

## Life Cycle & Sustainability

# Life Cycle Air Emissions External Costs Assessment for Comparing Electric and Traditional Passenger Cars

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### ABSTRACT

This study compares the environmental air emissions external costs of electric, gasoline, and diesel private passenger cars during their entire life cycle. The results provide the decision makers with a complementary and unconventional interpretation of the results of an ISO 14040-compliant life cycle assessment (LCA). Indeed, LCA results are often difficult to communicate and to be understood by the general public; on the other hand, an environmental external costs evaluation, where a single monetary value synthesizes the environmental impacts, can be easily understood, communicated to the broad public, and compared with taxes, incentives, and other economic tools. In the present study, we demonstrate that it is possible to carry out the application of a damage factor to the physical inventory flow. The application of this methodology to an Italian context leads to the conclusion that if we compare the 3 types of vehicles—electric, diesel, and gasoline—of an average midsize car (e.g., Volkswagen Golf), the electric version produces less external cost than the traditional internal combustion engine vehicles, considering both air pollution and climate change. The total life cycle air emissions externalities are 12.07 €/1000 km for the electric version, 21.30 €/1000 km for the gasoline vehicle, and 24.25 €/1000 km for the diesel vehicle. At the same time, the electric vehicle produces less external cost related to the air emissions considering both the entire life cycle and only the processes that occur in Italy. *Integr Environ Assess Manag* 2020;16:140–150. © 2019 The Authors. *Integrated Environmental Assessment and Management* published by Wiley Periodicals, Inc. on behalf of Society of Environmental Toxicology & Chemistry (SETAC)

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### INTRODUCTION

The transition toward electric vehicles (EVs), and in particular toward electric cars for private mobility, is seen as a great opportunity both for decarbonizing the transport sector and for improving air quality in highly populated urban areas (Girardi et al. 2015). In fact, EVs can rely on higher overall efficiency and zero tailpipe exhaust emissions. Understanding the environmental effects and trade-offs of replacing internal combustion engine vehicles (ICEVs) with EVs requires a life cycle approach (Gao and Winfield 2012). Of course, many comparative life cycle assessment (LCA) studies have been carried out as outlined by a recent review study (Nordelöf et al. 2014). Results from these comparative LCAs show that for most environmental impact categories (e.g., climate change, air acidification), electric cars are preferable to traditional cars (Girardi et al. 2015). On the

contrary, for some impact categories, diesel, gasoline, and electric cars perform almost the same, while for other categories (e.g., freshwater ecotoxicity), electric cars perform the worst (Hawkins et al. 2013). In this framework, the use of several indicators in life cycle impact assessment (LCIA) makes the interpretation and the identification of trade-offs difficult for policy and decision makers (Morel et al. 2018). Moreover, it should be underlined that most of the current ISO 14040 (ISO 2006) LCA impact categories make no distinction between different spatial impact categories. Impacts are summed up for phenomena at a global scale (e.g., climate change), at a regional scale (e.g., acidification) and at a local scale (Jeswani et al. 2010). This implies, for example, that emissions occurring in low-density areas from high stacks are considered as important as emissions occurring in high-density urban areas at ground level.

A way to overcome these difficulties could be to evaluate in monetary terms the external costs during the entire life cycle. Monetization could be seen as a bridge between environmental assessment and economic evaluation, providing a common and unified unit (Morel et al. 2018). Although the importance of coupling LCA and external costs evaluation in transport and energy environmental assessment has been recognized (Weidema et al. 2013), most

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studies use a flat damage function (i.e., €/t of emission) without taking into account any geographical differentiation (De Nocker et al. 1998; Weis et al. 2016), or they have used in the past the so called “impact pathway” approach (Schleisner 2000) also in the case of transport (Funk and Rabl 1999). However, in this way they have far exceeded an LCA study application for effort and expertise needed.

In the present paper, the authors present a methodology for applying damage factors (i.e., external costs per unit of emission) to life cycle inventory results (LCI) in line with the spirit of the new ISO 14008, “Monetary valuation of environmental impacts and related environmental aspect” (ISO 2019). The methodology we propose takes into account the year of emission (through economic parameters that represent the per capita income growth rate and the elasticity of marginal utility of consumption), the geographical area where the pollutant emissions take place, the average height of release, the population density of the area where the emissions take place, and the average level of income of the country in which the emissions take place. The methodology has been applied to carry out an environmental comparison among electric, gasoline, and diesel private passenger cars through their entire life. The results provide decision makers with an increased awareness of the problem and a complementary and unconventional interpretation of the results of an LCA comparable with economic instruments, such as taxes and subsidies.

## MATERIALS AND METHODS

Despite the fact that associating LCA and external costs evaluation is recommended (Weidema et al. 2013) and used in the research community, most of the applications make use of damage factors (expressed in €/t of pollutant emission) which refer only to the geographical area of the use phase of the product or service considered (De Nocker et al. 1998; Weis et al. 2016). Nevertheless, these damage factors do depend on the geographical context and, in particular, they are determined by physical (pollutants dispersion), societal (population density), and economical (willingness to pay [WTP] to avoid environmental damages) factors. Providing such estimates without considering the different geographical context of the various life cycle phases (e.g., where primary energy sources like crude oil are extracted) could lead to non-negligible errors. In other words, external costs cannot be considered a standard ISO 14040 impact category where contributions from different life cycle stages are summed up without taking into account the region where emissions and effects occur (Jeswani et al. 2010). The characterization factors of this particular impact category must be site-sensitive. Such characterization factors can be derived from the results of the NEEDS project (Schenler et al. 2009). NEEDS was an integrated project financed by the 6th Framework Programme of the European Union: Sustainable Energy Systems. The ultimate objective of NEEDS was to evaluate the full costs and benefits (i.e., direct and external) of energy policies and of future energy systems, both at the level of individual countries and for the enlarged European

Union as a whole. Among other results, the project provided damage factors (€/t of pollutant) for different countries, depending on the height of emission and year of emission (Bickel and Rainer 2005). The methodology that we propose for the application of the external costs to LCA starts from the damage factors produced by the NEEDS project.

The main steps of the proposed methodology are:

- To make an updated and detailed LCA of a midsize (c-segment in Europe) car with the 3 different engines (diesel, gasoline, and electric) offering as far as possible the same service in terms of performance and comfort;
- To regionalize both the life cycle processes (i.e., to assign a geographical reference to each process and subprocess involved in the LCA) and the LCA results in terms of environmental impacts;
- To update and detail the damage factors (€/t of pollutants emission) provided by the NEEDS project, on the basis of the most recent and comprehensive studies on external costs of energy carriers;
- To apply the damage factors to selected LCI results;
- To compare the 3 types of vehicles on the basis of the life cycle external costs of air emissions.

The following paragraphs will describe the main assumptions of the study according to the ISO 14040 goal and scope definition. The methodology for external cost calculation is described in the LCIA methodology paragraph as suggested in ISO 14040.

### *Goal of the study*

The goal of the study is to compare an electric, a diesel, and a gasoline midsize car used for urban journeys. To evaluate the external costs, the life cycle potential impacts have to be regionalized. In this study, we assume that the use phase of the vehicles takes place in Italy.

### *System description*

When comparing different vehicles, it is crucial to select vehicles that offer as far as possible the same services. As shown in a previous work (Girardi and Brambilla 2017), the market segment has little influence on the environmental comparison among electric, diesel, and gasoline cars. For this reason, the analysis developed in the present work focused on the 3 versions—electric-, gasoline-, and diesel-fuelled—of the Golf (Volkswagen) vehicle, considered as representative of the most widely used midsize vehicle on the market; it has been the most-sold vehicle model across Europe in 2016, 2017, and 2018 (Bekker 2019). Moreover, as shown in Table 1, the Golf is one of the few car models sold in the 3 versions (electric, diesel, and gasoline), comparable for the main technical characteristics (power, speed, maximum torque), comfort, and aesthetics. In other words, except driving range, the 3 models offer a similar service to the user. The performances of the vehicles are evaluated on the basis of urban driving cycle because EVs are characterized by almost-zero tailpipe emissions, and it is in the urban context that the benefits of electric transport would

Table 1. Main characteristics of the compared vehicles

Model	Fuel	Vehicle weight (kg) <sup>a</sup>	Battery weight (kg)	Max engine power (kw)	Max torque (Nm)	Urban range (km)	Urban fuel consumption <sup>b</sup>
Volkswagen e-Golf 2016	Electricity	1249	318	100	290	201	16.8
Volkswagen Golf TSI 1, 8 L, 4 cylinder	Gasoline	1344	—	125	270 @ 1600	531	9.4
Volkswagen Golf TDI 2, 0 L, 4 cylinder	Diesel	1397	—	110	320 @ 1750	637	7.8
Data source	—	c	c	c	c	d	d

<sup>a</sup>Vehicle weight, without passengers and load (curb weight); for the electric vehicle, battery weight is excluded.

<sup>b</sup>Expressed in kWh/100 km for the electric vehicle and in l/100 km for petrol- and diesel-fueled vehicles.

<sup>c</sup>Volkswagen; Wolfsburg (DE).

<sup>d</sup>USEPA 2016.

be most significant (and where the problem of air quality is more serious). Further, the limited range of EVs, which is considered one of the barriers to the acceptance of electric mobility, can be negligible in urban and suburban trips. Regarding the comparability of vehicles (i.e., the ability to offer the same services), it should be noted that a substantial portion of Italian drivers could move from conventional vehicles (internal combustion) to EVs (taking advantage of overnight recharge) without having to substantially change their traveling or commuting behavior (De Gennaro et al. 2014).

Regarding the dataset selection, we assumed that the use phase of the vehicles takes place in Italy, as well as the maintenance and end-of-life of vehicles and lithium-ion (Li-ion) battery. The vehicles' production phase and the battery pack assembly take place in Germany, where Golf, e-Golf, and its batteries are actually built, while the battery cell production occurs in South Korea.

#### Functional unit

The service provided by the vehicles is the distance driven. Accordingly, the functional unit of the study is based on the kilometers driven. More specifically, the functional unit selected for the study is 1000 km.

#### System Boundaries

The life cycle approach chosen for the study is the so-called "from cradle to grave" approach, and it considers:

- Vehicle production and dismantling;
- Electric battery production and dismantling;
- Complete energy carrier supply chains, including primary energy sources production;
- Vehicle use phase;
- Vehicle maintenance phase;
- Roads maintenance.

#### Data quality requirements and allocation procedure

The database used for background data is Ecoinvent v3, allocation at the point of substitution (Wernet et al. 2016).

Of course, for achieving a precise LCI, the technologies, the processes, and the energy carriers involved in the vehicles' life cycle must be geographically representative of the particular case study because these factors vary significantly moving from one place to another. For example, it is clear that air emissions associated with energy consumption strongly depend on the energy mix used and, hence, on where the process is located. For this reason, the original Ecoinvent datasets used have been modified not only according to the vehicles' weight but also according to where the vehicles' parts are produced. For example, the energy mix used in vehicles and battery assembly is German, while for the Li-ion cells is the South Korean one (where e-Golf cell batteries are produced) instead of the China electricity production mix of the original Ecoinvent dataset. For the same reason, the European markets for steel, glass, and other materials have been used in place of global markets. As far as possible, the study makes use of reliable and recent data related to the geographical scope of the study, as discussed previously.

#### LCIA methodology and type of impacts

In the present study, authors use only 1 impact category: air emissions external costs. The phenomena covered by this impact category are climate change and impacts on human health. In particular, impacts on human health cover effects like chronic mortality, infant mortality, acute mortality, and morbidity (Preiss et al. 2008). As mentioned in the *Introduction*, to use the air emissions externalities as an LCIA indicator, the impact assessment phase must be sensitive to the site-specific conditions that determine the indicator value. In other words, the particular characterization factors (ISO 2006) used—that in external costs evaluation, correspond to the damage factors for unit of pollutant emitted—must vary with site-specific conditions such as population density, meteo-climatic conditions, or background concentrations. According to this approach, there will not be a single characterization factor for each inventory flow (e.g., nitrogen oxides [NO<sub>x</sub>]), but at least 3 characterization factors for each geographical area considered in

the analysis: 1 for emission from high stacks (e.g., industrial chimney), 1 for low releases (e.g., tailpipe emissions), and 1 for average or unknown heights of release. Hence, the proposed methodology goes a little bit beyond the standard application established by the ISO 14040 standard, going toward a spatial differentiation (Jeswani et al. 2010). To allow the calculation of the environmental external cost indicator, the inventory physical flows shall be provided with geographical plus height of emission information, to apply the appropriate damage factors.

Furthermore, concerning processes occurring in Italy, a distinction between urban, suburban, and rural areas has been made as far as it has a relevant influence on the damage factor for emissions of fine inhalable particles with diameters 2.5  $\mu\text{m}$  and smaller (PM 2.5). In fact, the damage costs of PM 2.5 emissions, in the present study, depend also on the population density of the area where the emissions take place (distinguishing among urban, suburban, and rural areas). This upgrade aims at implementing the suggestions given in the guidelines of the European Commission about external costs evaluation in the transport sector in a European context (Korzhenevych et al. 2014).

As described, we assumed that the use phase of the vehicles takes place in Italy, as well as the maintenance and end-of-life of vehicles and Li-ion battery. The vehicles' production phase and the battery pack assembly take place in Germany, where Golf, e-Golf, and its batteries are actually built, while the battery cell production is located in South Korea, where the actual supplier is located. Regarding the energy carrier supply chain, the refining and distribution of the fossil fuels take place in Italy, while the extraction phase is allocated proportionally to each single nation from which Italian imports come. Electricity production takes place in Italy, except the import that is located in Europe.

As known, it is not possible to assign a geographical reference automatically. Commercial LCA software and databases are not designed to handle the geographical allocation of processes and, therefore, of the inventory flows. We refer specifically to SimaPro 8.3 and Ecoinvent 3.3 that have been used in the present work, but the statement can be extended to any existing LCA software and database. Considering that more than 10 000 processes were involved, a simplified approach was chosen, assigning a geographical reference to the processes that contribute more than 2% to the first rough estimation of overall air pollution externalities value, obtained using the Italian damage factors at average height of release for the entire life cycle.

Then, for all the involved processes (i.e., for all the processes contributing for more than 2% to the rough estimate of externalities), a height of release for air pollutants (high, low, average) and an area of release were assigned, distinguished by country: Italy, where cars are used and most of the electricity is produced; Germany, where cars are produced and the Li-ion battery is assembled; Libya, Algeria, Netherlands, Russia, mainly for upstream of fossil fuels; the 28 member states of the European Union (EU28) and rest of the world (RoW), for materials production. As discussed, the

Table 2. Geographical area used to estimate externalities

Geographical area <sup>a</sup>	Height of emission	Population density	Group name
Germany	High	undefined	DE_high
Germany	Low	undefined	DE_low
Germany	Average	undefined	DE_unknown
EU_27	High	undefined	EU_high
EU_27	Average	undefined	EU_unknown
Italy	Low	Urban	IT_low_urban
Italy	Low	Suburban	IT_low_suburban
Italy	Low	Rural	IT_low_rural
Italy	High	Rural	IT_high_rural
Rest of the world	Average	undefined	RoW_unknown
Russia	Average	undefined	RU_unknown
Libya	Average	undefined	LY_unknown
Algeria	Average	undefined	DZ_unknown
Netherlands	Average	undefined	NL_unknown
North Sea	Average	undefined	NOS_unknown

<sup>a</sup>For each geographical area (and for each height of release for pollutant), we used an appropriate damage cost (e.g., €/t emitted). For PM 2.5, a further distinction is made among urban, suburban, and rural areas due to the different effects on areas with relevant differences in population density. Each life cycle process has been assigned to a group (last column in table).

height of release has an influence on the pollutants' dispersion and on the related damage factors. For this reason, the geographical areas have been further subdivided into 3 classes corresponding to 3 different heights of release: high, low, average or unknown.

The list of the geographical areas considered is shown in Table 2, together with the relative group name. Each process of the life cycle has been assigned to one of the groups listed in Table 2.

Processes have been assigned to these groups using the analysis group function of the SimaPro software, while subsequent calculations needed for external costs evaluation have been performed offline outside the software.

#### *Monetary estimation: Update of damage factors for air emissions*

The methodology used to evaluate external costs in this study is built upon the NEEDS methodology (Desaigues et al. 2011) that provides damage factors (expressed in €/t of emitted pollutants) for the evaluation of the damage costs to human health and environment caused by local and regional air pollutants, ammonia (NH<sub>3</sub>), NO<sub>x</sub>, nonmethane volatile organic compounds (NMVOCs), PM<sub>2.5</sub>, sulfur dioxide (SO<sub>2</sub>) (Preiss et al. 2008) and by climate change (expressed as carbon dioxide equivalent, CO<sub>2</sub>eq).

As discussed, these damage costs are expressed as a function of country of emission, height of emission, and year

of emission. The country where the emissions take place is important because the damage factors depend on the meteorology, the population density, and the socio-economic conditions. The height of emission is taken into account within the dispersion modeling that allows the creation of source receptor matrices and, through the use of concentration response functions and exposed population data, allows for the damage costs evaluation. The year of emission influences the WTP for a year of life lost. Because it is a positive function of the per capita income, such willingness, for each given country, can vary with time depending on the growth of per capita income.

In our study, several upgrades were introduced compared with NEEDS methodology. First, the annual average growth rate has been updated because the values suggested in NEEDS were estimated before the recent world economic crisis. The average growth rate (2000–2017) for the EU28 aggregate has been estimated using the purchasing power parity (PPP) per capita gross domestic product (GDP) data from the World Economic Outlook database of the International Monetary Fund (IMF 2018). Therefore, the average growth rate assumed in this study for EU28 countries is 1.4%, well below the 2% value suggested by the NEEDS methodology.

Second, the damage factors have been adjusted to take into account the differences in the purchasing power between the countries involved in the study. The NEEDS damage factors are based on a European Union average value of lost years ( $VOLY_{2000}$ ) which is not adequate to represent the difference in income levels between the countries being analyzed, especially in the case of extra-European Union states, where, for example, natural gas and crude oil come from. For this reason, the average European Union damage costs have been transferred to the countries encompassed in this study using the following equation (Pearce and Howarth 2001; Lindhjem et al. 2011; OECD 2011; UNEP 2013):

$$V_{pt} = V_{s2017} \left( \frac{\bar{Y}_{pt}}{\bar{Y}_{s2017}} \right)^\varepsilon$$

where  $V_{pt}$  is the value in the policy site in year  $t$ ,  $V_{s2017}$  is the value in the study site (i.e., the year to which values to be transferred are referred to),  $\varepsilon$  is the income elasticity of the WTP for the good which value has to be transferred,  $\bar{Y}_{pt}$  is the per capita gross domestic product (a proxy for per capita income) expressed in PPP of the policy site in year  $t$  (constant values), and  $\bar{Y}_{s2017}$  is the PPP per capita GDP of the study site.

It is important to take into account income differences between the study and the policy site because income influences the WTP for an environmental good or health (i.e., it tends to increase as income increases).

If income levels are easily proxied by the per capita GDP (at PPP),  $\varepsilon$  is not easy to estimate because it can vary not only from country to country, but also from individual to individual on the basis of personal preferences and income

levels (Martini and Tiezzi 2014). In a recent handbook, Korzhenevych et al. (2014) assume  $\varepsilon = 1$ , although there is little empirical evidence for the fact that a change in the willingness to avoid ill health outcomes is exactly proportional to a change in income, as a unitary  $\varepsilon$  would imply. This is especially true when the difference in income between the policy and the study site is wide (Navrud and Ready 2007). Pearce and Howarth (2001) provide a list of values for  $\varepsilon$  for the valuation and transfer of values of a statistical life; such estimates span from 0.3 to 1.1. The income elasticity of WTP for the reduction of health-related risks is usually positive but lower than 1, as pointed out by Navrud and Ready (2007). More recent research (Navrud 2017) specifies that usually the income elasticity of WTP for different environmental, morbidity, and mortality impacts is often between 0.3 and 0.7.

It is worth highlighting the fact that, as shown by Flores and Carson (1997), a luxury good—as the environment and personal health are often regarded—can have an income elasticity of WTP lower than 1, while having, as the definition of a luxury good implies, an income elasticity of demand greater than unity. Because it seems plausible that the income elasticity of WTP varies with income levels, it has been assumed  $\varepsilon = 0.2$  for Western European countries (EU15) and  $\varepsilon = 0.5$  for the other EU countries, as empirically estimated by Desaignes et al. (2011) and suggested by Navrud (2009). Many studies highlight the fact that the income elasticity of the WTP for environmental goods or for reducing the risk of ill health outcomes tends to grow as per-capita income decreases; it has been pointed out (OECD 2016) that assuming an income elasticity of WTP equal to 1 (or lower) may underestimate the WTP in the case of low-income countries. Robinson et al. (2019) underline how, in high-income countries, the income elasticity of WTP is generally lower than 1, while in low-income countries, such elasticity is greater than 1. In other words, in low-income countries, the WTP per unit of risk reduction is more than proportional to changes in income because low-income individuals must allocate a larger share of their income to necessity goods. For these reasons, it has been assumed  $\varepsilon = 0.8$  for Russia, Belorussia, Ukraine, and North African and Middle East countries, which are characterized by a lower level of income per capita than most European countries, but they are still classified in the upper-middle and lower-middle income groups by the World Bank. For the RoW aggregate—a very heterogeneous group in terms of income per capita— $\varepsilon$  was assumed equal to 1.

The external costs caused by climate change do not depend on where emissions occur and are calculated using the European Environmental Agency (EEA) recommended values (Holland et al. 2011) and following updates (Holland et al. 2014).

## LIFE CYCLE INVENTORY

This section describes how the life cycle stages have been modeled. We also specify how main involved processes are assigned to each geographical group (see Table 2).

### Vehicles, battery production, and end-of-life

Concerning the construction phase, there are 3 aspects that are relevant: the vehicle's entire life-cycle mileage, the type and amount of the material used to build the vehicle, and the place where the vehicle (and the battery in the case of the EV) is built.

The mileage traveled during the vehicle's lifetime plays an important role in the LCA of a vehicle because a relevant part of total emissions is due to vehicle materials and production (Hawkins 2013). Most studies in literature use a low and similar mileage for all vehicles, usually 150 000 km (Gao and Winfield 2012; MacPherson et al. 2012; Szczepowicz et al. 2012; Hawkins et al. 2013; Sharma et al. 2013). This is, nonetheless, a critical aspect because making the hypothesis of equal and (a little too) short life implicitly gives a disadvantage to those vehicles, electric ones, for which the construction phase impacts more than for other vehicles. In this paper we use, as far as possible, a "realistic" total mileage. The mileage used in this paper is shown in Table 3, and it depends on the fuel type used by the vehicle and on the market segment (Weymar and Finkbeiner 2016). Another important factor for the life cycle of EVs is the predicted battery lifespan. In literature, many different assumptions are made about the battery lifetime and the consequent number of battery replacements during the vehicle's lifetime (Aguirre et al. 2012). For example, Ecoinvent v3.1 considers, for a vehicle lifetime of 150 000 km, a battery lifetime of 100 000 km (Del Duce et al. 2016) which is far below the warranty of several manufacturers (100 000 miles or ~160 000 km). Moreover, these works fail to provide any scientific evidence that encourages an assumption that the battery lifetime is limited to 100 000 or 150 000 km. The limited number of updated studies on aging of Li-ion batteries seem to indicate that, at present, the EVs' battery end-of-life (i.e., when they have lost 20%–30% of their capacity) could be reasonably set to 200 000 km (Friesen et al. 2015). In addition, a recent behavioral study (Saxena 2015) shows that batteries continue to meet the driver's daily travel needs well beyond a capacity fade of 80% and that most drivers would not perceive if the battery capacity fade is 80%, 70%, or 60% of the original energy capacity. As a consequence, drivers would continue to use the vehicle even if the battery has conventionally reached its commercial end-of-life. For this reason, we have assumed that battery lifetime corresponds

to vehicle lifetime as suggested in Girardi and Brambilla (2017) without giving any environmental credit for eventual battery second life.

Concerning the energy for vehicle assembly and battery assembly, we used a German mix (from Ecoinvent) because e-Golf and its battery are assembled in Germany, while for cell production the South Korean mix has been taken into account (where Golf battery cells are produced). For materials (steel, glass, etc.), we use Ecoinvent European markets. Cars and battery assembly have been assigned to DE\_high group while materials supply has been assigned to EU\_unknown group.

### Vehicle use phase

Concerning the use phase, as stated above, the performances of the vehicles are evaluated on the basis of an urban driving cycle because, as far as EVs are characterized by almost zero tailpipe emissions, it is in the urban context that the benefits of electrification of transport would be most significant. The use phase is assumed to take place in an Italian urban area.

When possible, this study makes use of real-world data. Energy consumption in the vehicles' use phase is derived from US Environmental Protection Agency data in order to have a homogeneous, reliable, and impartial data source. Moreover, to our knowledge, there is no publicly available European database on car fuel consumption which allows distinguishing between urban and rural driving-cycle fuel consumption. Available European data refer still to New European Driving Cycle that underestimates energy consumption, at least for ICEVs (Fontaras et al. 2017).

For exhaust emissions from ICEVs, the main emission factors are those used for the Italian Emissions Inventory (ISPRA 2016). Regarding non-exhaust emissions (fine particulates emissions due to tires, brakes, and road abrasion), they have been modeled on the basis of vehicles' weight as suggested in literature (Simons 2016). With regard to EV brake-wear emissions, we consider only 20% of the amount calculated on weight basis, due to the use of regenerative braking (Del Duce et al. 2016). It is worth highlighting that the use phase includes maintenance that, of course, occurs where the vehicles are supposed to be used (Italy in the present case study). Emissions during the use phase are assigned to the group IT\_Low\_Urban, while emissions from maintenance are assigned to IT\_Low\_suburban. Vehicle and battery end-of-life processes are assigned to IT\_unknown\_rural.

### Well to tank: Energy vectors production and distribution

The production of the energy vector used by each vehicle (the so-called well to tank phase) has a relevant role in the vehicle LCA (Gao and Winfield 2012). With regard to EVs, the recharging mix was built as marginal mix, according to the hypothesis proposed in Girardi et al. (2015) for the Italian scenario. The marginal technologies (i.e., the power plants that go into operation to satisfy the electricity request for mobility) have been modeled modifying suitable

**Table 3.** Lifetime mileage for vehicles analyzed

Model	Fuel	Lifetime mileage (km)
e-Golf (Volkswagen) 2016	Electricity	230 000
Golf TSI 1 (Volkswagen), 8 L, 4 cylinder	Gasoline	210 000
Golf TDI 2 (Volkswagen), 0 L, 4 cylinder	Diesel	240 000

Ecoinvent datasets on the basis of the average Italian efficiency of each technology considered (e.g., average efficiency for combined cycle power plants, gas turbine power plants) according to the data on the Italian electric system published by National Transmission System Operator Terna (Terna 2014). The emission factors for the regulated pollutants emissions ( $\text{CO}_2$ ,  $\text{NO}_x$ ,  $\text{SO}_2$ , particulates) of the thermal power plants are derived from the annual declarations of the Italian Eco-Management and Audit Scheme registered power plants (Girardi and Brambilla 2017). Concerning the geographical reference, electricity production takes place in Italy (hence, it is assigned to IT\_High\_rural), except the electricity import that is located in Europe (assigned to EU\_high).

Concerning traditional vehicles, for the upstream of fossil fuels, the mix of crude oil import used refers to the actual mix as published by the Italian government (Ministry of Economic Development, Directorate General for Mineral and Energy Resources 2015; Ministry of Economic Development, Directorate General for Security, Energy Supply and Infrastructure 2016) as well as the natural gas used in the Italian power plants. Hence, specific Ecoinvent datasets have been used for each production area (instead of European market), and the related emissions have been assigned to corresponding group: RoW\_unknown, RU\_unknown, LY\_unknown, DZ\_unknown, NL\_unknown, NOS\_unknown, IT\_unknown, where RU = Russia, LY = Lybia, DZ = Algeria, NL = Netherlands, and NOS = North Sea. The refining and distribution of the fossil fuels take place in Italy and are assigned respectively to IT\_High\_rural and to IT\_Low\_suburban.

## RESULTS

The implementation of the methodology presented in this paper leads to the quantification of the external costs due to both air pollution and climate change for the 3 vehicles analyzed. Regarding the air pollution externalities, the EV has lower externalities for all the considered emissions except  $\text{SO}_2$ . This is due to relevant  $\text{SO}_2$  emissions associated with the construction (and the related energy consumption) of the EV and the production of electricity for battery recharging. It is worth underlining that  $\text{SO}_2$  emissions during the use phase are negligible for all the vehicles analyzed.  $\text{PM}_{2.5}$  emissions are comparable across vehicles and in the use phase because they are mainly due to abrasion of tires, brakes, and road conditions.  $\text{NO}_x$  externalities are by far lower for the EV than for gasoline and diesel vehicles. NMVOC emissions externalities are appreciable only for the gasoline vehicle. Considering all the air pollution externalities together, the EV performs better than the others because its air pollution externalities are about 7 €/1000 km, while for diesel and gasoline, they are near 15 €/1000 km and 10 €/1000 km, respectively (see Figure 1).

If we consider only the externalities due to pollutant emitted in Italy, where the vehicles are used in our scenario, the advantage of an EV is even higher (Figure 2) because the diesel vehicle air pollution externalities are more than twice those of the EV.

Considering the externalities due to climate change, for example, emissions of greenhouse gases, the EV is again the best performer, with an associated external cost of 4.9 €/1000 km, by far lower than a diesel vehicle

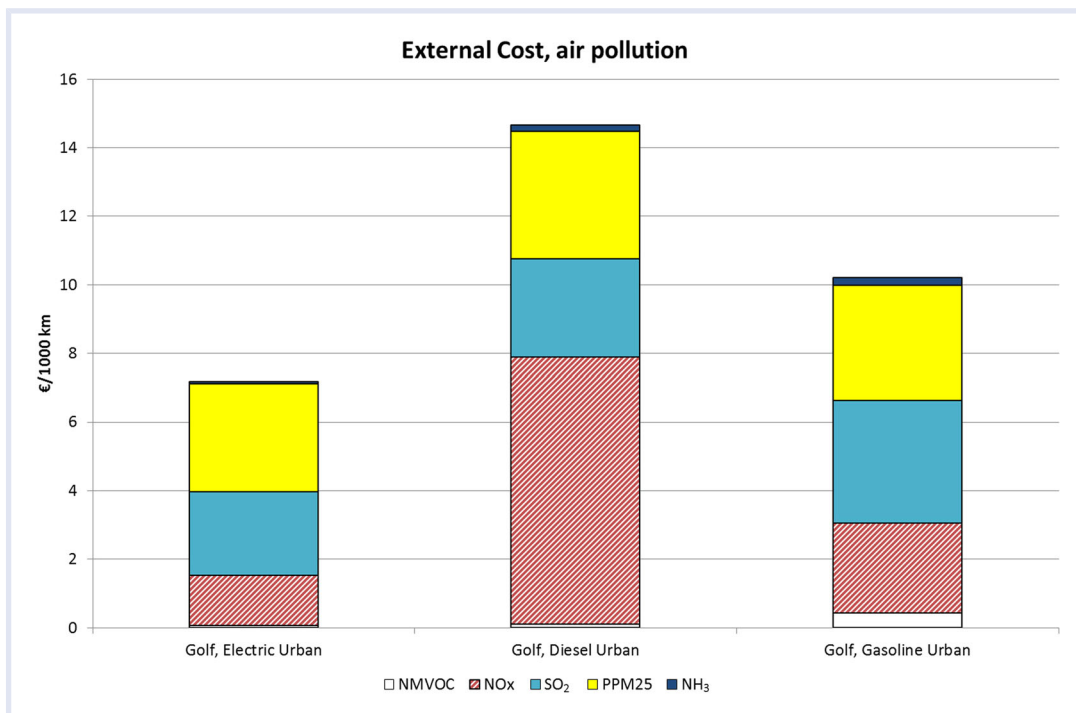


Figure 1. Air pollution external costs for the 3 vehicles, entire life cycle. NMVOC = nonmethane volatile organic compounds;  $\text{NO}_x$  = nitrogen oxides;  $\text{SO}_2$  = sulfur dioxide;  $\text{PPM}_{2.5}$  = fine inhalable particles with diameters 2.5  $\mu\text{m}$  and smaller;  $\text{NH}_3$  = ammonia.

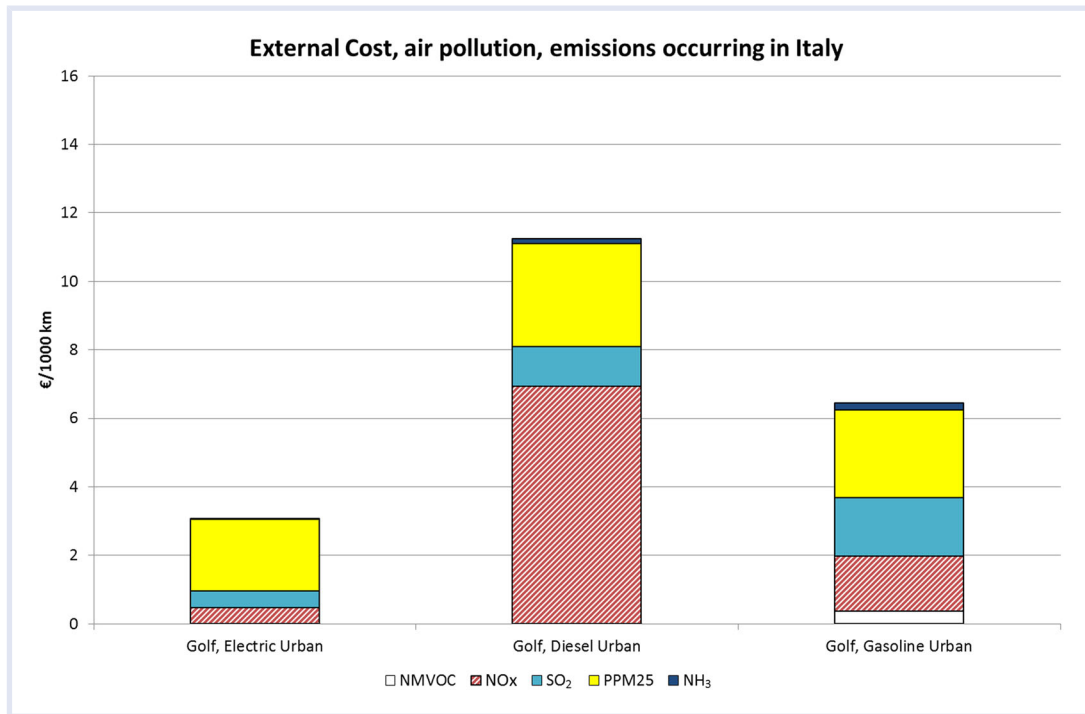


Figure 2. Air pollution external costs for the 3 vehicles, only pollutants emitted in Italy. NMVOC = nonmethane volatile organic compounds; NO<sub>x</sub> = nitrogen oxides; SO<sub>2</sub> = sulfur dioxide; PPM2.5 = fine inhalable particles with diameters 2.5 μm and smaller; NH<sub>3</sub> = ammonia.

(9.6 €/1000 km) or a gasoline vehicle (11.1 €/1000 km). The lower external costs of greenhouse gas emissions of the EV are due both to the higher efficiency of the energy pathway and to the higher penetration of natural gas and renewables in the energy mix. The diesel vehicle performs better than the gasoline vehicle due to higher efficiency. Considering all

the externalities together (climate change and air pollution), EVs perform better than conventional vehicles (Figure 3). Moreover, while for electric and diesel vehicles, about 60% of the total external costs are due to air pollution, for a gasoline vehicle, 50% is due to air pollution and 50% to climate change.

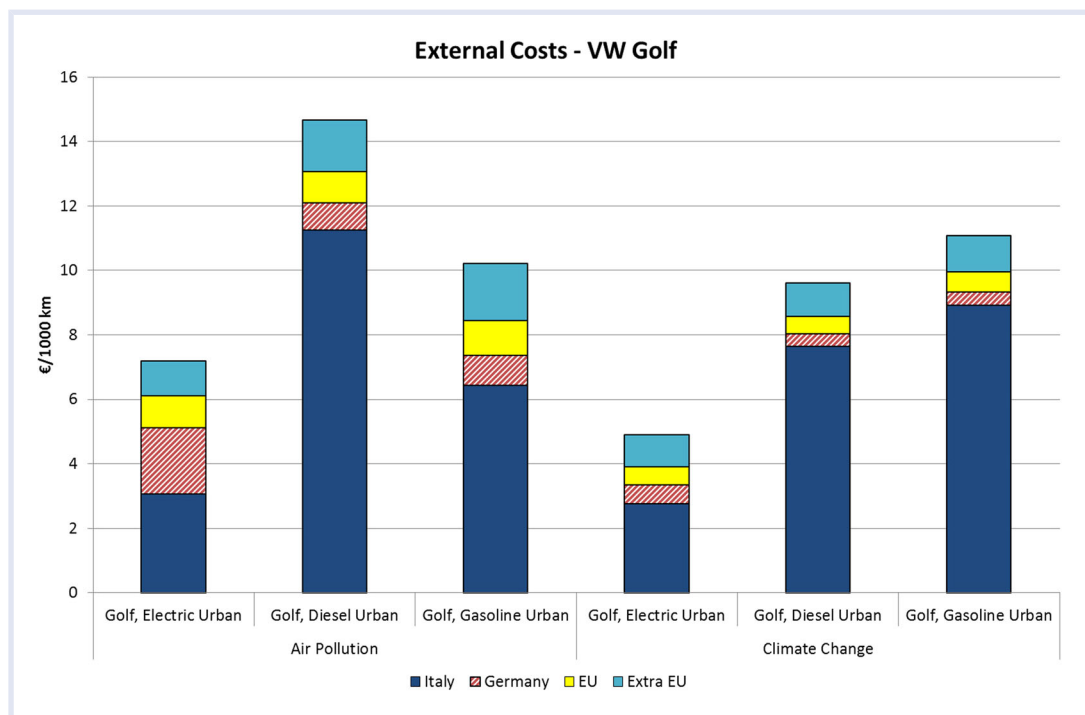


Figure 3. Overall external costs for the 3 vehicles, detail for geographic areas.



**Table 4.** Scenarios considered for sensitivity analysis and their effect on externalities expressed as €/1000 km

Vehicle, scenario	Air pollution	Climate change	Total
Golf-Electric, Urban	7.18	4.89	12.07
Golf-Diesel, Urban	14.66	9.61	24.27
Golf-Gasoline, Urban	10.21	11.08	21.30
Golf-Electric, Rural	5.66	5.24	10.89
Golf-Diesel, Rural	9.37	7.16	16.54
Golf-Gasoline, Rural	5.88	8.14	14.02
Golf-Electric PV, Daytime Urban	6.79	3.95	10.74
Golf-Electric CCNG, Urban	6.79	4.93	11.72
Golf-Electric, Urban (150 000 km <sup>a</sup> )	9.00	5.72	14.72
Golf-Diesel, Urban (150 000 km <sup>a</sup> )	15.96	10.38	26.33
Golf, Petrol Urban, (150 000 km <sup>a</sup> )	11.14	11.64	22.78

PV = photovoltaic; CCNG = combined cycle natural gas.

<sup>a</sup>Considering 150 000 km to be total life mileage for both vehicles and batteries.

## SENSITIVITY ANALYSIS

A first sensitivity analysis shows that also considering a less powerful gasoline car (2017 Golf, 1.4 cc version) or reducing the conventional vehicle (both gasoline and diesel) consumption and relative emissions reduces the gap but does not change the ranking among the 3 types of vehicles.

Moreover, other sensitivity analysis have been performed: using a rural driving cycle instead of an urban driving cycle; using an electric mix for recharging batteries based on

**Table 5.** Total life cycle air emissions external costs for electric, diesel, and gasoline Golf (Volkswagen) in urban driving cycle obtained from sensitivity analysis using VSL instead of VOLY

€/1000 km	Air pollution	Climate change	Total
Golf-Electric, Urban	30.61	4.89	35.50
Golf-Diesel, Urban	57.54	9.61	67.15
Golf-Gasoline, Urban	48.88	11.08	59.97

VSL = value of statistical life; VOLY = value of lost years.

**Table 6.** Damage factors used for sensitivity analysis on climate change external costs

€/1000 km	Holland et al. 2011, central	Holland et al. 2014, max	Holland et al. 2014, min	Korzhenyevych et al. 2014, max	Korzhenyevych et al. 2014, min	Korzhenyevych et al. 2014, central
Electric	5.2	5.9	1.5	34.7	9.9	18.6
Diesel	9.9	11.2	2.8	65.3	18.7	35.0
Gasoline	11.3	12.8	3.2	75.1	21.5	40.2

photovoltaic electricity during daytime; using an electric mix for recharging batteries based only on fossil fuel, and, in particular, on combined-cycle natural gas power plants; and considering 150 000 km total life mileage for both vehicles and batteries. Again, although affecting the gap of externalities among vehicles, none of the mentioned scenarios changes the ranking among the 3 kind of vehicles (see Table 4).

Another relevant hypothesis of the study is the use of value of lost years (VOLY) instead of value of statistical life (VSL) to monetize the air pollution effect on mortality. The use of VOLY is more correct; VSL is relevant for accidental deaths, and it is not appropriate for air pollution mortality (Bickel and Friedrich 2005). Nevertheless, for the sake of completeness, we also tested the use of VSL instead of VOLY. Because damage costs based on VSL are not available in the NEEDS project, we used the VSL damage costs suggested by the EEA (Holland et al. 2014), applying the value transfer methodology described above. Again, although externalities rise for all the vehicles considered, the relative ranking is not affected by the use of VSL instead of VOLY (Table 5).

Finally, sensitivity analysis has been carried out on the climate change damage factor (€/t CO<sub>2</sub>eq emitted). In addition to the central value suggested by the EEA (Holland et al. 2011), we used the minimum and maximum values (Holland et al. 2014). Moreover, we also used the CO<sub>2</sub>eq damage factor indicated by Korzhenevych et al. (2014) for 2025, discounted back to 2017 using a discount factor of 3% and converted into 2017 prices using the Eurozone GDP deflator. Resulting damage factors present a high range of variability, as shown in the sensitivity analysis represented in Table 6. The sensitivity analysis shows that the climate change external costs avoided due to an EV can increase up to 30 €/1000 km when compared with a diesel vehicle and up to 40 €/1000 km when compared with a gasoline vehicle.

## CONCLUSIONS

There are 2 main conclusions that we can draw from the present work. The first is methodological; we demonstrate that it is possible to carry out the application of damage factors to the physical inventory flows. The application of the air emissions external costs damage factors to LCI results gives further information to decision makers, in addition to those of standard ISO 14040 LCIA results. Applying appropriate damage factors to air pollutants, taking into account the place and the height of emissions, it is possible to obtain a detailed estimation of externalities. This result allows decision makers to manage 1 aspect, that is, to

understand where external costs occur and to compare them with economical performances of products and with potential economical tools for their promotion (e.g., taxes and subsidies). The second conclusion comes with the application of this methodology, namely, the case study of electric, gasoline, and diesel average private cars in the Italian scenario. To this end, a complete LCA of an electric, gasoline, and diesel Golf has been carried out, considering an urban driving cycle. Emissions of PM 2.5, NO<sub>x</sub>, SO<sub>2</sub>, NH<sub>3</sub> NMVOC, CO<sub>2</sub>eq have been taken into account for externalities evaluation.

The application of this methodology to the Italian context leads to the conclusion that if we compare the 3 types of vehicle—electric, diesel, and gasoline—of an average mid-size car (e.g., Golf), the electric version produces fewer external costs than the traditional internal combustion engine vehicles, considering both air pollution and climate change. At the same time, the EV produces less external cost related to air emissions considering both the entire lifecycle and only the processes that occur in Italy. The total life cycle air emissions externalities are 12.1 €/1000 km for the EV, 21.3 €/1000 km for the gasoline vehicle, and 24.3 €/1000 km for the diesel vehicle.

Of course, air emissions-related externalities are neither the only externalities nor the only environmental externalities. For example, typical externalities of private transport are accidents, or concerning environmental externalities, noise, which is different for an EV and a conventional vehicle. Nevertheless, the methodology and its application give decision makers useful and handy information on the main environmental issues of the use of private cars (i.e., climate change and air pollution).

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**Data Availability Statement**—Data associated with this research are available on request from the corresponding author, Pierpaolo Girardi, at pierpaolo.girardi@rse-web.it.

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