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# Estimation of economic costs of air pollution from road vehicle transportation in Turkey

# Türkiye'de karayolu taşımacılığından kaynaklanan hava kirliliğinin ekonomik maliyetinin tahmini

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**Abstract**

Valuation of the economic, social and environmental impacts of air pollution has become crucial for the benefit-cost analysis of pollution restriction strategies, which serve as a foundation for establishing priorities for action. This paper focuses on the estimation of total external costs caused by road transport-related air pollutants using an integrated evaluation methodology combining air quality modelling, engineering science and economics. Emission factors and transport network characteristics were used to compute emissions from the road transport that is followed by economic valuation approaches adopted from international case studies and used for calculating the economic costs of air pollution in Türkiye. The results showed that total external costs of air pollution in Türkiye in 2018 ranged between 37,500 euros which is computed for CO emissions and 2,686 million euros computed as an upper limit for NOx emissions. Regarding the social costs of $CO_2$ emissions, the values range between 31 million euro and 1,427 million euro, the former represents the low value estimate while the latter is the high value estimate. The findings indicate that the impact of emissions from road transport on environment and society can be substantial in Türkiye. Therefore, some regulations are necessary to reduce transport emissions and to sustain socio-economic welfare.

**Keywords**: Road transport, Air pollutants, Economic valuation, External costs, Türkiye

**Özet**

Hava kirliliğinin sosyo-ekonomik ve çevresel etkilerinin değerlendirilmesi, eylem önceliklerini belirlemek için bir temel oluşturan kirlilik kontrol stratejilerinin maliyet-fayda analizi için çok önemlidir. Bu makale, hava kalitesi modelleme, mühendislik ve ekonomiyi birleştiren entegre bir değerlendirme metodolojisi kullanarak karayolu taşımacılığıyla ilgili hava kirleticilerinin neden olduğu toplam dışsal maliyetlerin tahminine odaklanmaktadır. Karayolu taşımacılığından kaynaklanan emisyonların hesaplanmasında emisyon faktörleri ve ulaşım ağı özellikleri kullanılmış olup uluslararası örnek çalışmalardan uyarlanan ekonomik değerleme yaklaşımları takip edilerek Türkiye'deki hava kirliliğinin ekonomik maliyetinin hesaplanmasında kullanılmıştır. Sonuçlar, 2018 yılında Türkiye'de hava kirliliğinin toplam dışsal maliyetinin CO emisyonları için hesaplanan 37,500 avro ile $NO_x$ emisyonları için üst sınır olarak hesaplanan 2,686 milyon avro arasında değiştiğini gösterdi. $CO_2$ emisyonlarının sosyal maliyetleri ile ilgili olarak, değerler 31 milyon avro ile 1,427 milyon avro arasında değişmektedir. Bunlardan ilki düşük değerli tahmini, ikincisi ise yüksek değerli tahmini temsil etmektedir. Bulgular karayolu taşımacılığından kaynaklanan emisyonların çevre ve toplum üzerindeki etkisinin Türkiye'de önemli olabileceğini göstermektedir. Bu nedenle, ulaşım emisyonlarını azaltmak ve sosyo-ekonomik refahı sürdürmek için bazı düzenlemeler gereklidir.

**Anahtar kelimeler**: Karayolu taşımacılığı, Hava kirleticiler, Ekonomik değerleme, Dışsal maliyetler, Türkiye

## 1 Introduction

Emission reductions from air pollutants and greenhouse gases plays a significant part in supporting sustainable development. Road transport is one of the main drivers of increasing levels of greenhouse gases and air pollution [1, 2]. Climate change is known to be worsened by greenhouse gases such as carbon dioxide ($CO_2$), nitrous oxide ($NO_2$), and methane ($CH_4$). Local air pollutants including carbon monoxide (CO), volatile organic compounds (VOCs), particulate matter (e.g. Pb), nitrogen oxide ($NO_x$), and sulphur oxide ($SO_x$) have negative impacts on the environment and society [3]. Road traffic exposure raises all-cause mortality, asthma, deteriorated lung function, unfavourable birth outcomes and paediatric cancer [4]. Damage to materials and building structures, crop losses and additional costs for harming the ecosystem and biodiversity are some of the costs associated with air pollution. The considerable rise in private automobile ownership, which is partly fuelled by the construction of large-scale metropolitan highways, has resulted in increased transport-related pollution, and energy use. Nearly the whole global population participates in transportation activities, with road transport making up the biggest portion of such activities.

Continuous economic and population growth and high levels of energy consumption that rely on a carbon-intensive fuel mix, together with a road-dominated transportation

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system have caused large increases in greenhouse gases (GHG) and local air pollutant emissions in Türkiye. GHG emissions have followed economic growth path in Türkiye and have been decoupled particularly in recent years [5]. According to Organisation for Economic Co-operation and Development (OECD) [5: p:25], Türkiye experienced the largest increase in GHG emissions among OECD members and transportation industry is one of the major sources of GHG emissions. Türkiye's cities have been identified as one of the worst examples of excessive levels of air pollution found in Europe [5]. In Türkiye, roads are estimated to account for 25% of future infrastructure investment, with railroads accounting for 9%. This will lead to increasing congestion and air pollution levels, particularly in the biggest urban areas and regions of Türkiye. Regarding transportation sector, government needs to encourage a modal shift from private roads to public transportation through application of integrated urban and transportation planning, promotion of alternative fuels and renewal of the existing vehicle fleet structure.

Different methodological techniques and software can be used to evaluate the air pollution caused by vehicle emissions and forecast the air quality in short and long-term scenarios. Basically, there are two types of models to evaluate the transport-related emissions: The bottom-up and the top-down models. The former approach is based on the models that provide data on motor vehicle pollutants at the street scale. In such models, first, road transport characteristics (e.g. traffic volumes, vehicle speeds) including transport mode and types of vehicle involved are specified. Based on this information, total emissions causing air pollution are calculated for each transport mode (e.g. car, bus, rail) and vehicle type (e.g. share of diesel vehicles, age and technology of vehicles). Dispersion and transformation models can be used for the estimation of air pollution concentrations and the population exposed to a certain concentration [6]. The bottom-up method has been used in different Europe-wide studies such as UNITE [7]; NEEDS [8, 9]; HEATCO [10]; CAFE CBA [11]; ExternE [12].

More recently, air pollution monitoring systems based on remote sensing data have been utilised for the assessment of transport-related air pollution [13]. Input data requirements of these models are road traffic characteristics, vehicle emissions, whether data, terrain geometry and location of receptors. Air pollution assessment is carried out based on three different spatial scales: Local (only covering the pollutants including $CO_x$, $NO_x$, $SO_x$) [14, 15], urban (mainly primary pollutants) [16], and regional (primary and secondary pollutants such as $O_3$, $NO_2$, and others) [17]. An alternate way for estimating the socioeconomic and environmental effects of exposure to air pollution and for valuing with specific costs in the case of death and morbidity is the top-down approach. This strategy demands the availability of comprehensive exposure data for each country for the specific air pollutants (such PM2.5 or PM10). It is also necessary to have information on how each mode of transportation and vehicle category contributes to total pollution concentrations for each pollutant. This approach was used in the previous studies including INFRAS/IWW [18, 19] and World Health Organisation (WHO) [20]. The reviews of different methods used for modelling air pollution impacts are provided in Carruthers et al. [21], Chang and Hanna [22], Jerrett et al. [23], WHO [24], Sellier et al. [25], Conti et al. [26].

Despite the continuous growth in GHG emissions, Türkiye did not set a mitigation target in 2020 [5]. Nevertheless, the Country puts forward a mitigation target for the year 2030 as part of its Intended Nationally Determined Contribution (INDC) under the UN Framework Convention on Climate Change. Türkiye's National Climate Change Strategy and Action Plan, which established a framework for action with short- and long-term goals and a list of initiatives for reducing GHG emissions and responding to climate change, has been put in place [5]. Low emission zones were included to the 2008 transportation legislation, where they will be governed locally. The government has created a plan to increase domestic demand by making infrastructural expenditures and providing incentives for the adoption of low-emission vehicles [5]. However, Türkiye lacks a thorough framework for evaluating the energy savings from diverse industries (such as transportation) brought about by various policies or infrastructure upgrades. The government has not created strategies or measures for evaluating progress, and there is no assessment of the discrepancy between policy aims and actions taken. [27]. Therefore, the present study will contribute to the literature by focusing on the valuation of external effects of air pollutants caused by road transportation using an integrated assessment approach.

## 2 Material and methods

The overall transport cost evaluation methodology from CE Delft, INFRAS & Fraunhofer ISI [28] is given in Figure 1. As shown in Figure 1, each vehicle category's emission parameters and travel characteristics must be provided as input data. Calculating the transport-related pollutants brought on by additional vehicle miles is the first step utilizing this data. Then, it is computed how much of each type of pollutant is being released by vehicles. The final stage is to place a monetary value on various air pollution sources and distribute the overall external costs across the various vehicle categories. In Türkiye, road-based transportation account for around 40% of the national $NO_x$ and %13 of the NMVOC emissions [29]. As highlighted by EEA [30, 31], other potential sources of local air pollution are CO, $N_2O$, $NH_3$, Pb, $PM_{10}$ and $PM_{2.5}$. Therefore, these pollutants will be used to calculate the costs of air pollutants from road transport in Türkiye. Following the calculation of emissions of all air pollutants per vehicle category, this information is integrated with external air pollution costs for each of the air pollutants to calculate the average costs per vehicle-km by each transportation mode. The sub-sections below present the emission estimation methodology and the methods for evaluating the costs of emissions.

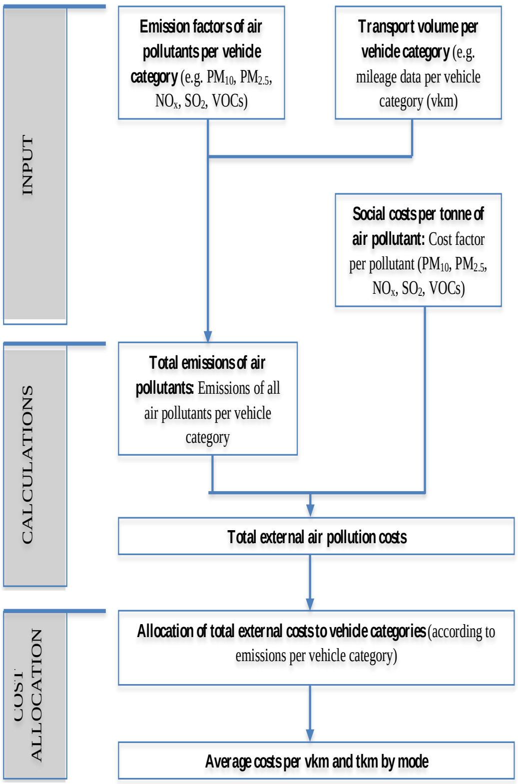


**Figure 1.** Methodology for air pollution cost evaluation

### *2.1 The emission estimation methodology*

Activity data refers to statistics about the numbers, volumes, and amounts of a particular activity, such as the amount of fuel used and the number of kilometers travelled annually by a particular vehicle category. Many different potential data sources that might be used to compile an inventory. The governmental departments, academic institutions, and organizations that already gather data from many economic sectors are national data sources, and they are also where users may find the most relevant information. Because there is no published work on the emission factors computed for the road transport sector in Türkiye, the emission factors computed by EEA [31] were adopted to the Turkish case. The emission estimation methodology of EEA [31] includes the emissions of $NO_x$, CO, $CO_2$, NMVOC, $N_2O$, $CH_4$, $NH_3$, PM and others. Emission factors for NMVOC, CO, $NO_x$, PM, $N_2O$, PM, $NH_3$, and $CO_2$ were calculated using the Tier 1 method that estimates the emissions in [g/kg]. In the Tier 1 approach, the emissions are calculated based on the following equation:

$$E_i = \sum_j \left( \sum_m FC_{j,m} \times EF_{i,j,m} \right) \quad (1)$$

Where $E_i$ is the emission from pollutant type i [g], $EF_{i,j,m}$ represents the consumption of fuel-specific emission factor of pollutant i for vehicle type j and fuel type m [g/kg]; and $FC_{j,m}$ is the fuel consumption of vehicle category j using fuel type m [kg]. Automobiles, heavy-duty vehicles, L-category vehicles, and light commercial vehicles should all be taken into account. The fuels to be considered are diesel, petrol, natural gas and LPG. The Tier 2 method, on the other hand, is based on the fuel used by different vehicle types and their emission standards. As a result, the number of cars and the annual mileage for each technology must be provided. These vehicle-kilometres data are multiplied by the Tier 2 emission factors given by the equation below.

**Table 1.** Emission factors for different vehicle classes (in g/km)

| Category | Vehicle class | CO (g/km) | NMVOC (g/km) | $NO_x$ (g/km) | $N_2O$ (g/km) | $NH_3$ (g/km) | Pb (g/km) | $CO_2$ lube (g/km) | $PM_{2.5}$ (g/km) |
|---|---|---|---|---|---|---|---|---|---|
| Petrol PC[1] | PRE-ECE | 37.300 | 2.780 | 2.780 | 0.010 | 0.002 | 0.000 | 0.663 | 0.002 |
| | EURO3 | 1.797 | 0.099 | 0.093 | 0.002 | 0.034 | 0.000 | 0.464 | 0.001 |
| | EURO4 | 0.628 | 0.052 | 0.058 | 0.002 | 0.034 | 0.000 | 0.398 | 0.001 |
| | EURO5/6 | 0.628 | 0.052 | 0.058 | 0.001 | 0.012 | 0.000 | 0.398 | 0.001 |
| Diesel PC | EURO3 | 0.089 | 0.029 | 0.772 | 0.009 | 0.001 | 0.000 | 0.464 | 0.039 |
| | EURO4 | 0.092 | 0.014 | 0.580 | 0.010 | 0.001 | 0.000 | 0.398 | 0.031 |
| | EURO5 | 0.043 | 0.009 | 0.550 | 0.004 | 0.002 | 0.000 | 0.398 | 0.002 |
| | EURO6 | 0.046 | 0.009 | 0.323 | 0.004 | 0.002 | 0.000 | 0.398 | 0.002 |
| Petrol LCV[2] | Conventional | 25.500 | 3.440 | 3.090 | 0.010 | 0.003 | 0.000 | 0.663 | 0.002 |
| | EURO3 | 5.050 | 0.189 | 0.129 | 0.028 | 0.030 | 0.000 | 0.464 | 0.001 |
| | EURO4 | 2.010 | 0.128 | 0.064 | 0.013 | 0.030 | 0.000 | 0.398 | 0.001 |
| | EURO5/6 | 1.300 | 0.096 | 0.064 | 0.001 | 0.012 | 0.000 | 0.398 | 0.001 |
| Diesel LCV | Conventional | 1.340 | 0.133 | 1.660 | 0.000 | 0.001 | 0.000 | 0.663 | 0.356 |
| | EURO3 | 0.473 | 0.094 | 1.030 | 0.009 | 0.001 | 0.000 | 0.464 | 0.078 |
| | EURO4 | 0.375 | 0.035 | 0.831 | 0.009 | 0.001 | 0.000 | 0.398 | 0.041 |
| | EURO5/6 | 0.075 | 0.035 | 0.496 | 0.004 | 0.002 | 0.000 | 0.398 | 0.001 |
| Petrol HDV[3] | Conventional | 59.500 | 5.250 | 6.600 | 0.006 | 0.002 | 0.000 | 1.990 | 0.000 |
| Diesel HDV | Conventional | 2.040 | 0.717 | 9.280 | 0.029 | 0.003 | 0.000 | 0.486 | 0.394 |
| | EURO3 | 1.209 | 0.223 | 5.158 | 0.005 | 0.003 | 0.000 | 0.486 | 0.106 |
| | EURO4 | 0.086 | 0.009 | 3.183 | 0.012 | 0.003 | 0.000 | 0.486 | 0.019 |
| | EURO5/6 | 0.086 | 0.009 | 1.082 | 0.034 | 0.010 | 0.000 | 0.486 | 0.010 |
| Busses | Conventional | 5.710 | 1.990 | 16.500 | 0.029 | 0.003 | 0.000 | 2.650 | 0.909 |
| | EURO3 | 2.670 | 0.409 | 9.380 | 0.001 | 0.003 | 0.000 | 0.861 | 0.207 |
| | EURO4 | 0.223 | 0.022 | 5.420 | 0.012 | 0.003 | 0.000 | 0.265 | 0.046 |
| | EURO5/6 | 0.223 | 0.021 | 1.800 | 0.036 | 0.010 | 0.000 | 0.265 | 0.024 |
| Motorcycles | EURO2 | 7.170 | 0.900 | 0.317 | 0.002 | 0.002 | 0.000 | 0.221 | 0.014 |

***Source:*** EEA [5] [1]Personal Cars [2]Light Commercial Vehicles [3]Heavy Duty Vehicles

$$E_{i,j} = \sum_k (N_{j,k} \times M_{j,k} \times EF_{i,j,k}) \tag{2}$$

Where $E_i$ is as defined previously, $N_{j,k}$ is the total number of vehicles in country's vehicle fleet of category j and technology k; $M_{j,k}$ is the annual average distance driven per vehicle of category j and technology k [km/veh]; $EF_{i,j,k}$ is the technology-based emission factor of pollutant i for vehicle type j and technology k [g/veh-km]. This was estimated for the vehicle categories j including automobiles, heavy-duty vehicles, light commercial vehicles, and L-category vehicles. For each vehicle technology, the Tier 2 emission factors are provided in grammes per vehicle-kilometer. Table 1 contains technology and fuel-specific emission factors for NMVOC, CO, $N_2O$, $NO_x$, Pb, $NH_3$, $CO_2$ lube and $PM_{2.5}$. It is clear from Table 1 that newer vehicle types use less gasoline and emit fewer pollutants than older vehicle classes (such as PRE-ECE and EURO3).

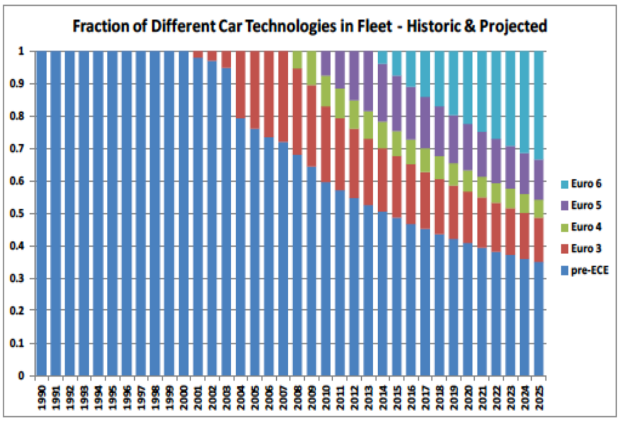


**Figure 2.** Road transport vehicle fleet characteristics in Türkiye, 1990-2025

The structure of the vehicle fleet is another factor needed to determine emissions per vehicle category. This is provided in Table 2 representing different classes for each vehicle category. More than 40% of cars are PRE-ECE, 17% are EURO3 and EURO6, and the remaining vehicles are EURO4 and EURO5, according to Table 2. (Figure 2).

Türkiye's percentage of diesel-powered vehicles has significantly expanded over the previous few decades, rising from 6.8% to 37% between 2005 and 2018. [5]. Diesel's contribution to road transportation's overall energy consumption rose from 56% in 2005 to 69% in 2015 [5]. The number of vehicles listed in Table 3 was multiplied by an average of 3,817 km for motorcycles, 13,107 km for automobiles, 28,172 km for light commercial vehicles, 50,114 km for heavy duty vehicles, and 50,141 km for buses to determine the total average vehicle kilometers for the year 2018 (see Turkstat vehicle statistics). Because the vehicle fleet structure data including diesel PC, petrol PC, diesel LCV, petrol LCV, diesel HDV, petrol HDV, motorcycle and bus is from the year 2018, total average vehicle kilometers were calculated for the year 2018 (Table 3). The data that was used to adopt emission factors is from EEA [5] and data on vehicle fleet structure and average vehicle kilometers is from EU-RTR Project [29] and TurkStat [32]. The latter data show the annual average vehicle kilometers on the road network for the year 2018. And the vehicle fleet structure represents the total number of different vehicle types given for the year 2018. 2018 data was used in the study as the vehicle fleet structure was provided with the given detail for the subject year comprising all petrol and diesel PC, LCV, HDV, bus and motorcycle (Table 2). Post-2018 data on vehicle statistics exist but does not classify the vehicle fleet structure as given in Table 2.

**Table 2.** Vehicle fleet structure in Türkiye, 2018

| Class | Petrol PC | Diesel PC | Petrol LCV | Diesel LCV | Petrol HDV | Diesel HDV | Bus | Motorcycle |
|---|---|---|---|---|---|---|---|---|
| PRE-ECE | 1487411 | 0 | 58489 | 106286 | 1283691 | 1003073 | 93965 | - |
| EURO3 | 588046 | 1744332 | 23123 | 42020 | 0 | 396564 | 37149 | - |
| EURO4 | 311319 | 1241462 | 12242 | 22246 | 0 | 209946 | 19667 | - |
| EURO5 | 484273 | 1555756 | 19043 | 34605 | 0 | 326582 | 30593 | - |
| EURO6 | 588046 | 1744332 | 23123 | 42020 | 0 | 396564 | 37149 | - |
| Total | 3459095 | 6285882 | 136020 | 247176 | 1283691 | 2332728 | 218523 | 3211328 |

**Source:** EU-RTR Project [29]; TurkStat [32]
**Note:** The EU-RTR Project (2012) is the data source for the classifications of Petrol PC and Diesel PC. Accordingly, the percentage distribution of PC classes are: PRE-ECE: 43%; EURO3: 17%; EURO4: 9%; EURO5: 14%; EURO6: 17%. The same percentage distribution was applied LCVs, HDVs and busses in order to redistribute the total vehicle number according to vehicle classes.

**Table 3.** Total average vehicle kilometres (thousand km) on the road network, Türkiye, 2018

| | Petrol PC | Diesel PC | Petrol LCV | Diesel LCV | Petrol HDV | Diesel HDV | Bus | Motorcycle |
|---|---|---|---|---|---|---|---|---|
| PRE-ECE | 19,495,494 | 0 | 1,647,741 | 2,994,282 | 64,330,877 | 50,268,009 | 4,711,494 | - |
| EURO3 | 7,707,521 | 22,862,964 | 651,433 | 1,183,786 | 0 | 19,873,399 | 1,862,683 | - |
| EURO4 | 4,080,452 | 16,271,839 | 344,876 | 626,710 | 0 | 10,521,211 | 986,127 | - |
| EURO5 | 6,347,370 | 20,391,292 | 536,474 | 974,883 | 0 | 16,366,328 | 1,533,975 | - |
| EURO6 | 7,707,521 | 22,862,964 | 651,433 | 1,183,786 | 0 | 19,873,399 | 1,862,683 | - |
| Total | 45,338,358 | 82,389,060 | 3,831,956 | 6,963,448 | 64,330,877 | 116,902,346 | 10,956,962 | 12,257,639 |

*2.2 Economic valuation methods*

In economic analysis, the health impacts born by the transport sector are considered as transport externalities. With the aid of Figure 3, the basics of road vehicle externalities can be explained.

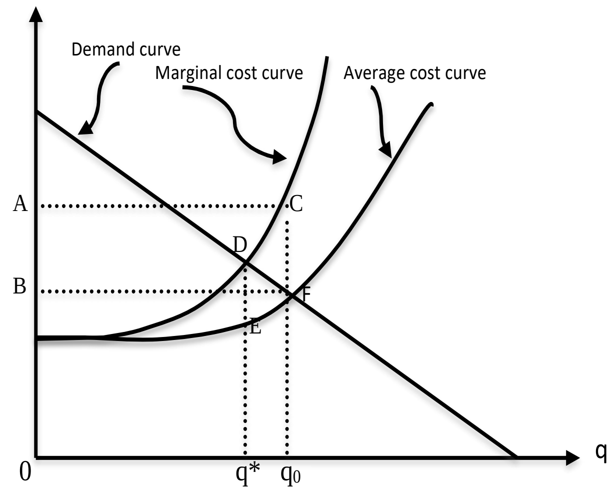


**Figure 3.** External costs of road transport

The figure shows the quantity of road usage on one axis and the unit cost of using the roads on the other. The unit cost of utilizing the road is a function of the demand curve D(q), which depicts the demand for its use. The average cost curve provides the costs that consumers incur when utilizing the road. The most important elements of this cost are travel time, perceived accident risks and other transport costs (e.g. fuel taxes). There are two types of externalities on the road network: externalities that are internal to road transport users (e.g. travel time, damage to road infrastructure, accident risks) and externalities external to road users (air and noise pollution, barrier effects etc.) that also affect the society as a whole. According to economic theory, the proper operation of a particular market is the single factor that determines whether expenses are internal or external [33] One of the key causes of market failure is external costs. Because of existence of the externalities, the social marginal costs will become higher. From the view of economic approach, there is a potential for efficiency improvement through the internalisation of externality costs, through regulatory measures, taxation and pricing schemes.

At F, when average cost and demand intersect with $q_0$ vehicle km driven in the zone and a unit cost of B, equilibrium will be reached. Beyond this point, using the road would cost the user more than it would benefit him/her, hence they would not use it. In this equilibrium, CF is the marginal social cost. Due to the additional vehicle on the road, the social marginal cost is equal to the individual cost plus the cost of the time spent by all other vehicles. Because of the externalities that are external to road users, the equilibrium at F is suboptimal. The optimal demand q* will be created by a highway toll or congestion fee equal to section DE. Point D where social marginal cost curve intersect with the demand curve is the optimal solution to the society.

Social marginal costs can be identified by deriving total user costs or experimentally by site observations or macro model simulations [33]. In order to assess the effectiveness of both new and current transportation networks, it is crucial to understand the monetary costs of the health impacts associated with transportation emissions that are borne by society. A monetary value has been attributed to these health impacts of air pollution through a number of studies in order to evaluate the proposed or implemented policy interventions. Three different types of methodologies that are commonly used for the valuation of externalities from transport emissions. These include: (a) the damage cost, (b) the avoidance cost and (c) replacement cost approaches [34]. These methodologies are explained briefly.

*2.2.1 Damage cost approach*

This method values the damage that is experienced by people as a result of the existence of externalities (such as the health impacts stemming from transport related noise and pollution resulting from transport vehicles). The two approaches used to value the external costs are willingness to accept (WTA) and willingness to pay (WTP), since market values for the experienced damage are not readily available. The Value of Statistical Life (VOSL) is the basis of the traditional methodology used to evaluate mortality or the risk of mortality. Individuals' willingness to pay (WTP) for a marginal decrease in the risk of premature mortality is summed up to determine the value of loss of life (VOSL), which is quantified at the societal level. Assume that each person has an expected utility function, denoted by ‘EU’, which links the utility of consuming over a specific time period, denoted by ‘U(y)’, and the risk of passing away during that time period, which is ‘r’:$EU(y,r) = (1-r)U(y)$. The individual's WTP to keep the same expected utility as the amount of risk decreases from r to r' is the answer to the of the following equation: $EU(y-WTP, r') = EU(y,r)$; and VOSL is the marginal rate of substitution between consumption and death risk reduction such that $VOSL = \partial WTP/\partial r$.

*2.2.2 Avoidance cost approach*

The avoidance cost approach determines the cost to reach a policy target (e.g., greenhouse gas reduction targets), with a focus on external cost valuation elements (e.g., shadow pricing). An avoidance cost function is used to assess the cost of delivering an additional level of environmental quality (such as reducing an extra amount of carbon dioxide).

*2.2.3 Replacement cost approach*

The replacement cost method is primarily concerned with estimating the value of a potential externality given the costs of substituting the externality's negative effects. Given that there are no accurate estimates of damage or avoidance costs, this method is typically employed to value external costs. A person who has health problems experiences multiple losses of utility: Not only is there suffering or discomfort as a result of the sickness, but there is also a loss of consumption (and leisure) time spent on mitigation activities for existing and prospective morbidity. The aggregate of three separate categories can be used to calculate the economic costs of the health impacts of air pollution: *1. Resource costs:* Direct non-medical and medical costs for treating the negative health impacts of air pollution, including additional costs; *2. Opportunity costs:* Related to the indirect expenses of lost productivity and/or leisure time as a result of health effects; *3. Disutility costs:* Related to the difficulty, pain, and concern brought on by the sickness.

In a recent study, EC [34] focused on the four types of impacts to assess the effects of transportation-related air pollution.: (1) health effects: exposure to air pollutants as nitrogen oxides (NOx), particles (PM2.5, PM10), and others is linked to a high risk of cardiovascular and respiratory disorders. These unfavourable health effects result in health care costs, reduced productivity at work, and even death (2) crop losses: Agricultural crops may be damaged by ozone and other acidic types of air pollutants (such as $NO_x$ and $SO_2$), which could result in a decrease in crop production. (3) material and building damage: Two different types of damage to materials and structures can be brought on by air pollutants like $SO_2$ and $NO_x$. These are (a) particle and dust contamination of building surfaces (b) materials and building facades are harmed by corrosion processes brought on by acidic chemicals (4) biodiversity loss: the damage to ecosystems include (a) acidification of water, soil, and precipitation (e.g., NOx, SO2) (b) eutrophication of ecosystems and negative impacts on ecosystem services ($NO_x$, $NH_3$). These environmental damages may result in a decline in biodiversity. The costs of these damages can be calculated by using the response functions. Several studies express their findings in terms of relative risk (RR), which is the ratio of reported incidents at two various stages of exposure. To quantify damages, this RR needs to be translated into concentration response function, namely the exposure response function [35]. Some examples of RR estimates are provided in WHO [24] and EC [34].

Because the literature on the calculation of air pollution costs in Türkiye is scarce, I used the benefit transfer approach, which is targeted at transferring knowledge from previously researched situations to other locations where the information is lacking. This data is presented in Table 4, which shows the value of air pollution expenses from several studies. The benefit transfer strategy multiplies the unit values by the economic disparity between the policy nation and the study nation, which is given as (see [34])

$$WTP_{ps} = WTP_{ss}\left(\frac{I_{os}}{I_{ss}}\right)^{\varepsilon} \quad (3)$$

where $WTP_{ps}$ is the WTP value transferred to the study area, $WTP_{ss}$ is the WTP at the study area, $I_{os}$ and $I_{ss}$ are the income or economic output at other and study areas, and ε is the income elasticity of WTP. In terms of income elasticity, EC [34] advised a value of 0.8, which was also adopted in the current study. According to EC [34], the subject value is based on a comprehensive OECD meta-analysis in which the income elasticity of WTP for environmental and health-related commodities ranges between 0.7 and 0.9.

It has already been demonstrated that the level of efficiency for automobiles and light commercial vehicles in Türkiye is comparable to that of vehicles registered in European nations [36]. As a first attempt, the WTP values obtained for Poland [34] were transferred to the Turkish situation using the formula in eq. (3). Poland was selected as the reference country considering that both countries of Poland and Türkiye have shown similar trends for the GDP per capita in the post-1995 period (Figure 4). Regarding CO estimates, the value reported for Finland (see [10]) was transferred to the Turkish case using eq. (3) and inflated to 2018 prices.

**Table 4.** Estimates of air pollution costs from different studies

| | EU-RTR [29] | EC-DG MOVE | EC [32] | | | HEATCO [10] | |
|---|---|---|---|---|---|---|---|
| | **Türkiye*** | **EU-average** | **EU-average** | **Poland** | **Türkiye**** | **Finland** | **Türkiye***** |
| Pollutants | 2010 prices (€) | 2010 prices (€/tonne) | 2016 prices (€/tonne) | 2016 prices (€/tonne) | 2018 prices (€/tonne) | 2000 prices (€/tonne) | 2018 prices (€/tonne) |
| CO | - | - | - | - | - | 12.5 | 10.63 |
| NMVOC | 10 | 1566 | 1200 | 700 | 852 | - | - |
| $NO_x$ | 2,278 | 10640 | 17000 | 11500 | 13987 | - | - |
| $NH_3$ | 5,443 | - | - | - | - | - | - |
| $PM_{2.5}$ (Urb) | - | 270178 | 381000 | 282000 | 342999 | - | - |
| $PM_{2.5}$ (Sub-Ur) | - | 70258 | 123000 | 91000 | 110682 | - | - |
| $PM_{2.5}$ (Rural) | - | 28108 | 70000 | 52000 | 63247 | - | - |

*Marginal abatement costs
** Estimated using the benefit transfer approach, in which Poland 2016 values have been transferred to the Turkish context and then inflated to 2018 using CPI data from TurkStat [32] for the 2016-2018 period.
*** Estimated using the benefit transfer approach, in which Finland 2000 values have been transferred to the Turkish context and then inflated to 2018 using CPI data from TurkStat [32] for the 2016-2018 period.

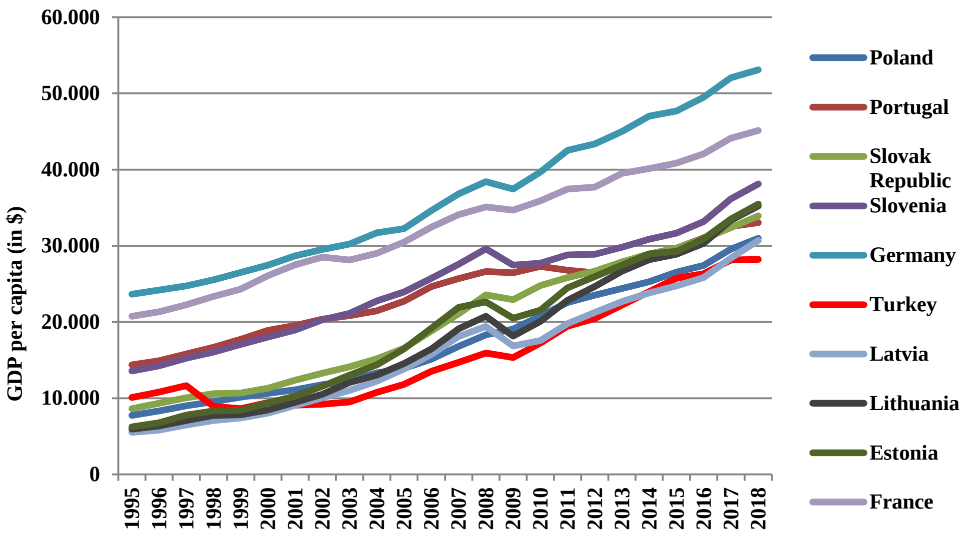


**Figure 4.** Comparison of GDP per capita across different European countries between 1995-2018 period

*2.2.4 The valuation of $CO_2$ emissions*

Fuel combustion accounts for 81% of greenhouse gas (GHG) emissions ($CO_2$), while agriculture is the main source of methane production. The social cost of carbon aims to assign a price on the physical harm that an additional tonne of GHG emissions will eventually cause. In order to lessen this harm in the future, society must be prepared to pay now. As a result, the social cost of carbon (SCC) is the fall in wellbeing caused by even a minor increase in carbon emissions [37]. The SCC estimations are extremely uncertain due to the unpredictability of future period emissions and climate change. Tol [38] reviewed numerous significant studies on climate change and estimated a median value of €4 and a mean value of €25 per tonne of carbon emissions released. These estimates, however, are cautious because the research only considers damage that can be quantified with an acceptable degree of certainty. Furthermore, longer floods and more frequently hurricanes with greater concentrations of energy are excluded since there is insufficient evidence to indicate a link between global warming and these effects [10].

As an alternative, HEATCO [10] cites research by Watkiss et al. [39] that developed shadow price values for carbon by taking into account projections of carbon's future development costs for damage and abatement. While avoidance cost estimates are centred on the UK state's long-term goal of reducing $CO_2$ emissions by 60% by 2050, damage costs were evaluated broadly. The latter is acknowledged to be consistent with the EU aim of keeping global warming to a growth of no more than 2°C over pre-industrial levels (HEATCO D5 [10]). Kuik et al.'s [40] meta-analysis work is widely recognized in the literature for confirming the significant diversity in SCC estimations. The study conducted by DEFRA [42] is cited by Kuik et al. [41] in the CASES Project as the most recent policy focused study on the estimation of social costs of carbon. It is highlighted that DEFRA [42] transparently integrates the findings from several Integrated Assessment Models (IAM), and the policy framework in which the values are used is clearly established.

The International Energy Agency (IEA) study, which provided a sub-2°C world scenario as stated by the Paris agreement, was cited in the World Bank Report [43]. The International Energy Agency (IEA) [44] produced specific scenarios for nine global regions in order to lessen reliance on imported fossil fuels, decarbonize power, improve energy effectiveness, and cut emissions in the industrial, transportation, and building construction sectors. The ETP 2012 2°C scenario investigates the technological possibilities for achieving a sustainable development centred on increased energy efficiency and a well-balanced energy system with a higher emphasis on the use of renewable energy sources and fewer air pollutant emissions. The 2°C scenario aims to reduce energy-related $CO_2$ emissions by more than half (relative to 2009) in 2050 and ensure that they keep decreasing after that. Regarding the 2°C scenario, carbon prices in 2030 rise to \$100/t$CO_2$ in the OECD countries and \$75/t$CO_2$ in Brazil, Russia, China and South Africa accompanied by a reduction in fossil fuel subsidies in the industrial and electricity sectors. The specifications on the IEA scenarios can be followed in IEA (2012). Table 5 presents global marginal abatement costs of $CO_2$ emissions from IEA [44] as cited by Hood [45].

**Table 5.** Global marginal abatement costs based on the IEA 2°C scenario (2012 prices)

| Year of Emission | Values, \$/t$CO_2$ |
|---|---|
| 2010-2020 | 30-50 |
| 2020-2030 | 80-100 |
| 2030-2040 | 110-130 |
| 2040-2050 | 130-160 |

Source: IEA [44] as cited by Hood [45]

**Table 6.** Average pollutant concentrations for different transport vehicle category, Türkiye, 2018

| Category | CO (tonne) | NMVOC (tonne) | $NO_x$ (tonne) | $N_2O$ (tonne) | $NH_3$ (tonne) | Pb (tonne) | $CO_2$ lube (tonne) | $PM_{2.5}$ (tonne) |
|---|---|---|---|---|---|---|---|---|
| Petrol Car | 752,409.7 | 55,901.0 | 55,963.6 | 236.8 | 614.2 | 0.83 | 23,719.7 | 75.5 |
| Diesel Car | 5,460.3 | 1,254.3 | 45,684.0 | 530.1 | 121.3 | 1.50 | 34,299.8 | 1,482.0 |
| Petrol LCV | 47,544.6 | 5,949.5 | 5,273.7 | 40.7 | 48.8 | 0.01 | 2,004.8 | 6.4 |
| Diesel LCV | 4,969.2 | 607.0 | 7,781.3 | 24.9 | 9.9 | 0.03 | 3,643.1 | 1,186.4 |
| Petrol HDV | 3,827,687.2 | 337,737.1 | 424,583.8 | 386.0 | 122.2 | 0.38 | 128,018.4 | 0.0 |
| Diesel HDV | 130,595.1 | 40,848.0 | 641,665.7 | 2,887.5 | 596.3 | 1.17 | 56,814.5 | 22,487.1 |
| Bus | 32,853.4 | 10,230.7 | 106,670.4 | 272.6 | 55.9 | 0.19 | 15,250.7 | 4,795.4 |
| Motorcycle | 87,887.3 | 11,031.9 | 3,885.7 | 24.5 | 23.3 | 0.01 | 2,708.9 | 171.6 |

## 3 Results and discussions

### *3.1 Results from air pollution estimations*

The mean concentrations of pollutants in tonnes for each vehicle type can be determined using vehicle fleet data (Tables 2 and 3) and emission factors computed for different road vehicle classes (Table 1), as shown in Table 6. Because there is no research on the distribution of transport emissions for the peak and off-peak periods, to disaggregate the average pollutant concentrations into peak and off-peak traffic periods, the ratios developed in the ExternE project and the rates provided by Casey [46] were used (see Apendix for the average traffic distributions).

The examples of the disaggregated average pollutant concentrations are shown in Table 7. Finally, the percentage distribution of pollutants is given in Figure 5. From Table 7, off-peak emissions are lower than peak emissions as expected for all the categories of air pollutants. CO and NMVOC emissions are the highest for petrol car, $NO_x$ and $PM_{2.5}$ are the highest for bus, and $NO_2$ is the highest for diesel car. From Table 6, it can be noted that HDVs generate the highest levels of emissions including CO, NMVOC, $NO_x$, $N_2O$, $CO_2$ lube and PM. The passenger cars, on the other hand, produce high levels of $NH_3$ and Pb emissions. Diesel LCVs and motorcycles are generally associated with the lowest level of emissions.

From Figure 5, cars, LCVs and HDVs are responsible from the highest percentage of CO emissions, which is followed by $NO_x$ and NMVOC. Regarding busses, $NO_x$ emissions have the highest share that is followed by CO, $CO_2$ and NMVOC. In all vehicle categories, Pb and $PM_{2.5}$ have the lowest shares in the total concentration of pollutants.

However, it should be emphasized that the population's exposure to fine particles in Türkiye is higher than the averages for the EU and OECD, and that the costs of premature mortality due to exposure to outdoor PM2.5 and ozone have increased since 2005 [5, 48]. The main sources of $PM_{2.5}$ emissions are coal-based heating systems, particularly low-quality fuel and burning systems, and industrial and mobile sources [49]. Since there is no detailed data, these health impacts of $PM_{2.5}$ emissions at the local level were not analysed; however, based on the availability of data, this analysis can be conducted in the future.

### *3.2. Results from air pollution cost estimation*

The data in Table 4 was coupled with mean pollutant concentrations calculated for each vehicle type in Table 5 in order to determine air pollution costs related to each different vehicle category. The findings are presented in Table 6 (see also Figure 6), Table 7 and Table 8 where upper and lower limit values represent the pollution costs valued at high and low prices. If there is single cost factor, the corresponding values are represented as average. Finally, the cost factor of $PM_{2.5}$ computed for sub-urban areas (Table 4) was used to compute the cost values of $PM_{2.5}$ in Table 7. From Table 7, it can be seen that the cost of CO, NMVOC, $NO_x$ emissions are the highest for HDVs; and the cost of $NH_3$ and $PM_{2.5}$ are the highest for passenger cars as well as the diesel HDVs. In common, the emission costs are lower for the diesel vehicles compared to petrol vehicles. Diesel LCVs have the lowest cost estimated for CO, $NH_3$ and NMVOC; petrol LCVs have the lowest cost regarding $NO_x$, and petrol HDVs have the lowest cost regarding $PM_{2.5}$ emissions.

**Table 7.** Disaggregated emission concentrations for public and private vehicles, Türkiye, 2018

| Category | CO (tonne) | NMVOC (tonne) | $NO_x$ (tonne) | $N_2O$ (tonne) | $PM_{2.5}$ (tonne) |
|---|---|---|---|---|---|
| Petrol Car-Peak | 391,253.0 | 30,186.5 | 29,101.0 | 132.6 | 37.8 |
| Petrol Car-Off-Peak | 361,156.7 | 25,714.5 | 26,862.5 | 104.2 | 37.8 |
| Diesel Car-Peak | 2,839.4 | 677.3 | 26,039.9 | 296.8 | 933.7 |
| Diesel Car-Off Peak | 2,621.0 | 577.0 | 19,644.1 | 233.2 | 548.3 |
| Petrol LCV-Peak | 24,723.2 | 3,212.7 | 2,742.3 | 22.8 | 3.2 |
| Petrol LCV-Off Peak | 19,968.7 | 2,736.8 | 2,531.4 | 17.9 | 3.2 |
| Diesel LCV-Peak | 2,584.0 | 327.8 | 4,435.3 | 14.0 | 747.5 |
| Diesel LCV-Off Peak | 2,385.2 | 279.2 | 3,346.0 | 11.0 | 439.0 |
| Bus-Peak | 20,369.1 | 6,547.7 | 59,735.4 | 149.9 | 2,973.1 |
| Bus-Off Peak | 12,484.3 | 3,683.1 | 46,935.0 | 122.7 | 1,822.3 |

**Note:** Calculations are based on the information given in Table 6 and the ratios provided in Bickel et al. [47] and Casey [46]

**Table 8.** Estimated marginal air pollution costs (in cent€/km) for public and private vehicles in Türkiye, 2018

| | CO | | NMVOC | | | $NO_x$ | | $NH_3$ | $PM_{2.5}$ |
|---|---|---|---|---|---|---|---|---|---|
| | Average | Upper | Average | Lower | Upper | Average | Lower | Average | Average |
| Petrol Car | 0.009 | 0.053 | 0.053 | 0.001 | 0.863 | 0.568 | 0.272 | 0.007 | 0.009 |
| Diesel Car | 0.000 | 0.001 | 0.001 | 0.000 | 0.388 | 0.255 | 0.122 | 0.001 | 0.100 |
| Petrol LCV | 0.007 | 0.066 | 0.067 | 0.001 | 0.962 | 0.633 | 0.304 | 0.007 | 0.009 |
| Diesel LCV | 0.000 | 0.004 | 0.004 | 0.000 | 0.781 | 0.514 | 0.247 | 0.001 | 0.943 |
| Petrol HDV | 0.032 | 0.224 | 0.227 | 0.003 | 4.616 | 3.036 | 1.456 | 0.001 | 0.000 |
| Diesel HDV | 0.001 | 0.038 | 0.039 | 0.001 | 9.898 | 6.510 | 3.123 | 0.007 | 2.745 |
| Bus | 0.002 | 0.040 | 0.040 | 0.001 | 6.808 | 4.478 | 2.148 | 0.003 | 2.422 |
| Motorcycle | 0.008 | 0.077 | 0.078 | 0.001 | 0.443 | 0.292 | 0.140 | 0.002 | 0.155 |

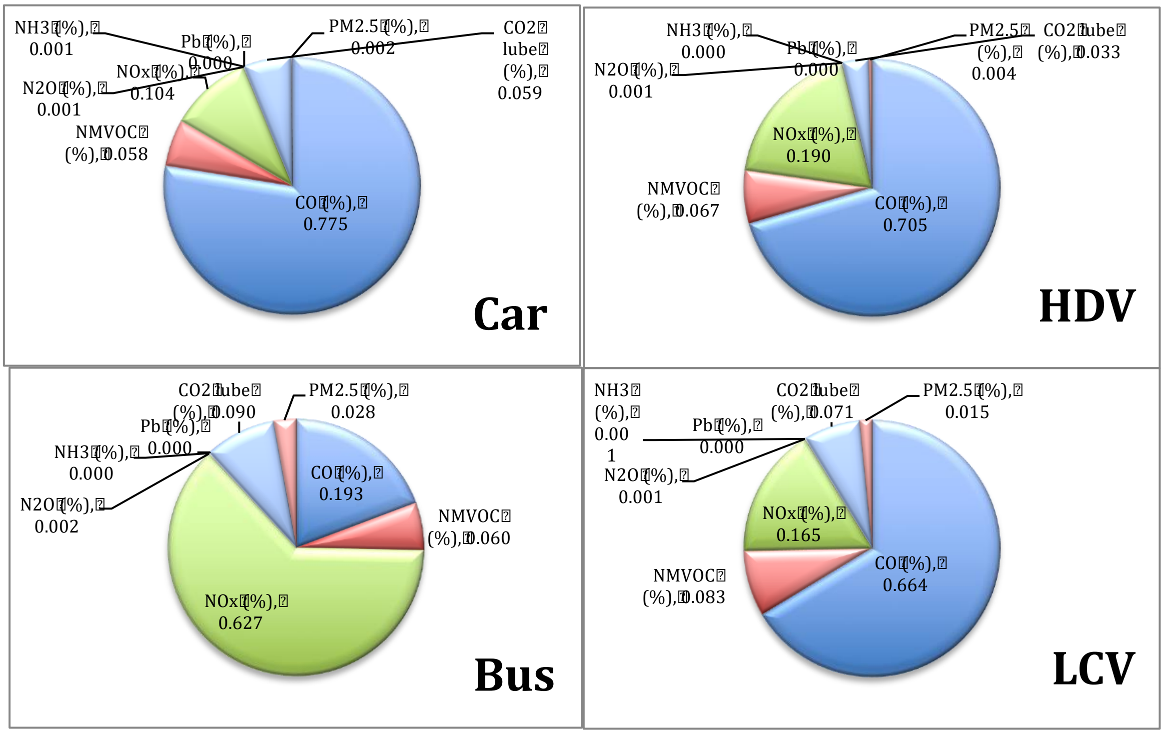


**Figure 5.** Percentage distribution of air pollutants across different vehicle categories (car, HDV, LCV, bus)

**Table 9.** Social costs of carbon, Türkiye (2018 prices)

| Category | $CO_2$ (m tonnes) | Low values (€) | Central values (€) | High values (€) |
|---|---|---|---|---|
| Petrol Car | 12.838 | 404,424,601 | 539,232,802 | 674,041,002 |
| Diesel Car | 18.565 | 584,817,754 | 779,757,005 | 974,696,257 |
| Petrol LCV | 1.009 | 31,805,919 | 42,407,892 | 53,009,865 |
| Diesel LCV | 1.834 | 57,797,853 | 77,063,803 | 96,329,754 |
| Petrol HDV | 27.184 | 856,321,758 | 1,141,762,344 | 1,427,202,930 |
| Diesel HDV | 12.064 | 380,035,291 | 506,713,721 | 633,392,152 |
| Bus | 6.473 | 203,915,071 | 271,886,762 | 339,858,452 |
| Motorcycle | 4.027 | 126,881,753 | 169,175,670 | 211,469,588 |

**Table 10.** Marginal social costs of carbon (in cents€/km), Türkiye (2018 prices)

| Category | Low values | Central values | High values |
|---|---|---|---|
| Petrol Car | 0.89 | 1.19 | 1.49 |
| Diesel Car | 0.71 | 0.95 | 1.18 |
| Petrol LCV | 0.83 | 1.11 | 1.38 |
| Diesel LCV | 0.83 | 1.11 | 1.38 |
| Petrol HDV | 1.33 | 1.77 | 2.22 |
| Diesel HDV | 0.33 | 0.43 | 0.54 |
| Bus | 1.86 | 2.48 | 3.10 |
| Motorcycle | 1.04 | 1.38 | 1.73 |

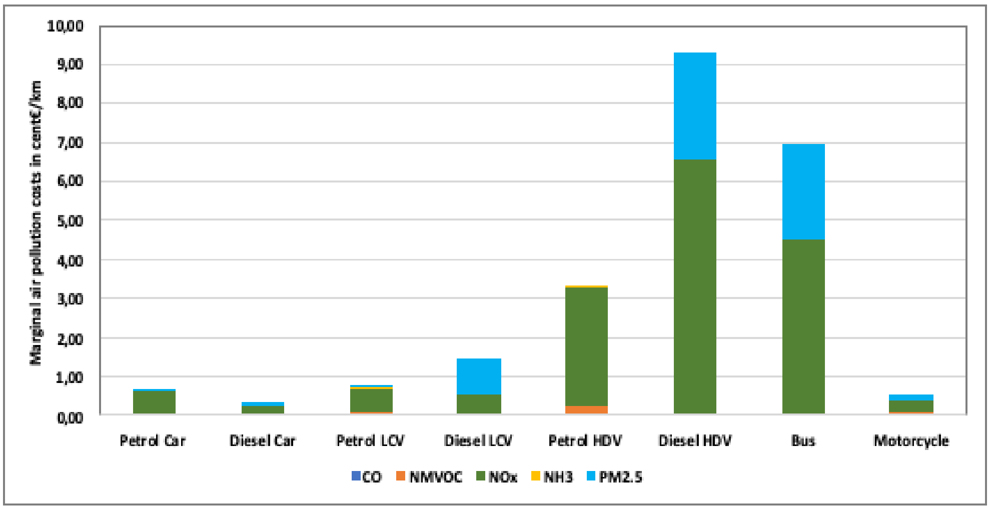


**Figure 6.** Air pollution costs across different vehicle categories (car, HDV, LCV, bus, motorcycle)

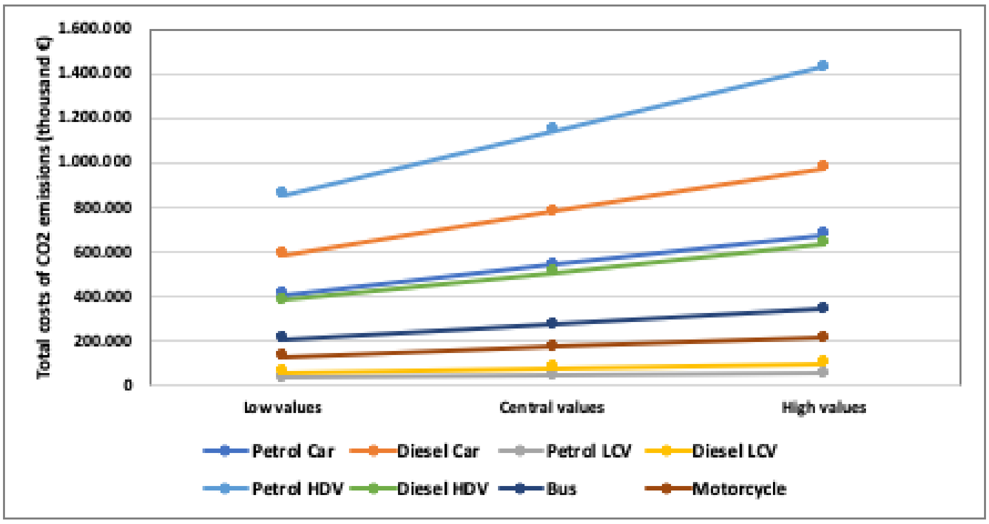


**Figure 7.** Social costs of carbon across different vehicle categories (car, HDV, LCV, bus, motorcycle)

For the evaluation of $CO_2$ emissions, a cost factor reflecting the global shadow value was used considering that greenhouse gas emissions have global environmental impacts such as ozone depletion and global warming. As presented in Table 5, carbon values derived by IEA [44] was utilised in the current analysis. These figures have been adjusted to 2018 World prices using OECD and non-OECD averages of general CPI information. In Türkiye, carbon dioxide emissions from transportation totalled 84.6 million tonnes in 2017 [48]. I utilized the % share distribution of CO2 lube in Table 6 for the breakdown of carbon dioxide emissions according to road vehicle type. Based on the $CO_2$ emissions computed for each vehicle category as given in the first column of Table 9 and shadow prices of carbon in Table 5, the values of $CO_2$ emissions in 2018 are computed in Tables 9 and 10. From Table 9 (see also Figure 7), the highest costs of $CO_2$ emissions were estimated for petrol HDVs followed by diesel car and petrol car and diesel HDVs. The lowest cost values were estimated for petrol and diesel LCVs.

## 4 Conclusions

Valuation of external costs of road transport emissions has become a key component in evaluating the socio-economic and environmental costs of air pollution as it enables evaluation of the effectiveness of cost-benefit analyses of pollution control methods and serves as a foundation for the development of an appropriate policy framework. It was found that the main air pollutants CO, NMVOC, $NO_x$, $N_2O$, $CO_2$ and PM are the emissions that are commonly associated with road vehicle transport in Türkiye. The air pollution may have different spatial impacts with excessive contamination on the highways and busy roads and slighter impacts on other sites. Therefore, the application of a bottom-up model would be more relevant in such a case for detailed assessment and estimation of peak and off-peak levels of air pollution for the identification of problematic sites that are prone to heavy traffic and significantly pollute the air.

There needs to be a validation of the current model by using a local data to understand whether there is overestimation or underestimation of the modelled air pollutant concentrations. The reason may be the different sources of uncertainties that can be related to the structure of transport vehicle flows, the conditions of emission concentrations such as climatic factors (e.g. temperature, wind speed) and characteristics of road and the nearby landscape. Therefore, more detailed data would be essential for the estimation of many of the significant parameters. Among these, the data from an operational transportation model (e.g. vehicle flows, average vehicle speeds, vehicle kilometers), spatial distribution of population, the emission factors from different road vehicles can be prioritised. The development of such data at the local and national levels, investigation of local cost variables (such as valuation of air pollution at the local and global levels), and assessment of the effects of mitigation measures for externalities should all be the goals of future research. Due to the variety of road classes, road vehicles and vehicle kilometers, it might be claimed that the marginal external costs determined in the current study may be underestimated. Another reason may be the assumptions being made for the estimation of external costs undertaken in the study. The lack of data and research on the estimation of external costs of transport-related air pollution in Türkiye has limited the analysis of the current study. An in-debt examination of the externality effect assessments at the local/regional scale, as well as the compilation of disaggregated data for the model's inputs, will thereby improve the estimations of external costs of air pollution.

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**Conflict of interest**

The author declares that there is no conflict of interest.

**Similarity rate (iThenticate):** 19%

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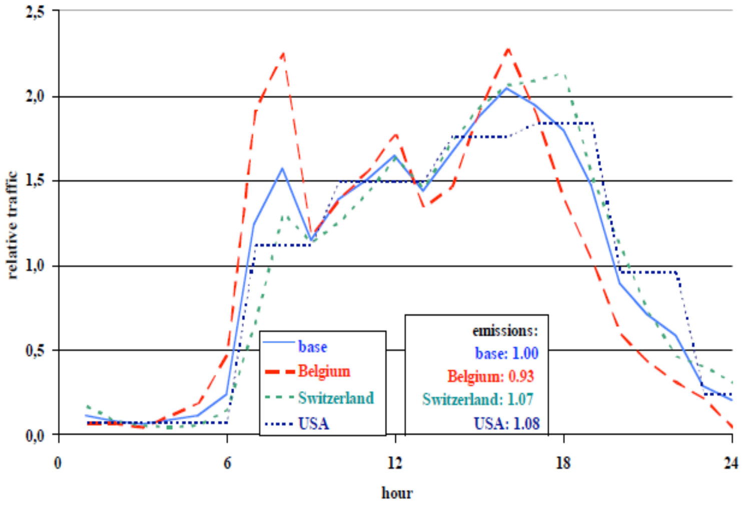
## Appendix

**Table A1.** Relative traffic distribution $ptf_{i,h}$ during the day

| Hour | $Ptf_{i,h}$ | Hour | $Ptf_{i,h}$ | Hour | $Ptf_{i,h}$ | Hour | $Ptf_{i,h}$ |
|---|---|---|---|---|---|---|---|
| **1** | 0.12 | **7** | 1.29 | **13** | 1.40 | **19** | 1.28 |
| **2** | 0.08 | **8** | 1.78 | **14** | 1.60 | **20** | 0.86 |
| **3** | 0.05 | **9** | 1.16 | **15** | 1.93 | **21** | 0.58 |
| **4** | 0.08 | **10** | 1.33 | **16** | 2.17 | **22** | 0.39 |
| **5** | 0.13 | **11** | 1.50 | **17** | 1.99 | **23** | 0.31 |
| **6** | 0.32 | **12** | 1.71 | **18** | 1.76 | **24** | 0.18 |
| | | | | | | **average** | 1.00 |

Source: ARTEMIS (2007)




Source: ARTEMIS (2007)

**Figure A1.** Average Traffic Distributions (relative to the Hourly Average) Representative of 3 Countries, Belgium, Switzerland, USA, and Relative Base (Average) Distribution