro per year.

Calculation of the external cost of transport has a long tradition, and the information has been used to evaluate investments in transport infrastructure and to design policy instruments to achieve an efficient level of transport in different parts of the world (c.f. Delucchi, 2000; Bickel et al., 2006; Iwata and Arimura, 2009; Proost and Van Dender, 2012). As illustrated by the recent paper by Santos (2017), the information can also be used to design road fuel taxes in order to internalize road transport externalities.[2] Zimmer and Koch (2017) show that recent proposals on changes in fuel taxation in Europe, if implemented, could reduce health damaging emissions while at the same time contributing

[1] There is a tax exemption for heavy machinery using diesel in some sectors of the economy, for example farming.

[2] The results in this paper are based on the EcoSense model. The following is a summary of the description of the input to the modelling provided in the paper: "Van Essen et al. (2011) update values from Preiss et al. (2008) and Maibach et al. (2008), who in turn update theirs from Bickel et al. (2006), who base their values on the ExternE set of projects and the ExternE's EcoSense System, which is an integrated software tool where emission sources are distinguished by administrative region, economic activity and emission height".

to achieving the EU climate policy goal for 2020. They however do not account for where in Europe the emissions are reduced.

An important aspect in a discussion of the pros and cons of diesel cars is that the impact on public health of emissions from transport and other sources varies greatly in Europe. Amann et al. (2011) provides an illustration of this and discuss the need for differentiated target setting to reach cost efficient reductions of air pollution in Europe. The results of the calculations reveal that the health impacts of NOx emissions are relatively low in Sweden and the other Nordic countries. This is a finding supported by a recent study investigating the impact of ‘Dieselgate’ (Jonson et al., 2017). Hence, although diesel emissions may be an important health problem in parts of Europe, this appear not be the case in Sweden. The mentioned studies however only account for the impact of secondary particles from NOx using regional dispersion modelling. As discussed in Jonson et al. (2017), to account for the full impact of the emissions of diesel cars, the effects on a local scale from the directly emitted exhaust particles and NO_2 also has to be assessed.

To assess local impacts, other dispersion models are needed. Studies have revealed that it is important for the result that such models are adapted to local conditions. Michiels et al., (2012) and Jensen et al., (2008) both found that the estimated exposure using the Eco-Sense model, developed in the EU funded ExternE-projects, deviate from models that are country specific with better resolution. This is probably especially problematic for the Nordic countries since, according to Jensen et al. (2008), the deviation is large. The Danish models with higher resolution predicts marginal unit costs per km for transport that are 40 -70% of the Eco-Sense Transport model for rural areas, and 60-90% for the city center of Copenhagen. The importance of the assumptions used in the dispersion and exposure modeling is also found in two recent Swedish studies (Nerhagen, 2016; Nerhagen and Andersson-Sköld, 2018).

The purpose of this paper is to use external cost calculations to provide a quantification of the difference in health costs between a scenario where 10 percent of the car fleet used in Stockholm is replaced by brand new gasoline cars and a scenario where 10 percent of the car fleet is replaced by brand new diesel cars. The calculation is based on detailed dispersion and exposure modeling in Stockholm, the capital and most populated city. The models used have been developed by the Swedish Meteorological and Hydrological Institute (SMHI) and they also performed most of the original exposure calculations as described in Nerhagen et al., (2009).

The paper is organized as follows. In the next section, we describe the international discussion on the air pollution problem in general, to transport and diesel in particular, and current policies to reduce air pollution. We then describe the air pollution policy context and the research undertaken in Sweden on the influence of cars and different fuels on air quality. After that we present results for external cost calculations for the scenarios described above. Based on this assessment the conclusion for Sweden is that the introduction of new cars entails improvements in air quality and that there is a somewhat higher external cost for diesel cars due to the emissions of NOx. However, even in Stockholm the cost is lower than the current additional yearly excise tax placed on new diesel cars. This implies that the tax is much too high for cars used on the countryside and many urban areas in Sweden that are generally very sparsely populated by international standards. We also discuss some of the uncertainties related to these kinds of calculations concluding that even if we assume higher emission factors, this does not change our results. A final recommendation is therefore that with varying geographical conditions it is important to base policies on good empirical assessments of air pollution impact tailored to the specific situation and problem.

2. International policy work on air pollution and the problem with diesel

International agreements on air pollution has a long history, starting with the problems related to “acid rain” that was destroying forests and causing fish loss in lakes in the Northern Hemisphere. To solve this problem, 32 countries signed the UNECE Convention on Long-range Transboundary Air Pollution (CLRTAP) in 1979, creating the first international treaty to deal with air pollution. CLRTAP soon provided a forum for Parties to negotiate and agree on binding obligations to reduce emissions. To date, it has been extended by eight protocols containing legally binding targets for emission reductions. Over the years, the number of substances covered has been gradually extended, notably to ground-level ozone, persistent organic pollutants, heavy metals and PM. The Joint Task Force on the Health Aspects of Air Pollution was established in 1998 within the Convention to assess the health effects of such pollution and to provide supporting documentation.

Local air pollution has mainly been the responsibility for national governments. One event that put this problem in focus was the London Smog in 1952. This and other similar episodes resulted in the UK Government introducing its first Clean Air Act in 1956. In 1961 the UK established the world's first coordinated national air pollution monitoring network, called the National Survey[3]. Limits for small airborne particles[4] (PM_{10}) was established in the US around 1990. In the EU, air quality has been one of the main environmental policy concerns since the late 1970s. Current work is guided by an air quality directive with established air quality standards, for example new $PM_{2.5}$ objectives introduced in 2008 targeting the exposure of the population to fine particles. These objectives are set at the national level and are based on measurements in urban background locations to best assess the $PM_{2.5}$ exposure to the general population.

The World Health Organization (WHO) contributes to international policy work by reviewing scientific evidence on the health effects of air pollution and providing evidence-based guidance to policy-makers on how to protect public health. Initially, the heavy industry and residential heating using coal and oil was the most important problem. However, with increased car use the influence of emissions from transportation on local air pollution grew. As discussed in Proost and Van Dender (2013), this has resulted in regulations giving rise to cleaner fuels as well as cleaner engines. The reason for the reductions in emissions in Europe is new emission standards for vehicles. From 2011, new cars must comply with the standard Euro 5 and from 2015 with Euro 6. Euro 6 implies additional requirements on diesel vehicles, especially for nitrogen oxides. Hence, additional reductions in some emissions are to be expected if new cars in the future will comply with the established limit values also in real driving conditions. However, it is a concern that in parts of the world the rise in vehicle use may outweigh the lower emissions per unit of traffic, Proost and Van Dender (2013) mention the development in megacities in developing countries in particular.

According to the WHO, the pollutant with the largest impact on human health is $PM_{2.5}$ (fine particles). $PM_{2.5}$ includes pollutants such as sulfate, nitrates and black carbon, which penetrate deep into the lungs and in the cardiovascular system, therefore posing the greatest risks to human health.[5] WHO guideline limits for annual mean of $PM_{2.5}$ are 10 μg/m^3, the level where no severe damage is expected. A recent worldwide assessment concludes that 92% of the world’s population lives in places

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[3] It monitored black smoke and sulphur dioxide at around 1200 sites in the UK (https://uk-air.defra.gov.uk/networks/brief-history). At the same time, this also became a policy issue in the US, but it was not until 1970 that a Clean Air Act was adopted at the federal level.

[4] In this paper we use PM when we refer to airborn particulate matter and NOx for nitrogen oxides.

[5] Recent evidence suggests that emission PM is considered to be the main source for external health costs. The effects of PM on human health depend on size and content. The size decides where in the breathing system the PM will deposit, as well as the probability that it will deposit at all or just follow the air out again. PM larger than 10 µm mainly gets stuck on the nose hairs etc. and do not enter the breathing system to a significant extent. This is why only PM smaller than 10 µm are included in the regulations, i.e., PM10. PM smaller than 2.5 µm (PM2.5) generally passes down through the throat. Even smaller PM, in the range of 0.05 µm, can penetrate all the way into the alveolar region and the smallest can even pass directly into the blood system following the gas molecules. The possibility of breathing the PM out again is highest in the size range 0.3 µm. Thus, the particle size is probably of largest importance for the toxicity of PM. The content of the PM deposited in different parts of the breathing system is also important, as, e.g., soot PM from exhaust may have larger toxicity than similar sized PM of other content.

where air quality levels exceed this limit value (WHO, 2016). It is not however exceeded in Sweden and the estimated deaths and years of life lost due to air pollution are among the lowest in the world.

The main difference between a gasoline and a traditional diesel engine is that diesel engines have relatively high fuel efficiency, thereby producing less CO_2 emissions per vehicle kilometer, but create more health threatening exhaust emissions, e.g., PM and NOx. Gasoline cars on the other hand emit more organic compounds (VOC) than diesel cars. Over time, technological improvements such as diesel particle filters have reduced some of the emissions but not others, while catalysts reduce NO emissions at the expense of increased emission of NO_2 and NH_3. Another difference between these two technologies is the type of PM emissions. According to Eriksson and Yagci (2009), the formation of PM is different in the two technologies, resulting in a difference in the sizes, composition, and number of the PM[6]. Whether diesel emissions have another chemical composition, and therefore are more harmful than gasoline emissions, has been studied by IARC (International Agency for Research on Cancer). Their conclusion is that it is likely that that diesel emissions are carcinogenic but that this could also be the case for gasoline emission (Benbrahim-Talla et al., 2012).

Mayeres and Proost (2013) is an example of a study where the performance of the two fuels is compared. Their conclusion is that the excise on diesel cars in Belgium should be much higher than the excise on gasoline for two reasons: diesel is more polluting than gasoline and more importantly, through the better fuel efficiency, diesel cars contribute less fiscal revenue per mile. In the paper, the external cost for emissions from a VW Golf is computed, based on emission factors from test cycles. The value used for the environmental damage per tonne for $PM_{2.5}$ is €135,475, €6640 for NOx and €3192 for NMVOC, based on the results from Michiels et al. (2012). These are averages for Belgium based on the assumption that 1.7% or the emissions are emitted in urban metropolitan areas, 21.1% in other urban areas and 77.2% in non-urban areas. According to Mayeres and Proost (2013), the external cost of diesel cars is in general higher than for gasoline cars, irrespective of Euro standard. Although emitting less CO_2, the external cost is higher due to the difference in conventional pollution. This result is modified somewhat by the findings of Michiels et al. (2012). Using a different dispersion model, they find that the outcome will depend on the amount of ozone reduction resulting from NOx emissions. For Euro 5 cars this effect is dominant, so diesel cars outperform gasoline cars, yet the same result is not found for newer cars.

3. Air pollution and transport policy in Sweden

Sweden was one of the countries where the eco-system was badly affected by acid rain and was therefore one of the initiators behind the CLRTAP. Since then, air pollution has been in focus in environmental policy and in national scientific research, hence general assessments of emissions and air quality therefore has a long history. In Stockholm, a first comprehensive study called SHAPE, the Stockholm Study on Health Effect of Air Pollution and their Economic Consequences, where dispersion modeling was used, was done in 1998. Since then, detailed dispersion modeling has been used as input to several health impact studies (Johansson et al., 1999; Nerhagen et al., 2005; Johansson and Eneroth, 2007; Bergström, 2008; Nerhagen et al., 2009; Johansson, Burman and Forsberg, 2009; Johansson et al., 2012; Gidhagen et al., 2013), also for environmental impacts (Klingberg et al., 2014) and for the evaluation of policies, for example congestion charging in Stockholm (Johansson, Burman and Forsberg, 2009; Eliasson et al., 2009; Nerhagen and Janhäll, 2015). The National Transport Administration (NTA) has also commissioned research and regularly produced data on the exhaust emissions from transport that are used as input to the calculations.

Over time, when new vehicles replace old ones, the total emissions will be reduced. In Table 1 we have compiled the results from SHAPE and a number of other Swedish studies to illustrate how the

[6] The latter has been discussed as an alternative to the current measurement of particulate mass, since this measure does not reveal the importance of ultrafine PM. This has not been realized, instead black carbon is measured in addition to $PM_{2.5}$ and PM_{10}.

emissions from single vehicles and the vehicle fleet has changed over time. The emission modelling in National Transport Administration (2017) is based the HBEFA[7] model and represents average estimates for the kilometers driven in the whole country. This report also includes a forecast over how the emissions from the vehicle fleet will change over time until 2030. These estimates are a weighted average of traffic emissions on urban and rural roads. 35% of vehicle kilometers travelled are assumed to take place in urban areas (cities)[8]. As described in the introduction, until a few years ago gasoline has been the fuel used in most cars.

Table 1
Estimates of emissions factors (g/vkm) for the car fleet in Sweden 1998-2030

Source	**Type of car**	**CO_2**	**NOx or NO_2**	**PM**	**HC**
SHAPE (1998)	Car without catalyst – gasoline 1990		2.4	0.019	
SHAPE (1998)	Car with catalyst – gasoline 1997		0.2	0.0053	
Ericsson (2001)	Volvo 940 with catalyst – gasoline	244	0.278	-	0.031
Ericsson (2001)	VW Golf with catalyst - gasoline	242	0.185	-	0.027
Nerhagen et al. (2009)*	Average fleet in Stockholm 2003	-	0.50	0.014	-
Nerhagen et al. (2005)	Average fleet in Stockholm - gasoline 2000	359	0.91	0.02	
NTA (2017)	Average fleet urban areas - gasoline 2016	210	0.22	0.0014	0.65
NTA (2017)	Average fleet urban areas - gasoline 2020	190	0.15	0.0013	0.55
NTA (2017)	Average fleet urban areas - gasoline 2030	150	0.09	0.0013	0.45
Nerhagen et al. (2005)	Average fleet in Stockholm – diesel 2000	266	1.24	0.18	
NTA (2017)	Average fleet urban areas - diesel 2016	140	0.43	0.0072	0.04
NTA (2017)	Average fleet urban areas - diesel 2020	130	0.28	0.0044	0.04
NTA (2017)	Average fleet urban areas - diesel 2030	110	0.11	0.0030	0.04

* Regarding the modelling in Stockholm, an update of the models in 2006 resulted in a reduction in emission factors for PM with 80% while the EF for NOx increased somewhat according to Nerhagen et al. (2009). Hence, as with all modelling, the input to these kinds of calculations contributes to the uncertainties in the final estimates.

From the results in Table 1, we can see that the emissions of NOx appear to have been reduced considerably since the 1990s while for PM and CO_2 a reduction took place around 2010. This is about the time when policy measures were introduced in Sweden to support the introduction of low emission vehicles. In our calculations we focus on PM and NOx. However, from the table it is clear that according to the National Transport Administration (2017) the emissions factors for CO_2 and HC are much lower for diesel cars. This is something that also needs to be considered in a complete assessment of the benefits and costs of using the different fuels. What is also clear is that there are variations in these estimates, which is due to the composition of the vehicle fleet and the traffic situation for which the calculation is made. There also appear to be uncertainties, for example the estimate for gasoline cars for PM and CO_2 varies quite substantially between different studies.

Another important thing to note is that the calculations are based on emissions factors from test cycles and hence the calculations will be an underestimate of the "true" external cost. However, this is the approach used in other studies such and Michiels (2012) and Mayres and Proost (2001, 2013) and to compare the estimates, we need to do the same. Furthermore, as discussed in the Section 5, there are also differences in emissions between test cycles and real driving for gasoline cars.

Due to these uncertainties, the exact influence of car use and the influence of different types of fuel on air quality, as opposed to other sources, is generally unknown. In Stockholm, heavy duty

[7] Handbook Emission Factors for Road Transport, see http://www.hbefa.net/e/index.html.

[8]This is a higher share of urban driving than the assumption in Mayeres and Proost (2013) despite Sweden being a more sparsely populated country. An explanation could be that the statistical definition of urban areas in Sweden includes conurbations over 200 inhabitants.

vehicles for example also make an important contribution. What we do know from measurement data collected by the Swedish Environmental Protection Agency is that the reduction in emissions for single vehicles described in Table 1 has resulted in large reductions from transport, especially for NOx and NMVOC (Swedish Environmental Protection Agency, 2015). It has also influenced air pollution concentrations. According to the Environmental Division of the City of Stockholm, urban background concentrations of NOx have been reduced by 70 percent since the beginning of the 1980s and 60 % for NO_2 in the city (SLB Analysis, 2017). This has also resulted in the limit values established in the EU air quality directive being met in most parts of the country (European Environmental Agency, 2016). However, despite the progress, local air pollution from road traffic is still in focus resulting in policy proposals such as low emission zones for both heavy and light duty vehicles, the possibility to ban the use of studded tires, subsidies to support the introduction of electric vehicles and the transformation of the vehicle fleet used in public transport and finally, modified vehicle taxation to provide incentive for the use of low emission vehicles. For the two largest cities congestion charging has also been implemented to curb traffic.

An important reason for this continued focus on air quality in Sweden is the ambitious implementation of the EU air quality directive. As described in Coria et al., (2015) evaluation of the air quality standards is done at street level.[9] This is contrary to the approaches recommended by the WHO and the method for health impact assessment adopted internationally where the focus is on reductions in urban background. Sweden has also adopted a stricter limit value for NO_2. According to Coria et al., (2015) it is 90 µg/m^3 per hour not to be violated more than 175 times per year. The corresponding limit value according to the EU air quality directive is 200 µg/m^3 per hour, not to be exceeded 18 times per year.[10] Since concentrations are considerably reduced with distance to road both horizontally and vertically, making assessments based on measurements in street canyons as in Coria et al. (2015) will overestimate the societal benefit of emissions reductions.

Of particular interest for this paper is that in 2017, the government presented a proposal for a so-called Bonus Malus system (Ministry of Finance, 2017). According to this system cars are to be taxed according to the emissions of CO_2 above 95 g/vkm. For diesel cars and light duty vehicles, a fixed cost to cover the difference in tax income due to lower energy use per km is added as well as a fixed cost of

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[9] The modelling done in this paper is based on data from Hornsgatan in Stockholm which is one of the streets with the highest concentrations of pollutants at street level in the whole of Stockholm, and possibly also in Sweden.

[10] The assessment of the air pollution situation is complicated and guidance on how to implement the requirements in the Air quality directive has therefore been provided in recent years, see for example http://ec.europa.eu/environment/air/quality/legislation/pdf/IPR_guidance1.pdf. In this guide the representativeness is for example discussed in the following way: "Where several individual exceedance situations (e.g. different exceedances observed by traffic stations and/or predicted at the roadside by model within the same city) have been grouped into one macro exceedance situation, the source appointment presented must be relevant to each of the individual exceedance situations and be applicable to the monitoring station or modelled location with the maximum concentration/number of hours exceeding the limit value. If there is significant difference in source appointments across the individual exceedance situations, Member State should consider whether it is legitimate to group them into a macro exceedance situation or whether it would be better to split them into smaller groups."

In Annex III of the air quality directive it is stated that sampling points directed at the protection of human health shall be sited in such a way as to provide data on the following:

— the areas within zones and agglomerations where the highest concentrations occur to which the population is likely to be directly or indirectly exposed for a period which is significant in relation to the averaging period of the limit value(s),

— levels in other areas within the zones and agglomerations which are representative of the exposure of the general population,

furthermore, that sampling points shall, where possible, also be representative of similar locations not in their immediate vicinity.

250 SEK per year for new cars. This is added to make up for the difference in external environmental costs of higher PM and NOx emissions compared to gasoline vehicles. This cost component was introduced about ten years ago after recommendations in the Government Official Report 2004:63. The argument then was mainly the problem with higher PM emissions from diesel cars and it was stated that this addition could be reduced if the difference in emissions between diesel and gasoline cars were reduced.

Since this will be the case with the new Euro 6 demands, the most recent inquiry suggested that the addition should not be required (Government Official Report 2016:33). The government however, on the basis of recommendations from the Swedish Environmental Protection Agency, has decided to keep it with reference to the differences in NOx emissions. No impact assessment to inform this decision has been done, probably because Sweden lacks a tradition of doing impact assessments of in the area of environmental policy (OECD, 2014; Wallström and Söderqvist, 2016; Hansson and Nerhagen, 2017; Nerhagen, Forsstedt and Hultkrantz, 2017; Andersson et al., 2018). In the new proposal, only cars running on alcohol and gas have exemptions on the CO_2 related tax. This differential treatment of bio-diesel has given rise to criticism in the referral process (The Swedish National Financial Management Authority, 2017).

4. Emissions and external health costs for diesel and gasoline cars in Greater Stockholm

The aim of these calculations is to assess the influence on public health of new gasoline cars compared to new diesel cars. Table 2 provides an illustration of the difference between the emissions from the average car fleet in Stockholm in 2003 as in Nerhagen et al. (2009) and new cars. For new cars we use estimates from testing results from an evaluation commissioned by the Swedish Transport Agency (Köhler, 2014). As expected, the emission factors for the car fleet in Stockholm in 2003 are much higher than the average estimates in Köhler (2014) for both diesel and gasoline cars. According to these results, the emissions from new diesel cars compared to similar gasoline cars are higher for NOx but lower for PM.

Table 2
Estimates of emissions factors used in calculations (g/vkm)

Source	**Type of car**	**PM**	**NOx or NO2**
Nerhagen et al. (2009)	Average car fleet in Stockholm 2003	0.014	0.50
Köhler (2014)	Test positive ignition - gasoline	0.00049	0.016
Köhler (2014)	Test compression ignition - diesel	0.00014	0.15

The approach we use is the impact pathway approach that has been developed in the EU-funded ExternE projects (Friedrich and Bickel, 2001; Bickel and Friedrich, 2005).[11] [12] In this method, the calculation is based on an assessment of the impacts of the emissions on the environment and/or human health and the economic values placed on these impacts. As discussed in Viscusi and Gayer (2005), several aspects need to be accounted for when performing these calculations in practice. An example is nonlinear relationships in the exposure modeling since the pollutants have very different chemical and physical properties (Muller and Mendelsohn, 2007; Jensen et al., 2008). [13] Another is

[11] The US Environmental Protection Agency (EPA) has developed a similar tool referred to as BenMAP (Environmental Benefits Mapping and Analysis Program) which is used in regulatory impact assessments.

[12] Following the revision of the EU Thematic Strategy on Air Quality, WHO coordinated two projects, REVIHAAP (Review of Evidence on Health Aspects of Air Pollution) and HRAPIE (Health Risks of Air Pollution in Europe), to provide the latest scientific evidence on the health effects of all pollutants regulated in current directives. An impact assessment and a cost-benefit analysis (CBA) based on these updated exposure-response relationships have also been carried out (Holland, 2014b).

[13] In most applications linear relations are assumed (Small and Kazimi, 1995; Olsthoorn et al., 1999; Bickel et al., 2006; Jensen et al., 2008). Why this is a reasonable assumption for most part of the chain is discussed at

that, since studies providing estimates of relative risk in most cases have used measurement values from fixed monitoring stations in the urban background (e.g., rooftops), concentration levels corresponding to such locations should be the basis for the extrapolation (Ostro, 2004).

A formal description of how the external health cost calculations are done in the impact pathway approach is given by equation (1)[14]. It describes the yearly cost (benefit) due to an increase (reduction) in concentration, C, from a change in emissions of a certain pollutant from a specific source:

$$\text{External health cost} = \Delta\text{yearly exposure} \cdot \text{effect} \cdot \text{value} =$$

$$= (\Delta C_{a;i} \cdot POP) \cdot (B_{a;j} \cdot P_{i;j}) \cdot V_j \qquad (1)$$

where

$\Delta C_{a;i}$ = change in average annual concentration at urban background for pollutant i ($\mu g/m^3$)

POP = population exposed to $\Delta C_{a;i}$

$B_{a;j}$ = baseline annual health impact rate in population for health impact j (number of cases per inhabitant)

$P_{i;j}$ = effect on health impact j per $\mu g/m^3$ of pollutant i (relative risk)

V_j = value of health impact j.

This calculation has to be done separately for each pollutant since the effect estimates $P_{i;j}$ (the exposure-response functions) differ. The cost calculated for each pollutant i and each health impact j can then be added up to arrive at the total yearly health cost for the change in emissions from each source, such as different fuels or different types of vehicles.

To calculate the expected health impacts, we use the results from the EU/WHO HRAPIE project (Holland, 2014a) as summarized in Table 3. In the CBA undertaken by Holland (2014b), not all of the functions recommended in HRAPIE were used. The reason for this was lack of exposure data and the risk of double counting in the case of ozone and NO_2 regarding chronic mortality. The functions are classified using a system where those for which confidence is highest are given an "A" rating, and "B" for which less confidence in the scientific evidence. Furthermore, an asterisk "*" is added to the rating for effects that are additive. To be noted is that it is assumed that NO_2 has an impact on acute mortality, although this is a pollutant mostly discussed in relation to respiratory problems. The recommended exposure response function is in this case based on the results from one single study. Additional assumptions needed are a baseline for the population where we have assumed 1010 per 100,000 as in Nerhagen et al., (2009) and Nerhagen, Bellander and Forsberg (2012). For deaths due to chronic exposure, 11.2 years of life are assumed to be lost, and for deaths due to acute exposure, 1 year. The monetary values in Table 3 are the same as those used in Nerhagen et al. (2015).

length in Small and Kazimi (1995) and Bickel et al. (2006). Small and Kazimi (1995) argue that for small changes in emissions, these relationships can be assumed to be linear while other studies have shown mixed results (Muller and Mendelsohn, 2007; Jensen et al., 2008). However, assuming linearity implies that only minor changes from the current state can be evaluated. The reason for this is that both economic values and exposure-response relationships are likely to change the further we move away from the current situation (Viscusi and Gayer, 2005).

[14] This is a modification of the equation used in Ostro and Chestnut (1998).

Tabell 3
Exposure response functions and monetary values used in the external cost calculation

Impact	Rating	Population	Exposure metric	RR (95% CI) per 10 μg/m^3	Monetary value (SEK$_{2012}$)
All cause mortality from chronic exposure as life years lost or premature deaths	A*	Over 30 years	$PM_{2.5}$, annual average	1.062 (1.040 – 1.083)	1,095,000 per VOLY
Respiratory hospital admissions	A*	All ages	$PM_{2.5}$, annual average	1.0190 (0.9982 – 1.0402)	22,800 per admission
All cause mortality from acute exposure	A*	All ages	NO_2 annual mean	1.0027 (1.0016-1.0038)	1,095,000 per VOLY
Respiratory hospital admissions	A*	All ages	NO_2 annual mean	1.0180 (1.0115 – 1.0245)	22,800 per admission

In the calculations we use the results from the study in Greater Stockholm by Nerhagen et al. (2009)[15]. To evaluate the impact on air quality when parts of the vehicle fleet is renewed we will assume that 10% of the vehicle km driven (600 million km) are replaced by new cars, either gasoline or diesel. The external cost calculations for this reference case, with the car fleet in 2003, are presented in Table 4. Presented in the columns are the inputs where the first is central to the calculation, the expected population exposure. The last column is the output in terms of the estimated total external cost divided by the vehicle km of 600 million.

If we instead calculate the external cost per ton, we get estimates of about 183,591 euro for PM and about 8,393 euro for NOx (including the cost for secondary PM). This is in the range of the estimates used in Mayeres and Proost (2013). However, while their cost estimates represent traveling mostly on the countryside in Belgium, our estimate is from the most densely populated area in Sweden. For traffic on the countryside and smaller cities in Sweden, the external cost will be much lower than these estimates since the population density is lower.

Table 4
External cost (EC) calculation of emissions from 10 per cent of average car fleet in Greater Stockholm in 2003.

	Population exposure (person μg/m^3)	Baseline	Relative risk (RR) per 1 μg/m^3	Monetary value (SEK$_{2012}$)	External cost/vkm
PM – chronic mortality	15,015	0,1	1.0062	1,095,000	0.0192
PM – respiratory hospital admission	15,015	0,1	1.0190	22,800	0.0059
NOx – acute mortality	441,358	0,1	1.0027	1,095,000	0.0220
NOx – respiratory hospital admission	441,358	0,1	1.0180	22,800	0.0030
Nitrates – chronic mortality	13,461	0,1	1.0062	1,095,000	0.0172
Average vehicle fleet 2003					0.0673

To calculate the cost when the average cars in 2003 are replaced by new ones, we use the emission factors from Köhler (2014). Our calculation is based on the assumption that these cars have the same driving pattern as the average car in Nerhagen et al. (2009), hence contributing in the same

[15] Details about the dispersion and exposure modeling are provided in Johansson and Eneroth (2007) and Bergström (2009).

way to population exposure. Some basic data used, and the results of the calculations regarding the differences in emissions, are presented in Table 5. By replacing 10 percent of the vehicles with new cars the emission reduction is about 8 tons for PM and between 200 and 300 ton for NOx.[16] The difference between the two alternative fuels is that with the use of new gasoline cars instead of diesel, the total emissions for PM will be 0.21 tons higher while it will be 79,4 tons lower for NOx.

Table 5
Assumptions in calculations and emission reductions if new cars replace average car fleet in 2003 in Greater Stockholm

	New gasoline cars	**New diesel cars**
Area	Greater Stockholm	Greater Stockholm
Population	1,405,600	1,405,600
Vehicle km used in calculation (million)	600	600
PM emissions with average emission factors from Nerhagen (2009) (ton)	8.2	8.2
PM emissions with emissions factors from Köhler (2014) (ton)	0.294	0.084
PM **reduction** with new cars (ton)	7.91	8.12
NOx emissions with average emission factors from Nerhagen (2009) (ton)	303	303
NOx emissions with emissions factors from Köhler (2014) (ton)	10.6	91
NOx **reductions** with new cars (ton)	292.4	212

In Table 5 we see that the total emissions with new cars will be reduced quite substantially. This will in turn have an impact on population weighted concentrations for the two pollutants. In Table 6 we have compiled the results for the calculation of the expected impact on population weighted exposure from the reductions in total emissions with the use of new cars. We assume that the relative reduction in emissions has the same impact on the estimated population exposure. Hence, the PM reduction in emissions from gasoline cars of 8.106 ton equalizes a reduction in emissions of 99 per cent. Using the information in Nerhagen et al. (2009), and the conversion in Bergström (2008) used to arrive at an estimate of the number of people exposed to 1 $\mu g/m^3$ to simplify the health impact calculation, we can calculate the reduction in exposure in Greater Stockholm. A reduction in emissions of 99 per cent for gasoline cars will reduce the population exposure with 13,601 person $\mu g/m^3$, from 14,100 to 499. Based on the results in Bergström (2009), we can also calculate the impact on the population in the rest of Europe due to reduced concentrations of PM and secondary PM from the emissions of NOx (nitrates). Hence for example, for PM we have to add an additional reduction in exposure of 883 person $\mu g/m^3$.

[16] Our calculation is based on NOx and not NO_2 which implies a slight overestimation of the change in concentrations.

Table 6
Exposure estimates of PM and NOx from emissions in Stockholm, average car fleet 2003 vs new cars

Emissions	**Total emission (ton)**	**Exposure Greater Stockholm (person µg/m^3)**	**Exposure in rest of Europe (person µg/m^3)**
Exhaust PM from 10% of average fleet 2003	8.2	14,100	915
Reduction with new gasoline cars	7.910	13,601	883
Reduction PM with new diesel cars	8.116	13,956	906
NOx from 10 % of average fleet 2003	303	441,358	0
Reduction with new gasoline cars	292.4	425,918	0
Reduction with new diesel cars	212	308,805	0
Secondary PM from 10 % of average fleet 2003		471	12,990
Reduction with new gasoline cars		439	12,536
Reduction with new diesel cars		329	9,089

In Table 7, we have calculated the cost for the emissions from new cars based on the results from Table 6 regarding the reduction in population exposure. This time we do separate calculations for gasoline and diesel cars. Again, the first column reveals the input to the calculation which is the estimated population exposure. Since this is much lower compared to the reference case in Table 4, the estimated external cost per vehicle km in the last column is also much lower. We also find that the cost for new diesel cars is higher compared to new gasoline cars. The main reason for this is the difference in estimated exposure and hence cost from the emissions of NOx. It is important to note that part of this difference is due to NOx contributing to the formation of secondary PM which mainly influence the air pollution concentrations, and hence the population, outside the area of Greater Stockholm.

Table 7
External cost (EC) calculation of emissions for new cars, gasoline and diesel.

	Population exposure with new cars (person µg/m^3)	**Baseline**	**Relative risk (RR) per 1 µg/m^3**	**Monetary value (SEK$_{2012}$)**	**External cost/vkm (SEK$_{2012}$)**
Gasoline cars total					0.0022
PM – chronic mortality	531	0,1	1.0062	1,095,000	0.0007
PM – respiratory hospital admission	551	0,1	1.0190	22,800	0.0002
NOx – acute mortality	11,440	0,1	1.0027	1,095,000	0,0006
NOx – respiratory hospital admission	11,440	0,1	1.0180	22,800	0.0001
Nitrates – chronic mortality	486	0,1	1.0062	1,095,000	0.0006
Diesel cars total					0.0129
PM – chronic mortality	144	0,1	1.0062	1,095,000	0.0002
PM – respiratory hospital admission	144	0,1	1.0190	22,800	0.0001
NOx – acute mortality	132,553	0,1	1.0027	1,095,000	0.0066
NOx – respiratory hospital admission	132,553	0,1	1.0180	22,800	0.0009
Nitrates – chronic mortality	4,043	0,1	1.0062	1,095,000	0.0052

In Table 8, we have summarized the results from the three different external cost calculations in Table 4 and 7. The results reveal that the external cost is much lower for new cars compared the average vehicle driven in 2003. Comparing gasoline with diesel, we find that the additional benefit from driving new gasoline cars instead of diesel cars is 0.0107 SEK/vkm. This equals a benefit of 161

SEK, or about 16 euro, per year for a car driven 15000 km. Policy wise this implies that an extra yearly tax of 250 SEK (25 euro) for new diesel cars is too high in comparison to the external cost imposed by them if they are driven in Stockholm. For diesel cars used mainly outside Stockholm the cost is much too high. In this case, only the difference in the external cost due to the formation of nitrates is relevant since the difference in PM exposure is negligible. For this pollutant the additional cost of a diesel car driven 15 000 km is 0.0046*15000=69 SEK, i.e. 7 euro. For a complete assessment of the impact of diesel, policy makers should also account for other benefits such as reduced CO_2 emissions.

Table 8
Summary of calculations of external costs and estimated yearly costs (SEK_{2012})

	Average car fleet in 2003	**New petrol cars**	**New diesel cars**
PM, External cost/vkm	0.0251	0.0009	0.0003
NOx, External cost/vkm	0.0250	0.0007	0.0075
Secondary PM, External cost/vkm	0.0172	0.0006	0.0052
Total cost, External cost/vkm	*0.0673*	*0.0022*	*0.0129*
Cost per year for cars driven 15000 vkm	1,010	33	194

Based on these results we can conclude that there are significant differences in external cost between different geographical areas in Sweden, but therefore also between Sweden and other parts of Europe. Stockholm is most likely the city with the highest external cost for car driving in Sweden although much lower than the cost in urban areas in Belgium. According to Michiels et al. (2012) the average estimate for $PM_{2.5}$ emissions in these areas is euro 432,000 per ton. Therefore, these results also underline the conclusion in Amann et al. (2011), namely that the harm done by emissions varies over Europe and therefore need to be accounted for in the discussion of cost-efficient emission reductions. In the case of emissions from cars, introducing similar policy measures as in other larger cities in Europe may not be worth the while in many parts of Sweden health wise since the population density is very low by international standards.

5. Uncertainties

There are several uncertainties related to these kinds of calculations and we will discuss some of them here in the order as the calculation is done, hence emissions, dispersion and exposure modeling and health impact calculation. As for the monetary values used, the motivation for these are presented in Nerhagen et al. (2015). They are in line with estimates used in calculations at the EU level where they have been discussed at length (for example Holland, 2014b), we will therefore not repeat that discussion here but focus on other inputs into these calculations.

Regarding emissions, ‘Dieselgate’ highlighted the difference in emission factors for cars between real driving conditions and the type-approval values presented by the manufacturer. The latter are based on a standardized test cycle called NEDC. That there are differences has been known for a long time. According to an assessment of 2001-2011 new gasoline as well as diesel passenger cars sold in Germany (Mock et al., 2012), there was a gap between type-approval and real-world fuel consumption/CO_2, a gap that had increased from about 8% to 21%. According to the report, reasons for this are 1) that the experimental design allows for differences in the protocol presently used by many developers to minimize the CO_2 emissions within the standard, 2) an inability of the current test cycle to represent all the different real-world driving conditions that exist, and 3) an increasing share of vehicles equipped with air conditioning systems. According to a Norwegian study another explanation relevant in Nordic countries is cold starts (Hagman and Amundsen, 2013).

Information about the difference between test cycle emission factors and more realistic driving condition for VW diesel cars are also presented in two Swedish studies by Ecotraffic (2012 a, b), work commissioned by the Swedish NTA. Table 9 summarizes the results from tests on specific vehicles.

As can be seen, the table confirms that that fuel consumption and emissions of PM, NO_x, and CO_2 are higher in real driving conditions than in the test cycle. The discrepancy however is much larger than that reported in Mock et al. (2012), for fuel consumption almost twice as high. Moreover, the results point to something not discussed in relation to 'Dieselgate' and that is that there are differences also for other important emissions.

Table 9
Emission factors for different types of VW diesel cars (source: Ecotraffic 2012 a, b)

	Fuel consumption (l/100 km)	**CO_2 (g/vkm)**	**NOx (g/vkm)**	**PM (g/vkm)**	**Comment***
Diesel A, 2010 - Type approval values	4.4	115	0.17	0.00016	Euro 5 DPF
Diesel A, 2010 - Artemis urban	7.9	188	0.49	0.00070	Euro 5 DPF
Diesel B, 2010 - Type approval values	4.6	120	0.09	0.00032	Euro 5 DPF
Diesel B, 2010 - Artemis urban	8.0	210	0.75	0.00071	Euro 5 DPF
Diesel C, 2011 - Type approval values	4.9	129	0.16	0.00041	Euro 5 DPF
Diesel C, 2011 - Artemis urban	7.7	199	0.99	0.0013	Euro 5 DPF

* According to the description in the text.

In our view, these results illustrate the need to be aware and make assessments of the uncertainties on emission factors in general in order to support efficient policy making. The problem of focusing on emissions from a single source or a single pollutant such as NOx, is that outdoor air pollution is a mixture of multiple pollutants originating from a myriad of natural and anthropogenic sources. Transport, power generation, industrial activity and biomass burning are important anthropogenic sources. The mix of pollutants in outdoor air also varies substantially over space and time due to sources and the effect of atmospheric processes, including oxidation and weather.

This need to account for different sources of emissions as well as and changes over time is illustrated by a comparison we have done to validate our results against a more recent health impact assessment of emissions from diesel cars in Greater Stockholm (Johansson et al., 2012). Both emissions and exposure calculation are performed by SLB-analysis, a department at the Environment Division in the City of Stockholm. [17] An overview of the two studies is presented in Table 10. The comparison reveals that inputs to the calculation such as inhabitants change over time, but also the assumptions used regarding traffic, for example the emissions. According to the results, the total emissions of cars has decreased between 2003 and 2010 in Greater Stockholm while the emissions for heavy duty vehicles, where diesel is mainly used, has increased quite substantially. Hence, these results illustrate the problem with focusing solely on diesel cars. From an economic perspective, the Bonus Malus system results in differential treatment between diesel as a fuel used in new cars versus diesel as a fuel used in heavy duty vehicles which may not be an efficient way to improve air quality in Stockholm.

[17] We have not been able to verify if this is a trend or not although dispersion and exposure modeling has been done in more recent studies. That the input to the calculations are seldom reported is a problem we have often encountered, two illustrations of this are Lövenheim (2014) and SLB-analys (2016).

Table 10
Inputs and results in two studies with dispersion and exposure modeling for Greater Stockholm

	Nerhagen et al. (2009)	Johansson et al. (2012)
Year of calculation	2003	2010
Area	Greater Stockholm (35kmx35 km)	Greater Stockholm (35kmx35 km)
Population	1,405,600	1,628,528
PM exhaust total emissions (ton)	82 (LDV) 40 (HDV)	61 (LDV) 62 (HDV)
PM pop.weighted conc. (ug/m3)	0.136 (all vehicles)	0.132 (all vehicles)
NOx emissions (ton)	3029 (LDV) 2645 (HDV)	2613 (LDV) 3459 (HDV)
NOx pop.weighted conc. (ug/m3)	5.68 (all vehicles)	6.43 (all vehicles)

Another uncertainty is related to the dispersion and the exposure modeling. In this study we have assumed that the relative reduction in emissions has the same impact on the population weighted exposure. According to Johansson et al. (2012) however, a reduction of PM emissions of 33 tons (26%) would only reduce the population weighted concentration with 0.025 $\mu g/m^3$ (19%). Hence, this is not a linear relationship as we have assumed in our calculation. One reason for this difference is probably that Johansson et al. (2012) also include HDV which to some extent travel on roads farther away from populated areas relative to cars. For NOx on the other hand, they report the same percentage decrease in concentrations as in emissions (4 %). Hence, based on this we believe we have used a reasonable assumption. However, it is clear that the assumptions and data used in the dispersion and exposure modeling will influence the results of the external cost calculations.

The relative risk estimates used in the calculations will also have a significant influence on the results. This is discussed both in Nerhagen et al., (2009) and Johansson et al., (2012). In the latter a comparison is made between using exposure response functions for NOx (Nafstad et al., 2004) compared to $PM_{2.5}$ (Jerret et al., 2004) when calculating effects on chronic mortality. This comparison reveals the calculations based on NOx result in the number of extra deaths being six times higher than if their estimate for PM from is used. NOx has been used in many Swedish health impact assessments since it is considered to be a good indicator for PM emissions, an assumption endorsed by the Swedish EPA. However, as discussed in Johansson et al. (2012), the ratio between NOx and exhaust PM is different today than a number of years back. Therefore, using estimates for NOx to calculate the effects on mortality in the population is uncertain.

Finally, emissions from combustion also take part in chemical reactions, for example ozone formation. Nerhagen et al. (2009) focused on primary and secondary PM and therefore ozone formation is not included in the calculation in this paper. Ozone is a secondary pollutant and the concentration depend on sunshine, NO, NO_2, and organic species. Both the relatively high emissions of VOC from gasoline cars and the relatively high emissions of NOx from diesel cars contribute to ozone formation, even though the amount of NOx and VOC already in the air will also have a large impact on the result. In Sweden, ozone formation is more, but not totally, dependent on NOx emissions, compared with central Europe, where VOC is of larger importance. However, as discussed in Fridell et al. (2014), in Sweden the conditions for ozone formation differ from other parts of the world and the impact of minor changes in emissions is likely to be small. We therefore conclude that our example reveals the most important differences in health impacts between the emissions from gasoline versus diesel cars.

6. Diesel use in the transport sector in Sweden, is it a problem – discussion

In this paper we have used information on external costs to investigate if the new system for vehicle taxation, discouraging the use of diesel cars, can be motivated on economic grounds. We have described and applied the impact pathway approach, a method where external costs are calculated based on information on emissions, dispersion, population exposure, health impacts and the monetary value placed on these impacts. The approach is based on many assumptions and does not cover all aspects. Still, it has been applied for several years and rests on results from the most recent scientific research in each discipline.

We performed our calculations based on the results from a previous study done in Stockholm in 2009. To compare the costs on public health from diesel versus gasoline cars, we assumed that for ten percent of the vehicle km travelled in Stockholm new cars were used. Based on this we concluded that the PM emissions are higher for gasoline cars while the NOx emissions are lower. In contrast to the recent study by Jonson et al. (2017) on the impact of "Dieselgate", we calculate the health impacts on both the local and the regional scale. The latter due to the dispersion of both primary and of secondary PM from NOx (nitrates). According to our calculations, the additional cost of driving a diesel vehicle is 0.0107 SEK/vkm, resulting in a yearly cost for a car driven 15000 km in Stockholm of 169 SEK (17 euro). This is a result similar to the one in Mayeres and Proost (2013) although their estimate concerns the average emissions for the vehicle fleet in Belgium while ours is an estimate for probably the most populated area in Sweden.

We also find that the external cost is significantly reduced with the introduction of new cars, irrespective of fuel used. Regarding the additional yearly tax placed on diesel cars we find that this is generally too high for cars driven in Stockholm, although it may be relevant for the inner city. However, for other parts of Sweden we expect it to be much too high due to lower population densities close to the roads. We have also discussed the uncertainties in these calculations and among other things there is the question of assuming linear relationships for the exposure-response functions in parts of Sweden where the air pollution levels are very low by international standards. Hence, based on this and given that we have not included the positive aspects of using diesel, and especially bio-diesel, from reduced CO_2 emissions, we conclude that diesel as a fuel in Sweden is more a blessing than a curse.

The emission data presented in this paper also highlights the need to undertake situation specific empirical assessments in the design of policy instruments to curb air pollution. Comparing the results in the study we have used in our calculations with a more recent health impact assessment for Stockholm, we find that the problem with NOx now appear to be due to an increase in the emissions from heavy duty vehicles while the emissions for cars (light duty vehicles) has decreased. Hence, the expected impact of the additional yearly tax on the air pollution concentrations is somewhat uncertain. Regrettably, the decision on this tax in the new proposed Bonus-Malus system does not appear to be prepared based on such an assessment. This is most likely due to the lack of tradition in Sweden, as found in recent studies (Radaelli, 2010; OECD, 2014; Nerhagen and Forsstedt, 2016; Nerhagen, Forsstedt and Hultkrantz, 2017), of doing regulatory impact assessments.

A final remark is that, as is evident for our calculations and the discussion on uncertainties in this paper, these kinds of calculations are based on a large number of assumptions. In the Swedish health impact assessments currently done, transparency is lacking which poses problems for research and evaluation and therefore also for efficient policy making. In the design of policy instruments such as Air Quality Standards targeting air pollution from cars, it is for example important not to focus on evaluation of concentrations in street canyons since they are comparatively high and not likely to be representative for the exposure of the general population. Models and calculations used for impact assessments therefore need to be tailored to the purpose and revealing the assumptions used. If done carefully, these calculations can provide information on how different aspects influence the outcome of a certain policy choice, what impacts are large and what are small.

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marginal costs[18] for using the country's national infrastructure, i.e. roads, railways, airports and sea infrastructure. This work has resulted in three reports on the calculation of the external cost of transport air pollution (Nerhagen m fl., 2015; Nerhagen, 2016 and Nerhagen and Andersson-Sköld, 2018). I am grateful to Mattias Haraldsson and Lars Hultkrantz for useful comments and suggestions.

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