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LIFE CYCLE EXTERNALITIES VERSUS EXTERNAL COSTS:
THE CASE OF INLAND FREIGHT TRANSPORT IN BELGIUM

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ABSTRACT :

This paper proposes a case study on Belgium in which externalities and external costs of inland freight transport modes in Belgium are compared for the year 2012. The well-known Life Cycle Assessment methodology is used to identify the updated specific Belgian externalities related to three categories of negative impacts: climate change, photochemical ozone formation and particulate matter formation. The obtained values of externalities are then compared to the related external cost values. The objective is to determine if these two tools can be used interchangeably. We find that road transport has the maximum impact for every environmental impact indicator, with rail freight transport presenting the minimum one. We identify that each category of negative impact on the environment does not represent the same percentage of global externalities and external costs. In the analysis of the environmental impact per mode, it is observed that, when external costs are considered instead of externalities, the impact of road transport is slightly increased compared to both impacts of rail and inland waterways. Using externalities and average external costs interchangeably for estimating the impact of specific transport categories on the environment may thus lead to different results and different related policies.

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1. Introduction

Transportation contributes to the economic development of societies, but is also responsible for negative impacts on climate, natural environment and human health. These negative impacts can occur at a local level like air pollution or at a global level like climate change and they can be assessed through the determination of the values of externalities or of external costs.

The objective of this paper is to provide updated values of externalities and external costs for the case study of Belgium in 2012. With the liberalization of the Belgian rail freight market, data on energy consumption has become a competitive advantage and is therefore more difficult to obtain. Although there is available data until 2014 for inland waterways (IWW) and road transport, the most recent values on energy consumption for rail freight transport in Belgium are from 2012. For the sake of coherence, our analysis is consequently based on data from 2012. A comparison is done on three categories of negative impacts related to atmospheric change caused by human-beings: climate change, photochemical ozone formation and particulate matter formation. The focus on these three issues has been voluntary done to limit the size of the study. Moreover, the Life Cycle Assessment (LCA) methodology has been recognized for the evaluation of these three externalities, whereas it is not still the case regarding other external effects like for instance accidents damage, congestion or noise (Fries and Hellweg, 2014; Meyer et al., 2017). Hence, the environmental impact indicators required to determine these impacts still need to be amended from a methodological point of view. Therefore, these environmental impacts are beyond the scope of this paper.

LCA allows a holistic view of externalities whereas external costs provide an economic estimation of the negative effects of transport. The comparison of externalities and external costs values is interesting to identify how uncertainties in methodological issues (e.g. monetization of externalities) may lead to divergent policy decisions depending on whether externalities or external costs are considered to evaluate the negative impact of transport. If externalities and external costs do not provide similar results, the use of one or another tool could lead to divergent policy actions (for instance taking measures at a global level for climate change, instead of focusing on more local measures for air pollution).

The climate change issue is considered since it has become an important societal topic in the recent years. The interest for climate change is in particular illustrated by the organization of international conventions and events, such as the Kyoto Protocol or the regular United Nations Climate Change Conferences. The Paris agreement which entered into force in November 2016 is a further illustration of the willingness to limit climate change effect with the objective of "keeping a global temperature rise this century well below 2 degrees" (UNFCCC, 2016).

Climate change is produced by the emission of greenhouse gases (GHG) such as carbon dioxide (CO_2), methane (CH_4) or nitrous oxide (N_2O) which have the capacity to absorb infrared radiation, increasing the radiative forcing in the atmosphere and therefore causing the increase of the global temperature of the planet. As shown from 2012 data in Table 1, the transport sector is responsible for 24.2% in the European Union (EU-28) and 34.3% in Belgium of the total GHG emissions (excluding emissions or removals from land use, land use change and forestry (LULUCF)) (Eurostat Statistics,

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2017). If international aviation and navigation are excluded, these percentages fall to 19.5% and 21%, respectively. In Belgium, this difference is mainly explained by the international navigation which contributed 13.87% of total GHG emissions due to the presence of the Port of Antwerp: the second port in Europe in total maritime freight volume. Within transport sector, road transport represents 72% in the EU-28 and 50% in Belgium while if international bunkers are excluded, road transport produces most of the GHG emissions, accounting for 95% in the EU-28 and 97% in Belgium.

Particulate matter and ozone exposure are nowadays considered as major environmental health problems in most cities. In the EU-28, 391,000 premature deaths in 2015 were caused by the long-term exposure to $PM_{2.5}$ (particulate matter of a diameter of 2.5 µm or less), 76,000 due to NO_2 (nitrogen dioxide) and 16,400 due to tropospheric ozone. In Belgium, the premature deaths in 2015 attributed to $PM_{2.5}$, NO_2 and tropospheric ozone are 7400, 1500, and 220, respectively (European Environment Agency, 2018). Transport represents an important source of air pollution, especially for particulate matter and nitrogen oxides (NO_X). Particulate matter can be emitted directly from vehicles (primary particulate matter) or be formed in the atmosphere from precursor pollutants such as sulphur oxides (SO_X), NO_X, ammonia (NH_3) or Volatile Organic Compounds (VOC). The tropospheric ozone is formed from other precursor pollutants such as NO_X and Non-Methane Volatile Organic Compounds (NMVOC) by photochemical reaction under the influence of solar radiation. It should be noted that stratospheric ozone is beneficial to the environment because it filters ultraviolet radiation from the sun.

As shown from 2012 data in Table 2, road transport was the main source of NO_X emissions, representing 38.13% and 48.33% of the total emissions in the EU-28 and Belgium, respectively. Moreover, transport was a major source of NMVOC with 12.02% in the EU-28 and 7.89% in Belgium of the total emissions. For primary $PM_{2.5}$, transport constitutes 14.27% in the EU-28 and 18.22% in Belgium of the total emissions. For particles with a diameter of 10 µm or less (PM_{10}), it accounts for 12.86% in the EU-28 and 17.65% and for SO_X, it constitutes 2.01% and 1.95% of the total emissions in the EU-28 and Belgium, respectively. The non-road transport emissions of SO_X are bigger than road transport emissions in both EU-28 and Belgium, as a result of the highest sulphur content in the gas-oil used in navigation. In Belgium, road transport was responsible for 17% of the total emissions of carbon monoxide (CO) in 2012, accounting for 1% the other modes of transport (European Environment Agency, 2014). The mentioned pollutants are produced during fuel combustion, but other non-exhaust emissions of particulate matter, including $PM_{2.5}$ and PM_{10}, are emitted from the wear of brakes, tyres and road surface in road transport and the abrasion of brakes, wheels and rails in rail transport. The air pollutant emissions from road transport have decreased over the years as a result of the implement of the "Euro" emission standards defined in a series of European Union directives staging the progressive introduction of increasingly stringent standards.

In this paper, we compute the updated externalities of transport for Belgium using the Life Cycle Assessment (LCA) methodology and convert them in the related external costs based on unit external cost values recognized in the Update of the Handbook on External Costs of Transport (Ricardo-AEA, 2014). We compare both externalities and external costs to identify if they provide similar results in terms of policy decision-making. Belgium is an interesting case to study since the

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country is one of the least performant European countries in terms of air quality (European Commission, 2015). Belgium is also chosen for its significant freight flows on road, rail and IWW networks and the current lack of specific Belgian data on freight transportation (Troch et al., 2015). The contribution of the paper consists in the collection and generation of updated data regarding Belgian transport and its related externalities and external costs. A comparison of the use of externalities and external costs in terms of policy effect is also provided.

The paper is structured as follows: we start by reviewing the relevant literature on externalities assessment through LCA and on external costs of freight transport. The methodologies used for computing the externalities and external costs values on the Belgian case study are explained in Section 3. Results are then provided and discussed in Sections 4 and 5. Conclusions are drawn in the last section.

***Table 1**. Greenhouse gas emissions (excluding LULUCF) in the EU-28 and Belgium in the year 2012. Source: Eurostat Statistics (2017).*

	Including international bunkers		Excluding international bunkers	
	EU-28	Belgium	EU-28	Belgium
Non-transport sectors	75.77%	65.73%	80.46%	78.95%
International aviation	2.77%	2.88%		
International navigation	3.06%	13.87%		
Cars	10.51%	10.08%	11.16%	12.11%
Light duty trucks	2.12%	1.99%	2.25%	2.39%
Heavy duty trucks and buses	4.55%	4.81%	4.83%	5.78%
Motorcycles	0.22%	0.12%	0.23%	0.14%
Other road transport	0.01%		0.01%	
Railways	0.15%	0.09%	0.16%	0.10%
Domestic aviation	0.34%	0.01%	0.36%	0.01%
Domestic navigation	0.37%	0.29%	0.39%	0.35%
Other transport	0.13%	0.13%	0.14%	0.16%

***Table 2**. Air pollution in the EU-28 and Belgium in the year 2012. Source: Eurostat Statistics (2017).*

		NOx	NMVOC	$PM_{2.5}$	PM_{10}	SOx	NH_3
EU-28	Non-transport sectors	54.60%	87.98%	85.73%	87.14%	97.99%	98.28%
	Road transport	38.13%	10.88%	12.29%	11.26%	0.14%	1.71%
	Non-road transport	7.27%	1.13%	1.98%	1.60%	1.87%	0.01%
Belgium	Non-transport sectors	46.11%	92.11%	81.78%	82.35%	98.05%	98.59%
	Road transport	48.33%	7.34%	16.22%	15.81%	0.25%	1.41%
	Non-road transport	5.55%	0.55%	1.99%	1.84%	1.70%	0.00%

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2. Literature review

The LCA methodology constitutes an effective tool to quantitatively analyse and compare the environmental impacts of inland freight transport. The system perspective of the LCA methodology implies the need to analyse not only the direct processes related to the transport activity such as energy consumption and exhaust emissions, but also the processes connected with the electricity and fuel production, vehicles and infrastructure. Several studies have already applied the LCA methodology to different modes of inland freight transport (Spielmann and Scholz, 2005; Facanha and Horvath, 2006, 2007; Chester and Horvath, 2009, 2010; Spielmann et al., 2007; Fries and Hellweg, 2014; Jones et al., 2017). LCA and life cycle costs methods have also been applied to passenger transportation, e.g. for rail passenger transport on the Turkish case study (Banar and Özdemir, 2015). Van Lier and Macharis (2014) perform the LCA of IWW transport in Belgium. However, as highlighted in Caris et al. (2014), a detailed analysis on a Belgian level to compare the inland transport modes is lacking to date.

External costs are the monetary valuation of externalities. They are side effects of transportation. According to Maibach et al. (2008a, 2008b), "they are costs to society and – without policy intervention – they are not taken into account by the transport users". They allow comparing different kinds of externalities in the same unit. Their values enable the objective of the European Union which is to integrate external costs in transport pricing policies (European Commission, 2018). Internalizing external costs allows pricing at the right social cost, leading to an efficient allocation of resources.

However, the values rang up to several orders of magnitude, e.g. the economic valuation of GHG emissions vary up to six orders of magnitude (Nocera et al., 2015). According to Nocera et al. (2018), the uncertainties that explain this range can be classified into three main macrocategories: (1) technical related to the methodology required to quantify the amount of GHGs emitted, or expected to be emitted; (2) economic related to the coexistence of adaptation and mitigation approaches and to the conversion of the environmental and social impacts into economic ones; and (3) decisional related to the decision-making process and to the relationship between the various decision makers.

External costs have been widely studied in the transport literature, both from a scientific and from a project-based perspective (Mostert and Limbourg, 2016). Even if some studies concentrate more on internalization policies (Beuthe et al., 2002; Macharis et al., 2010; Moliner et al., 2013; Agarwal et al., 2015; Austin, 2015; Dente and Tavasszy, 2018) or on optimization objectives related to externalities (Musso and Rothengatter, 2013; Zhang, 2013; Zhang et al., 2015; Santos et al. 2015; Mostert et al., 2017a, 2017b), most of research related to external costs consists in applying the methodological valuation tools for determining specific numerical values of external costs (Forkenbrock, 1999, 2001; Sansom et al., 2001; Mayeres et al., 2001; RECORDIT by Schmid et al., 2001; Beuthe et al., 2002; INFRAS/IWW, 2004; CAFÉ by AEA Technology Environment, 2005; Bickel et al., 2006a, 2006b, 2006c; Maibach et al., 2008a, 2008b; Macharis et al.,2010; Janic and Vleugel 2012; Cravioto et al., 2013; Pérez-Martínez and Vassallo-Magro, 2013; Ricardo-AEA, 2014; Agarwal et al., 2015; Austin, 2015; De Langhe, 2017).

Our paper relates to the latter category, by proposing an updated version of the externalities and external costs of transport related to climate change, photochemical oxidant formation and particulate matter formation for the case study of Belgium.

3. Methodology

3.1. EVALUATION OF EXTERNALITIES: LIFE CYCLE ASSESSMENT

The environmental impact of freight transport is determined using the LCA methodology, which is a structured and comprehensive methodology standardised by ISO Standards 14040:2006 and 14044:2006 (ISO, 2006a, 2006b) at the international level. Moreover, at the European level, the ILCD Handbook (European Commission, 2010) is a reference to perform a LCA.

The LCA methodology allows the analysis of complex systems like freight transport, providing a system perspective analysis that enables the study of environmental impacts through all the stages of the transport system (transport operation, vehicles and infrastructure) including their related supply chains, from raw material extraction, through production and use, and finally disposal. Furthermore, LCA methodology allows quantifying all relevant emissions and consumptions, as well as the related environmental and health impacts and resource depletion issues that are associated with transport. This analysis gives us clues about how to improve the environmental performance of transport. A LCA study comprises four phases: goal and scope definition, inventory analysis, impact assessment and interpretation.

The first stage of a LCA is the goal and scope definition, which in this paper is twofold: to compare the environmental impacts of the different inland freight transport modes in Belgium, and to analyse the environmental impacts of the modal splits obtained for the Belgian case. Moreover, an important element to be defined is the functional unit, which is the reference unit to which the material and energy flows included in the life cycle processes are referenced. The functional unit chosen in our study is "one tonne-kilometre (tkm) of freight transported in the different modes of inland freight transport". The second stage of a LCA is the Life Cycle Inventory (LCI) analysis, which consists in the collection and compilation of data on the processes included in the scope of the study. In our case, we have collected data for the inland freight transport system from the main rail freight stakeholder in Belgium (i.e. Infrabel and LINEAS), from the Ecoinvent v3 database, and from literature sources. Road and railway infrastructure are shared between passenger and freight transport, therefore the environmental impacts related to the construction, maintenance and disposal of the infrastructure must be allocated proportionally by the use of passenger and freight transport.

The third stage of a LCA is the Life Cycle Impact Assessment (LCIA), where the information collected in the LCI is translated into environmental impacts through the use of science-based models. A selection of the impact categories covering the most relevant environmental issues for the analysed system has to be carried out. An impact category includes impact category indicators (quantifiable representation of an impact category) and characterisation models, which convert an assigned LCI

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result to the common unit of the impact category indicator through the use of characterisation factors (Hauschild and Huijbregts, 2015; ISO, 2006a, 2006b). The impact category indicators can be midpoint or endpoint depending on the level of the impact pathway at which they are placed. The impact pathway is the cause-effect chain of an environmental mechanism, which is the system of physical, chemical and biological processes for a given impact category, linking the LCI results to category indicators and to category endpoints (Rosenbaum et al., 2018a; ISO, 2006a, 2006b). Therefore, the midpoint impact categories are placed in a point of the impact pathway between the LCI flow and the endpoint categories. Overall, the elementary flows from the LCI (e.g. pollutants or resources and energy flows) can be characterised using specifics characterisation factors (which belong to a characterisation model) and therefore be converted into the common unit of midpoint impact category indicators related to midpoint impact categories (e.g. radiative forcing as Global Warming Potential (kg CO_2 equivalents) for climate change, tropospheric ozone concentration increase (kg NMVOC equivalents) for photochemical ozone formation and intake fraction for particles (kg PM_{10} equivalents) for particulate matter formation). The midpoint category indicators can be multiplied by damage factors and aggregated into endpoint impact category indicators (e.g. damage to human health, damage to ecosystem diversity and resource scarcity).

For this research study, all calculations for the LCIA were made with the SimaPro 8.0.5 software using the LCIA method ReCiPe 2008, hierarchist version (version V1.12 / Europe). ReCiPe 2008 is a LCIA method including 18 midpoint impact categories with their respective characterisation models and factors and midpoint impact category indicators. Moreover, most of these midpoint impact indicators can be multiplied by damage factors and aggregated into three endpoint impact categories. Therefore, through the application of ReCiPe 2008, the resources consumed and contribution of the pollutants emitted by freight transport and determined in the LCI can be analysed using midpoint impact categories such as climate change, photochemical oxidant formation or particulate matter formation. Then, the influence of most of these midpoint impact categories can be evaluated (i.e. all except marine eutrophication and water depletion due to methodological limitations in ReCiPe 2008) in terms of endpoint impact categories such as damage to human health, damage to ecosystem diversity and damage to resource availability, which are related to the areas of protection of human health, natural environment and natural resources, respectively (Goedkoop et al., 2013). Fig. 1 represents the relations between the LCI, the 18 midpoint impact categories and the three endpoint impact categories of the LCIA method ReCiPe 2008. The impact category climate change influences both areas of protection human health and natural environment.

LCA studies applied to transport mainly focused on air emissions, especially CO_2, CO, NO_X, SO_2, NMVOC and particulate matter (Spielmann and Scholz, 2005; Facanha and Horvath, 2006, 2007; Chester and Horvath, 2009, 2010; Van Lier and Macharis, 2014; Jones et al., 2017). Therefore, we have considered that the following midpoint environmental impact categories cover the most relevant environmental problems on freight transport: climate change (kg CO_2 eq.), photochemical oxidant formation (kg NMVOC eq.) and particulate matter formation (kg PM_{10} eq.). The inventory emissions of NO_X emissions to air have been included in our study due to its importance as precursor for tropospheric ozone formation and particulate matter.

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Moreover, the endpoint categories have been analysed as well. ReCiPe 2008 assesses damage to human health using the indicator disability-adjusted loss of life years (DALY), which encompass the number of years of life lost and the number of years of life disable. The damage to ecosystem diversity is assessed using the indicator loss of species in a certain area during a year (species × year). The damage to resource availability is assessed using the indicator increased cost, which is expressed in a monetary unit ($) and it is based on the surplus costs of future resource production in the future (Goedkoop et al., 2013). Please note that comparing endpoint categories leads to a greater uncertainty than comparing midpoint categories, due to a more complete modelling of impact pathways (Kägi et al., 2016). Therefore, these results on endpoint damages should be interpreted with caution because of the uncertainty related to the methodology. Finally, the fourth stage of a LCA is the interpretation of the results obtained in the LCI and the LCIA. The interpretation can be accompanied by sensitivity and uncertainty analysis to make the conclusions of the study more robust (Hauschild, 2018).

Fig. 2 shows the system boundaries considered in our study for LCA of rail, IWW and road freight transport. Looking at the year 2012, road transport was the dominant mode of the three major inland freight transport modes in Belgium, accounting for 64.5% of the total inland freight expressed in tkm. Rail transport was responsible for 14.6% and IWW accounted for 20.9% of the total inland freight transported in Belgium (Eurostat Statistics, 2017).

Figure 1. *Diagram of the Life Cycle Impact Assessment method ReCiPe 2008 applied on inland freight transport. Source: Adapted from Goedkoop et al. (2013).*

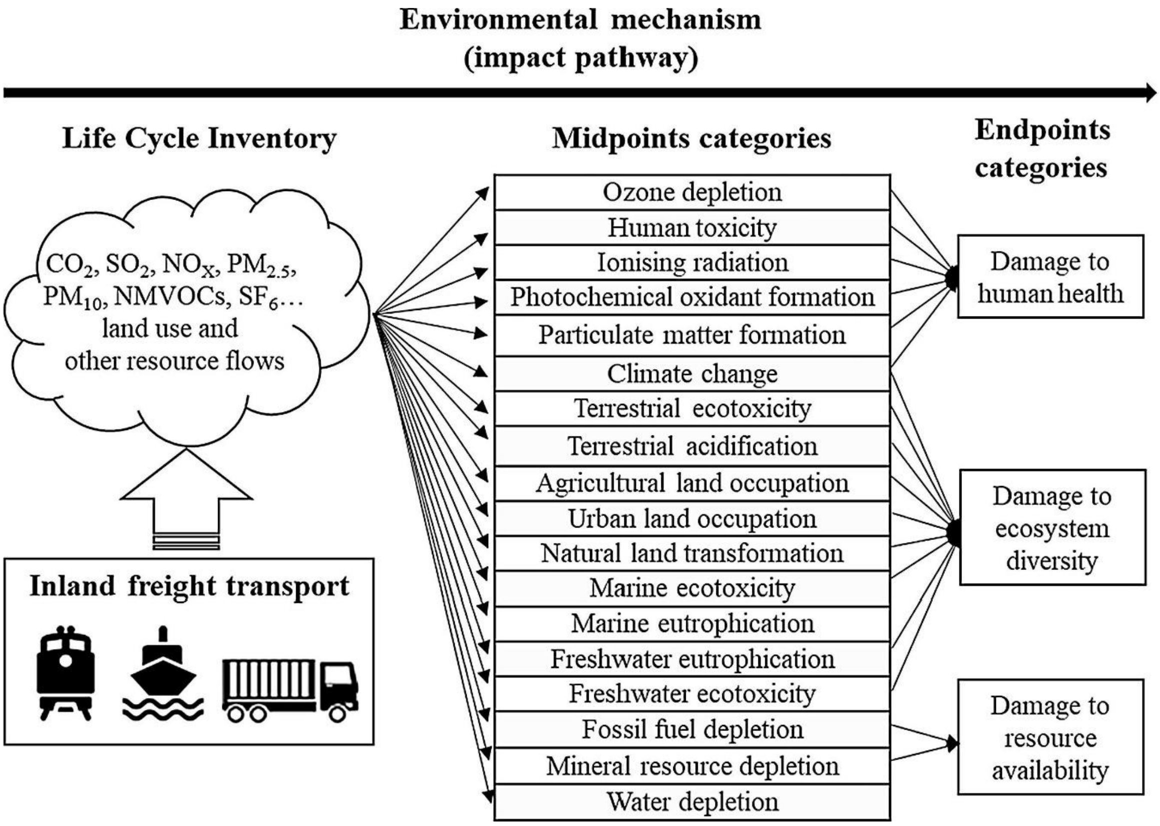

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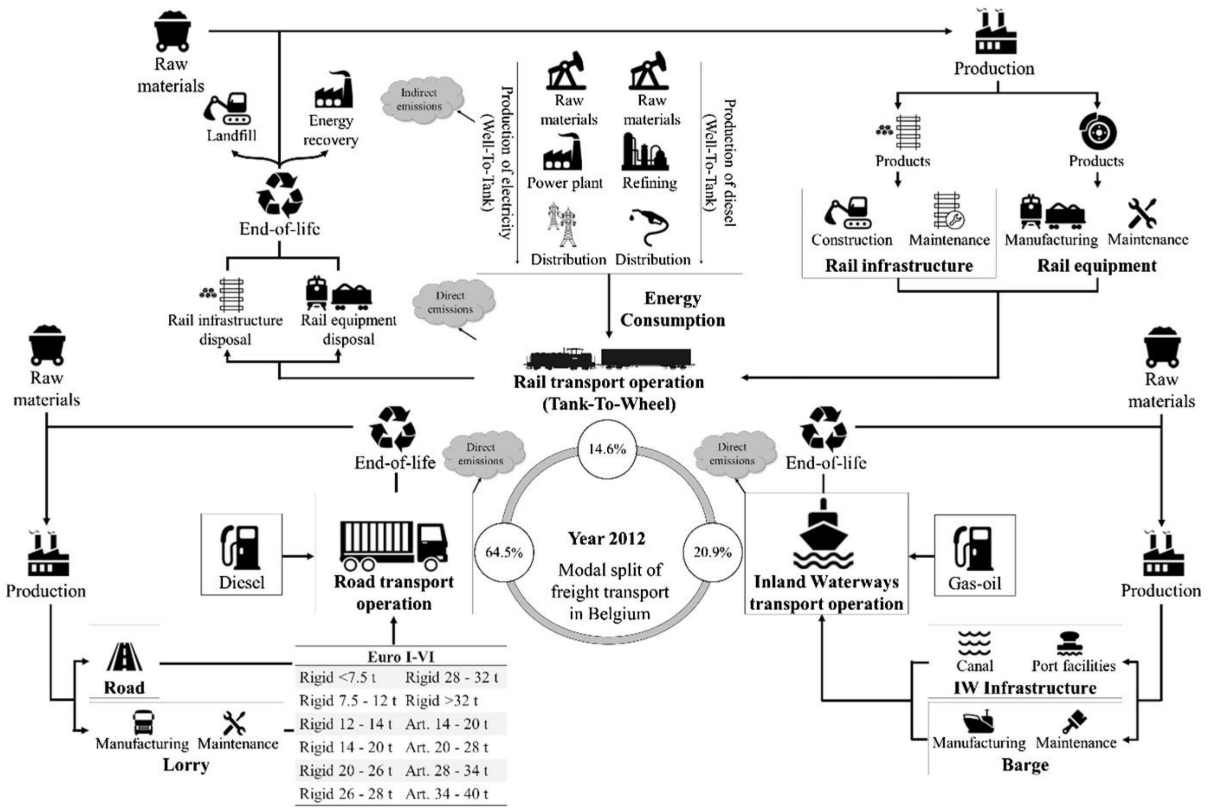
Figure 2. *Boundaries considered in the Life Cycle Assessment of the inland freight transport system. Source: Merchan et al. (2017).*



3.1.1. RAIL FREIGHT TRANSPORT

We have collected specific information related to the rail freight transport system in Belgium from both literature sources and directly through the use of questionnaires from the Belgian railway infrastructure manager (Infrabel) and the main rail freight operator in Belgium (LINEAS). The rail freight transport system has been divided in three sub-systems: rail transport operation, rail infrastructure and rail equipment (i.e. locomotives and wagons). The sub-system rail transport operation includes the processes that are directly connected with the train activity. In diesel trains, exhaust emissions to air from diesel locomotives and the indirect emissions from diesel production are taken into account. In electric trains, both the sulphur hexafluoride (SF_6) emitted during conversion at traction substations related to electricity consumption and the indirect emissions from electricity generation are considered. Moreover, rail transport operation includes the direct emissions to soil from abrasion of brake linings, wheels, rails and overhead contact lines in both types of traction. Furthermore, the LCA approach followed in our study allows the analysis of the environmental impacts related to the construction or manufacturing, maintenance and disposal of rail infrastructure and vehicles (Spielmann et al., 2007).

The electricity supply mix used by electric trains plays an important role in their environmental impact. Thereby, depending on the energy split of the country (i.e. the share of nuclear or natural gas power for example), the environmental impacts of the electric trains varies. Therefore, the electricity supply mix used for the electric trains in Belgium for the year 2012 has been determined

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according to Eurostat data (Eurostat Statistics, 2017). The electricity supply mix has been calculated using the domestic production of Belgium and exports and imports of electricity from France, The Netherlands and Luxembourg. Analogously to Belgium, the electricity imports from France, The Netherlands and Luxembourg have been modelled considering the supply mix of the exporting countries. The processes related to electricity supply mix from Ecoinvent v3 database (Weidema et al., 2013) have been taken as a model to translate the data from Eurostat on energy sources to the technologies used in the electricity production in our study. Table 3 presents the electricity supply mix used in our study for electric trains and transhipment processes.

The energy consumption of electric and diesel trains calculated for the year 2012 in Belgium was 427 kJ/tkm and 650 kJ/tkm, respectively (Merchan et al., 2017). However, in this study we have used the energy consumption of the Belgian traction mix in 2012, which includes an 86.3% of electric trains and 13.7% of diesel trains. Thus, 368 kJ of electricity and 89 kJ of diesel (including shunting activity) were needed per tkm in Belgium. These values of energy consumption represent the final energy consumption during transport operation (Merchan et al., 2017). The direct emissions produced during the rail transport operation have been calculated using the emission factors of Spielmann et al. (2007), which do a selection of emission factors from others authors.

Table 3*. Electricity supply mix of Belgium, France, The Netherlands and Luxembourg in the year 2012. Sources: Eurostat Statistics (2017) and Weidema et al. (2013).*

Energy source (%)	Belgium	France	The Netherlands	Luxembourg
Nuclear, pressure water reactor	41.88	75.42	3.24	
Natural gas	22.18	2.66	38.58	11.38
Hard coal	4.99	2.84	14.19	
Oil	0.37	0.71	1.26	
Hydro, pumped storage	1.41	0.90		5.21
Hydro, run-of-river	1.79	9.72	0.09	5.65
Hydro, reservoir, alpine region		1.86		
Wind, < 1 MW turbine, onshore	0.09	0.21	1.10	0.09
Wind, > 3 MW turbine, onshore	0.29	0.05	0.14	
Wind, 1–3 MW turbine, offshore	0.09		0.46	
Wind, 1–3 MW turbine, onshore	2.47	2.35	1.81	0.29
Co-generation, biogas	0.43	0.08	0.71	0.21
Co-generation, wood chips	2.24	0.18	1.58	
Treatment of blast furnace gas	1.45	0.36	1.87	
Treatment of coal gas	0.06	0.11	0.19	
Import from France	8.96		0.00	0.00
Import from Luxembourg	1.67			
Import from The Netherlands	9.63			
Import from Belgium		0.49	4.00	11.91
Import from Germany		0.22	24.39	65.26
Import from other countries		1.84	6.38	

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Table 4. *Specific fuel consumption (g/tkm) of a truck depending of the load factor (LF).*

Heavy Duty Truck	Fuel Consumption (g/km)		
	LF 0% (Empty truck)	LF 100% (Full truck)	LF 50%
Rigid < 7.5 t	110	119	114
Rigid 7.5–12 t	143	166	154
Rigid 12–14 t	164	199	181
Rigid 14–20 t	164	199	181
Rigid 20–26 t	182	248	215
Rigid 26–28 t	192	311	251
Rigid 28–32 t	192	311	251
Rigid > 32 t	192	311	251
Art. 14–20 t	164	199	181
Art. 20–28 t	182	248	215
Art. 28–34 t	192	311	251
Art. 34–40 t	192	311	251

3.1.2. ROAD TRANSPORT

Similarly to the rail transport system, the road transport system comprises the sub-systems road transport operation, road infrastructure and truck. In order to consider the influence of the load weight in the specific fuel consumption of the truck, the average fuel consumption during the road freight transport activity has been determined in two stages. First, we have determined the fuel consumption per kilometre of the vehicle (g/km) using the fuel consumption and methodology from EcoTransIT (2014). We have translated the fuel consumption factors of EcoTransIT (2014) from five truck gross vehicle weight (GVW) categories to the twelve truck GVW categories used in our study. Table 4 presents the specific fuel consumption (g/km) calculated for a truck with a load factor of 50%. The choice of a load factor of 50% is because this is the load factor of an average cargo in road transport including empty trips (EcoTransIT, 2008).

In a second stage, the energy consumption has been converted from g/km to g/tkm dividing by the actual payload of each GVW class. The actual payload of each truck GVW class has been calculated by multiplying the maximum payload by a load factor of 50% (see Table 5). It should be noted that the fuel consumption in g/km increases with the size of the truck (see Table 4), but in g/tkm decreases (see Table 5) due to increased payload capacity with the GVW category. Finally, in order to calculate the average fuel consumption of road freight transport for the year 2012, the tkm moved by each truck GVW category have been used to calculate a weighted arithmetic mean. The annual transport performance (i.e. tkm) has been calculated multiplying the vehicle-kilometres (i.e. vkm) by the actual payload of each GVW class. The annual vehicle-kilometres have been determined using the population of heavy duty trucks and the mileage (km/year) of road freight transport in Belgium of each GVW class. The values of population of heavy duty trucks and the mileage (km/year) of road freight transport in Belgium considering the lorry GVW category used in our study are from COPERT database.

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Table 5 shows the distribution by tkm of road freight transport in Belgium by GVW category in 2012. The lorry GVW category articulated of 34–40 t represents approximately 75% of the road freight transport performance in Belgium in 2012. The average diesel consumption calculated for road freight transport with a load factor of 50% in the year 2012 was 23.22 g/tkm or 994 kJ/tkm (considering that diesel net calories are 42.8 MJ/kg).

The exhaust emissions produced during the road transport operation have been determined using the calculated diesel consumption and the emission factors from two sources. For fuel dependent emissions such as CO_2 and heavy metals, the emission factors of Spielmann et al. (2007) have been used. For other pollutant emissions dependent on the engine emission technology (CO, NMVOC, NO_X, N_2O, NH_3 and $PM_{2.5}$ for example), the tier 2 emission factors from EMEP/EEA air pollutant emission inventory guidebook 2013 (Ntziachristos and Samaras, 2014) have been used (Merchan et al., 2017). In the year 2012, the trucks with an emission engine technology conventional represented the 4% of the Belgian heavy duty market, the Euro I a 7%, the Euro II a 19%, the Euro III a 26%, the Euro IV a 21% and the Euro V a 22%. Since the emissions related to the engine technology are dependent on the truck GVW category as well, 72 different types of trucks have been considered in the year 2012 (twelve truck GVW categories split in six emission engine technologies). In order to determine an average emission, following the same methodology as for energy consumption, the tkm moved by each truck GVW category and emission engine technology have been used to calculate a weighted arithmetic mean.

Table 5*. Maximum payload, actual payload and average fuel consumption of road transport using a load factor of 50% in Belgium in the year 2012.*

Heavy Duty Truck	Maximum Payload (t/vehicle)	Actual Payload LF 50% (t/vehicle)	Fuel consumption LF 50% (g/tkm)	Share of freight transport performance 2012 (% of tkm)	Contribution to average fuel consumption in 2012 (g/tkm)
Rigid < 7.5t	2	1.01	113	0.65%	0.73
Rigid 7.5–12 t	5	2.51	61	1.84%	1.13
Rigid 12–14 t	7	3.51	52	0.24%	0.12
Rigid 14–20 t	9.7	4.85	37	3.26%	1.22
Rigid 20–26 t	13.7	6.85	31	4.95%	1.55
Rigid 26–28 t	16.4	8.19	31	0.03%	0.01
Rigid 28–32 t	18.4	9.19	27	1.21%	0.33
Rigid > 32 t	19.7	9.86	25	11.58%	2.95
Art. 14–20 t	12.6	6.32	29	0.27%	0.08
Art. 20–28 t	17.1	8.54	25	0.20%	0.05
Art. 28–34 t	21.5	10.76	23	0.47%	0.11
Art. 34–40 t	25.3	12.66	20	75.31%	14.94
TOTAL	–	–	–	100%	23.22

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3.1.3. INLAND WATERWAYS TRANSPORT

In the same way, the IWW system consists of three sub-systems: IWW transport operation, IWW infrastructure, and barge. For the IWW infrastructure, the Port of Antwerp and the Belgian IWW have been used as references. The average fuel consumption of IWW transport calculated for the year 2012 was 288 kJ/tkm (Merchan et al., 2017). The exhaust emissions produced during the IWW transport operation have been calculated using the emission factors of Spielmann et al. (2007) and the calculated fuel consumption. The Directive 2009/30/EC established a sulphur content of gas-oil used by barges of 10 ppm in the year 2012, being the same sulphur content than conventional road-transport diesel, which is used by both rail and road transport.

3.2. EVALUATION OF EXTERNAL COSTS: MONETIZATION

Even if, Calthrop and Proost (2008) emphasised the need for more research on establishing external costs, to the best of our knowledge, only Janic (2007, 2008) determines generic external cost functions for rail and road transport. Evaluations of external costs of land transportation are highly dispersed. In the U.S., Forkenbrock (2001) compare external costs for intercity general freight truckload trucking and rail freight transportation whereas Quinet (2004) analyses external transport cost estimates of European studies and shows that the main differences come from the specificity of the situation under review and the type of cost calculated.

Indeed, evaluation of transportation external costs depends on parameters such as location, congestion, vehicle characteristics, meteorological condition, accidents, noise, water pollution, etc. Regarding to the location parameter, the particular interest to urban zones mainly lies in the higher concentration of traffic, the intensity of exposition, as well as the number of people exposed. Assessing the economic costs of congestion involves a number of parameters and assumptions (Maibach et al., 2008a, 2008b). The vehicle characteristics (Euro standards) but also specific parameters related to the externalities (e.g. the speed and the loading of a vehicle) should be taken into account. The interaction between road transport emissions and street structures (e.g. the slope on which the vehicle evolves) also plays an important role (Bagienski, 2015). Moreover, the translation of the obtained results into policy measures is a challenging task (Van Essen et al., 2007).

The section aims at monetizing these external effects, to identify the total Belgian external costs of rail, road, and IWW which relate to climate change, photochemical oxidant formation and particulate matter formation. These particular impact categories have been chosen because they represent the most important contributors to the negative impact of transport on human health.

The valuation of Belgian external costs of transport is done in two phases. First, the tkm transported in Belgium for the year 2012 (reference year for externalities' calculations) are identified for each mode of transport. The impacts of these flows are then valued in Euros by multiplying them with the unit external costs values (per tkm). Finally, these monetary values are summed for each mode. A comparison of these results to the total externalities values in DALYs related to climate change, photochemical oxidant formation and particulate matter formation is then provided in order to identify if the two tools (externalities in DALYs and external costs in Euros) provide similar or

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divergent conclusions. DALY is "a summary measure of population health that accounts for both mortality and nonfatal health consequences" (Salomon, 2014).

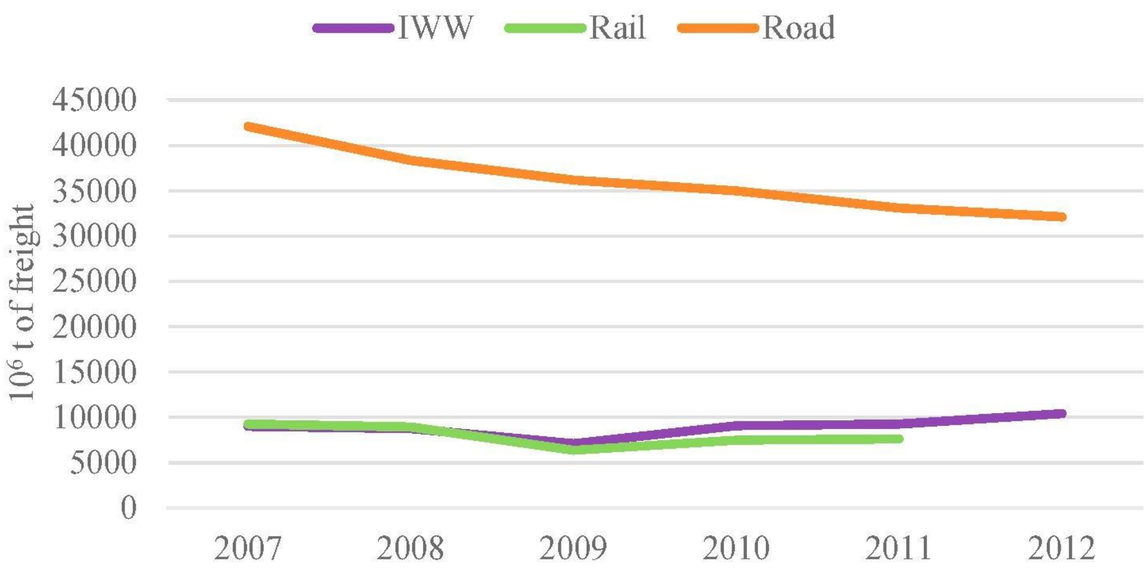
Table 6 provides a summary of the considered unit external costs values for rail, road and IWW transport. Unit externalities for Belgium have been computed in the framework of this study (see Table 8) and are expressed per tkm.

Unit external costs related to photochemical oxidant formation and particulate matter formation in Belgium are directly taken from Ricardo-AEA (2014). Unit external costs for climate change are computed using the central value for CO_2 computed in Ricardo- AEA (2014), on the basis of the meta-study of Kuik et al. (2009). The avoidance costs are computed with the target to stabilize global warming at 2 °C (maximum CO_2 equivalent concentration in the atmosphere of 450 ppm) by 2025, which corresponds to the goal supported and formalized in the 2016 Paris Agreement of the United Nations Framework Convention on Climate Change (UNFCCC).

Table 6. *External costs for rail, road and IWW transport. Source: Ricardo-AEA (2014).*

	External cost value
Climate change	90 €/tonne of CO_2 eq.
Photochemical oxidant formation	3228 €/tonne of NMVOC eq.
Particulate matter formation	67,278 €/tonne of PM_{10} eq.

Figure 3. *Freight flow evolution per mode. Source: Eurostat Statistics (2018).*



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4. Results

4.1. FREIGHT FLOWS IN BELGIUM

Rail, road and IWW freight flows in Belgium for the year 2012 are retrieved from Eurostat statistics (2018a, 2018b, 2018c). Road and IWW flows in tkm for the year 2012 in Belgium are available on Eurostat statistics (2018a, 2018b), accounting for 32,105 million tkm and 10,420 million tkm, respectively. For rail transport, the most up-to-date data in tkm are only available for the year 2011 (Eurostat Statistics, 2018c). The rail flows in tkm for 2012 have been estimated by increasing the 2011 flows by a factor of 3.15%, resulting in 8125 million tkm. This ratio has been computed based on the evolution of flows from 2007 to 2011 for each mode of transport. Fig. 3 shows that rail flows in tkm have the same trend than IWW flows in tkm, but in a lower range of variation.

As shown in Table 7, between 2009 and 2010, and 2010 and 2011, rail flows increases respectively corresponded to a relative increase of 35% and 28% of the evolution of IWW flows in tonnes. Since IWW flows in tonnes increased by 10% between 2011 and 2012, it is assumed that rail also increased by an average ratio of 70% of these 10%, i.e. by 7% between 2011 and 2012.

Table 7*. Freight flows variations between two consecutive years in tkm in Belgium. Sources: Eurostat statistics (2018a, 2018b, 2018c).*

	2007–2008	2008–2009	2009–2010	2010–2011	2011–2012
Rail variation	-3.58%	-28.60%	17.29%	1.57%	
Road variation	-8.86%	-5.69%	-3.24%	-5.41%	-3.03%
IWW variation	-2.89%	-18.97%	27.98%	2.00%	12.64%

4.2. LIFE CYCLE ASSESSMENT OF INLAND FREIGHT TRANSPORT MODES IN BELGIUM

Table 8 presents the environmental impact assessment results of the transport processes in Belgium in the year 2012. These transport processes are the following: rail freight transport considering the Belgian traction mix of the year 2012 (86.3% of electric trains and 13.7% of diesel trains), average road transport with a load factor of 50% and IWW transport.

These results update the global values of externalities of inland freight modes in Belgium related to the midpoint impact categories photochemical oxidant formation, particulate matter formation and climate change, for the year 2012.

Table 9 shows the unit value of externalities in terms of damage to human health (DALY) for each mode of transport. These values have been calculated from the results of the midpoint impact indicators for Belgium (see Table 8) using damage factors of the LCIA method ReCiPe 2008 (Goedkoop et al., 2013).

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Table 10 highlights the externalities in terms of DALYs for each category and mode, as well as the total externalities in Belgium for the year 2012. As mentioned in the methodology, these results consist in the unit DALYs per tkm, multiplied by the number of tkm transported in 2012 in Belgium (Eurostat Statistics, 2018a, 2018b, 2018c).

***Table 8**. LCIA results of 1 tkm of rail, road and IWW transport in Belgium in 2012.*

		Unit	Rail (/tkm)	Road (/tkm)	IWW (/tkm)
Midpoint category	Climate change	kg CO_2 eq.	6.42×10^{-2}	1.13×10^{-1}	7.47×10^{-2}
	Photochemical oxidant formation	kg NMVOC eq.	3.13×10^{-4}	8.72×10^{-4}	5.34×10^{-4}
	Particulate matter formation	kg PM_{10} eq.	1.33×10^{-4}	3.09×10^{-4}	1.92×10^{-4}
Endpoint category	Damage to human health	DALY	1.45×10^{-7}	2.65×10^{-7}	1.66×10^{-7}
	Damage to ecosystem diversity	species × year	6.64×10^{-10}	1.20×10^{-9}	7.89×10^{-10}
	Damage to resource availability	$	4.04×10^{-3}	7.26×10^{-3}	3.65×10^{-3}
Inventory	Nitrogen oxides emissions to air	kg NO_X	2.38×10^{-4}	7.17×10^{-4}	4.61×10^{-4}

***Table 9**. Unit externalities for road, rail and IWW transport in DALYs.*

	Rail (DALY/tkm)	Road (DALY/tkm)	IWW (DALY/tkm)
Climate change	8.99×10^{-8}	1.58×10^{-7}	1.05×10^{-7}
Photochemical oxidant formation	1.22×10^{-11}	3.40×10^{-11}	2.08×10^{-11}
Particulate matter formation	3.47×10^{-8}	8.03×10^{-8}	4.99×10^{-8}

***Table 10**. Total externalities in Belgium for the year 2012.*

2012	Rail	Road	IWW	TOTAL
Climate change in DALYs	730.39	5072.59	1094.10	6897.08
Photochemical oxidant formation in DALYs	0.10	1.09	0.22	1.41
Particulate matter formation in DALYs	281.92	2578.03	519.96	3379.91
TOTAL	1012.41	7651.71	1614.27	10278.40

***Table 11.** Total external costs in Belgium for the year 2012 – reference scenario.*

2012 (reference)	Rail	Road	IWW	TOTAL
Climate change in Euros	46,943,419	326,507,850	70,053,660	443,504,929
Photochemical oxidant formation in Euros	8,208,712	90,369,668	17,961,496	116,539,876
Particulate matter formation in Euros	72,697,904	667,427,699	134,599,058	874,724,661
TOTAL	127,850,035	1,084,305,216	222,614,214	1,434,769,466

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4.3. EXTERNAL COSTS OF INLAND FREIGHT TRANSPORT MODES IN BELGIUM

This section summarizes the total external costs for each mode of transport for one year. The analysis is based on the flows in tkm transported respectively by rail, road and IWW for the year 2012. Table 11 provides the total external costs in Euros for each category and mode, as well as the total external costs in Belgium for the year 2012.

At a first look, it is easily observable that externalities (Table 10) and external costs (Table 11) of transport do not provide exactly the same type of results. This will be further analysed in the discussion section 5.2 Externalities versus external costs of inland freight transport modes in Belgium. After the study of Mayeres et al. (2001), these results update the global values of external costs of inland freight modes in Belgium related to climate change, photochemical oxidant formation and particulate matter formation for the year 2012.

5. Discussion

5.1. EXTERNALITIES OF INLAND FREIGHT MODES IN BELGIUM

Fig. 4 shows a comparison of the results obtained in the environmental impact assessment of 1 tkm of freight transported by rail, road and IWW in Belgium in the year 2012 (from Table 8). Since each indicator is expressed in different units, and to facilitate the interpretation of the results, all the scores of an indicator have been divided by the highest score of the indicator, which represents the maximum impact of the indicator. Therefore, the lowest value represents the mode of transport with less impact and the highest value represents the maximum impact. Road transport has the maximum impact in every environmental impact indicator, with rail freight transport presenting the minimum. IWW transport is in intermediate situation. The only exception is the endpoint environmental impact category damage to resource availability, where rail transport shows a greater impact than IWW transport.

For the LCIA of the Belgian rail freight transport, all life cycle phases of rail freight transport operation, electricity generation for electric trains (i.e. the calculated electricity supply mix in Belgium of the year 2012), diesel production for diesel trains, railway infrastructure and rail equipment (i.e. locomotive and wagons) are taken into account. Similarly, for the LCIA of IWW transport, all life cycle phases of IWW transport operation, fuel production, IWW infrastructure (including canals and port facilities), and barge are included. For the LCIA of road transport, all life cycle phases of road transport operation, diesel production, road infrastructure, and truck are included. Fig. 5 shows the distribution of the environmental impacts (see the absolute values in Table 8) between the different stages of freight transport (i.e. transport operation, electricity generation, fuel production, infrastructure and vehicle) obtained in the LCIA of 1 tkm of freight transported by rail, road and IWW in Belgium in the year 2012.

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Due to uncertainties regarding the LCIA results, a discussion of the most significant differences between transport modes has been performed. Within the Belgian rail freight transport, on the one hand the transport operation is the main source of impact for diesel trains in the indicators climate change, photochemical oxidant formation and particulate matter formation as a result of the exhaust emissions of diesel locomotives. On the other hand, the electricity generation is the main source of impact for electric trains in the indicators climate change. In the case of IWW transport, the infrastructure sub-system (especially the port facilities demand) is the main source of impact in the indicators climate change and particulate matter formation. For road freight transport, the transport operation stage is the main source of impact in the indicators climate change, photochemical oxidant formation and particulate matter formation.

Regarding the indicator at midpoint level climate change, the generation of the electricity used by electric trains is the main source of impact for the Belgian rail freight transport. It should be noted that the 86.3% of rail freight transport was produced by electric traction in Belgium in 2012. Thereby, the GHG emitted by the natural gas and coal power plants represent the 19% and 9% of the total impact of freight trains in this indicator, respectively. In Belgium, the natural gas power plants contributed 22.18% of the total electricity supply mix and the coal power plants were responsible for 4.99% in the year 2012. Additionally, diesel trains constituted the 13.7% of the total freight trains in Belgium in 2012, representing the GHG emitted as direct emissions from diesel trains the 10% of the total impact in the indicator climate change. Diesel trains are also responsible for 2% of indirect GHG emissions from the production of diesel used in transport. Focusing on the contribution of railway infrastructure to the indicator climate change, the production of steel, concrete and gravel are the main sources of GHG emissions. The production of the steel used in the manufacturing of the rolling stock is the main contributor to climate change for the vehicle stage. In the case of road transport, the main source of impact in the indicator climate change is the GHG emissions emitted as exhaust emission by the trucks in the transport activity. Moreover, the petroleum refining to obtain the diesel is major source of impact. For IWW transport, the main source of impact for this indicator is related to the production of materials such as concrete and steel used in canals and port facilities.

For the indicator photochemical oxidant formation, the exhaust emissions during transport operation of NO_X and NMVOC from diesel trains, barge and trucks are the main contributor in this indicator. The tropospheric ozone is formed from other precursor pollutants such as NO_X and NMVOC by photochemical reaction under the influence of solar radiation.

For the indicator particulate matter formation, the exhaust emissions of $PM_{2.5}$ and NO_X from diesel trains, trucks and barges are a major source of impact in this indicator. The impact generated by the transport infrastructure is important in this indicator due to the emissions of SO_2 and particles during the production of materials used in the transport infrastructure such as steel, gravel and concrete. Moreover, the production of electricity used by electric trains from fossil fuels such as coal and natural gas power plants is a main source of impact for freight trains. Particulate matter can be emitted directly from vehicles (primary particulate matter) or be formed in the atmosphere from precursor pollutants such as SO_X, NO_X, ammonia or VOC.

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Focusing on the indicators at endpoint level, the electricity generation is the main contributor for rail freight transport in the indicators damage to human health, ecosystems diversity and resource availability. This finding corroborates the recommendation of Banar and Özdemir (2015) to better improve the LCA results of rail freight transport by shifting the electricity mix to cleaner energy sources. For IWW transport, the main source of impact for the endpoint indicators is the infrastructure demand. In the case of road transport, the transport operation stage is the main source of impact in the indicators damage to human health and ecosystems diversity as a result of the exhaust emissions of trucks and the diesel production is the main source of impact for the endpoint indicator damage to resource availability.

An uncertainty analysis performed with the Monte Carlo method has been carried out to study the robustness of the LCIA results obtained. The Monte Carlo analysis is the most common uncertainty propagation method used to analyse uncertainties in LCA (Igos et al., 2018). The accuracy of Monte Carlo analysis increases as the number of iterations grows, but it is difficult to predict how many iterations are sufficient (Rosenbaum et al., 2018b). Therefore, it has been decided to set at 10,000 the number of iterations in all the analyses. This number of iterations is considered sufficient to achieve a representative analysis and is widely used in other LCA studies (Igos et al., 2018). All calculations of this uncertainty analysis have been performed with the Monte Carlo implementation of SimaPro 8.0.5 software (Pré, 2013) using the LCIA method ReCiPe 2008 (hierarchist, version V1.12/Europe).

Tables 12–14 present the uncertainty distribution obtained for the LCIA of rail, road and IWW transport, respectively. The uncertainty analysis includes the mean, median value, standard deviation (SD), coefficient of variability (CV) and standard error of mean (SEM) using a 95% confidence interval.

Figs. 6–8 show a bar chart with the uncertainty ranges per impact category considering a 95% confidence interval (i.e. 95% of the results obtained in the Monte Carlo simulation are within the range) for rail, road and IWW transport, respectively. For rail freight transport (see Fig. 6), the midpoint indicator climate change shows the lowest uncertainty, while the other indicators present similar uncertainty ranges. For road transport (see Fig. 7), the endpoint indicators damage to ecosystem diversity and damage to resource availability present the higher uncertainty. For IWW transport (see Fig. 8), all the indicators present similar uncertainty ranges. However, the midpoint indicators have higher uncertainty ranges compared to rail and road transport.

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Figure 4. *Environmental impact assessment of 1 tkm of freight transported using rail, road and IWW in Belgium in 2012.*

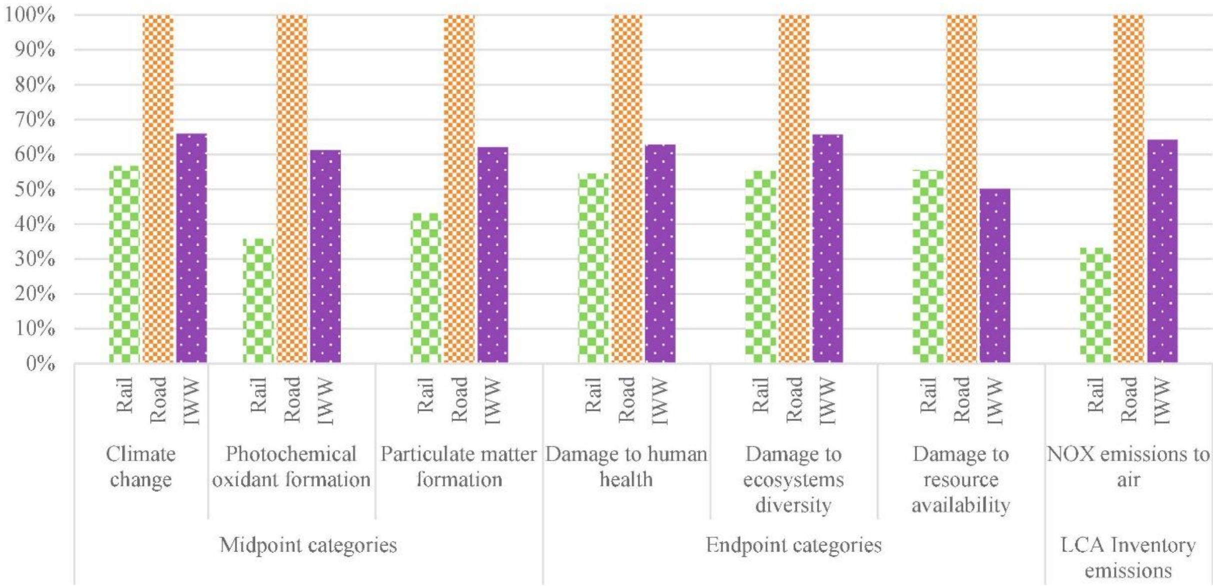
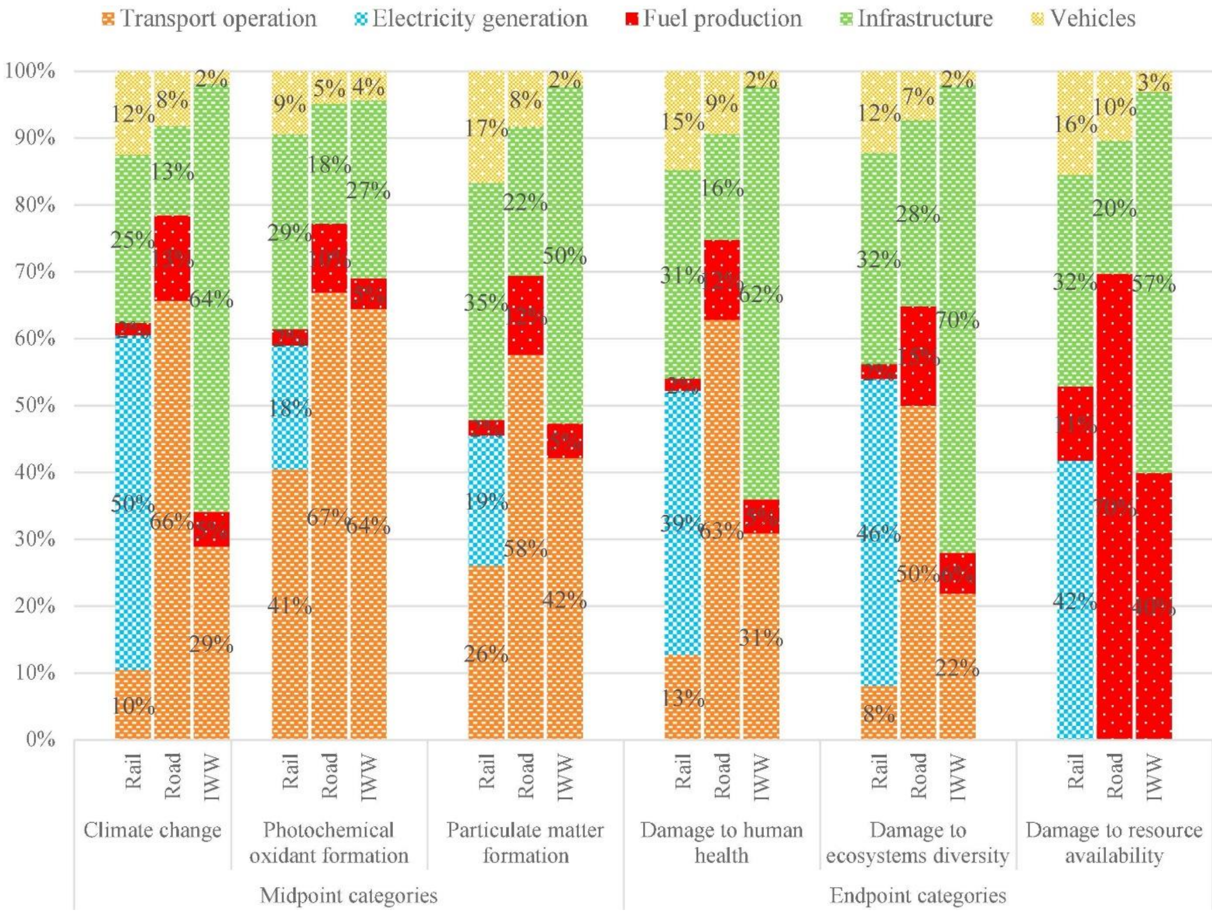


Figure 5. *LCIA of 1 tkm of freight transport using rail, road and IWW in Belgium in 2012.*



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5.2. EXTERNALITIES VERSUS EXTERNAL COSTS OF INLAND FREIGHT TRANSPORT MODES IN BELGIUM

This section discusses whether externalities in DALYs or external costs in Euros can be used interchangeably to evaluate the negative impacts of transport on its environment. Based on the Belgian case study, Fig. 9a and b respectively provide a comparison of the proportion of externalities and external costs per category of negative impact and per category of mode of transport.

On Fig. 9a, it is noticed that each category of negative impact on the environment does not represent the same percentage of global externalities and external costs. Indeed, dealing with externalities at endpoint level implies that climate change and particulate matter formation have the most important relative impact on human health. This is due in the case of climate change to the highest emissions per tkm ofkg CO_2 eq. in comparison with kg PM_{10} eq. and kg NMVOC eq., and in the case of particulate matter formation to the greater detrimental effect that particles have on human health, so which have a damage factor greater than CO_2 and NMVOC. The impact of photochemical oxidant formation in DALYs for the Belgian case is negligible compared to the two other categories because of the lower emissions per tkm of kg NMVOC eq. and the lower harmful effect considered on human health of NMVOC emissions compared to CO_2 and PM_{10}. Focusing only on externalities would therefore imply that climate change and particulate matter formation represent a bigger issue than photochemical oxidant formation. Dealing with external costs highlights the importance of the negative impact of particulate matter formation and climate change issues, leaving photochemical oxidant formation issues as a more negligible issue.

When comparing the externalities and external costs per mode of transport on Fig. 9b, we observe more similar results. Indeed, the most negative impact is generated by road transport, followed by IWW and then rail transport, in both valuations. However, some differences are still observed between the two valuation tools. The impact of road transport is slightly increased compared to both impacts of rail and IWW when external costs are considered instead of externalities. In practice, this highlights that the valuation of the impact of externalities in terms of money is higher for road than for rail and IWW.

Results presented in Fig. 9b are coherent with the results of the INFRAS study (CE Delft, 2011) which highlight that, regarding freight transport external costs related to air pollution at the European level, road has the major impact followed by IWW and by rail. According to CE Delft (2011), the contribution of road, IWW and rail freight transport to air pollution external costs for EU-27 plus Norway and Switzerland is respectively 93.7%, 3.9% and 2.4%. Climate change costs also present the same pattern since most of the impact at the European level is generated by road, followed by IWW and then by rail. For the specific case of Belgium, CE Delft (2011) suggests that total external costs are generated for 74.4% by road, 8.4% by rail and 17.2% by IWW, which again supports our conclusions. Our results are based on the external cost values given in Ricardo-AEA (2014), which synthetizes the knowledge regarding external costs by reviewing several recognized scientific works like the series of projects ExternE studies with the Impact Pathway Approach (Bickel and Friedrich, 2005), the HEATCO study (Bickel et al., 2006a), the CAFE-CBA study (AEA Technology Environment,

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2005) and the NEEDS project (Preiss and Klotz, 2007). External cost values are therefore generated based on the currently best available data.

Results of Fig. 9a and b stand for the monetary values attributed to externalities in the framework of this study. Nevertheless, external costs related to GHG emissions (climate change) are known to vary in a certain range, in order to express the uncertainty regarding the effects of these externalities in the future (Tol, 2013; Ricardo-AEA, 2014). To account for this uncertainty, we now develop a short what-if analysis to determine how the modification of the unit external cost values of climate change influences the results observed above. This is done by changing the unit external cost value to 168 €/tonne of CO_2 eq. (worst-case scenario) and to 48 €/tonne of CO_2 eq. (best-case scenario), based on the range of external cost values identified in Ricardo-AEA (2014).

The updated values of total external costs are given in Table 15 (worst-case scenario) and Table 16 (best-case scenario).

This sensitivity analysis tends to confirm the results observed on the reference scenario. When comparing externalities and external costs in terms of their category (climate change, photochemical oxidant formation, particulate matter formation), we observe in the best-case scenario that focusing on external costs leads even more to policies aiming at reducing particulate matter formation (which represent around 71% of the total external costs against 61% in the reference case) while looking at externalities leads to policies aiming at reducing climate change. In the worst-case scenario, the focus is still on policies related to particulate matter formation (48% of the costs), even if the climate change impact represents a bigger proportion of the total costs (46%) than in the reference scenario (31%).

When comparing externalities and external costs per mode of transport, we observe that each mode represents a similar proportion of the global externalities or external costs, whatever the considered scenario (best-case, worst-case or reference case). Road transport contributes to around 75% of the road externalities and external costs, while rail and IWW transport respectively contribute to around 9% and 16%.

Results highlight that when it comes to evaluate different categories of negative impacts of transport on the environment, the proportion of each category differs between the total computed externalities and external costs. Using externalities and average external costs interchangeably for estimating the impact of specific transport categories on the environment may therefore be dangerous since it may lead to significant different results. On the contrary, the evaluation of the proportion of total externalities and total external costs per mode of transport provides rather similar results, meaning that global unit external costs and externalities per mode are similar. This means that externalities and external costs per mode of transport (sum of climate change, photochemical ozone formation and particulate matter formation) can be used interchangeably for determining the average impact of these modes of transport.

When the focus is on the evaluation of negative impacts of land transport for policy-related decision purpose at a global level (i.e. the sum of the negative effects of rail, road and IWW, see Fig. 9a), it is not correct to interchangeably use externalities and average external costs. Indeed, some categories

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of negative impact may be over- or under-represented compared to the others, depending on the used method. Practically, if externalities are taken into account, policy measures may lead to the development of measures aiming mainly at reducing climate change effect, while a focus on external costs would rather lead to an increase of measures aiming at reducing air pollution related to particulate matter formation. While climate change effects have to be managed at the global level, pollution matters mainly refer to the local level. The use of externalities and average external costs as evaluation tools of the negative impact of land transport on the environment may therefore lead to completely different types of policy measures.

The analysis of Fig. 9b is valuable to specific domains of transport research like operations research. Indeed, the results highlight that externalities and average external costs provide similar trends in optimization models when the objective is to identify the modal split of different modes of transport while minimizing the environmental impact. Indeed, the distribution of the environmental impact between land modes is rather similar, whatever the choice of tool. The use of externalities or average external costs related to a same category of negative impacts should therefore lead to similar relative modal split results in operations research models applied to Belgium (Santos et al., 2015; Mostert et al., 2017a, 2017b).

Figure 6. *Uncertainty analysis of 1 tkm of rail freight transport in Belgium in 2012 considering a 95% confidence interval.*

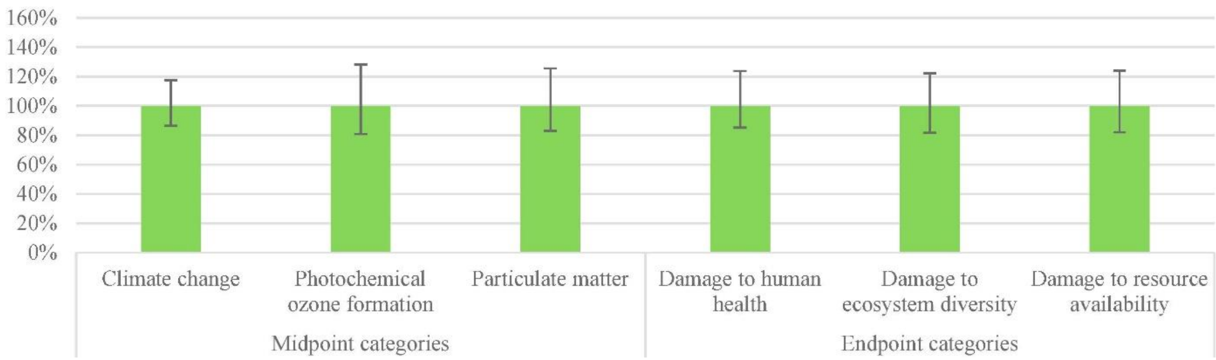

Figure 7. *Uncertainty analysis of 1 tkm of road freight transport with a load factor of 50% in Belgium in 2012 considering a 95% confidence interval.*

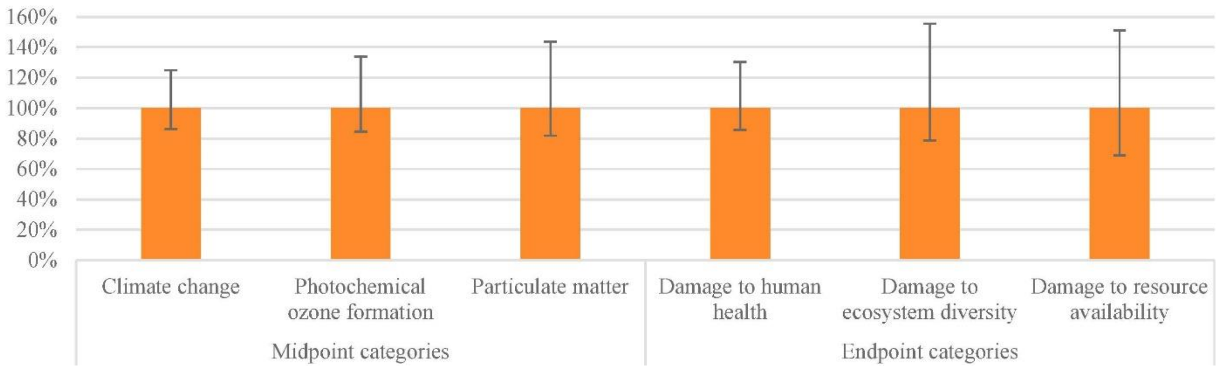

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***Figure 8**. Uncertainty analysis of 1 tkm of IWW transport in Belgium in 2012 considering a 95% confidence interval.*

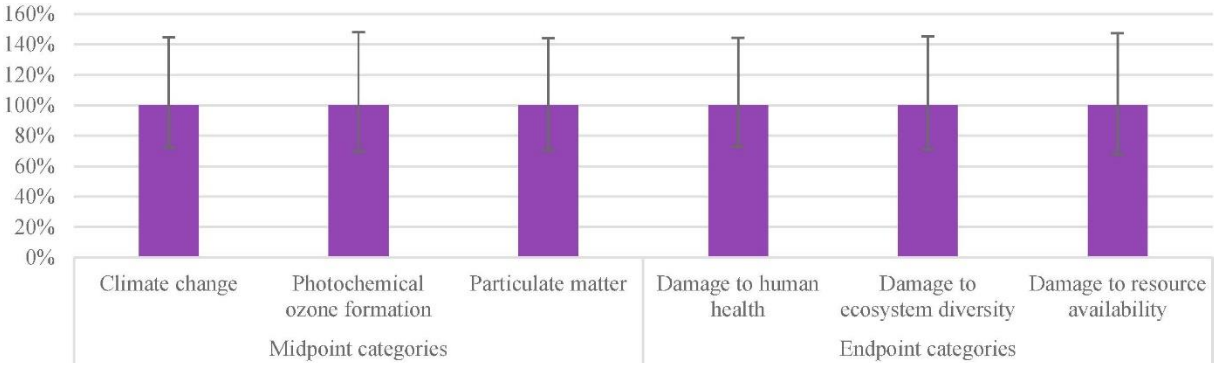

***Figure 9.** (a) Freight flow evolution per category of negative impact. (b) Freight flow evolution per category of transport mode.*

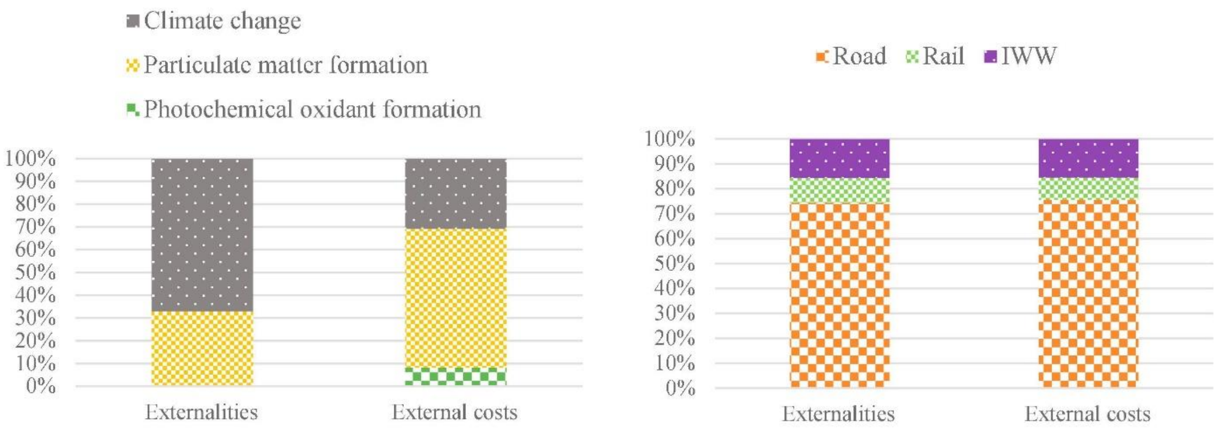

***Table 12.** Uncertainty analysis of 1 tkm of rail freight transport in Belgium in 2012.*

	Impact category	Unit	Mean	Median	SD	CV (%)	Confidence interval 95%		SEM
							2.5%	97.5%	
Midpoint category	Climate change	kg CO_2 eq	6.41×10^{-2}	6.38×10^{-2}	5.05×10^{-3}	7.88	5.52×10^{-2}	7.49×10^{-2}	5.05×10^{-5}
	Photochemical oxidant formation	kg NMVOC eq	3.12×10^{-4}	3.09×10^{-4}	3.69×10^{-5}	11.83	2.50×10^{-4}	3.96×10^{-4}	3.69×10^{-7}
	Particulate matter formation	kg PM_{10} eq	1.33×10^{-4}	1.32×10^{-4}	1.43×10^{-5}	10.76	1.09×10^{-4}	1.65×10^{-4}	1.43×10^{-7}
Endpoint category	Damage to human health	DALY	1.45×10^{-7}	1.43×10^{-7}	1.58×10^{-8}	10.92	1.22×10^{-7}	1.77×10^{-7}	1.58×10^{-10}
	Damage to ecosystem diversity	species × year	6.64×10^{-10}	6.61×10^{-10}	6.79×10^{-11}	10.22	5.40×10^{-10}	8.07×10^{-10}	6.79×10^{-13}
	Damage to resource availability	$	4.04×10^{-3}	4.01×10^{-3}	4.30×10^{-4}	10.63	3.29×10^{-3}	4.97×10^{-3}	4.30×10^{-6}

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Table 13. *Uncertainty analysis of 1 tkm of road freight transport in Belgium in 2012.*

	Impact category	Unit	Mean	Median	SD	CV (%)	Confidence interval 95%		SEM
							2.5%	97.5%	
Midpoint category	Climate change	kg CO_2 eq	1.13×10^{-1}	1.11×10^{-1}	1.11×10^{-2}	9.80	9.61×10^{-2}	1.39×10^{-1}	1.11×10^{-4}
	Photochemical oxidant formation	kg NMVOC eq	8.71×10^{-4}	8.51×10^{-4}	1.06×10^{-4}	12.23	7.21×10^{-4}	1.14×10^{-3}	1.06×10^{-6}
	Particulate matter formation	kg PM_{10} eq	3.09×10^{-4}	2.99×10^{-4}	4.70×10^{-5}	15.23	$2.45 \times I10^{-4}$	4.30×10^{-4}	4.70×10^{-7}
Endpoint category	Damage to human health	DALY	2.65×10^{-7}	2.60×10^{-7}	3.04×10^{-8}	11.49	2.23×10^{-7}	3.38×10^{-7}	3.04×10^{-10}
	Damage to ecosystem diversity	species × year	1.20×10^{-9}	1.15×10^{-9}	2.30×10^{-10}	19.13	9.07×10^{-10}	1.79×10^{-9}	2.30×10^{-12}
	Damage to resource availability	$	7.27×10^{-3}	7.09×10^{-3}	1.52×10^{-3}	20.86	4.89×10^{-3}	1.07×10^{-2}	1.52×10^{-5}

Table 14. *Uncertainty analysis of 1 tkm of IWW transport in Belgium in 2012.*

	Impact category	Unit	Mean	Median	SD	CV (%)	Confidence interval 95%		SEM
							2.5%	97.5%	
Midpoint category	Climate change	kg CO_2 eq	7.48×10^{-2}	7.32×10^{-2}	1.35×10^{-2}	18.00	5.27×10^{-2}	1.06×10^{-1}	1.35×10^{-4}
	Photochemical oxidant formation	kg NMVOC eq	5.33×10^{-4}	5.20×10^{-4}	1.06×10^{-4}	19.84	$3.62 X 10^{-4}$	7.70×10^{-4}	1.06×10^{-6}
	Particulate matter formation	kg PM_{10} eq	1.92×10^{-4}	1.89×10^{-4}	3.51×10^{-5}	18.30	1.33×10^{-4}	2.72×10^{-4}	3.51×10^{-7}
Endpoint category	Damage to human health	DALY	1.66×10^{-7}	1.62×10^{-7}	3.00×10^{-8}	18.05	1.19×10^{-7}	2.35×10^{-7}	3.00×10^{-10}
	Damage to ecosystem diversity	species × year	7.88×10^{-10}	7.71×10^{-10}	1.48×10^{-10}	18.82	5.49×10^{-10}	1.12×10^{-9}	1.48×10^{-12}
	Damage to resource availability	$	3.64×10^{-3}	3.57×10^{-3}	7.35×10^{-4}	20.21	2.42×10^{-3}	5.26×10^{-3}	7.35×10^{-6}

Table 15. *Total external costs in Belgium for the year 2012 – worst-case scenario.*

2012 (worst-case)	Rail	Road	IWW	TOTAL
Climate change in Euros	87,627,715	609,481,320	130,766,832	827,875,867
Photochemical oxidant formation in Euros	8,208,712	90,369,668	17,961,496	116,539,876
Particulate matter formation in Euros	72,697,904	667,427,699	134,599,058	874,724,661
TOTAL	168,534,332	1,367,278,686	283,327,386	1,819,140,404

Table 16. *Total external costs in Belgium for the year 2012 – best-case scenario.*

2012 (best-case)	Rail	Road	IWW	TOTAL
Climate change in Euros	25,036,490	174,137,520	37,361,952	236,535,962
Photochemical oxidant formation in Euros	8,208,712	90,369,668	17,961,496	116,539,876
Particulate matter formation in Euros	72,697,904	667,427,699	134,599,058	874,724,661
TOTAL	105,943,107	931,934,886	189,922,506	1,227,800,499

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6. Conclusion

This study provides updated values of externalities and external costs for inland freight transport on the Belgian case study regarding three negative effects: climate change, photochemical ozone formation and particulate matter. Externalities and external costs are then compared to identify if they can be used interchangeably in terms of policy decision-making.

LCA and external costs results show that road transport has the maximum impact in every environmental impact indicator, with rail freight transport presenting the minimum one.

In Belgium, rail freight transport is performed by both electric and diesel traction, but the use of electric trains is much greater, representing 86.3% of the total rail freight in 2012. Hence, the electricity generation is the main contributor for rail freight transport in the indicators at endpoint level damage to human health, ecosystems diversity and resource availability. For IWW transport, the main source of impact for the endpoint indicators is the production of materials such as concrete and steel used in canals and port facilities. In the case of road transport, the transport operation stage is the main source of impact in the indicators damage to human health and ecosystems diversity. The exhaust emissions of trucks and the diesel production are the main source of impact for the indicator damage to resource availability.

The findings of this case study highlight that externalities and average external costs may be two different tools to evaluate the negative impact of inland transport on the environment, providing divergent results. Indeed, each category of negative impact on the environment does not represent the same percentage of global externalities and external costs. In the analysis of the environmental impact per mode, it is observed that, when external costs are considered instead of externalities, the impact of road transport is slightly increased compared to both impacts of rail and IWW. Using externalities or external costs in a global manner (i.e. by summing externalities or external costs of rail, road and IWW) may therefore lead to different decisions in terms of policy. External costs or externalities nevertheless provide similar trends when it comes to evaluate the environmental impact per mode of transport. Even if external costs allow comparing different kinds of negative impacts of transport in the same unit, they still do not completely reflect the same reality as externalities do. Externalities and external costs should therefore be used in a complementary way.

In Belgium, externalities of road transport are practically internalized through specific policy measures such as the introduction of the Viapass tax for trucks in 2016. Even if the introduction of this tax practically disfavours road transport flows (by making road transport supporting part of its external costs), it is still not high enough to lead to the optimal environmental distribution of flows between road and intermodal transport (Mostert et al., 2017b). This shows that a clear definition of updated externalities and external costs values is really necessary to correctly cope with environmental policy measures.

The results obtained in this paper are valid for Belgium. These results do not necessarily directly have applicability for other case studies. Indeed, different characteristics between countries have to be considered such as e.g. different degrees of electrification in rail transport or trucks with different emission engine technologies. However, the presented methodology is directly applicable to evaluate externalities and external costs of other geographical cases.

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