Mind the price gap: How optimal emission pricing relates to the EU CO_2 reduction targets

Benjamin Kickhöfer[1,*], **Amit Agarwal**[2]

[1] Institute of Transport Research, German Aerospace Center (DLR), Berlin, Germany

[2] Transport Systems Planning and Transport Telematics, Technische Universität Berlin, Germany

* Corresponding author: benjamin.kickhoefer@dlr.de

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Abstract

From the transport economic literature it is known that optimal pricing of (environmental) externalities improves the urban system. In contrast to theory-based optimal pricing strategies, real-world policy setting often follows so-called 'backcasting' approaches where certain goals are set, and policy measures are implemented in order to reach those goals. This paper aims to compare these two approaches in a simulation environment. It identifies, for a specific case study, the price gap between the toll levels obtained from optimal emission pricing and the toll levels resulting from the goal to reduce global greenhouse gas emissions in the transport sector by 20% until 2020 with respect to 1990 levels. For this purpose, an optimal emission pricing strategy is applied to a real-world scenario of the Munich metropolitan area in Germany. The highly differentiated tolls relate to individual exhaust emissions, i.e. they are calculated using damage cost estimates from the literature and vary over time of day, with traffic situation, and with vehicle type. The results indicate that the desired reduction in CO_2 emissions is not reached, and that the initial damage costs estimates need to be multiplied by a factor of 5 in order to reach the goal, yielding a price of 350 EUR/ton CO_2. When aiming at a decrease of the overall emission costs by 20% (CO_2 *and* local pollutants), the initial cost estimates need to be multiplied by a factor of 10. Furthermore, it is shown that the major contribution to the overall emission reduction stems from behavioral changes of (reverse) commuters rather than from urban travelers; under some circumstances, urban travelers even increase their CO_2 emission level. Hence, the study rises awareness that there might be conflicting trends for different types of pollutants and different types of individuals: Pricing emissions does not necessarily result in a reduction of all pollutants or of the emissions levels of all users. This shows how agent-based simulations can be used to provide valuable insights and decision support in such possibly counter-intuitive situations.

Keywords: Sustainable transport, Emissions, Air pollution, Optimal pricing, Backcasting, Agent-based modeling

1 Introduction

Growing motorization and urban sprawl has led to significant increases in transport-related negative effects (emissions, congestion, accidents, noise etc.). For instance, passenger and freight transport in Europe has grown substantially between 1990 and 2010, and the corresponding CO_2 emissions have increased by about 20% (Schoemaker et al., 2012; European Environment Agency, 2016). Knowing the possible negative impacts of climate change, the European Union (EU) and the international community have agreed on the need to reduce global greenhouse gas (GHG) emissions in order to limit global warming below 2° Celsius (European Commission, 2011; FCCC/CP/2015/L.9/Rev.1, 2016). To achieve this goal, the directive 2008/101/EC (2008) sets the goal to reduce global GHG emissions in the transport sector by at least 20% until 2020 with respect to 1990 levels. In the light of the above, the EU has launched several regulation schemes[1]: (a) emission trading system (b) use of renewable energy sources (c) reduction in the energy use of buildings and industries and (d) improvement in fuel and vehicle technology.

Looking at the historic trends for the EU-28[2] , the reduction in GHG emissions from all sectors has already reached the 20% reduction goal; however, an increase of 10-15% is observed for GHG emissions from road transport (European Environment Agency, 2016, online data code:*env_air_gge*). Future forecasts indicate that passenger and freight transport might grow more than 80% by 2030 with respect to 1990 levels (Schoemaker et al., 2012).

For the transport sector, the EU regulations from above mainly concentrate on improvements in fuel and vehicle technology in order to balance the increase in demand (Romm, 2006). As a consequence, the average CO_2 emissions from new cars registered in 2014 are as low as 123.4 $g\ CO_2/km$, below the 2015 target of 130 $g\ CO_2/km$ (EEA, 2015). However, these numbers are questionable as the

[1] These are taken from the EU 2020 climate and energy package http://ec.europa.eu/clima/policies/strategies/2020/index_en.htm.

[2] See http://europa.eu/about-eu/countries/member-countries/ for a complete list of all EU member countries.

growing gap between "type-approved" (emissions tests under laboratory conditions) and on-road CO_2 emissions from the vehicles indicates (Mock et al., 2014; EEA, 2014). That is, improvements in vehicle and fuel technology might not be effective under on-road conditions, which reduces the chance to reach the GHG emissions targets. Additionally, the improvements in fuel and vehicle efficiency (if actually happening) implicitly lead to a reduction in the generalized costs of travel. This, in turn, can counteract the positive impact of the technology improvements through rebound (or takeback) effects[3] (see, e.g., Divjak, 2009; Parry and Small, 2005; Barla et al., 2009).

For these reasons, many researchers have criticized the technology-oriented policy setting of the EU and pointed out the important role of regulatory demand- and supply-side policies in order to reach the CO_2 reduction goals (see, e.g., Emberger, 2015; Banister and Hickman, 2009; EEA, 2008; Parry et al., 2014). In contrast to relatively 'hard' traffic restrain policies in the central areas of cities (see Elmberg, 1972; Buehler and Pucher, 2011; Fernandes et al., 2014; Zhou et al., 2010; Cai and Xie, 2011, for real-world examples), pricing schemes offer a less restrictive and more dynamic opportunity of managing transport-related problems in cities. From a theoretical point of view, optimal pricing seems to be a very effective measure to move towards a more efficient utilization of capacities and resources (Verhoef, 2001). Also, it would allow the technological improvements to unfold their full potential (May, 2013). However, only few pricing schemes have been implemented in the real-world, e.g. in Singapore, London, Stockholm (Eliasson et al., 2009), Gothenburg (Börjesson and Kristoffersson, 2015) and Milan (Rotaris et al., 2010).

In real-world politics, the use of so-called 'backcasting' approaches (Geurs and van Wee, 2000, 2004; IWW et al., 1998) is more common than implementing pricing strategies. The idea behind this concept is to set political goals, and implement a number of policy measures in order to reach these goals. For instance, it is used to achieve the 2025 CO_2 reduction targets for the UK (Hickman et al., 2009). With the current trends, chances to achieve these targets were slim and therefore, several policy pathways were identified to help reduce transport-related CO_2. Overall, there is some indication that there exists a price gap between the actual costs of reducing the CO_2 emission in the transport sector and the existing estimates on the social cost of carbon[4]: Liu and Santos (2015) find that even the highest estimates of the

[3]The rebound effects are mainly categorized in direct and indirect rebound effects (IPCC, 2014; Thomas and Azevedo, 2013). The former relates to the increase in demand because of a decrease in travel costs due to an efficient vehicle; e.g., a fuel-efficient car will have lower operating costs which may increase the vehicle kilometer traveled. The latter is the effects from re-spending the savings due to increased efficiency on other goods or services; e.g., spending fuel savings on vacation. The combined effect is called economy-wide rebound effects.

[4]The social cost of carbon (or marginal damage cost of carbon emission) is defined as the net present value of the impact of one additional Ton (ton) of carbon over the next 100 years which is

social cost of carbon from the literature is not able to justify the mass introduction of low/zero emission vehicles/fuel technologies. They can only be justified if the social cost of carbon is revised upwards.

The present study picks up on this observation and aims to compare the price levels obtained from an optimal pricing strategy to those resulting from the backcasting approach. The former typically aims to quantify damage costs (= social costs), whereas the latter only implicitly defines avoidance (= abatement/mitigation) costs, depending on the chosen pathway (see, Watkiss et al., 2005; Link et al., 2014; Maibach et al., 2008, for a detailed discussion on damage and avoidance cost). In the light of the above, the questions arise

a) to what extend pricing schemes, in particular optimal pricing of air pollution externalities, would contribute to the EU 2020 goal, and

b) how (additional) prices would need to be set in order to reach this target. Or, in other words, how different the price levels of a (best-practice) damage cost approach are compared to the backcasting approach.

Thus, in a first step, the present study applies an existing optimal pricing scheme for exhaust emissions to a real-world scenario of the Munich metropolitan area in Germany similar to work by Kickhöfer and Nagel (2016); Agarwal and Kickhöfer (2016). In a second step, the paper attempts to identify the necessary additional prices, as multiples of the original damage cost estimates, in order to reach the EU 2020 CO_2 reduction targets. The remainder of the paper is organized as follows: Section 2 illustrates the methodology and research approach in more detail. The scenario set up for a real-world case study is exhibited in Section 3, and the results are analyzed in Section 4. Limitations of the presented approach and their potential influence on the results are discussed in Section 5. Finally, Section 6 concludes the study and identifies possible directions for future research.

2 Methodology

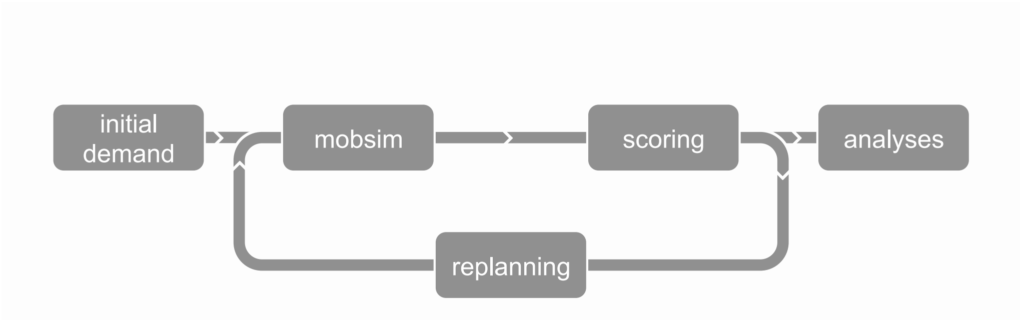
2.1 Simulation platform – MATSim

MATSim[5] is an modular open-source transport simulation framework designed to simulate large-scale scenarios. It is therefore chosen for all simulation runs. Physical boundary condition (network data), initial demand (daily plans of all individual travelers (or agents), see Figure 1) and various configuration parameters are minimal inputs.

emitted to the atmosphere today (Watkiss et al., 2005; Downing et al., 2005).

[5]See www.matsim.org; for detailed information, please refer to Horni et al. (2016b).

Figure 1: MATSim cycle (Horni et al., 2016a)



In an iterative co-evolutionary process, every agent in the simulation learns and adapts to the system. This process is composed of the following three steps:

1. **Mobility simulation**: Daily plans of all individuals are executed simultaneously on the network. The network loading algorithm in the MATSim is so called queue model (Horni et al., 2016c), which can simulate large-scale scenarios in reasonable computation time.[6]

2. **Plans evaluation**: In order to model the choice between multiple potential daily plans, executed plans of all agents are evaluated using a utility function, indicating the performance (or score) of the plan. A plan's utility (S_{plan}) is represented by:

$$S_{plan} = \sum_{q=0}^{N-1} S_{act,q} + \sum_{q=0}^{N-1} S_{trav,mode(q)} \tag{1}$$

where N is the number of activities, $S_{act,q}$ is the utility from performing activity q and $S_{trav,mode(q)}$ is the (typically negative) utility for traveling to activity q. In short, the utility earned from performing an activity is given by[7]

$$S_{act,q} = \beta_{dur} \cdot \ t_{typ,q} \cdot \ln(t_{dur,q}/t_{0,q}) \tag{2}$$

where $t_{dur,q}$ and $t_{typ,q}$ are actual and typical durations of activity q, respectively. β_{dur} is the marginal utility of activity duration. $t_{0,q}$ is the minimal duration, which essentially has no effect as long as dropping activities is not allowed. The simplified mode-specific utility from traveling by car or public transport (PT) following Nagel et al. (2016) is described by:

$$\begin{aligned} S_{car(q)} &= \qquad\qquad \beta_{trav,car(q)} \cdot t_{trav,q} + \beta_m \cdot \gamma_{d,car(q)} \cdot d_{trav,q} \\ S_{PT(q)} &= C_{PT(q)} + \beta_{trav,PT(q)} \cdot t_{trav,q} + \beta_m \cdot \gamma_{d,PT(q)} \cdot d_{trav,q} \end{aligned} \tag{3}$$

[6] In this study, the traditional *"first-in-first-out"* traffic dynamics of the queue model is used (see Agarwal et al., 2015, 2016, for more details and the resulting fundamental diagrams).

[7] See Nagel et al. (2016) for a more detailed description.

where $t_{trav,q}$ and $d_{trav,q}$ is the travel time and distance between activity q and $q+1$. $C_{pt(q)}$ is the alternative specific constant (ASC) of public transport (PT).

3. **Plans re-planning**: After executing and scoring plans, a new plan is generated for a predefined share of agents. The new plan is generated by modifying an existing plan according with respect to predefined choice dimensions. The new plan is then executed in the next iteration.

2.2 Toll calculation and internalization

For the calculation of time-dependent, link- and vehicle-specific exhaust emissions, the paper uses a tool developed by Hülsmann et al. (2011) and further improved and extended by Kickhöfer et al. (2013). It models warm and cold-start emissions; the latter are generated during the warm-up phase of vehicles, whereas the former are generated while driving. The most relevant parameters for their calculation are parking duration, distance traveled, vehicle characteristics, engine type, road category and the speed of vehicles. At the end of every link and for every vehicle, cold-start and warm emissions are calculated based on the traffic states (free flow or stop and go) on the link. The essential parameters such as parking duration, traveled distance, road category, speed of vehicle etc., are derived from the simulation. With these, the HBEFA[8] database provides the resulting exhaust emission values differentiated by type of pollutant.

In order to convert these emissions into vehicle-specific toll values, a marginal social cost (MSC)[9] pricing approach, developed by Kickhöfer and Nagel (2016) is used. It converts the time-dependent, vehicle-specific emissions into a toll by using emission costs factors from the literature (see Table 1). Thus, in the simulation, every time an agent leaves a link, the agent consequently pays the toll equivalent to the emissions produced by her. As a reaction to the toll, agents learn and adapt their behavior within the iterative learning cycle (see Section 2.1). Consequently, agents' decisions are based on MSC, and the external effect is internalized.

The internalization of externalities through optimal pricing can, e.g. in agent-based transport simulations, be used to identify the upper bound of possible efficiency gains in a transport system (see, e.g., Kickhöfer and Nagel, 2016; Kaddoura et al., 2015; Agarwal and Kickhöfer, 2016). However, the calculation of dynamic, vehicle-specific emissions is very complex and time consuming, and especially for environmental externalities, it remains unclear whether the cost factors (see, e.g.,

[8] 'Handbook Emission Factors for Road Transport', Version 3.1, see www.hbefa.net

[9] The marginal social costs are the sum of marginal private costs (MPC) and marginal external costs (MEC) (see, e.g., Walters, 1961; Turvey, 1963). In absence of any pricing, the MATSim utility functions includes only marginal private costs i.e. time and money spent for traveling between planned activities (see Equation 1).

Table 1: Emission cost factors. Source: Maibach et al. (2008).

Emission type	Cost factor (EUR/ton)
CO_2	70
NMHC	1,700
NO_x	9,600
PM	384,500
SO_2	11,000

Table 1) can be determined in a way that they actually represent damage costs. The idea of the present study therefore is to identify the potential price gap between the toll levels obtained from an optimal pricing strategy and the backcasting approach to achieve the EU emissions reduction target. For that purpose, the emission cost factors from Table 1 are increased by a multiplication factor following a parametric approach. In the remainder of this paper, this factor is referred to as "emission cost multiplication factor (ECMF)". The increased emission costs are then charged to the agents who eventually consider them in their decision making.

2.3 Problem simplification

Due to the complex nature of the research problem, the following simplifications are made:

a) GHG emissions from the road transport and from the whole transport sector (excluding international aviation) follow similar trends over the previous decades (European Environment Agency, 2016, online data code: *env_air_gge*). Therefore, it is assumed that a 20% reduction in GHG emissions is required from road transport.

b) In the context of global warming and road transport, the objective of reduction in the GHG emissions is translated to a reduction of CO_2 emissions since CO_2 is a major component among the gases released during the combustion of fossil fuels.

c) The objective is to reduce the EU's GHG emissions, however, the same objective is used for the Munich metropolitan area (MMA) in Germany.

d) The travel demand data is available for the survey year (12/2001-12/2002) and therefore, the proposed approach is applied to this demand (Follmer et al., 2004) rather than forecasting the demand to year 2020.

With the above simplifications, the research problem is reduced to the estimation of avoidance costs to reduce the CO_2 emissions by 20% for MMA with respect to the survey year.

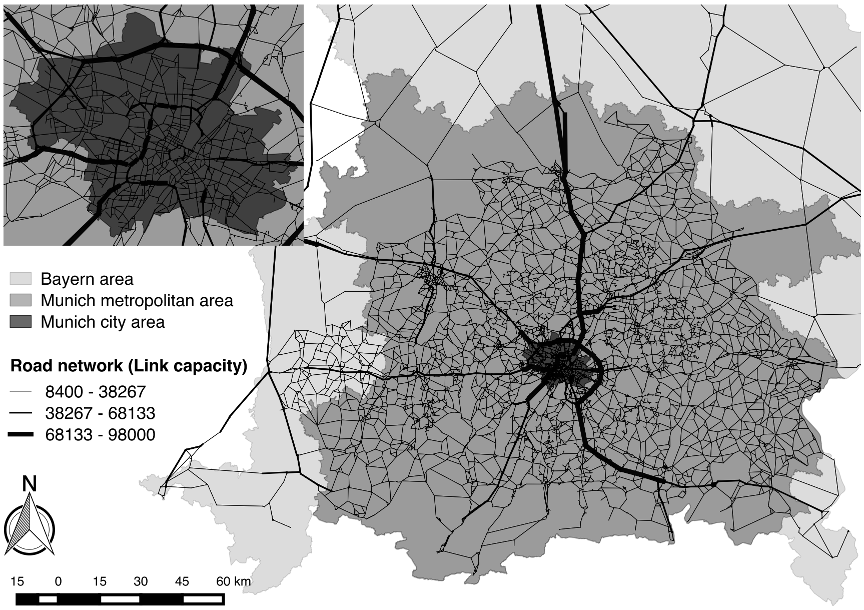


Figure 2: Munich city (inset) and metropolitan area.

3 Real-world scenario: Munich

In this section, the set-up for the scenario of the Munich metropolitan area (MMA) is illustrated shortly. Figure 2 shows the territorial border of Munich city and Munich metropolitan area (MMA). The initial scenario was created by Kickhöfer and Nagel (2016) and further modified by Agarwal and Kickhöfer (2016). In the present study, the latter is used.

Network The network data in the form of VISUM[10] (Municipality of Munich; RSB, 2005) is converted into a MATSim network (see Figure 2).

Demand The demand is based on three different data sources, resulting in four sub-populations: urban, commuters, reverse commuters and freight. A realistic activity-based demand for each of the sub-population is created as shown in Table 2. The table also shows the number of individuals for each sub-population. Urban travelers are confined to Munich city area only whereas Munich metropolitan area is populated by (rev.) commuters and freight trips. For computational reasons, 1% of total population is used for the present study. Network flow and storage capacities are adjusted accordingly. In the simulation, only car mode is simulated on the network, all other modes are assumed to run emission free and without capacity constraints. Therefore, in the present study, all modes other than car are depicted as *non-car* travel modes however an agent can switch mode between car and public transport (PT) as described further in re-planning strategies. Following the study

[10] 'Verkehr In Städten UMlegung', see www.ptv.de

Table 2: User groups in the Munich metropolitan area.

User group	Data source	#Agents [m]	Travel modes
Urban	Follmer et al. (2004)	1.4	car, PT, bike, walk,ride
Commuters	Böhme and Eigenmüller (2006)	0.3	car, PT
Rev. commuters		0.2	
Freight	ITP and BVU (2007)	0.15	car

Table 3: Behavioural parameters.

Parameter	Value	Unit
Source: Kickhöfer (2014)		
Marginal utility of activity duration (β_{dur})	+ 0.96	$utils/h$
Marginal utility of traveling by car ($\beta_{trav,car}$)	– 0.00	$utils/h$
Marginal utility of traveling by PT ($\beta_{trav,PT}$)	– 0.18	$utils/h$
Monetary distance rate by car ($\gamma_{d,car(q)}$)	–0.30	EUR/km
Monetary distance rate by PT ($\gamma_{d,PT(q)}$)	–0.18	EUR/km
Marginal utility of money (β_m)	– 0.0789942	$utils/EUR$
Approximate average $VTTS_{car}$	+ 12.15	EUR/h
Approximate average $VTTS_{PT}$	+ 14.43	EUR/h
Source: Agarwal and Kickhöfer (2016)		
ASC for urban PT	– 0.75	$utils$
ASC for commuters/reverse commuters PT	– 0.3	$utils$

by Agarwal and Kickhöfer (2016), the present study also uses two different PT modes and consequently two ASCs for each PT mode. All behavioral parameters and the approximate average Values of Travel Time Savings (VTTS)[11] are listed in Table 3.

Re-planning strategies Two re-planning strategies are used in order to allow agents to react towards the different pricing schemes: route choice and mode choice. In every iteration, 15% agents switch route, 15% agents switch mode[12], and rest of the agents chose a plan from their existing choice set according to multinomial logit (MNL) model. After 80% of the iteration, agents only chose from their fixed choice set.

[11]The VTTS is defined as the individual willingness-to-pay for reducing the travel time by one hour. For linear utility functions, it is the ratio of the marginal utility of travel time and the marginal utility of money. The former is the sum of the dis-utility for traveling $\beta_{trav,mode(q)}$ and the negative utility of time as a resource $-\beta_{dur}$. Please note that the person-specific VTTS in MATSim can vary significantly with the time pressure which an individual experiences. This is because of the non-linear utility function for performing activities, influencing the actual value of β_{dur} (see Kaddoura and Nagel, 2016, for further details).

[12] According to the Agarwal and Kickhöfer (2016), an urban traveler can switch mode between car and slower PT (speed 25 km/h) whereas, commuters and reverse commuters can switch mode between car and faster PT (speed 50 km/h).

Simulation procedure Figure 3 exhibits the simulation procedure for the different scenarios under consideration. A base case simulation is run for 1000 iterations and its output is then used as input for the different policy cases:

- The base case is continued for 500 more iterations and is referred to as "Business As Usual"(BAU) case. This is the reference case for comparison.
- Six different emission cost multiplication factors (ECMF), namely 1.0, 5.0, 10.0, 15.0, 20.0, and 25.0, are considered and one simulation is set up and for each ECMF by running it for 500 iterations.

In each of the pricing schemes, the ECMF are set to the above mentioned values to increase the highly differentiated tolls for the agents by that factor. The reaction of the agents under various ECMF is analyzed next.

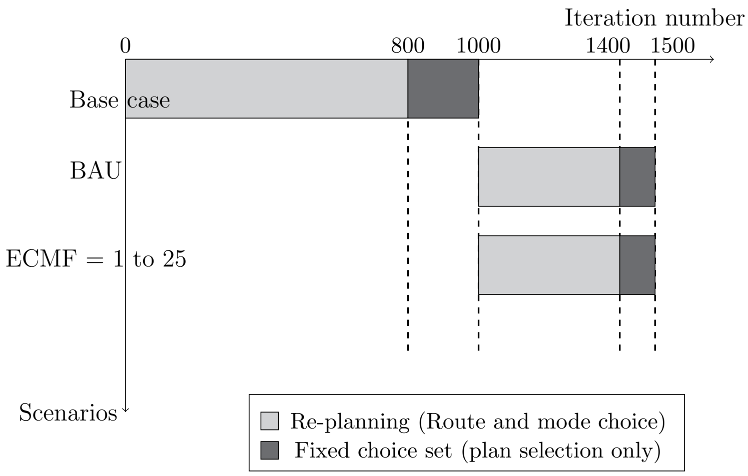


Figure 3: Iteration flow for different scenarios.

4 Results

The presentation of the results is performed from two different angles: (a) based on the geographical area, e.g. city area or metropolitan area, and (b) based on the sub-population (also called user group), namely urban, (rev.) commuters, and freight.

4.1 The amplitude of emissions costs

Table 4 shows that the absolute daily emission costs caused by all sub-populations for the whole area amounts to 3.71 m *EUR*. Though, freight trips represent roughly

[13]Please note that since the MMA already includes values inside the city area, only the values for MMA and "rest" sum up to the total values.

Table 4: Daily emission costs for the BAU scenario. The numbers indicate absolute costs (in $EUR \cdot 10^6$), and relative shares in brackets (in %). All values are scaled to the full population.

Sub-populations for the whole area				
	urban	**(rev.) commuter**	**freight**	**total**
	a	**b**	**c**	**= a+b+c**
Total emissions costs	0.20 (5.47)	0.96 (25.95)	2.55 (68.58)	3.71 (100)
Number of trips [m]	1.27 (62.66)	0.60 (29.52)	0.16 (7.82)	2.04 (100.00)
Total car distance [m km]	7.81 (11.35)	43.31 (62.95)	17.68 (25.7)	68.8 (100)
Area[13]				
	Munich city	**MMA**	**rest**	**total**
	a	**b**	**c**	**= b+c**
Total emissions costs	0.38 (10.24)	1.73 (46.63)	1.98 (53.37)	3.71 (100)
Number of links	4804 (11.45)	35317 (84.21)	6624 (15.79)	41941 (100)
Total car distance [m km]	14.04 (20.35)	45.86 (66.66)	22.94 (33.34)	68.8 (100)

7.82% of all car trips, they contribute to approximately 68.58% of the emission costs because freight vehicles emits more emissions than other vehicles and have longer travel distances (mean and median trip distances are 111 and 69 km, respectively). On the other hand, the share of urban car trips is 62.66% of all car trips, but these contribute to only 5.47% of total emission costs. When looking at the emission costs inside the Munich city area, it appears that only 10.24% of the total costs are accumulated here, but urban travelers are responsible for more than half of these costs (i.e., 0.20 m EUR out of 0.38 m EUR). The emission costs inside MMA (including the emission costs inside Munich city area) is four times higher than those in the Munich city area; the total distance traveled by car/truck inside MMA is three times more than that of the total distance traveled inside the Munich city. Clearly, for the conventional petrol/diesel vehicles, the traveled distances remain the crucial factor of total emission costs.[14] Figure 4 exhibits that almost the entire costs caused by urban travelers are caused inside Munich city, whereas the share of emission costs from freight inside Munich city is rather small. Furthermore, one can observe that most of the emission costs caused by (rev.) commuter is emitted inside the metropolitan area, but outside of Munich city. Freight is responsible for most of the emission costs, causing the major share outside the metropolitan area.

4.2 Changes in emission costs

Before analyzing the changes in CO_2 emissions, the impact of the different pricing schemes on total emission costs is analyzed. As Figure 5 shows, the overall emission costs by tendency decrease with increasing ECMF. This reduction in emission costs

[14]Please note that the traffic related to Munich city is included in this case study i.e., the private and commercial traffic of surrounding urban areas are not included in it.

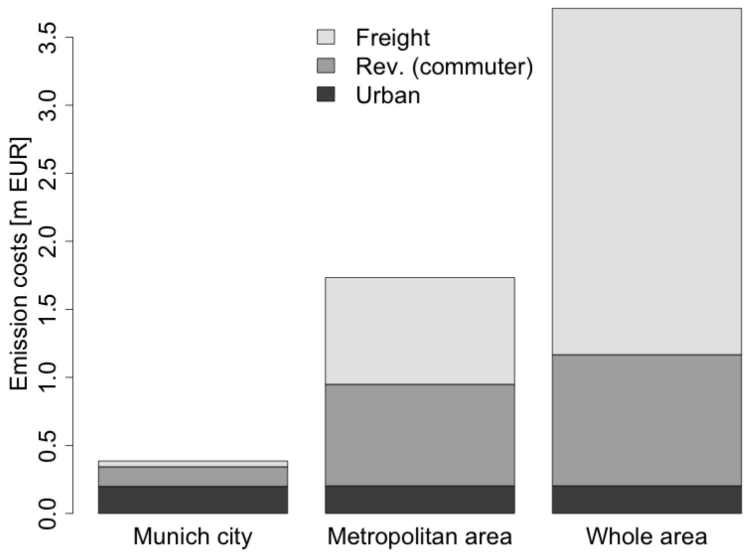


Figure 4: Contribution of sub-populations to emission costs in different regions.

is a combined effect of re-routing and modal shift towards environmentally friendly modes. As shown in Table 5, the modal shift is the driving force behind these savings. (Rev.) commuters are better off by already shifting to PT at low values of ECMFs. In contrast, emission costs caused by urban travelers first decrease marginally (about 0.08%), then increase (about 2%) for ECMF = 5 and then decrease again. The significant decrease in the car share of (rev.) commuters (see Table 5) leads to capacity relief in the network and makes car travel more attractive again. As a consequence, the car share for urban travelers increases and ultimately results in higher emission costs at ECMF = 5. With higher factors, the tolls for urban travelers become so high that even after further relief in the capacities, urban travelers are better off by changing to PT. This, in consequence, diminishes the emission costs of urban travelers. For freight transport, where only route choice is allowed, the decrease in emission costs is – as expected – by far smaller than for the other sub-populations. In Munich city, however, freight-related costs strongly decrease compared to the other areas at higher levels of ECMF. This effect will be discussed more when looking at different pollutant types in the next section. For now, let it suffice to say that it has to do with a relief in network capacities, and the associated shift from stop and go to free flow traffic conditions and by tendency to shorter routes (in Munich city only).

Overall, for the whole area and all sub-populations, ECMF and caused emission costs are inversely proportional to each other, i.e. an increase in the ECMF yields a decrease in emission costs. However, this effect stagnates at higher values of the ECMF (> 10). The goal of a 20% reduction of emission costs (local pollutants *and* CO_2) can be achieved for the whole area with a cost factor of 10; a cost factor of 5

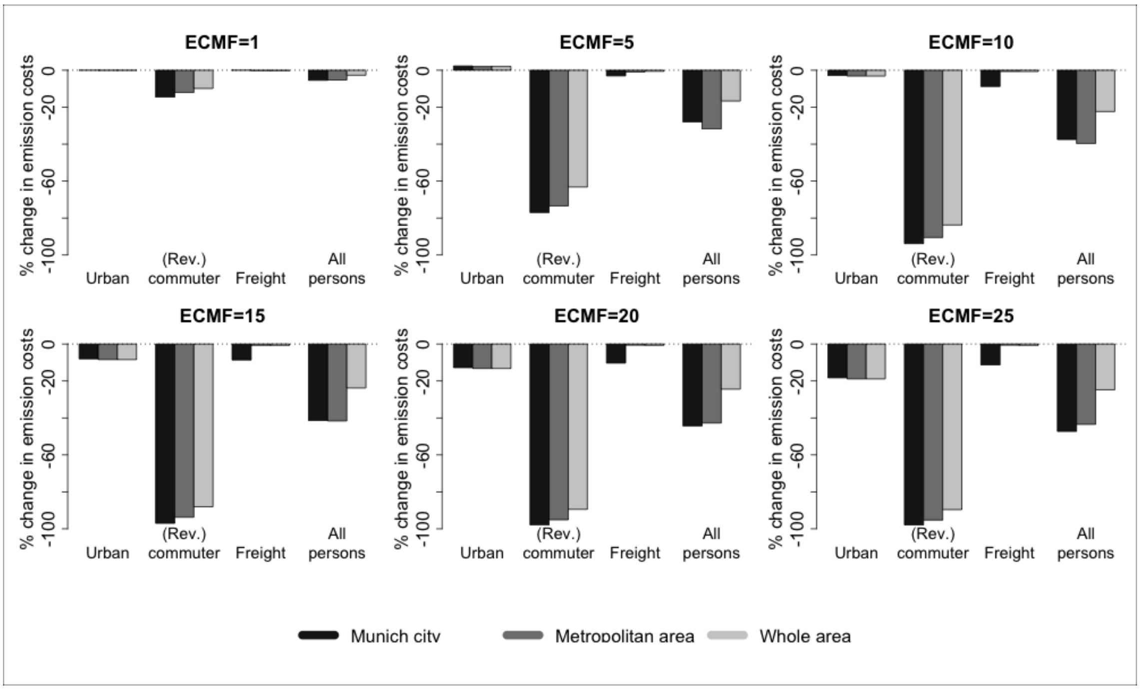


Figure 5: Relative change in emission costs by sub-population and area.

Table 5: Change in car trips (in percentage points) with respect to BAU for various ECMF.

User group	BAU	Emissions cost multiplication factor					
		1	5	10	15	20	25
Urban	22.98	0.22	1.39	1.14	0.66	0.20	-0.41
(Rev.) commuter	65.57	-7.04	-44.96	-59.57	-62.71	-63.61	-63.67
Freight	100.00	No change					
Total	30.72	-0.79	-5.06	-7.29	-8.12	-8.63	-9.15

is needed to achieve this target inside Munich city and the metropolitan area.

4.3 Changes in pollutant types

Following the overall interpretation from above, the effects on different types of pollutants is presented next. Section 4.3.1 exhibits the changes in CO_2 emissions for the sub-populations in different areas; Section 4.3.2 summarizes the effect of ECMFs on Non-Methane Hydrocarbons (NMHC), which is associated with building the greenhouse gas Ozone.

4.3.1 Changes in CO_2

Figure 6 shows the relative change in CO_2 levels for various ECMF. The overall trend is similar to the change in emission costs (see Figure 5), i.e., for (rev.) commuter, CO_2 decreases significantly with an increase in the ECMF and then becomes stationary after ECMF = 15. For freight, the decrease in CO_2 levels is very small except in

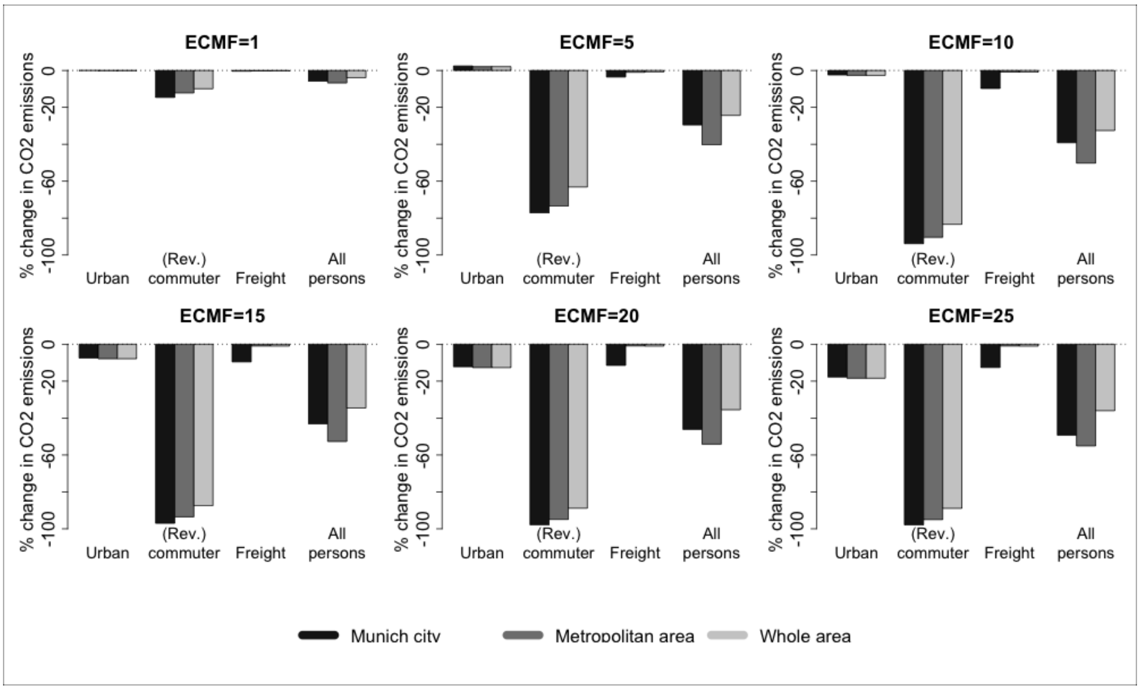


Figure 6: Effect of ECMF on CO_2 emissions by sub-population and area.

city area because, freight reroutes and avoid links inside city area or shift to shorter distance routes. In contrast, for urban travelers, CO_2 level remains almost same at ECMF = 1, increases at ECMF = 5 and afterwards, decreases with an increase in the ECMF. The increase at ECMF = 5 is due to the capacity relief effect (see Section 4.2).

Interestingly, the EU emission reduction target (20% reduction in CO_2 emissions) can be achieved at ECMF = 5 (light grey bar on the right). Recall that for a 20% reduction in the total emission costs, an ECMF = 10 or higher was needed for whole area and ECMF = 5 or higher for Munich city and the metropolitan area, respectively. Thus, a toll five times higher than using the damage cost estimates results in 20% lower CO_2 levels. Consequently, the implicit avoidance costs of CO_2 with this very measure amount to 350 EUR/ton (see Table 1 for the initial cost factors).

4.3.2 Changes in NMHC

The emission level of NMHC emissions mainly depends on the fuel type, engine type, age of the vehicle and vehicle speed (Haszpra and Szilágyi, 1994). NMHC emissions are higher for the cold-starts than for a warmed up vehicle (Schmitz et al., 2000; Hoekman, 1992).

For the BAU scenario, urban travelers contribute to about 39% of total NMHC emissions because they (a) travel relatively shorter distance (average distance = 6.11 km), and (b) perform multiple trips in a day, whereas (rev.) commuters and freight only perform 2 and 1 trip(s) per day, respectively. All pollutants except NMHC show similar trends as CO_2; the changes in NMHC emissions for urban and freight user groups show an exceptional trend, which is presented next.

Figure 7a shows the effect of the different pricing schemes on NMHC levels for urban travelers and freight, aggregated for whole scenario. The following points can be observed:

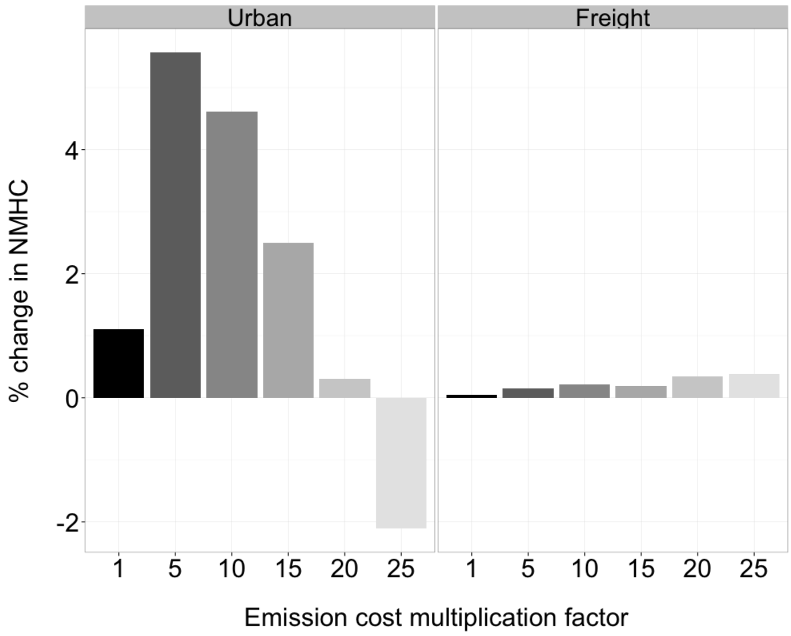
1. **Urban:** As discussed above, pricing emission increases the number of urban car trips (see Table 5) and decreases their average car distance (see Figure 7b). That is, some of the PT users with short trip distance are better off by shifting to car mode. This eventually results in higher NMHC emissions for urban travelers. On the contrary, at ECMF = 25, even after the decrease in average trip distance, the NMHC costs is reduced by more than 2% due to a significant drop in car share.

2. **Freight:** The freight sub-population is different than all other sub-populations. The average trip distances decrease with increasing ECMF, but NMHC emissions increase. The average trip distance of freight trips is very high (average distance = 111 km), therefore, it is less likely that the small change in average trip distance will impact the NMHC emissions significantly. Furthermore, the freight vehicle fleet, fuel type, age of the vehicle do not vary in the scenario. Thus, the reason for the increase in NMHC results from freight trips shifting from motorways to local roads where the engine of trucks works in a different environment.

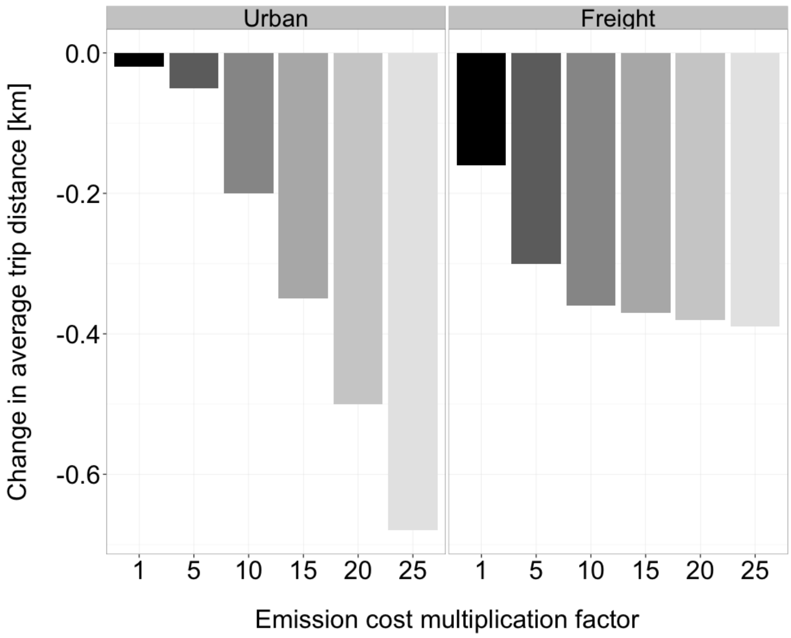
Overall, the analyses show that the CO_2 reduction target may be achieved at ECMF = 5. However, this may also lead to some adverse effects due to the changes in the local pollutants such as NMHC, which eventually counteracts the CO_2 reduction by increasing other greenhouse gas levels through the building of Ozone.

5 Discussion

The ongoing efforts to cut global GHG emissions face various road blocks, such as the growing divergence between vehicle emissions under laboratory and real-world conditions, continuous economic (and thus transport) growth in urban agglomerations, or rebound effects counteracting technological improvements. Research on sustainable transportation and several real-world examples have shown that pricing schemes can help to reduce transport-related externalities such as GHG emissions. However, in order not to harm the economy, estimates for damage cost are required,



(a) Relative change in NMHC levels.



(b) Absolute change in average trip distance.

Figure 7: Change in NMHC levels and average trip distance for urban and freight sub-populations with respect to BAU. Values are aggregated for scenario.

which need to be in the same order of magnitude as the external effect. Since the uncertainty range for these costs is very high, the exact determination of external environmental and health costs is close to impossible. Additionally, the cost factors for local pollutants vary highly depending on the number of affected individuals or buildings eventually leading to very complex pricing schemes (see, e.g., Kickhöfer and Kern, 2015). In such situations it can be useful to define goals on the political level, which implies setting the avoidance costs depending on the measures implemented to reach the goals. In this paper, a parametric backcasting approach is applied to a real-world case study to determine the avoidance costs to achieve a 20% reduction in CO_2 emissions from motorized individual transport. In the following, the assumptions and simplifications for this case study are discussed in terms of the influence on the overall results.

Share of road transport The GHG emissions from road transport follow a similar trend as the overall transport sector over previous decades. However, the share from road transport is continuously increasing. If this trend continues, the required avoidance cost would be higher than the values estimated in this chapter.

Base and projected year demand The urban travel demand is synthesized using detailed survey data from 12/2001 to 12/2002 (MiD 2002; Follmer et al., 2004). Because of the absence of detailed demand and vehicle fleet data from the reference year 1990, the emission reduction target for CO_2 of 20% it applied to the survey year. The following qualitative statements can be made at this point:

a) CO_2 emissions from road transport in the years 2001-2002 (survey year) are approximately 20% higher than the 1990 level; that is, in order to reduce the CO_2 emissions by 20% from 1990 levels, the real objective would be to cut emissions approximately by 33% from the survey year. Consequently, the avoidance charge would be higher than the estimated values in this paper.

b) The share of GHG emissions from road transport is continuously rising. On the other hand, until 2020, developments like advances in the vehicle and fuel technology might push into the opposite direction, depending on the market penetration of these technologies. Assuming that an aggressive intake of the vehicle and fuel technologies would compensate the increasing emissions from future growth and behavioral rebound effects, then cost estimates from this chapter are required to achieve the EU emission reduction target.

Estimated price for CO_2 The base damage cost estimate for CO_2 used in this study is 70 EUR/ton (see Table 1) with a lower and a upper bound of 15 EUR/ton

and 280 *EUR/ton*, respectively (Krewitt and Schlomann, 2006; Maibach et al., 2008). This base value is on the upper end compared to estimates from other studies (Maibach et al., 2008, pp. 262-263; Tol, 2005). The proposed backcasting approach finds that the base estimate for CO_2 needs to be increased by a factor of 5 (i.e. 350 *EUR/ton*), which is even higher than the highest estimates from the literature. Clearly, it might be that emission reduction in other sectors can be achieved at lower avoidance costs.

6 Conclusion and outlook

This paper determined the price gap between toll levels derived from optimal emission pricing and toll levels implicitly resulting the EU 2020 CO_2 reduction target. First, an existing optimal emission pricing approach was applied to a real-world scenario and changes in emissions and cost levels were evaluated. Second, in order to obtain the possible avoidance toll levels, different emission cost multiplication factors (ECMF) were used to modify the initial toll levels. The results of these scenarios were compared to the base scenario.

It was shown that ECMF = 10 is required to reduce total emission costs by 20%, whereas ECMF = 5 is enough to obtain a 20% reduction in CO_2 levels. That is, damage costs estimates from the literature have to be multiplied by a factor of 5 to achieve the EU 2020 CO_2 reduction target. Hence, this paper estimates the (avoidance) cost of CO_2 to 350 *EUR/ton*, which is significantly higher than available estimates from the literature where the damage cost approach is typically used.

The highest contribution to the emission reduction came from (rev.) commuters and their modal shift from car to PT mode. Urban travelers, however, shifted to car because of the resulting relief in road capacities. Only at very high toll levels, the car share of urban travelers decreased. For freight traffic, significant improvements in the emission levels were observed in the city area above ECMF = 5. Furthermore, the investigation of emission levels indicated that because of the increase in the number of short urban car trips, Non-Methane Hydrocarbon (NMHC) levels by tendency increased. Similarly, for freight, an increase in NMHC was observed because of route shifts from motorways to local and distributor roads. NMHC can contribute to increased Ozone levels, which again is a greenhouse gas.

Overall, this paper provided valuable insights about differences in price levels, potential different outcomes for the various types of pollutants and groups of travelers. Pricing schemes for emissions might not necessarily result in a reduction of all pollutants or of the emission levels of all users. It shows how agent-based simulations can be used for quantifying the results and for decision support in such possibly counter-intuitive situations. In future research, pricing other externalities of transport (e.g. congestion, noise, accidents) should be included. Additionally, the analysis is planned to be carried out for a greater region (e.g. Germany, EU) in order to test the variability of results.

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