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Prospective life-cycle assessment of sustainable alternatives for road freight transport

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ABSTRACT

This study investigates decarbonization pathways for the road freight transport sector by evaluating three alternatives to conventional diesel trucks: trucks powered by biofuels, battery electric trucks, and fuel cell trucks with hydrogen. A prospective life cycle assessment of these options is conducted under two policy scenarios for decarbonization across 12 distinct regions over the century. Employing a cradle-to-grave approach, the assessment covers activities from fuel and electricity production to the end-of-life of truck components. Findings reveal that, in eight of the 12 regions examined, an early transition to battery electric trucks could increase life-cycle greenhouse gas emissions by up to 70 % by 2030 compared to the continued use of conventional diesel trucks, underscoring the significance of liquid fuels for short to medium-term decarbonization. However, in the long term, as electricity mixes and hydrogen production are decarbonized, battery electric trucks and hydrogen fuel cell trucks emerge as superior alternatives in all regions, emitting, at least, 29 % less greenhouse gases than trucks powered by biofuels, and 45 % less than diesel trucks. The optimal transition from conventional diesel trucks to trucks powered by biofuels and, subsequently, to battery electric trucks and/or hydrogen could avoid 134–204 Gt CO_{2eq} worldwide and prevent a temperature rise of 0.22–0.33 °C compared to the diesel-based scenario. This emphasizes the crucial role of appropriate policies for the timely transformation of the road freight transport sector. Despite this, battery and hydrogen trucks can impact human health up to 239–1200 % more than diesel, which is not to be taken lightly.

1. Introduction

According to the European Environment Agency, road transport is responsible for approximately 20 % of the European Union's total greenhouse gas (GHG) emissions, with road freight transport (RFT) contributing around 25 % of these [1]. RFT emissions are projected to increase by 22 % by 2050 if current policies persist, further exacerbating the global climate crisis [2]. Despite this, ignition combustion vehicles (ICVs) will likely continue being employed in the short to medium-term. As an example, regions such as Germany, which had previously restricted the sale of ICVs after 2035, have reconsidered the regulation to secure the ongoing presence of ICVs, as long as they use low-CO_2 emissions fuels [3].

Currently, the transport sector is in a critical transition, with many global policies striving for its gradual decarbonization through technological innovation [4,5]. One of the goals proposed by the European Commission as a means of reaching carbon neutrality by 2050 is to achieve a 40 % reduction of emissions in the transportation sector by 2030 [6]. Despite this, petroleum-based fuels are still the leaders in the transportation sector, with renewable energy sources only accounting for around 3.7 % of energy supply, a 93 % of which comes from biofuels [7].

In this context, the penetration in the market of low-carbon fuels such as biofuels, synthetic fuels, dimethyl ether, and electric vehicles (EVs), must be part of the integral solution to achieve the aforementioned climate objectives [8–10]. Among these options, this study analyses those with a higher technology readiness level (TRL) and efficiency, namely biofuels and electric vehicles. Other alternatives for heavy trucks, such as ammonia, are disregarded [11].

Biofuels are a promising alternative for long-distance RFT. Unlike light vehicles, which operate over short distances and whose energy demand can be met more easily, heavy-duty vehicles used for RFT require a greater amount of energy to operate and, therefore, need fuels with higher energy density [12]. Advanced biofuels fit squarely in this application, since they exhibit a similar energy content to diesel, plus the

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List of abbreviations

BD	Biodiesel
bioICTs	Biofuel-based trucks
BETs	Battery electric trucks
CP	Current policies
dICTs	Diesel-based trucks
EVs	Electric vehicles
FCT	Hydrogen fuel cell truck
GHG	Greenhouse gas
GWP	Global warming potential
HH	Humal health impact
HVO	Hydrotreated vegetable oil
IAMs	Integrated assessment models
ICV	Ignition combustion trucks
ICV	Ignition combustion vehicles
LCA	Life cycle assessment
LCI	Life cycle inventory
MRS	Mineral resource scarcity
NZ	NetZero policies
p-LCA	Prospective life cycle assessment
RFT	Road freight transport
RCPs	Representative concentration pathways
SCG	Share of cumulated GHG
SSPs	Shared socio-economic pathways
TEEA	Techno-economic and environmental assessment

ability to use existing engines, storage systems and infrastructures with minimal or no modifications [13].

On the other hand, the development and adoption of electric trucks is now advancing rapidly as part of a broader trend towards the electrification of the transportation sector. Battery electric trucks (BETs) can reach travel distances of up to 900 km and capacities of up to 14 tons of cargo [14,15]. They also have the advantage of releasing no combustion emissions during operation and are expected to have lower operating costs than diesel trucks over their lifetime owing to lower maintenance costs [16,17]. Electric trucks can also consist of hydrogen fuel cell trucks (FCTs), which offer several advantages to BETs, including greater range and reduced vehicle weight [18].

Although electric trucks are frequently considered to be more sustainable than diesel trucks (dICTs), they are not free from life cycle emissions. As an example, life cycle CO_2 emissions of BETs can vary between 0.3 and 2.0 times those of dICTs [19], depending on the energy source used to charge the batteries. If the electricity comes from renewable sources such as wind or solar power, these emissions may be lower than those of dICTs, yet the use of fossil energy can lead to the opposite situation. Furthermore, in addition to indirect emissions associated with electricity production and battery manufacturing and recycling, BETs generate significant non-exhaust emissions by braking, road wear, and tire wear, which are higher than those of ignition combustion trucks (ICTs) due to the additional weight of the batteries [20,21]. Similarly, hydrogen FCTs can also release important amounts of GHGs depending on the methods used to produce and transport the hydrogen.

Informing decision-makers about the most sustainable vehicle technologies and fuels for the RFT sector requires a comprehensive and transparent comparison of all relevant emissions of the various options. This calls for using life cycle assessment (LCA), which allows quantification of the environmental burdens associated with the life cycle of the various alternatives, including fuel production, truck manufacturing and operation.

Various studies comparing biofuel and electric trucks have been presented to date, finding different results depending on the type of feedstock used for fuels and the primary energy sources used to generate the electricity for the batteries [19,22–24]. Sathre et al. [23] compared BETs with electricity generated from biomass, wind, and solar, with ICTs based on dimethyl ether. They found that, when based on a renewable mix, BETs have much lower GHG emissions than ICTs using diesel or dimethyl ether. Conversely, Ternel et al. [24] explored the use of biofuels, fossil fuels, and EVs on medium-size cars in 2030, finding that, in the European region, biofuel trucks have lower GHG emissions than fossil-based cars and EVs. Other works have demonstrated that the geographical region, the fuel production process, the feedstock, and the methodological assumptions employed can strongly affect the final outcome of the study [22–25]. Finally, Bai et al. [18] compared BETs and FCTs sourced exclusively from renewable energy as an alternative for dICTs, demonstrating a lower GWP impact from BETs than the other options.

Most of these works assume that truck technologies and the electricity mix will remain the same as today in the long term, thus disregarding the effect of improved efficiencies or the use of new decarbonization technologies. Hence, although these contributions provide valuable insights into the current environmental performance of transportation technologies, they may not capture the long-term impacts and implications of today's technological choices.

To avoid this, here, we resort to the prospective life cycle assessment (p-LCA) concept. Opposite to traditional LCA, p-LCA explicitly includes the effects of future changes in LCA datasets' background system (i.e., in all activities not explicitly modelled). This is done by generating future life cycle inventories based on the results obtained from Integrated Assessment Models (IAMs) for various socioeconomic and environmental policies. Hence, p-LCA considers potential changes affecting environmental impacts over time, thus ensuring that long-term decisions are not made based only on today's assets.

To the best of our knowledge, few contributions have resorted to p-LCA in the context of the RFT sector [22,26]. In a pioneering work, Van den Oever et al. [22] compared two types of BETs (i.e., hybrid and 100 % electric) with diesel and biofuel trucks in Europe for 2030 and 2050, finding that ICTs based on biofuels obtained through the Fischer-Tropsch synthesis could achieve lower impacts than BETs and diesel trucks. On the other hand, Ren et al. compared different penetration scenarios for alternative fuels, aiming to predict changes over GWP impacts [26]. Finally, Campos-Carriedo et al. applied an instrument provided by the European ecodesign directive to assess whether fuel cell passenger cars and fuel cell heavy duty trucks are sustainable options between 2020 and 2030 [27].

In this contribution, we aim to assess the environmental performance of three key alternatives for decarbonizing the RFT sector over the century: ICTs powered by biofuels (bioICTs), BETs and hydrogen FCTs, informing on their potential benefits or disadvantages. To accomplish this, we perform a p-LCA of the RFT sector in different regions across the globe, utilizing a cradle-to-grave approach, i.e., covering all activities from fuel production until the end-of-life of the truck components. The comparison of the impacts incurred by these three alternatives with those stemming from the business-as-usual scenario, dominated by diesel trucks (dICT), allow us to quantify the reductions that could be achieved in CO_2 emissions through appropriate policies for the deployment of each of these technologies in the different regions studied. In addition, we also assess impacts on mineral resource scarcity (MRS) and on human health (HH), thus providing a comprehensive assessment of the wide implications of the deployment of these technologies in each region.

This study extends previous research in different ways. On the one hand, we do not focus on a single geographical zone but rather study regional variations across the globe. On the other hand, we go beyond carbon emissions and calculate impacts on two additional indicators strongly related to the transport sector. Finally, we focus on biofuels obtained from vegetable oils instead of biomass gasification, therefore exploring the potential advantages of other biofuel synthesis routes. Overall, our results can identify viable pathways towards low-carbon emissions in the RFT sector across the globe, informing decision-makers about the benefits of supporting the most sustainable technologies according to the development level at the time.

2. Methodology

This work studies the environmental performance of the RFT sector over the course of the century. The aim is to provide a comprehensive understanding of the consequences of replacing dICTs with bioICTs, BETs or FCTs on the environment and on human health in the long term and in different regions. It is expected that the corresponding impacts will differ over time since both, technologies and policies are expected to experience changes during the century. However, LCA databases are myopic to these changes since they only provide environmental exchanges and life cycle inventories based on the current landscape. To overcome this limitation, we will introduce the following methodology, with p-LCA at its core (see Fig. 1).

The starting point is the information provided by IAMs, which forecast the potential evolution of the energy system, the sources of GHGs, and the mitigation technologies used under different socio-economic narratives and climate policies. Specifically, we select two scenario projections from the REMIND IAM [28] and use the corresponding results to inform other elements of the methodology.

Next, we generate a set of LCA databases, one for each year and scenario considered, that will be used in the final p-LCA. We capitalize again on the information provided by the IAM for the selected scenarios and use the *premise* toolbox [29] to modify the Ecoinvent v3.8 LCA database accordingly [30]. The resulting databases include additional activities considering the temporal evolution of ICTs, BETs, and FCTs. In addition, we enter additional biofuel production activities into the databases and use them as options for the fuel consumed by bioICTs. To this end, we combine models for growing various biomass feedstocks and simulations for two fuel production processes. From these options, only the most promising alternative is kept for further analysis (see section 2.3 for further details).

With the models of the four options for the RFT sector at hand (i.e., dICTs, bioICTs, BETs and FCTs), we next perform a p-LCA of each alternative adopting a cradle-to-grave approach, i.e., considering various life cycle stages spanning from fuel production, fuel use, truck construction, and the end-of-life of the truck components. Calculations are based on meeting the annual demand for the RFT between 2020 and 2100 in the whole world, divided into 12 separate regions for consistency with the IAM employed.

Further details about the different elements of the methodology are described in the ensuing subsections.

2.1. Integrated assessment models

Integrated assessment models (IAMs) are crucial tools for providing insight and assessing the options and consequences of different long-term GHG emission reduction strategies [29,31]. In these models, scenarios are built by combining two essential pieces of information. On the one hand, IAMs include a series of narratives describing different future trajectories of global socio-economic conditions, i.e., the so-called shared socio-economic pathways (SSPs). On the other hand, various temperature scenarios that could result from implementing different policies [32,33], the so-called representative concentration pathways (RCPs), can also be considered. There are five socioeconomic narratives (i.e., SSP1 to SSP5), and four RCPs spanning a wide range of radiative forcing in 2100 (i.e., 2.6, 4.5, 6.0, and 8.5 W/m^2) [34].

In this contribution, we capitalize on a technology-based IAM [28], REMIND, which can effectively represent how energy, economy, land, and climate systems interact and change over time. The advantage of relying on this type of model is that it can ascertain the evolution of certain technological parameters that we can use to inform later steps of our methodology. Within this IAM, we select SSP2, representing a middle-of-the-road socio-economic narrative, and concentrate on two RCPs: 3.4 and 1.9. The RCP 3.4 scenario corresponds to the continuation of current policies (CP), leading to an increase in global temperature ranging from 2 to 2.4 °C by 2100. On the other hand, RCP 1.9 aims to limit global warming to below 1.5 °C by reaching net zero around 2050 (NZ) [35], thus aligned with the aspirational goal of the Paris Agreement [36]. We will simply use CP and NZ to refer to the two scenarios considered since both rely on the same SSP.

2.2. Creating life cycle assessment databases for future years

The process of integrating IAM scenarios into the LCA database to represent future activities is outlined in Fig. 2. In this process, we capitalize on the *premise v1.5.0* tool, an open-source Python library, running on BrightWay2 V.0.8.7 [37]. This tool has been designed to modify the Ecoinvent v3.8 database by incorporating IAM scenario data concerning five key energy-intensive sectors: electricity generation, cement and steel production, road freight, and passenger transport, as well as conventional and alternative fuel supplies. Moreover, *premise* includes additional activities for emerging and future technologies not originally available in the LCA database, such as hydrogen from biomass gasification, synthetic fuels, and heavy-duty trucks [29].

We depart from the results obtained from REMIND under CP (i.e. REMIND-SSP2-Base) and NZ (i.e., REMIND-SSP2-PkBudg500) narratives in the 2020–2100 period. These consider changes in both technology (e.g., vehicle weight) and policies (e.g., fostering the use and

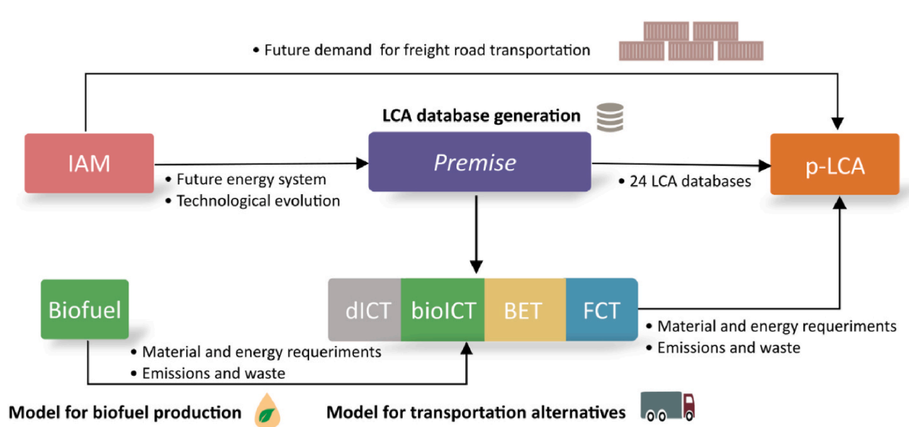


Fig. 1. Methodology overview.

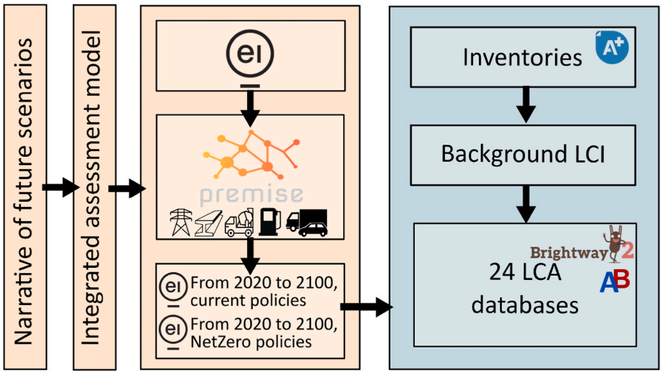


Fig. 2. Prospective life cycle assessment framework. LCI: Life cycle inventories; LCIA: Life cycle impact assessment; LCA: Life cycle assessment; GWP: Global warming potential; HH: Human health; MRS: Mineral resource scarcity.

development of renewable sources). We then generate new LCA databases for the period between 2020 and 2100. Databases are created at a 5-year interval until 2050 and subsequently at a 10-year interval from 2050 to 2100, thus resulting in a total of 12 databases for each scenario (i.e., 24 databases in total). This temporal discretization is chosen because most of the technological improvements considered in *premise* for the RFT sector vary only until 2050. The only exceptions are the electricity mixes and the hydrogen markets of the different regions, which do vary until 2100 (Fig. 3).

2.3. Modelling road freight transportation technologies

Models for ICTs, BETs, and FCTs are sourced from the LCA databases generated with *premise* [29]. This implies there are seven models for each truck type (i.e., one for each year considered until 2050). These models are based on long-haul trucks with a capacity of 40 t, a payload of 14 t, and an autonomy of 800 km with a full tank/battery. For ICTs, the driving mass of the truck is set to 14.9 t. However, the driving mass of BETs will vary over time because of the improvement in batteries and energy efficiency. For some context, the battery required to achieve the same autonomy as an ICT would weight 7.8 t in 2020 (corresponding to an energy density of 120 Wh/kg) but only 1.6t in 2100 (350 Wh/kg) [29]. As a result, the weight of BETs would vary between 22.8 t in 2020 and 16.4 t in 2100, ultimately affecting energy consumption and fugitive emissions. Acknowledging that the energy density of existing lithium batteries only reaches values of around 250 Wh/kg [38], the use of other technologies such as lithium-sulfur batteries will allow us to reach energy densities higher than 600 Wh/kg [39]. Further information for the different truck models is provided in Table S10 in the Supplementary material [40].

In addition, the environmental impact of BETs and FCTs is highly dependent on the composition of the electric mix used to charge their batteries and the method for hydrogen production. To account for this, we source the evolution of the electricity mix and the market for hydrogen in the different regions from REMIND [41], and integrate this information in all the activities related to ICTs, BETs, and FCTs in the 24 LCA databases: from the manufacture of the truck components to the generation of electricity and hydrogen.

On the other hand, the standard model for dICTs in *premise* is used as a basis to create two alternative models for bioICTs: (i) one using a 20 % blend of biodiesel (BD) with diesel, and (ii) another model fully relying on hydrotreated vegetable oil (HVO) as fuel. To this end, we assume that BD and HVO have similar characteristics to diesel. This allows us to obtain alternative bioICT models by simply replacing an equivalent amount of diesel in the ICT models for the different years considered.

Note that both BD and HVO can be obtained from different biomass sources. Hence, to simplify subsequent analyses, we first shortlist the most promising combination of biomass feedstock and biofuel type (i.e., BD vs HVO) by means of a preliminary techno-economic and environmental assessment (TEEA) of a set of alternatives (Fig. 4). This preliminary selection process is based on global (i.e., not region-specific) LCA models for 2020 and considers the use phase of biofuels in bioICTs, as will be explained next.

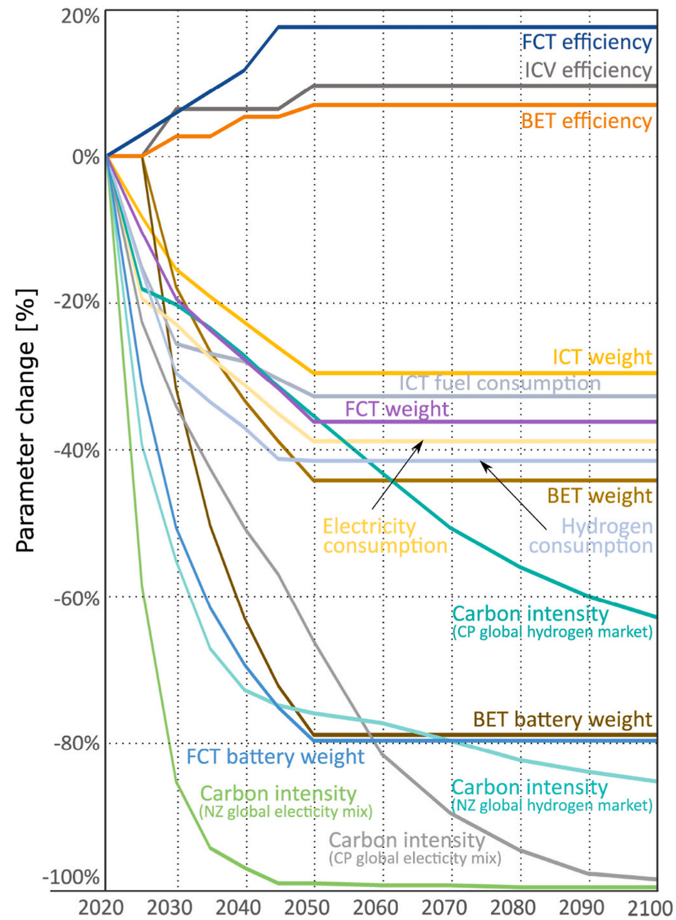


Fig. 3. Change in selected parameter values used to model the global road freight transport sector over time. ICT efficiency: Efficiency from tank-to-wheel [%]; BET efficiency: Efficiency from battery-to-wheel [%], FCT efficiency: Efficiency from tank-to-wheel [%] ICT weight: Total mass excluding driver and cargo [kg]; BET weight: Total mass of the truck excluding driver and cargo, but including batteries [kg]; FCT weight: Total mass of the truck excluding driver and cargo, but including tank and batteries [kg]; ICT fuel consumption: Tank-to-wheel energy consumption of ICTs [kJ/km]; Electricity consumption: Battery-to-wheel energy consumption of BETs [kJ/km]; Hydrogen consumption: Tank-to-wheel energy consumption of FCTs [kJ/km]; Battery weight: Total battery mass in a truck [kg]; Carbon intensity (CP): GHG emissions of the corresponding activity under current policies, [kg $CO_{2eq.}$/MWh] for electricity mixes, [kg $CO_{2eq.}$/kg H_2] for hydrogen market; Carbon intensity (NZ): GHG emissions of the corresponding activity under current policies, [kg $CO_{2eq.}$/MWh] for electricity mixes, [kg $CO_{2eq.}$/kg H_2] for hydrogen market.

We start by modelling LCA activities for BD and HVO production from various feedstocks. This is done by combining LCA models to produce vegetable oils from different biomass sources with models for transforming these oils into either BD through transesterification or into HVO *via* hydrodeoxygenation.

For the former, we consider four biomass feedstocks: soybean, canola, palm, and microalgae. The first three feedstocks are selected because they are state-of-the-art sources for producing HVO [42]. In the case of microalgae, we highlight its lower land and water use requirements [43,44]. The LCA models for producing vegetable oils from soybean, canola, and palm are readily available in the Ecoinvent

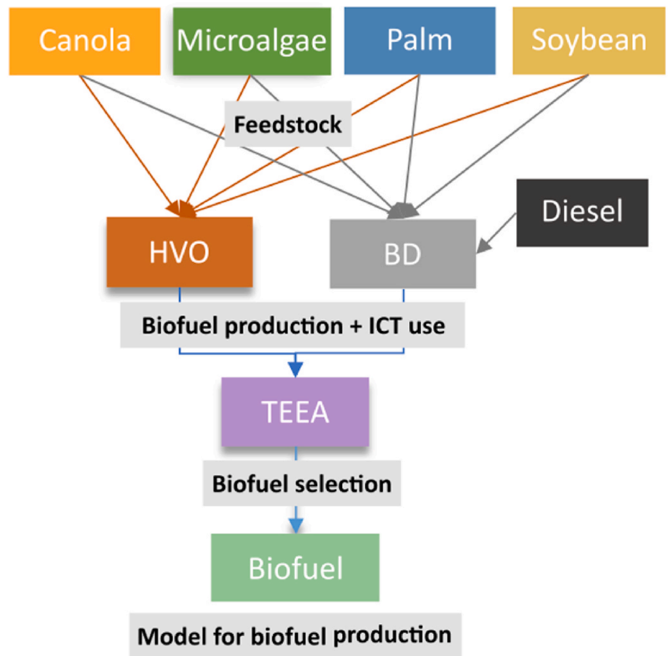


Fig. 4. Process flow diagram for the selection of the best performing biofuel. HVO: Hydrotreated vegetable oil; BD: Biodiesel; TEEA: Technoeconomic and environmental assessment.

database, while for microalgae oil, we source the LCA data from the literature [30,44].

Then, process simulation is employed to quantify the energy and material requirements for the two routes considered for biofuel production. These simulations, outlined in Figs. 5 and 6, are developed in Aspen Plus v12 based on the data from Apostolakou et al. [45] and Lee et al. [46] for BD production [45,46] and on data from Huo et al. [47] for the hydrodeoxygenation process [47]. The two simulations use triolein to mimic vegetable oil, regardless of the biomass feedstock [46,48,49]. This assumption, which allows us to perform only two (instead of eight) simulations, is underpinned by the fact that triolein is one major component of fatty acids for various vegetable oils [50,51], representing around 40–80 % of their mass [52]. Both simulations are optimized to obtain the design and operating conditions yielding the minimum total annualized costs, i.e., considering both investment and operation, as will be explained next. Energy integration (not shown in Figs. 5 and 6 for simplicity) is also applied to further improve the designs.

The outputs of these simulations are used to evaluate the economic and environmental performance of these two transformation routes. Specifically, biofuel production costs are calculated following the methodology proposed by Towler and Sinnott [53]. According to this, the total capital investment (TCI, in USD) is determined with Eq. (1), as given by the sum of the total direct costs and the total indirect costs:

$$TCI = ISBL_{2021} + W_{cost} + SD_{cost} + AP_{cost} + OSBL \tag{1}$$

Direct costs are determined by what is known as ISBL (Inside Battery Limits), which refers to the cost of procuring and installing all the process equipment that constitutes the new plant, including major process equipment, bulk items (such as piping, valves, wiring, instruments, structures, etc.), civil works, and installation costs. To determine this, individual equipment costs are calculated, and installation factors are applied to obtain the installed cost of the equipment.

The ISBL cost is then updated from the reference year (2015) to the present (2021) using the Chemical Engineering Plant Cost Index (CEPCI) (CEPCI values of 532.9 and 720.0, respectively). Warehouse costs (4 % of $ISBL_{2021}$), site development costs (9 % of $ISBL_{2021}$), and additional piping costs (4.5 % of $ISBL_{2021}$) are also included [54].

To determine indirect costs, the OSBL (Outside Battery Limits) is calculated. This represents the interactions with utility plants, such as electricity and water suppliers. For typical chemical projects, these costs are estimated to be between 20 % and 50 % of the $ISBL_{2021}$ costs (in this study, 29 % is assumed). Table S3 provides a summary of the TCI calculation.

Finally, the TCI is annualized as shown in Eq. (2), where R is the annual equivalent cost (USD/yr), i is the interest rate (6 %), and n is the number of years (15).

$$R = \frac{TCI}{\frac{(1+i)^n - 1}{i(1+i)^n}} \tag{2}$$

On the other hand, the OPEX (Operational Expenditure, in USD/yr) is calculated by considering the raw materials and utilities required, based on so-called variable and fixed operating costs (Eq. (2)). In addition, co-products are assigned a market price, which is deducted to

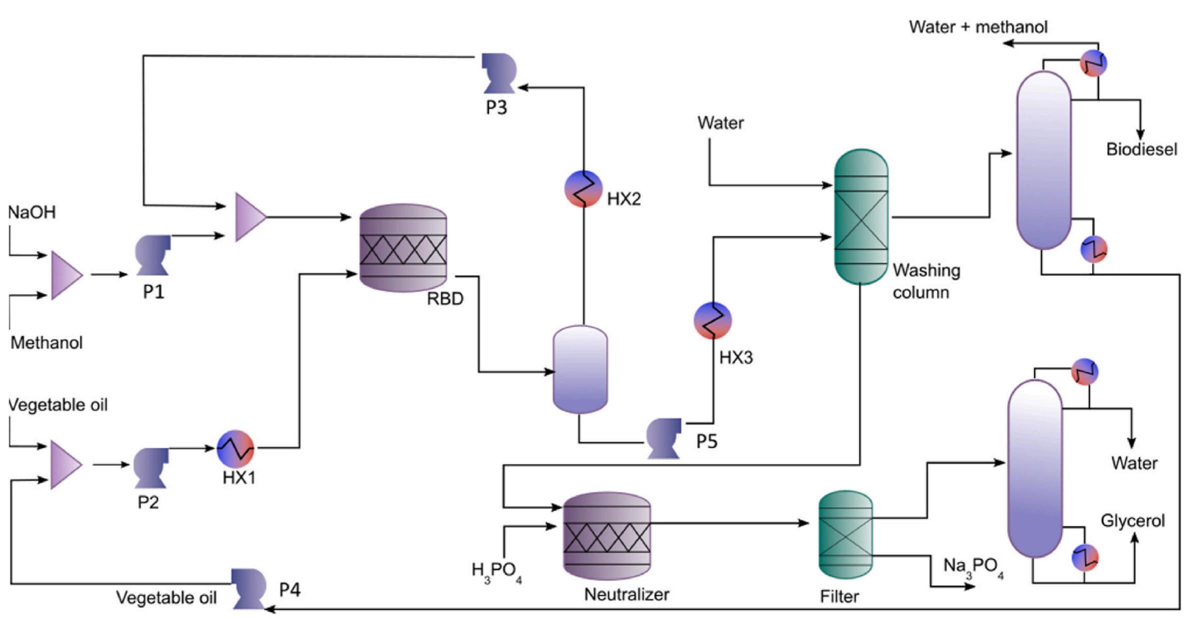


Fig. 5. Simplified process flow diagram for biodiesel production.

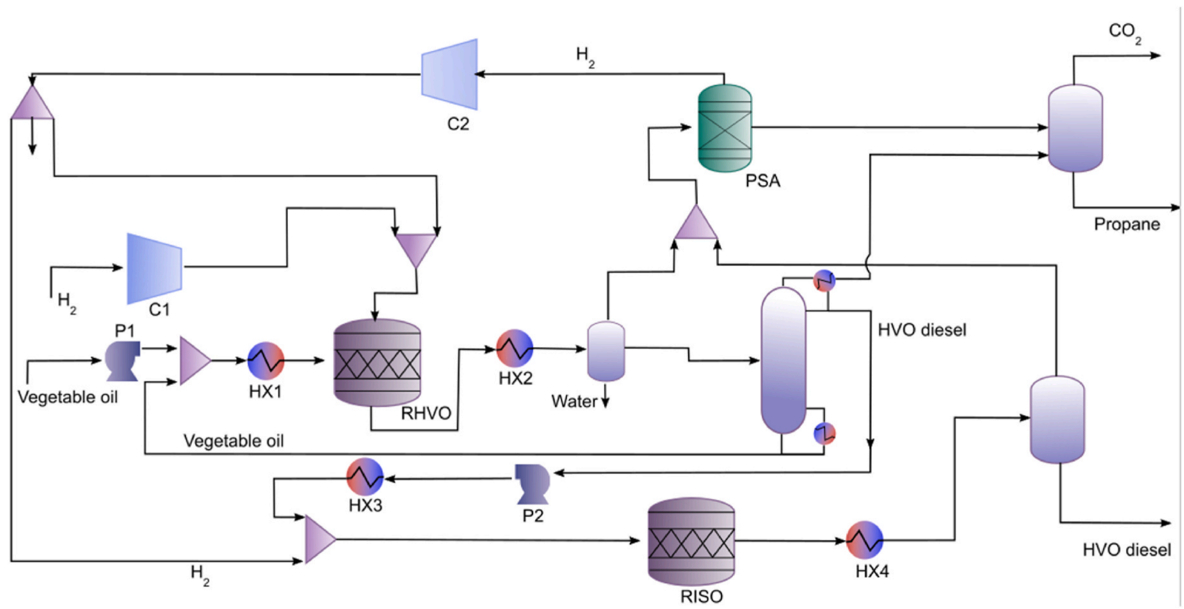


Fig. 6. Simplified process flow diagram for hydrotreated vegetable oil production.

obtain the final OPEX. These values are obtained assuming a production of 5273.4 kg/h of BD and 4347.8 kg/h of HVO during 8000 h of operation per year, as shown in Table S3. Specific values for the other variables and key assumptions used in the calculations are summarized in Tables S4–S6, while Table S7 provides a summary of the total biofuel production cost.

$$OPEX = Variable_{operating\ cost} + Fixed_{operating\ cost} - Revenues_{co\text{-}products} \quad (3)$$

The final biofuel product cost is determined by adding the OPEX obtained with Eq. (3) with the annualized capital investment (R) calculated using Eq. (2).

On the other hand, the environmental performance is based on LCA for the global region in 2020. Economic allocation is used to distribute the environmental impacts among the different co-products generated during the transesterification and hydrodeoxygenation processes.

With the results from the TEEA at hand, we finally screen the eight biofuel alternatives based on their GWP and production costs, selecting only the best candidate for further comparison with dICTs, BETs and FCTs. Results for this selection process are briefed later in section 3.1, while further details can be obtained from in the Supplementary material [40]. Finally, the selected alternative is used to develop bioICT models for the 12 years considered by modifying the existing dICT models from the databases created before.

2.4. Prospective life cycle assessment

Analogous to conventional LCA, p-LCA also follows the four-step framework outlined in ISO 14040 and 14044 standards [55,56].

In the first phase, the goal and the scope of the study are defined. The purpose of this contribution is to evaluate the impact of different alternatives for the RFT sector from cradle to grave, that is, considering impacts from activities dedicated to fuel (or energy source) production and distribution (i.e., cradle/well-to-tank), plus those incurred during the use phase in the vehicle (tank-to-wheel, i.e., combustion in the case of ICTs, electricity consumption in the case of BETs, and electrochemical reaction for FCTs). Note that we also cover fugitive emissions from brakes/tires and downstream emissions stemming from the end-of-life alternatives for the vehicle components (i.e., to-grave). This assessment is carried out for two possible future scenarios (i.e., CP and NZ) and 12 different years spanning from 2020 to 2100. In all the cases, the functional unit is the total demand for RFT activities expressed in kilometers traveled times ton transported per year (tkm/yr).

The second phase of the LCA identifies and quantifies the so-called life-cycle inventories (LCIs), i.e., the emissions, waste generated, and feedstock requirements associated with each RFT option p along its life cycle. For the case of dICTs, BETs and FCTs, LCIs are directly retrieved for the different years from the LCA databases created with *premise*. For bioICTs, mass and energy balances from the process simulations and previous studies [24,36] are translated into the corresponding LCIs using specific activities from these databases (see Tables S8-S9). This can be seen in Eq. (4), where the LCIs associated with resource or emissions e (LCI_e, e.g., kg CO_2 emitted per tkm using a bioICT) are obtained from the net consumption of raw materials (RM_r) and utilities (UT_u) obtained from the process simulations, and the corresponding LCI entry per unit of raw material and utility ($\omega^{RM}_{e,r}$ and $\omega^{UT}_{e,u}$, respectively), as sourced from the databases. Note that some of these parameters also vary depending on the year t (e.g., t = 2030), the geographical location g (e.g., g = Europe), and the corresponding IAM scenario s (e.g., s = Net Zero policies) considered in each case.

$$LCI_{e,g,p,s,t} = \sum_r RM_{r,p}\omega^{RM}_{e,r,g,s,t} + \sum_u UT_{u,p}\omega^{UT}_{e,u,g,s,t} \quad \forall e,g,s,t,p = bioICT \quad (4)$$

The third phase of the LCA involves assessing the damage produced by the LCIs into different environmental categories. This can be done with the state-of-the-art impact assessment model, ReCiPe 2016 [57], which allows to calculate impacts on different midpoint and endpoint categories. Among all the options available, we focus on three categories that previous research deemed key for the transport sector: global warming potential (GWP) [58,59], mineral resource scarcity (MRS) [60–62], and human health (HH) [63,64]. The rationale behind this choice is explained next.

RFT has a substantial impact on climate change, primarily due to its reliance on fossil fuels. Incorporating the GWP indicator in our analysis allows us to quantify the climate benefits of transitioning to more sustainable options in the RFT. BETs and FCTs offer the advantage of not having exhaust gases; however, the generation of electricity used to charge batteries and the production of hydrogen used in the fuel cells may introduce hazardous compounds. Including the HH indicator allows us to compare the impact of combustion gases with those associated with electricity and hydrogen production, as well as other non-exhaust emissions. Finally, it is crucial to consider the environmental challenges associated with the manufacturing, use, and disposal of batteries for BETs and FCTs, which involve materials such as nickel, lithium, cobalt, copper, and rare earth metals. To this end, MRS will be applied [60–62].

Eq. (5) is used to determine the impacts caused by each RFT alternative p on the three environmental categories b considered, in every region g, for every year t, and scenario s ($IMP_{b,g,p,s,t}$). To this end, we multiply LCIs obtained from Eq. (4) (e.g., kg CO_2 released per tkm transported) by a characterization factor ($CF_{b,e}$) translating LCIs into impacts on each category b. Characterization factors for the three indicators are based on a hierarchical perspective, which integrates environmental impacts over a 100-year time horizon. Finally, yearly volumes of load transported in each region, year and scenario ($TT_{g,s,t}$, in [tkm/yr]) are used to scale results up to the functional unit.

$$IMP_{b,g,p,s,t} = \sum_{e} LCI_{e,g,p,s,t} CF_{b,e} TT_{g,s,t} \quad \forall b, g, t, s, p \tag{5}$$

The yearly volumes of load transported utilized in Eq. (5) are obtained as follows. We first retrieve the energy consumption of the sector ($TTECT_{g,s,t}$, in [EJ/yr]) for the period 2020–2100 from the category labeled "Final Energy-Transport-Road" in REMIND [41]. Then, this value is divided by the current RFT energy consumption ($EC_{p,t}$, in [kJ/tkm]) of a typical dICT (see Eq. (6)). We consider a weighted average (75%–25 %) [65] between the fuel consumption of a long haul with a payload of 14 t (i.e., 729 kJ/tkm for 2020), and that of a regional delivery truck with a payload of 2 t (i.e., 2989 kJ/tkm for 2020) [66]. This EC represents the total energy required per tkm.

$$TT_{g,s,t} = \frac{TTECT_{g,s,t}}{EC_t} \quad \forall g, t, s \tag{6}$$

With the total impacts for every region, year and alternative at hand ($IMP_{b,g,p,s,t}$), it is possible to ascertain the cumulative GWP savings ($\Delta IMP_{b,g,p,s,t}$) that could be achieved up to a certain year t, by replacing dICTs with each of the sustainable alternatives (embedded in set SUS in Eq. (7)). Specifically, we consider four of such alternatives: (i) bioICTs, (ii) BETs, (iii) FCTs, (iv) and a utopic scenario where we deploy the best of the three options in every year, region and policy. It is this last alternative that allows us to explore the potential benefits on global warming of the timely introduction of appropriate environmental policies in each region.

$$\begin{gathered} \Delta IMP_{b,g,p,s,t} = \sum_{t'=2020}^{t} IMP_{b,g,p,s,t'} - \sum_{t'=2020}^{t} IMP_{b,g,p',s,t'} \\ \forall b = GWP, g, s, p = dICT, p' \in SUS, t = 2020, \dots, 2100 \end{gathered} \tag{7}$$

2.5. Limitations

This study conducts a comparative assessment of three alternatives for decarbonizing the RFT sector, yet it presents some limitations that are outlined next.

Firstly, we generate future LCA databases for the background activities following two possible scenarios from a specific IAM, REMIND. However, other models or scenarios might prioritize different technologies for energy generation and product manufacturing. This would result in different LCI values for the background activities that could potentially affect the conclusions drawn, although these are expected to maintain a similar trend to those observed in this study.

Similarly, a specific truck size is considered here to model the truck fleet, which can affect both fuel consumption and final emissions. Higher technological resolution, as explored elsewhere [26], could potentially lead to different results.

Along the same lines, we limit ourselves to studying a complete substitution of the truck fleet to capture the effects of the large-scale implementation of each technology. Even in the case of the utopic scenario, only a specific truck technology is allowed in every year. In practice, more than one technology will typically coexist until inferior options disappear from the market completely. Modelling technological diffusion entails greater complexity and would introduce additional uncertainty into the calculations. Hence, we stick to the simplest approach with the capacity of informing on the advantages and disadvantages of each technological option available in every region and in every time period.

On the other hand, biofuel production can vary according to the type of biomass used, for which we only consider a limited number of alternatives. In addition, the same crop yield is assumed for all the regions studied. This decision is based on the observation that the geographical variation of crop yields contributes marginally to GHG emissions of bioICTs (around 5 %, see section 3.2), which does not justify modeling regions separately. Other parameters, such as water and soil requirements, also vary across regions in practice, but are not included in this study due to the lack of specific data for all the regions considered.

For the proposed alternatives, we use the electricity mix and the hydrogen market, including the generation of renewable energy inside this parameter. In this regard, the choice of different technological options (e.g., the use of green hydrogen instead of the whole market) could potentially affect some results. This last issue is briefly discussed later in section 3.2.

Resource availability is not taken into account, implicitly assuming that the reserves of minerals, land, water, and metals will be sufficient to meet the demand from the proposed alternatives. However, previous studies stress that this might not be the case under certain assumptions [67,68].

Finally, we only consider the cost in the selection of the candidate biofuel and, even in this case, we stick to current values. Hence, we deliberately avoid undertaking a prospective economic evaluation of the different technologies to prevent the introduction of additional uncertainty. However, we acknowledge that economic considerations will have a strong impact on the decision to favor or hinder a specific technology in a given territory, together with other logistic aspects, also neglected here (e.g., the possibility to maintain or not the refueling infrastructure or the truck fleet).

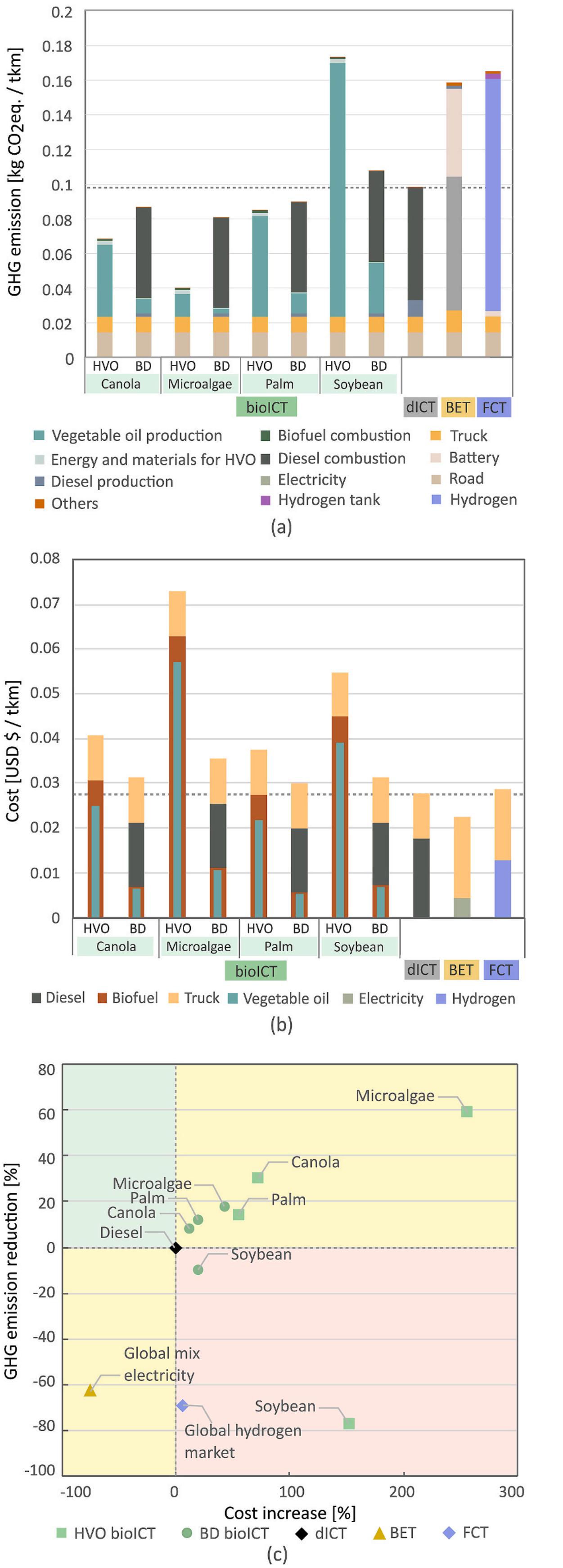
3. Results

3.1. Selection of the candidate biofuel

Fig. 7 shows the results obtained from the TEEA of the eight biofuel alternatives for bioICTs (see section 2.3). Results from dICTs, BETs and FCTs are also depicted for the sake of comparison. This analysis will allow us, not only to select the most promising biofuel, but also to provide more insight into the drivers behind the performance of the different options for the RFT sector.

As a general trend, we observe that bioICTs based on BD have relatively similar performance among them, and not far away from that of diesel. This is because these options are limited by the amount of BD that can be included in the blend. As an example, bioICTs fueled with canola BD and palm BD only improve the GWP of dICT by 12 % and 8 %, respectively, since the emissions from the diesel in the blends already account for 48 % and 64 % of the total GWP of these options (Fig. 7a). In turn, the cost increase for BD bioICTs is also more modest than for other options, e.g., 13 % and 8 % when using canola and palm BD (Fig. 7b). The case of soybean BD is also worth mentioning, since this crop achieves worse environmental (and economic) performance than diesel. This is primarily attributed to the farming stage, where factors such as a high land-use, machinery use, and transportation, penalize this alternative [69].

The fact that HVOs can be used alone (i.e., without diesel) in bioICTs is translated into a more distinct performance among themselves and compared with dICT. For instance, vegetable oil production represents 84 % of the total GWP for soybean, while it only accounts for 33 % of the total emissions for microalgae-based HVO. From a GWP perspective, bioICTs based on microalgae HVO are the best, followed by canola and



(caption on next column)

Fig. 7. Analysis of options for the road freight transport sector. (a) Breakdown of the main drivers of GHG emissions from cradle-to-grave. (b) Breakdown of the costs of the different alternatives, including fuel cost and vehicle cost, with an internal green bar providing the cost related to the production of the vegetable oil (biofuels only). (c) GHG emissions per tkm transported (y-axis) relative to diesel, versus the cost per tkm with respect to the cost of diesel (x-axis). GHG: greenhouse gases. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

palm HVO, and with soybean emerging as the worst performing HVO. BioICTs based on microalgae HVO stand out for their low GWP (60 % lower than diesel), which is achieved by combining the capture and storage of the carbon contained in the lipid-extracted algae residue, and the use of carbon from direct air capture as feedstock [44]. However, the production costs of algae HVO, 250 % higher than diesel, might limit the use of this alternative. On the other hand, bioICTs fueled with canola or palm HVOs demonstrate significant reductions in the GWP (30 % and 14 %, respectively), yet these options also present some economic challenges related to the production of vegetable oil, which can represent between 72% and 86 % of their total costs.

Hence, as can be seen in Fig. 7c, there is no single option improving dICTs simultaneously in cost and emissions (i.e., occupying the green quadrant), which suggests that any effort to make the sector more sustainable will entail economic efforts. When balancing out advantages and disadvantages, we find that bioICTs using canola HVO and palm BD achieve the largest ratio of GWP reduction to cost increase (0.65 and 1.1, respectively). Among these two options, bioICTs based on canola HVO demonstrate a superior capacity to reduce GHG emissions and, therefore, is the option selected as the bioICT for subsequent analyses.

In addition, we also study the performance of BETs and FCTs, and compare them with that of dICTs and bioICTs. We find that, although BETs and FCTs do not emit exhaust gases, their GWP is 50 % and 60 % larger than for dICTs, respectively, in 2020. In the former case, the main drivers for this are the electricity used as a power source and the production of the batteries, contributing 49 % and 32 % of the total lifecycle GHG emissions, respectively. In the case of FCTs, hydrogen production contributes 82 % of total lifecycle GHG emissions (Fig. 7a). In addition, results evidence that electricity is 75 % cheaper than the fuel cost for dICTs (Fig. 7b), yet this advantage diminishes when considering the total costs of BETs (i.e., including vehicle costs). In this case, BETs are still the most economical option for RFT, yet only 20 % cheaper than dICTs, and 45 % cheaper than bioICTs based on canola HVO. We also note that, although the market price of FCTs is lower than that of BETs, current hydrogen prices make this alternative to have 4 % higher costs than dICTs as a whole.

The Supplementary material contains further details on the techno-economic and the environmental assessment of all these options [40].

3.2. *Evolving impacts of road freight transport under different policies and regions*

Impacts from the RFT sector are expected to change dynamically over time due to the effect of variable factors like technological advancements, the evolution of electricity mixes and the penetration of different hydrogen production methods, which, in turn, have a strong regional component. Hence, this section will explore the reduction in GHG emissions that can be achieved by satisfying the demand for RFT with bioICTs, BETs or FCTs, rather than employing dICTs as mostly done today (Fig. 8). In the case of BETs and FCTs, specific results are presented for the different regions studied, as significant disparities are observed between territories in variable parameters such as the electricity mix, or the technologies used for hydrogen production. Conversely, only a global analysis is shown for bioICTs since, in our study, most parameters affecting their performance vary in the same magnitude across regions. An exception is the carbon intensity of the electricity mixes and hydrogen markets, which do differ regionally (see

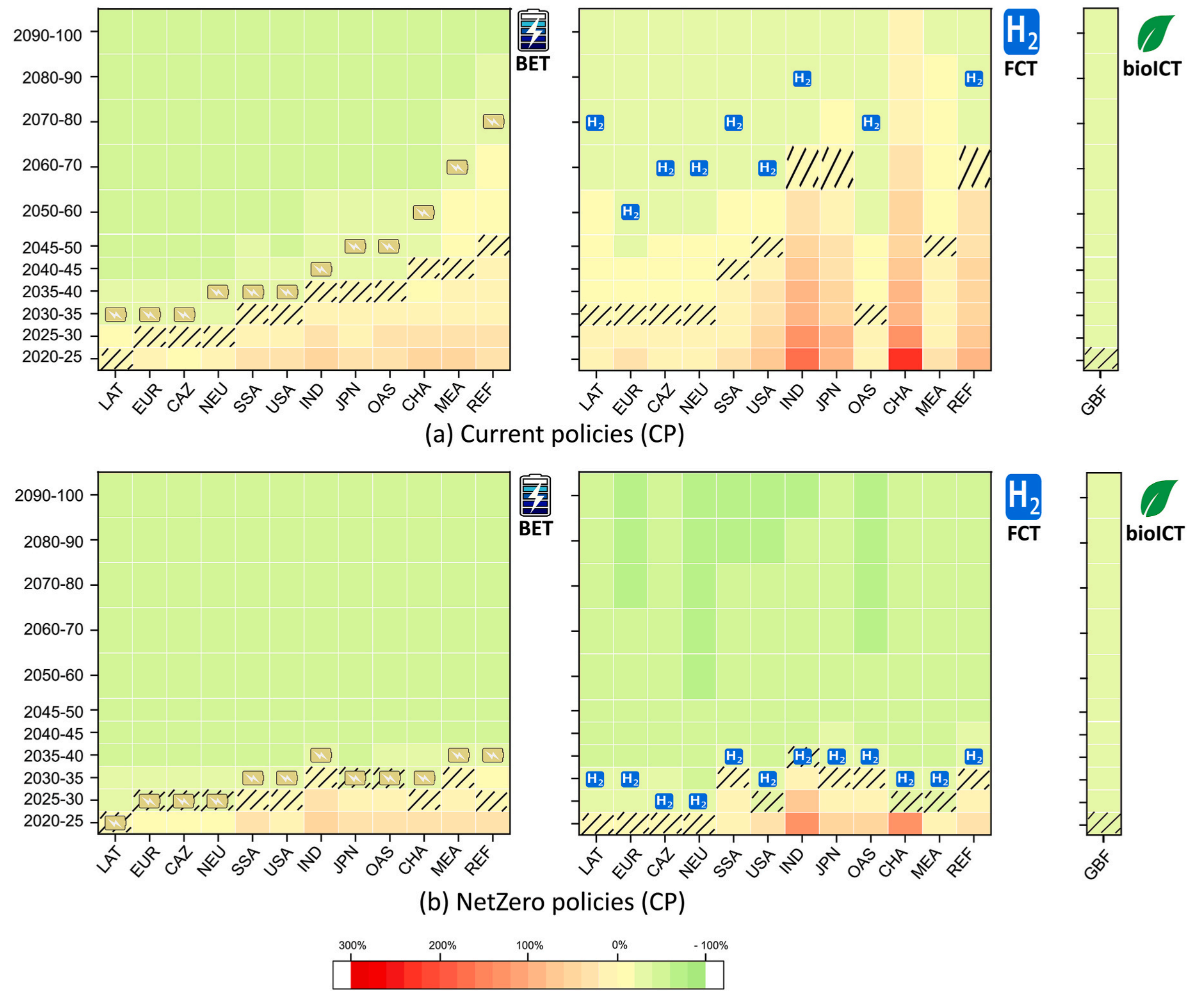


Fig. 8. Change in GHG emissions from road freight transport resulting from the substitution of dICTs by bioICTs, BETs or FCTs. The heat map illustrates the amount of GHG emission that would be avoided by replacing diesel trucks with electric, hydrogen or biofuel trucks under CP (subplot a) and NZ (subplot b) policies. The first twelve columns of the map depict emission reductions achieved with an electric truck (BET) for different regions, while the next twelve columns refer to hydrogen fuel cell trucks (FCTs). The final column, labeled "GBF," provides analogous information for biofuel trucks (bioICT) and represents worldwide results. In all the cases, cells marked with diagonal lines indicate the period in which GHG emissions per tkm from BETs, FCTs, or GBF become lower than those from dICTs, while the small battery (for BETs) and hydrogen (for FCTs) symbols indicate the period in which they become lower GHG emitters per tkm than bioICTs. GBF: Global biofuel (i.e., based on canola HBO); Region acronyms as in Table 1.

Tables 1 and 2). However, as explained before, this variation was found to contribute marginally to GHG emissions of bioICTs (i.e., around 5 %), which it does not justify modelling regions separately.

Results for the CP scenario (Fig. 8a) reveal that, by 2030, only four regions would exhibit lower GHG emissions (10%–19 %) by employing BETs instead of dICTs. These are Latin America (LAT), the European Union and United Kingdom (EUR), Canada, Australia and New Zealand (CAZ), and the rest of Europe (NEU). Among them, LAT and CAZ stand out for having a high share of renewable energy in their electricity mix since 2020–2025 (i.e., 76 % and 71 %, respectively), with a significant contribution from hydro (58 % and 48 %) (see Tables S12–S23 in the Supplementary material [40]). This makes BETs a better alternative than dICTs in these regions early in the period studied.

Results also suggest that the remaining eight regions are not prepared for the wide deployment of BETs yet. Regions like China and the Middle East show electricity mixes dominated by fossil sources in 2020–2025 (65 % in China and 92 % in Middle East) and would need to wait until 2040–2045 before BETs show lower GWP than dICTs. By then, their electricity mix would have 57 % (China) and 30 % (Middle East) renewable sources under CP.

Among all the regions studied, reforming economies (REF) is where BETs will take the longest to surpass dICTs in terms of GHG emissions (2045–2050). This region, with countries like Russia or Uzbekistan, departs from an electricity mix composed mainly of coal (26 %) and natural gas (33 %), and it will take until 2045–2050 to replace coal with additional natural gas. This explains why the reduction in GHG emissions obtained in 2100 under CP by substituting dICTs for BETs is less significant in REF than in other regions (e.g., 42 % in REF versus 50 % in EUR). Overall, each region examined follows a different ongoing process for decarbonizing electricity generation, which results in diverse electricity mixes over time and, ultimately, in divergent timelines before BETs can improve the GWP of dICTs. This reflects the importance of coordinating efforts between the different facets of the energy sector.

Regarding hydrogen FCTs, results show that they currently (i.e., 2020) have the highest GHG emissions among all the options considered, including dICTs, which confirms previous observations at the global level (Fig. 7a). This is due to the high share of fossil hydrogen in the market, with 42 % coming from natural gas, 27 % from coal and 22 % from oil, globally. In regions like China (CHA), where two-thirds of the hydrogen market is coal-based, emissions from FCTs in the 2020–2025 period are expected to be 2.2 times higher than those from dICTs, and FCTs remain the worst option until the end of the century. Actually, under CP, FCTs have the highest GHG emissions in all the regions until, at least, 2030, and it would take until 2070 to revert this

Table 1
Carbon intensity of electricity mix according to CP and NZ polices based on REMIND IAM.

		Current policies (CP) [kg CO_{2eq}/kWh]			NetZero (NZ) [kg $CO_{2\,eq}$/kWh]		
Region	**Acronym**	2030	2050	2100	2030	2050	2100
Canada, Australia, and New Zealand	CAZ	147	63	63	69	60	51
China	CHA	468	263	104	121	61	64
European Union and UK region	EUR	210	82	76	113	72	56
India	IND	503	170	60	342	60	63
Japan	JAP	566	396	105	349	140	75
Latin America	LAT	173	92	136	122	85	52
Middle east	MEA	498	382	56	315	57	60
Non-EU Europe countries	NEU	182	93	62	70	56	50
Other Asian countries	OASIA	436	205	56	256	54	57
Reforming economies	REF	516	506	152	258	95	62
Sub-Saharan Africa	SSA	313	136	71	168	60	61
USA	USA	319	100	75	144	87	62

Table 2
Carbon intensity of hydrogen gaseous market according to CP and NZ polices based on REMIND IAM.

		Current policies (CP) [kg CO_{2eq}/kg H_2]			NetZero (NZ) [kg CO_{2eq}/kg H_2]		
Region	**Acronym**	2030	2050	2100	2030	2050	2100
Canada, Australia, and New Zealand	CAZ	10.4	9.3	8.9	5.4	5.8	3.0
China	CHA	29.2	23.3	13.7	8.0	4.5	3.5
European Union and UK region	EUR	11.8	8.9	7.6	6.7	4.9	1.7
India	IND	33.3	20.4	7.2	22.8	4.2	2.0
Japan	JAP	23.4	17.8	10.0	13.3	6.5	4.9
Latin America	LAT	12.3	10.3	8.3	7.1	5.5	4.4
Middle east	MEA	12.6	12.3	9.4	7.4	4.9	4.6
Non-EU Europe countries	NEU	10.3	9.5	8.7	5.7	2.9	1.6
Other Asian countries	OASIA	21.6	18.2	7.8	12.3	4.2	1.8
Reforming economies	REF	14.0	13.1	12.1	13.0	6.0	5.6
Sub-Saharan Africa	SSA	12.6	11.1	7.7	12.0	4.9	2.0
USA	USA	19.6	12.6	8.1	6.6	4.6	2.4

trend in regions such as India, Japan or REF. Only by then, all hydrogen markets will exhibit a share of non-fossil hydrogen of, at least, 9 %.

On the other hand, the penetration of clean hydrogen will progress faster in certain regions like EUR, where FCTs are expected to achieve lower GHG emissions than bioICTs after 2050. By then, at least 25 % of hydrogen production will come from water electrolysis and biomass. Despite this, FCTs never become the least emitting option in this or any other region, as this privilege always belongs to BETs under CP.

The situation is very different under the NZ policy scenario (Fig. 8b), where BETs achieve lower GHG emissions than dICTs in all the regions by 2035, at latest. By then, the share of renewable would be, at least, 52 % in any region. This translates into GHG emissions 12%–37 % lower for BETs than for dICTs in all the regions: similar results to those under CP by 2050 (i.e., 15 years earlier).

A similar trend is observed for hydrogen FCTs thanks to the early adoption of lower-emitting technologies for hydrogen production, namely biomass gasification with carbon capture and storage (CCS) (market share between 29% and 65 % by 2035), water electrolysis (3%–75 %), and steam methane reforming with CCS (3%–78 %). Under NZ policies, hydrogen FCTs become the best alternative in regions like EUR starting from 2070 onwards.

As a curiosity, the case of EUR in the year 2100 allows the comparison between BETs and FCTs in a renewable fuel scenario, since 97 % of the hydrogen in the market comes from electrolyzers powered by a 100 % renewable electricity mix, which, in turn, is also used to charge the batteries in BETs. In this case, we observe that FCTs can achieve a 32 % lower GHG emissions than BETs, which contrasts with the results from the CP scenarios, where the performance of hydrogen FCTs was found to be 26 % worse than that of BETs for the same region and year.

Interestingly, our results demonstrate that bioICTs will have lower GHG emissions than dICTs throughout the century, regardless of the evolution of environmental policies (i.e., for both CP and NZ scenarios). An instant replacement of diesel by canola HVO, if possible, could reduce the GHG emissions from the RFT sector by 31 % as early as in 2020–2025. In fact, bioICTs are the best alternative in most regions during 2020–2025, emitting 29 % less GHGs than BETs and 30 % less than FCTs when considering all the regions simultaneously under CP. BioICTs also remain better than hydrogen FCTs for the whole century in Middle East and China under CP.

This situation changes over time as electricity mixes and hydrogen production decarbonize worldwide. By 2100, BETs become a very solid alternative, emitting, at least, 45 % less than dICTs and 29 % less GHGs than bioICTs regardless of the policy scenario. In contrast, policies play a fundamental role for FCTs, as they cause this alternative to go from improving dICTs by 46–67 % under the NZ scenario (surpassing also BETs and becoming the best alternative), to underperforming compared to bioICTs in MEA and China under CP. Hence, the early enforcement of more stringent policies for the power sector and hydrogen production will clearly favor a quick transition towards BETs and FCTs, while increasing the amount of GHG emissions avoided by the substitution. In turn, this also suggests that biofuels are a very promising interim solution as an electricity-based system for RFT is developed.

This idea is further explored in Fig. 9, where we show the evolution of the annual GHG emissions for the four options considered for the RFT for the European Union and United Kingdom (EUR), and China (CHA) under CP. A decreasing trend in GHG emissions is observed in both regions for all the alternatives. Although this is partially due to the decrease in the demand for RFT activities anticipated by REMIND [28], the improvement in technological parameters such as fuel consumption for ICTs, the cleaner electricity mixes for BETs, or the decarbonization of hydrogen production for FCTs also contribute to this trend.

According to our results for the EUR, BETs would represent a lower-GWP alternative to dICTs starting sometime before 2025. At that time, nonrenewable sources of the electricity mix are expected to represent less than 12 % of electricity generation. As a result, if CP persists, it will take until 2025–2030 for the cumulative GHG emissions released by BETs to be lower than those from dICTs. This means that an early widespread adoption of BETs (e.g., starting in 2020) would result in cumulated savings of 0.62 Gt CO_{2eq} emissions compared with the continuous use of dICTs by 2035 (see bars in the bottom part of the figure). A similar situation is observed for hydrogen FCTs, which remain a dirtier alternative than dICT until around 2030 and need to wait until 2045–2050 to outperform dICTs in terms of cumulative GHG emissions due to their poor performance during the earlier years.

On the other hand, bioICTs always perform better than dICTs: 31 % better in 2020 and 28 % by 2100. However, bioICTs are outperformed by BETs in 2035–2040 and by hydrogen FCTs in 2060–2070. By 2100, BETs would emit 44 % less GHGs than bioICTs per tkm. This means that the hypothetical replacement of all the dICTs in EUR by bioICTs or BETs in 2020, would translate into 6.6 and 9.1 Gton of CO_{2eq} saved, respectively, by the end of the century. Under CP, hydrogen FCTs will never surpass BETs as the best alternative. However, starting from 2060, FCTs are expected to perform similarly to biofuels, achieving a 31 % reduction

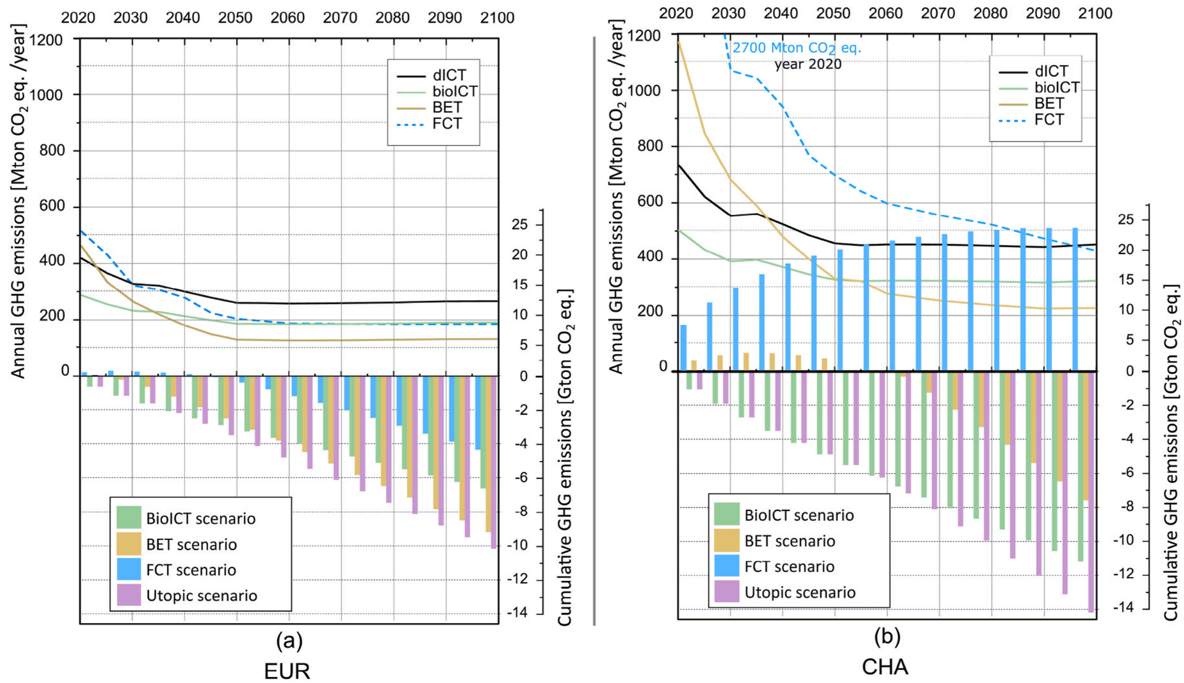


Fig. 9. Annual and cumulative GHG emissions in EUR (a) and CHA (b), under the CP scenario, using dICTs, bioICTs, BETs and FCTs. Lines in the upper y-axis (left-hand side) represent the annual GHG emissions of the four alternatives. Bars in the lower y-axis (right-hand side) depict cumulative GHG emissions avoided in each scenario with respect to those from the dICT scenario, grouped in 5-year periods.

in emissions compared to dICTs by 2100: equivalent to 4.2 Gton of cumulative CO_{2eq} emissions avoided, the lowest (i.e., more modest) among the three alternatives.

In this case, the utopic scenario would use bioICTs until the period 2030–2035, when they will be replaced by BETs, giving no use to hydrogen FCTs. This would allow to further increase in environmental savings up to 10.1 Gton CO_{2eq} avoided, 11 % larger than when using only BETs over the century. However, in practice, replacing the entire fleet of ICTs with BETs might take at least 20–25 years due to the long lifespan of vehicles [70,71]. Hence, these results suggest that the decarbonization foreseen in the CP of the European Union has already paved the way for the imminent adoption of BETs. In the meantime, any shift from dICTs to bioICTs would result in GHG emissions avoided in the range of 0–1 Gton CO_{2eq}. In turn, this would allow dedicating clean hydrogen to other applications such as the chemical industry [72].

The landscape would be different for EUR under the NZ scenario (Table S38). GHG emissions from FCTs equalize with those from BETs somewhere between 2020 and 2025 and remain similar until about 2060. After this, hydrogen FCTs improve at a faster pace and, by 2100, their GHG emissions become 31 % lower than those from BETs. This causes the utopic scenario under NZ policies to start with bioICTs until 2030, and then resorting to FCTs until the end of the century, although BETs could also be used during 2030–2060 with similar results (less than 10 % difference in the cumulative emissions of this period).

In China, the breakeven point between the GWP of dICTs and BETs under CP would only arrive during 2035–2040, when renewable sources based on solar, wind, and hydro would represent around 56 % of the electricity mix. This means a hypothetical replacement of all dICTs by BETs in 2020 would be counter-productive at first, resulting in larger cumulative GHG emissions early in the century, that would peak during 2030–2035 at 2.8 Gton CO_{2eq}. This is explained by the slow decarbonization expected for electricity under CP in China. Still, if BETs were maintained until the end of the century, 7.6 Gton CO_{2eq} would be saved in this region compared to the continuous use of dICTs.

The situation is even worse for hydrogen FCTs, since coal-based hydrogen in China accounts for 30 % of the worldwide hydrogen output in 2020. Indeed, the massive use of hydrogen from the market in the region, combined with a slow penetration of hydrogen production methods based on renewable sources under CP, means that this scenario does not offer any emissions savings compared to dICTs. Instead, the instantaneous replacement of dICTs by hydrogen FCTs in 2020 would be counterproductive, resulting in additional 24 Gton CO_{2eq} of GHGs released throughout the century.

BioICTs look more promising in this region, since the cumulative GHG emissions saved with this option by the end of the century would amount to 11.3 Gton CO_{2eq}, 33 % more than with BETs. This suggests that exploiting the current infrastructure and truck fleet through biofuels until the period where BETs have lower GHG emissions (utopic scenario) could significantly curb CO_2 emissions from the RFT sector. In the case of China, this would imply delaying the penetration of BETs until 2050–2055, reaching additional savings of 6.9 Gton CO_{2eq} compared with the BETs scenario (i.e., 23 % more emissions avoided), for a total of 14.6 Gton CO_{2eq} avoided over the century. These savings equal to 88 % of the emissions generated from 2020 to 2050 by dICT under CP.

If the utopic scenario under CP were adopted globally, by the end of the century, cumulative emissions from the RFT sector would be 135 Gton CO_2eq lower than with the continuous use of dICTs. To provide some context, this is equivalent to avoiding 0.22 °C of global temperature increase, which represents 10 % of the 2 °C limit proposed by the Paris Agreement [73] (see section S1in the Supplementary material for calculations [40]). This scenario is illustrated in Fig. 10, where we represent the RFT option with the lowest GHG emission in each of the 12 regions considered for different periods: 2020–2030 (panel a), 2030–2050 (panel b), 2050–2100 (panel c). Further results are provided in Tables S25–S36 in the Supplementary material [40].

This analysis reveals that, under CP, bioICTs would be the most favorable option in all regions during 2020–2030 (Fig. 10a). From 2030 onwards, the expected increase in the use of renewable energies over time (Tables S12–S23 in the Supplementary material [40]) will make BETs the most favorable option in seven regions (i.e., EUR, SSA, LAT, NEU, USA, CAZ, and IND). In the remaining five regions (i.e., REF, MEA,

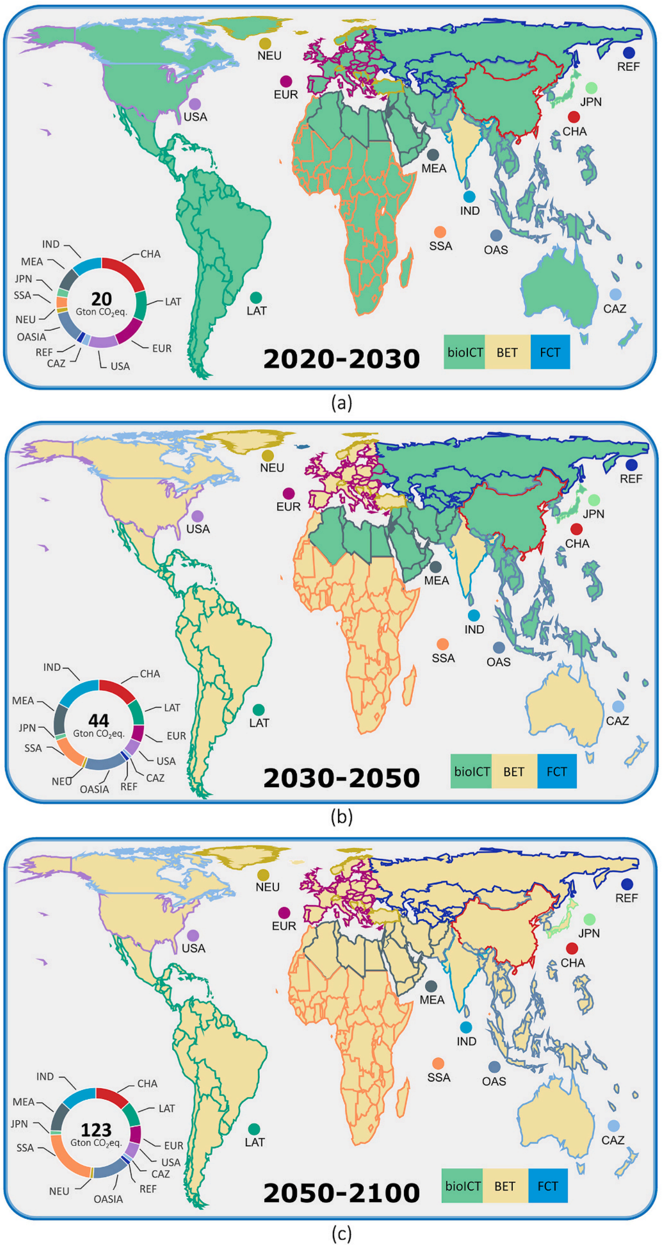


Fig. 10. Best alternative for road freight transport in each region based on GWP under the CP scenario. Each panel corresponds to a different period: a) 2020–2030, b) 2030–2050 and c) 2050–2100. The outline color of the countries indicates the region to which they belong, according to the circular icons that also provide the region acronym (see Table 1). In contrast, the fill color of the regions in each panel denotes the freight transport alternative that achieves the lowest GHG emissions in the region during that period: green for bioICTs, yellow for BETs, and blue for hydrogen FCTs. The pie charts in each panel provides the share of cumulated GHG emissions generated by each region during the corresponding period, with the inner number providing the global GHG emissions released worldwide. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

JPN, CHA, and OAS), this shift will only happen after 2050. As already discussed in Fig. 8, hydrogen FCTs would never become the most preferred option in any region under CP, although the faster decarbonization under the NZ scenario (Tables S25 to S36) will make it appear in EUR during 2025–2030, in IND during 2035–2040 and in regions such as JPN, where FCTs have a similar behavior to BETs in terms of GHGs, after 2060–2070. This would further increase the amount of GHGs avoided during the century compared to the continued use of dICTs to 204 Gton CO_{2eq}, which is equivalent to 0.33 °C of warming and 110 % higher than under CP.

The role played by certain regions in terms of emissions will change significantly throughout the century (i.e., see pie charts in Fig. 10). In 2020–2030, emissions from the RFT are dominated by China, which alone contributes 22 % of the cumulative emissions over the period. For context, the second region in terms of GHGs is EUR, which contributes almost half (13 %). However, China will drop to the fifth position in the last period (Fig. 10c), reducing its contribution from 30 % (2020–2030) to only 11 % (2050–2100).

At the other end of the spectrum, we find regions such as NEU, REF, CAZ, JAP and SSA, together responsible for 14 % of cumulative global emissions from the RFT sector during 2020–2030. However, as we move towards 2050–2100, SSA stops being one of the lowest contributors to become the largest emitter (30 %).

These changes in the share of emissions of each region are affected by population growth and increased freight activity. For instance, in SSA, the per-capita demand for RFT is expected to increase by 1200 % along the century: from 830 tkm/person in 2020 to 10380 tkm/person in 2100 [28]. This, combined with an anticipated population surge of up to 300 % by 2100 compared with 2020 [74], results in this region being responsible for 23 % of the total cumulative demand for RFT between 2020 and 2100. This underscores the significance of directing efforts toward decarbonization initiatives in the SSA region.

Conversely, in the EUR, the per-capita demand for RFT is expected to rise from 9630 tkm/person to 16040 tkm/person in the same period, yet this would be combined with a slight population decline of 7 % [75]. Overall, this translates into a remarkable change in the role of EUR in terms of cumulative GHG emissions: from being responsible for 13 % over the period 2020–2030, to representing only 7 % for 2050–2100 (and 8 % when 2020–2100 is considered).

Previous results correspond to the utopic scenario, yet the situation will be very different if regions ignore the policies that promote the integration of bioICTs, BETs and FCTs, and stick to the continuous use of dICTs. Following with the example of EUR, this would translate to a 68 % increase in GHG emissions by the end of the century. In this context, we note that the Green Deal stands out as a key initiative within the current policy framework of the European Union. This plan aims at reducing direct GHG emissions in the RFT sector by 40 % compared to 2005 levels by 2030, and by 98 % by 2050 [62]. Notably, the Green Deal assumes no "direct emissions'" are released from EVs owing to the absence of tailpipe emissions. Hence, considering direct emissions from dICTs and the demand for RFT for 2030 and 2050, meeting the proposed targets would require the integration of, at least, 26 % and 88 % of BETs by 2030 and 2050, respectively, in the vehicle fleet.

It is essential to recognize that although BETs show no exhaust emissions during operation, their overall life cycle still contributes to GHG emissions. Indeed, fulfilling the proposed energy demand of BETs to reach the Green Deal target under CP would result in an increase of 143 TWh of energy demand and 38 Mton CO_{2eq} emissions in 2030 considering the cradle-to-grave approach in the EUR. This represents 11 % of the GHG emissions that would be incurred in the same region and year using only dICTs, which highlights the significance of considering life cycle approaches for policy-making.

The results shown in Fig. 10 offer guidance for planning the optimal deployment of alternatives for RFT. Transitioning to these technologies will take time, requiring the development of necessary infrastructure and adjustments to the electricity mix and hydrogen sources to make them competitive in terms of GHG emissions compared to dICTs. In this context, it is crucial to emphasize the role of biofuels as an effective transitional solution, as they can use existing infrastructure for fuel distribution, as well as current truck technology.

Conversely, the implementation of BETs or FCTs presents additional challenges, both in transitioning from combustion technology in the vehicle fleet, to establishing the necessary infrastructure for refueling, which suggests a much slower replacement than for biofuels. In this regard, our results highlight BETs as a safer option compared to hydrogen FCTs, as the latter will require a rapid decarbonization (i.e., as given by NZ) policies, to become competitive.

3.3. Additional impacts associated with alternatives for road freight transport

Efforts to combat climate change might have undesired side effects on other aspects related to sustainable development. To address this and secure a wider view of the implications of fuel and truck substitutions, additional metrics are analyzed next.

We begin by assessing the impacts incurred by each RFT option on human health (HH) under different policies, regions, and years (Fig. 11). We represent unitary (instead of absolute) impacts (i.e., per tkm) to prevent region size from affecting the results. We find that dICTs show the best performance among the three options considered for the two regions shown (the European Union plus United Kingdom and China), regardless of the policy and year. These impacts remain almost constant over time, with a slight decrease of 12.3%–12.4 % between 2020 and 2050 due to the increase in ICT efficiency.

BioICTs follow, showing HH impacts around 20 % larger than those from dICT based on CP 2020 for both regions. This difference between bioICTs and dICTs is explained mainly by two factors. On the one hand, a larger amount of NO_x emissions is generated in bioICTs due to a higher combustion temperature in the presence of N_2. On the other hand, farming of the vegetable feedstock translates into higher N_2O emissions [76] and water consumption, which ultimately increases impacts on HH, too [57]. Similar to dICTs, HH impacts from bioICTs remain almost constant along the century (13.0%–13.4 % decrease between 2020 and 2050 under CP).

For the two regions explored in Fig. 11, HH impacts from BETs are between 2.1 (EUR) and 2.4 (CHA) times larger than those from dICTs under CP in 2020. In both regions, the main drivers for HH impacts are non-exhaust emissions and battery production, representing 74 % of total HH impacts in EUR in 2020, and 60 % in CHA for the same year. This means that replacing 26 % of RFT by BETs, as needed to achieve the Green Deal target in 2030, would result in a 47 % higher impact on HH than if using dICTs alone.

Finally, we find that HH impacts from FCTs are usually (i.e., for most years, regions, and policies) the largest. FCTs exhibit impacts ranging from 3 times (EUR) to 12 times higher than dICTs (CHA) under CP in 2020. These impacts are primarily associated with hydrogen production. In the case of CHA, the heavy reliance on coal gasification significantly contributes to the HH impact of hydrogen production, which is nearly three times greater than in the EU in 2020.

These results evidence the need to understand the drivers behind these impacts and devote significant research efforts to prevent electric technologies (i.e., BETs and FCTs) from becoming a threat for HH. For instance, non-exhaust emissions of toxic substances represent 15 % of the total HH impacts from BETs in EUR under CP in 2020. These emissions include dust and other metals released from brake wear [63], as well as particulate matter resulting from abrasion caused by friction between tires and the road [77]. These impacts could be reduced by integrating new metal alloys in the brake pads, currently designed with antimony, and reducing the weight of the trucks. In the case of FCTs, the emissions generated by the brakes represent up to 42 % less than BETs due to the use of lighter (i.e., with smaller capacity) batteries.

Indeed, batteries represent the other major contributor to HH impacts from BETs. In this case, the downsizing of batteries over time translates into less resource extraction and waste generation. In addition, improvements in battery energy density result in a reduction of the truck's weight, increasing battery economy, and reducing non-exhaust emissions. This, along with the penetration of renewables in electricity mixes, is observed to reduce HH impacts during the decades by 36 %: from $2.2 \cdot 10^{-7}$ DALYs/tkm in 2020 to $1.4 \cdot 10^{-7}$ DALYs/tkm in 2050 (EUR CP). In the case of EUR CP, this allows for partially closing the existing gap between BETs and dICTs: from 212 % in 2020 to 133 % in 2050. The situation is even better for NZ policies, where, by 2050, HH impacts from BETs would be "only" 53 % larger than those from dICTs. In the case of CHA, the high reliance on coal makes HH impacts from BETs 9 % higher than in EUR in 2020.

In addition, the deployment of renewable energy (e.g., photovoltaic and wind power), together with battery development and some technologies for hydrogen production (e.g., water electrolysis), leads to an increased demand for metals [78]. To address this issue, we assess the impacts of the four options considered for RFT on metal resource scarcity (MRS) in EUR under the two policies (Fig. 12). Again, we depict unitary impacts to avoid any bias stemming from demand volumes.

In 2020, we observe that battery materials represent 98 % of the total unitary impacts on MRS when BETs are used under the CP and NZ scenarios. As aforementioned, differences in MRS impacts observed between different regions are marginal since the same truck and battery production inventories are used for all regions. Existing variations stem from varying shares of renewable sources across regions, yet these are very small since the contribution of the electricity mix to MRS impacts is close to 2 % in 2020. As an example, replacing 2020 CP's electricity mix with that of 2050 CP in CHA would cause an increase in MRS of BETs

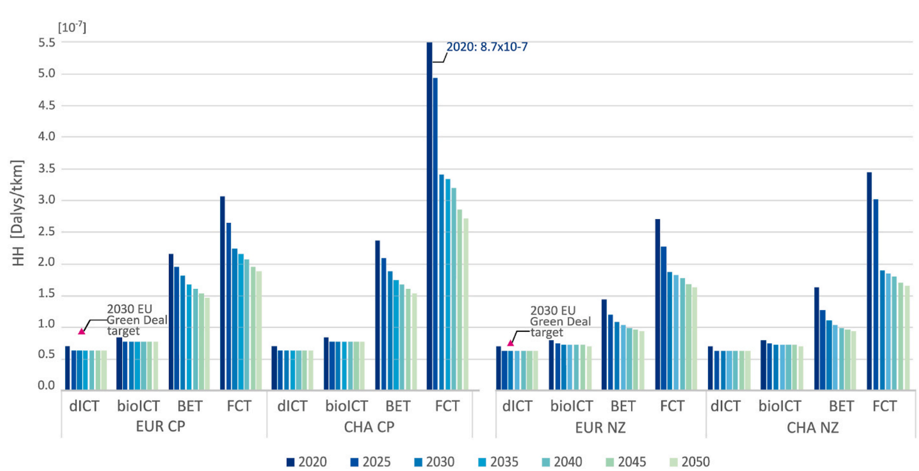


Fig. 11. Impacts on human health from freight road transport under current (CP) and NetZero (NZ) policies. EUR: European Union and UK; CHA: China; dICT: diesel-based ignition combustion truck; bioICT: biofuel-based ignition combustion truck; BET: Battery electric truck; FCT: Fuel cell hydrogen truck. The 2030 mark in the EUR region indicates the human health impact that would result from replacing 26 % of dICTs with BETs, as required to achieve the EU Green Deal target. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

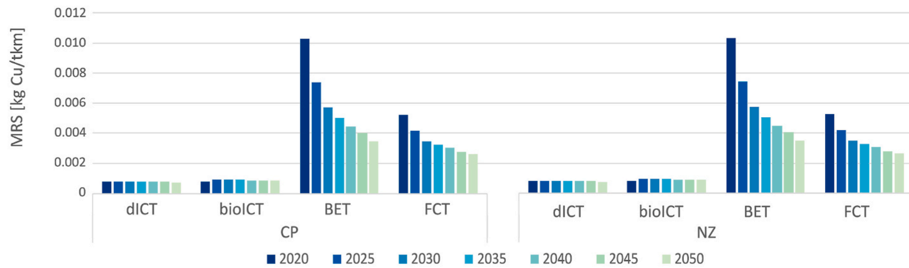


Fig. 12. Impacts on mineral resource scarcity from freight road transport under current (CP) and NetZero (NZ) policies. dICT: diesel-based ignition combustion truck; bioICT: biofuel-based ignition combustion truck; BET: Battery electric truck; FCT: Fuel cell hydrogen truck.

lower than 0.2 % despite the larger share of renewables.

Hydrogen FCTs are second, with impacts on MRS 48–52 % lower than those of BETs in 2020 under CP. These impacts stem mostly from different equipment units, such as the hydrogen storage tank in the truck (around 1.9 % of MRS impacts) and the battery included in the truck (around 9 %), even if smaller than those in BETs (1700 kg vs 15000 kg, both in 2020) [29].

The difference between the other two options, bioICTs and dICTs, is minimal (2%–11 %) and stems from the use of vegetable feedstock and renewable energy. However, these options, based on ICTs, have much lower impacts than electrified alternatives. For instance, BETs have MRS impacts 1200 % higher than dICTs in 2020, although this difference could decrease to 350 % if BETs achieve the improvements anticipated by 2100, such as reducing battery weight by 79 % (see Fig. 3). In the case of FCTs, MRS impacts are 573 % larger than dICTs in 2020, with this difference decreasing down to 243 % by the end of the century. This is mainly due to the reduction of battery weight and the inclusion of materials to reduce the weight of the truck structure.

Note that these improvements in BET and FCT technologies offset the additional demand for metals stemming from other echelons of the value chain. For instance, the penetration of renewable sources such as solar or wind, characterized by a strong reliance on metals [79], leads to increased impacts on MRS. This is especially relevant in the NZ scenario, where the more aggressive adoption of renewables causes the global electricity mix in 2050 to reach impacts on MRS 170 % higher than in 2020.

The demand for metals is a critical issue, as the availability of resources, overlooked in this study, can further limit the deployment of any alternative in practice (see section 2.5). For instance, battery production is currently concentrated in three regions: EUR, USA, and China. However, the implementation of BETs that we propose for an effective reduction in the GHG emissions of these regions would occur only by 2025, 2030, and 2040, respectively (Fig. 8). Until then, other regions for which we suggest an earlier adoption of BETs may need to import the technology. This might be the case of LAT, where BETs become the preferred option by 2030, i.e., 10 years earlier than in China.

This discourse is also valid for the availability of land and water for growing the corresponding amount of biomass to produce the necessary volume of biofuels that even if they are a better alternative in terms of GHG, their inclusion will be limited to the regional availability of these resources.

4. Conclusions

This contribution assessed the environmental performance of three key alternatives for decarbonizing the RTF along the century: trucks powered by biofuels (bioICTs) battery electric trucks (BETs) and hydrogen fuel cell trucks (FCTs). To this end, we divided the world into 12 regions and applied prospective LCA to each of them, utilizing a cradle-to-grave approach, i.e., covering all activities from fuel production until the end of life of the truck components. We also considered two scenarios for the evolution of environmental policies: the continuation of current policies (CP) and rapid decarbonization through net zero strategies (NZ).

We found BETs to become the best alternative for combating climate change in the RFT sector in all regions under CP at some point between 2030 and 2050. This leadership would start even earlier in regions such as Latin America (2020–2025) or the European Union (2025–2030) thanks to their high share of renewables. In contrast, other regions might need to decarbonize electricity generation or reduce the energy density of batteries before this technology becomes convenient. Specifically, we found that, in eight of the 12 regions considered (e.g., China or Japan), the use of BETs until 2030 would increase life-cycle GHG emissions by up to 70 % compared to the continued use of current dICTs (CP).

According to our results, the current market for hydrogen production prevents FCTs from improving the GHG emissions from dICTs. More worryingly: in some regions such as China, the evolution of the hydrogen market envisaged under CP would cause FCTs to emit 22 Gton CO_{2eq} more than if dICTs continued being used until the end of the century. This situation changes dramatically under NZ policies, where hydrogen FCTs have a similar performance to BETs in all regions until about 2045–2050, always becoming the best option by the end of the century (e.g., emitting 31 % less GHGs than with BETs in EUR).

Overall, this situation suggests that liquid fuels will continue to be an essential component for the decarbonization of the transport sector in the short to medium term, especially in regions less prepared for the early adoption of BETs or FCTs. For example, we found biofuels based on hydrotreated vegetable oil to outperform BETs in terms of GHG emissions by 11%–64 % in all regions up to 2030 under CP. In addition, biofuels can leverage existing infrastructure for fuel supply, which suggests that prioritizing the early adoption of bioICTs would maximize the amount of GHG emissions avoided by the end of the century. Specifically, we found that an optimal transition from dICTs to bioICTs, and then to BETs or FCTs, would translate into 134 Gt CO_{2eq} avoided worldwide, and prevent 0.22 °C of temperature rise under CP (204 Gt CO_{2eq} and 0.33 °C under NZ). Nonetheless, it is crucial not to overlook the significance of selecting the right feedstock and biofuel type, as conventional options like biodiesel from soybean can result in emissions that are 77 % higher than those from dICTs as well as impacts on land and water use could potentially increase.

On the other hand, BETs and FCTs are not exempt from negative side-effects. Despite having no exhaust emissions, the release of hazardous components during battery production, power or hydrogen production, and fugitive emissions from brakes or tires, cause larger impacts on human health than conventional dICTs for all regions and years studied (up to 239%-larger for BETs and 1200 % larger for FCTs). BETs also possess higher demands for metals, with impacts on mineral resource scarcity up to 13 times larger than for ICTs in 2020 in EUR. Given the scarcity of these materials, there is an urgent need to enhance technologies for material recovery to prevent material shortages from limiting the widespread adoption of BETs in the future.

When comparing BETs and FCTs, we found the former option to be a safer bet, as it is expected to perform better than dICTs and bioICTs in all the regions, regardless of the policy adopted. Despite this, FCTs have the potential to become the best alternative in all regions, but this will only happen if hydrogen production is rapidly decarbonized, an action that would also benefit BETs. Indeed, sticking to BETs (instead of FCTs) under NZ policies only results in additional 15 Gt CO_{2eq} released throughout the century (i.e., 7 % worse). Conversely, if actual policies move closer to CP, betting on FCTs instead of BETs would translate into the emission of additional 80 Gt CO_{2eq}, a 42 % worse than with BETs.

Overall, this contribution highlights the need for comprehensive approaches to assess the sustainability level of the different options for the transport sector, addressing not only exhaust emissions, but also a myriad of other environmental problems associated with fugitive emissions, electricity generation or material scarcity. Recognizing that decarbonizing the transport sector will be a gradual process, it becomes imperative to prioritize NZ policies that accelerate the wide deployment of clean energy technologies, while favoring the use of biofuels as an interim solution.

CRediT authorship contribution statement

Richard Cabrera-Jiménez: Conceptualization, Methodology, Software, Writing – original draft, Visualization. **Josep Maria Mateo:** Validation, Writing – review & editing. **Laureano Jiménez:** Resources, Writing – review & editing, Supervision, Project administration. **Carlos Pozo:** Conceptualization, Methodology, Validation, Writing – review & editing, Supervision.

Declaration of competing interest

As authors we have nothing to declare.

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During the preparation of this work the author(s) used ChatGPT in order to improve the readability. After using this tool/service, the author (s) reviewed and edited the content as needed and take(s) full responsibility for the content of the publication.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.rser.2024.115243.

Data availability

Data will be made available on request.

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