



# Inclusive impact assessment for the sustainability of vegetable oil-based biodiesel – Part I: Linkage between inclusive impact index and life cycle sustainability assessment

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## ABSTRACT

Sustainability of vegetable oil-based biodiesel has been a controversial issue since its invention as an alternative to conventional diesel. In this study, the Inclusive Impact Index (Triple I) is applied to evaluate the trade-off between advantages and disadvantages of the biodiesel system. Triple I is a single quantitative index for sustainability assessment, which is based on ecological footprint (EF), biocapacity (BC), ecological risk (ER), human risk (HR), cost (C) and benefit (B) under the life cycle (LC) approach. Due to the lack of appropriate guidelines for calculation, the application of Triple I is varied and limited. With respect to contribution to sustainability assessment of renewable energy for transportation, this study aims to propose a methodical estimation of Triple I by integrating its current principles within the Life Cycle Sustainability Assessment (LCSA) framework. The entire study is presented in two parts: Part I identifies and describes the methodological framework of the Triple I - LCSA integration, whilst Part II involves assumed exemplifications on the application of the Triple I framework attached with the case study of an inedible vegetable oil-based biodiesel system in Ha Long Bay, Vietnam. In the first part, an integration framework of the Triple I and LCSA was developed. Accordingly, human health impacts (DALY person<sup>-1</sup>) and ecosystem quality impacts (PDF m<sup>-2</sup> year<sup>-1</sup>) from IMPACT 2002+ and Lethal/Effective Concentration (LC/EC) (under ecotoxicity assessment) are adopted as HR and ER in Triple I, respectively. Life cycle-based EF and life cycle costing are used for EF and C and B, respectively. Under the newly developed framework, ecosystem impacts and human health impacts are firstly converted to a monetary value, then to global hectares in line with cost and benefit values. Following the success developed in the Triple I framework, this work introduces to the sustainable assessment community, especially the transportation sector, a convenient and easy-to-apply quantitative index. Furthermore, this paper also summarizes several state-of-art life cycle assessment techniques and provides brief information about biodiesel life cycle inventories and some potential impacts of the system.

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## 1. Introduction

Fossil fuel energy supply has steadily increased twofold from more than 5300 Mtoe in 1973 to around 11,110 Mtoe in 2014, providing more than 80% of total primary energy supply for four decades, despite increasing non-fossil energy (IEA, 2016a). This domination of fossil fuel is projected to continue until 2035 (BP p.l.c., 2016). Since fossil fuels are depletable, this will lead to a

massive future burden on natural resources. Furthermore, fossil fuel combustion is the key driver of the surge in global carbon dioxide (CO<sub>2</sub>) emissions, which reached 32 GtCO<sub>2</sub> in 2014 (IEA, 2016b). As carbon dioxide emissions are the major contributor to climate change, several substitutions of fossil fuel are of great interest to international communities regarding future energy guarantee and environmental and human well-being protection. Vegetable oil-derived biodiesel is considered as an ideal alternative to fossil diesel (petrodiesel) in the transport sector. This type of fuel is renewable and environmentally friendly, with the potential to mitigate climate change and cause less harm to human health (Achten, 2010). However, several disadvantages of biodiesel have also been indicated; for example, higher impacts on the ecosystem

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### Abbreviations

BC	Biocapacity
EF	Ecological footprint
ER	Ecological risk
DALY	Disability adjusted life year
GHG	Greenhouse gas
HR	Human risk
LCA	Environmental life cycle assessment
LCC	Life cycle cost
LCEF	Life cycle-based ecological footprint
LCI	Life cycle impact
LCIA	Life cycle impact assessment
LCSA	Life cycle sustainability assessment
PDF	Potentially disappeared fraction
S-LCA	Social life cycle assessment
TRIPLE I	Inclusive Impact Index

due to fertilizer and other agricultural chemical use (Achten, 2010), land use changes (Fargione et al., 2008), and higher net production costs (Rajagopal and Zilberman, 2007). Due to both pros and cons of biodiesel production and utilization, scholars have argued about net benefits and sustainable potential of biodiesel for years. To settle this controversy, biodiesel systems need to be evaluated with an appropriate sustainability assessment tool that can consider the trade-off between various positive and negative impacts of the system.

The term 'sustainable development' was first defined in the Brundtland report as 'a process of change in which the exploitation of resources, the direction of investments, the orientation of technological development, and institutional change are all in harmony and enhance both current and future potential to meet human needs and aspirations' (World Commission on Environment and Development, 1987). Sustainable development does not only mean to preserve the natural environment, but rather aims to promote an appropriate trade-off between social and economic development and environmental protection and to emphasize the responsibility of humankind for their future generations. With an effort toward a future-oriented society, various sustainability assessment methods were developed based on the 'triple bottom line - economic prosperity, environmental quality, and social justice' concept (Elkington, 1998). Out of those methods, Life Cycle Sustainability Assessment (LCSA) is a popular framework for the assessment which evaluates economic, environmental and social impacts from cradle-to-grave of a product system. As a framework, nevertheless, LCSA only means to provide guidelines for each impact assessment (Kloepffer, 2008; UNEP/SETAC, 2011). Therefore, results from the three dimensions of LCSA are mostly presented separately, and a cohesive sustainability benchmark is almost neglected in previous studies.

Another innovative sustainability assessment method is Inclusive Impact Index (Triple I). By integrating ecological footprint, cost and benefit, ecological risk and human risk assessments, Triple I seeks to evaluate and combine the three-dimensional sustainability over the whole life cycle of a product system into a single index (Otsuka, 2011). Triple I was applied in several previous sustainable studies in, for example, marine technologies: ocean nutrient enhancer (Otsuka, 2011), artificial upwelling (Yoshimoto et al., 2010), water purification (Duan et al., 2011), oil tankers (Yuzui and Kaneko, 2011); and the energy sector: biodiesel from jatropha oil and waste cooking oil (Nguyen et al., 2015) and power generation (Takahashi and Sato, 2015). Since Triple I aims to

provide a complementary result of the sustainability assessment, its outcome is easy to understand and highly applicable, especially in the policy-making process. However, the application of this index varied case by case and in most cases, a simplified Triple I (Triple I light) was used that omitted some parameters including ecological risk and human risk (Lee et al., 2011; Otsuka, 2011). This obstacle has been made due to the lack of feasible guidelines for the calculation of parameters of Triple I. Furthermore, even when the full-scale is of concern, the early application of Triple I only considered the risks/impacts on human health and the ecosystem in case of product release into the environment (Duan et al., 2011; Omiya and Sato, 2011; Yuzui and Kaneko, 2011). Impacts from other life cycle stages, for example, the production of raw materials and emissions from transportation were not considered. These are also fundamental issues in life cycle assessment studies. Consequently, Triple I requires a more sophisticated methodology to identify and analyze all factors in a product's life cycle system. Overall, since Triple I is a single-quantitative-index of the sustainability assessment throughout the life cycle of a product system, and LCSA is a framework providing a proper pathway for the assessment, it is possible to connect the two methods to obtain an all-inclusive result with a systematic approach.

Therefore, to contribute to the sustainability assessment of renewable energy for transportation, the authors aimed to propose a methodical estimation for Triple I through integrating with the LCSA framework which focused on vegetable oil-derived biodiesel. Moreover, some examples of life cycle inventory related to vegetable oil-based biodiesel were also targeted in this paper.

## 2. Life Cycle Sustainability Assessment (LCSA)

### 2.1. The concept of life cycle sustainability assessment

LCSA is a decision-making support tool based on a systematic approach towards all-in assessment of environmental, economic and social impacts from cradle-to-grave of a product (UNEP/SETAC, 2011). The well-known conceptual framework for the Life Cycle Sustainability Assessment (LCSA) is as follows (Finkbeiner et al., 2010; Kloepffer, 2008):

$$LCSA = LCA + LCC + SLCA \quad (1)$$

where *LCA* is (Environmental) Life Cycle Assessment, *LCC* is Life Cycle Costing, and *SLCA* is Social Life Cycle Assessment (S-LCA). Three parameters in this formulation represent the three key issues of the sustainability assessment, including environmental impacts, economic impacts, and social impacts and they need to be conducted in parallel.

### 2.2. Life cycle initiative methodological framework

Initially, a life cycle assessment framework was introduced and standardized for LCA studies under ISO 14040:2006 (ISO, 2006a). This framework, later on, has also been applied to both LCC and S-LCA (UNEP/SETAC, 2011).

In accordance with ISO 14040, there are four steps to conduct an LCA study: goal and scope definition, inventory analysis, impact assessment and interpretation. Key factors for the success of a life cycle study are to have explicit system boundaries followed by a thorough life cycle inventory, a proper selection of impact category and assessment method, and a precise evaluation of all results from the inventory and impact assessment phases.

Due to the presence of various life cycle assessment models (sections 2.3, 2.4, 2.5), the discovery and evaluation of the life cycle inventory have become the most challenging work in

environmental, economic and social life cycle studies. The inventory analysis phase involves recognizing all input-output flows of all chains in a product system. Input elements can be categorized as energy inputs, raw material inputs, and other physical inputs. Outputs can be products, co-products, wastes, and emissions to air, water and soil. Although the three life cycle assessments, including LCA, LCC and S-LCA may share the same elements of input-output flows in each life cycle stage, the data processing depends on the life cycle-based method, whether it is concerned with environmental impacts, economic value, or social implications.

The crucial milestone for the integration of the three assessments is they have to share the same system boundaries (Kloepffer, 2008) and inventory data. Table 1 shows shared phases and different sample indicators for life cycle assessment in life cycle impact assessment (LCIA) of the three models. The next three sections will summarize the main approaches and well-known methods developed for and applied in environmental, economic, and social and socio-economic life cycle assessments. The selection among those methods is flexible depending on research purposes.

### 2.3. Environmental life cycle assessment (LCA)

Among those three life cycle-based methods, LCA dealing with environmental aspects has been widely used as a reference for the environmental management schemes (Finkbeiner et al., 2010). LCA is a tool to measure 'potential environmental impacts throughout a product's life cycle'. Its framework was standardized in ISO 14040:2006 and ISO 14044:2006 (ISO, 2006a, 2006b) which has been extended to apply in LCC and S-LCA studies with certain revision. Regarding impact assessment in LCA, LCIA aims to quantify the potential environmental impacts of the entire product life cycle system. Principle components of LCIA are, for example, identification of impact categories, category indicators and characterization models, assignment of life cycle impact (LCI) results to the selected impact categories (classification) and calculation of category indicator results (characterization). Various methodologies were developed differently from each other, either in using midpoint or endpoint approaches or combining both methods and also in defined impact categories (JRC, 2010).

Some of the well-known LCIA methodologies are CML 2002, Eco-indicator 99, ReCiPe and MEEuP from Netherlands, Impact 2002+ and Swiss Ecoscarcity 07 from Switzerland, LIME from Japan, TRACI from the USA, and ESP 2000 from Sweden (refer to JRC (2010) for general information of each method). In general, those methods mostly consider impacts on climate change, human health, natural environment quality and natural resource use. However, each model applies different approaches and develops a varied set of indicators. Some adopt midpoint modeling (problem-oriented) including, for example, CML 2002, MEEuP and TRACI;

some employ endpoint modeling (damage-oriented) including Eco-indicator 99 and EPS 2000; and some combine the two approaches including Impact 2002+, LIME and ReCiPe, for instance (JRC, 2010). Selection of the model used mostly depends on the purposes of the study and its research location.

Furthermore, being based on the diversity in purposes of life cycle-based studies, several adapted and extended LCA methods have been developed, including:

- Greenhouse gas (GHG): LCA method focuses on identifying and accounting for total greenhouse gasses emitted over a product's life cycle and their climate change impact (Hondo, 2005);
- Life cycle-based ecological footprint (LCEF): Assessment expresses how much bioproductive area (either land or water) is needed to regenerate all the resources consumed and to absorb the waste formed by a population (Huijbregts et al., 2008). LCEF presents aggregate land area both directly and indirectly occupied across the whole product life cycle system. Indirect land occupation concerns nuclear energy use and CO<sub>2</sub> emissions from fossil energy use and cement burning;
- Ecologically-based LCA (Eco-LCA) aims to account for the use of ecosystem goods and services and its impacts on ecosystem quality over the life cycle of a product (Zhang et al., 2010);
- Life cycle risk assessment (LCRA) is a methodological approach that integrates the ordinary risk assessment with life cycle thinking to assess potential human health and ecological impacts throughout the life cycle of a system. It seeks to identify broadly and screen for potential human health and ecological impacts by incorporating life cycle stages of a product while analyzing 'multi-media environmental fate and transport, exposure, and effects on both ecological receptors and human health' (Eason et al., 2011).

### 2.4. Life cycle costing (LCC)

Life cycle costing is a tool to evaluate economic aspects of a product over its life cycle. LCC calculates total cost and benefit of a product over its life cycle. LCC consists of various elements, including initial capital costs, the lifetime of the asset, discount rate, operating and maintenance costs, disposal cost, information and feedback, uncertainty and sensitivity analysis. Huppes et al. (2004) applied four-level categories of cost to summarize various sophisticated types of costs ranging from general cost (1st level: budget cost and market cost, for instance) to specific expenses (4th level: materials, buildings, taxes, and wastewater treatment, for example) (Table 2). Those costs might be derived from private or social sources, from direct or indirect accounting, from tangible or intangible costs. To avoid double accounting between LCA and LCC,

**Table 1**

Examples of shared and separated phases between LCA, LCC and S-LCA under LCIA, adapted from Traverso et al. (2012) and UNEP/SETAC (2011).

Life cycle stages	Inventory	Indicators		
		LCA	LCC	S-LCA
Raw material acquisition	Inputs:	Embodied energy	Extraction costs	Salary per employee
Raw material transportation		Global warming potential	Manufacturing costs	Percentage of female workers
Production	Facilities & equipment	Human toxicity potential	Finishing costs	Percentage of females at the administration level
Product distribution	Raw materials	Photochemical oxidation	Waste disposal costs	Percentage of employees with limited contracts
Use	Energy	Acidification	Energy costs	Percentage of workers with yearly check up
Disposal & recycling	Natural resources	Eutrophication	Equipment costs	Number of accidents
	Manpower	Abiotic depletion	Revenues	Percentage of child labor
	Outputs:	Ozone layer depletion	Raw material costs	Number of discrimination cases
		Terrestrial ecotoxicity	Product costs	Social benefits per employee
			Labor costs	
			Transport costs	

**Table 2**  
Examples of cost categories (Hupples et al., 2004).

1st level: in economics	2nd level: life cycle stages	3rd level: activity types	4th level: exemplary elements
Budget cost	R&D	Development	Personnel equipment
Market cost	Primary production	Extraction	Loans (rent)
Alternative cost	Manufacturing	Purchase	Overheads
Social cost	Use	Sales	Materials
	Disposal management	Reuses	Disposal
		Management	Communication
		Design	Investment
		Agriculture	Food production
		Manufacturing	Service
		Public relations	Electricity
		Recycling	Office costs
		Administration	Building costs
		Research	Warranties
		Testing Packaging	Infrastructure
		Transport	Depreciation
		Maintenance	
		Waste treatment	
		Infrastructure	

the LCC method only considers true monetary input (cost) and output (revenue) flow of each unit process within a specific system boundary (Kloepffer, 2008). Therefore, within LCSA, intangible costs, such as alternative cost and social cost, are not quantified. Since LCC deals with the whole life cycle of a product, relevant costs can be either one-time payments, such as investing in infrastructure and purchasing facilities and equipment, or annual costs, such as labor cost, material costs, and taxes. To aggregate those costs, several methods were suggested, for example, Net Present Value of Cost (NPV), profit, payback time, steady state costs and average yearly cost, and inflation and discount rate (Hupples et al., 2004). Previous studies mostly suggested and used steady state costs/average yearly cost for the life cycle costing under life cycle assessment. Nevertheless, this paper recommended the use of net present values or discount rate cooperating with payback time to obtain more precise results regarding economic dimension. The application of those methods is discussed in section 4.3.

### 2.5. Social life cycle assessment (S-LCA)

The third and the most controversial component in LCSA is S-LCA, which evaluates direct social and socio-economic impacts of a product and service throughout its cradle-to-grave (UNEP/SETAC, 2009). Social and socio-economic impacts are site-specific and vary case by case. Indicators and assessment methods of LCSA are at their early stage and have not yet been standardized. Stakeholder involvement can be workers/employees, local communities, societies (national and international), consumers (either at the end-use state or within the supply chain), value chain actors and other groups, such as non-governmental organizations (NGOs) and public authorities/state (UNEP/SETAC, 2009). Common impact categories under S-LCA are, for example, human and indigenous rights, working conditions, cultural heritage, poverty, health and safety, and governance and political conflict (Eason et al., 2011; UNEP/SETAC, 2009). Table 1 presents some example indicators of S-LCA. The evaluation of social impacts is very complicated and easily overlaps with environmental impacts (human health impacts, for instance) and economic impacts (job creation and labor income). Therefore, the choice of S-LCA indicators under LCSA has to be thoroughly taken into account.

### 3. inclusive Impact Index (Triple I)

Inclusive Impact Index (Triple I) is a quantitative evaluation tool

to assess the sustainability of a system, which was developed by the Inclusive Marine Pressure Assessment and Classification Technology (IMPACT) Research Group in 2006 (Otsuka, 2011). Following the theory of LCSA, Triple I also considers environmental, economic, and social impacts along the entire life cycle of a studied system. This index employs ecological footprint, financial flow and environmental and social impacts (in terms of ecological risk and human risk) for the estimation, and integrates them into a single index of the world-average biological productive area for a given year, the so-called global hectares (gha) (Duan et al., 2011; Otsuka, 2011). Triple I (III) is determined by the following equation:

$$III = [(EF - BC) + \gamma ER] + \alpha[(C - B) + \beta HR] \quad (\text{gha}) \quad (2)$$

where  $EF$  is ecological footprint (gha),  $BC$  is biocapacity (gha),  $ER$  is ecological risk,  $C$  is cost (US \$),  $B$  is benefit (US \$),  $HR$  is human risk, and  $\alpha$ ,  $\beta$ , and  $\gamma$  are the conversion factors from economic value (US \$) to gha, from  $HR$  value to economic value (US \$), and from  $ER$  value to gha, respectively.

Moreover, the ratio ( $III^*$ ) between the burdens and the benefits within Triple I could be used as a proper reference under policy dimension (Eq. (3)) (Otsuka, 2011).

$$III^* = \frac{EF + \gamma ER + \alpha(C + \beta HR)}{BC + \alpha B} \quad (3)$$

To convert from economic value to global hectare, the ratio of total  $EF$  of the country or region, where the target system is implemented, to its gross domestic product (GDP) in the same year was applied (Otsuka, 2011) (Eq. (2)).

$$\alpha = \frac{EF_{region}}{GDP_{region}} \left( \text{gha US \$}^{-1} \right) \quad (4)$$

$HR$ , as a parameter of TRIPLE I, involves both human health impacts and social and socio-economic impacts. Therefore, the computation of  $\beta$  depends on which impact is under consideration.

Regarding  $\beta$  and  $\gamma$ , there are several available indices of  $ER$  (Potentially Disappeared Fraction (PDF), Lethal/Effective Concentration (LC/EC) and No Observable/Lowest Observable Effect Concentration (NOEC/LOEC), for instance) and  $HR$  (Disability Adjusted Life Years (DALY) and Years of Lost Life (YOLL) in terms of human health impacts, for example). Therefore, the conversion of  $ER$  and  $HR$  values to the desired values varies case by case. Section 4.5 produces some options for the conversion.



Triple I not only can assess the sustainability of a product system but also can be used as a global and transboundary tool to compare technologies and products among various countries due to the application of global hectares.

#### 4. Linkage between Triple I and LCSA in the assessment of vegetable oil-based biodiesel

##### 4.1. Integration framework of Triple I and LCSA for assessing biodiesel

Although Triple I is not a method developed under the LCSA scheme, there is a close relationship between Triple I and LCSA. Both Triple I and LCSA are making efforts to measure the sustainability of the entire life cycle of a product/service. They can support and boost each other to reach their final expected destination. Regarding LCSA, Triple I can be considered an optimal quantitative tool for the sustainability evaluation of the studied product/service. Initially, the cooperation of different life cycle approaches in Triple I requires a concurrent study goal, functional unit, and system boundaries. Then, flexible application of the three-dimension life cycle assessment in LCSA is a vital issue determining the success in Triple I final estimation.

Fig. 1 illustrates the Triple I estimation pathway using a relative life cycle assessment technique under LCSA. It is noteworthy that since biodiesel is considered an alternative to petrodiesel, all impacts, including those on the environment, human beings, and society, of petroleum's life cycle can be treated as a business-as-usual baseline. Therefore, the assessment of biodiesel potential always has to take into account the different impacts between biodiesel and petrodiesel. However, due to the lower energy content of biodiesel, from more than 35.6 MJ kg<sup>-1</sup> to less than 44 MJ kg<sup>-1</sup> (Atabani et al., 2013), than that of petrodiesel, approximately 45 MJ kg<sup>-1</sup>, fuel efficiency might be applied when comparing the two fuel sources.

##### 4.2. Life cycle-based ecological footprint

EF and BC in Triple I can be calculated using the LCEF approach. The entire LCEF is contributed by total EF of direct land use, carbon footprint, and nuclear energy footprint (Eq. (5)) (Huijbregts et al., 2008).

$$EF = EF_{direct} + EF_{CO_2} + EF_{nuclear} \text{ (gha)} \quad (5)$$

Nevertheless, EF, under the context of Triple I, only covers the total carbon footprint from the life cycle of a product system and direct land occupation over time of the system (refer to biocapacity section below for the calculation method). Carbon footprint ( $EF_{CO_2}$ ) is the indispensable forest area needed to absorb all the CO<sub>2</sub> emitted into the air due to the burning of fossil fuel, cement production, and land use change. The total carbon footprint of a product system is calculated as follows (Huijbregts et al., 2008):

$$EF_{CO_2} = \sum_{t=0}^n P_{CO_2,t} \times FI_{CO_2} \text{ (gha)} \quad (6)$$

where  $n$  is the project lifetime,  $P_{CO_2,t}$  is total greenhouse gas emissions in year  $t$  (t CO<sub>2</sub>),  $FI_{CO_2}$  is the footprint intensity of CO<sub>2</sub> (gha (t CO<sub>2</sub>)<sup>-1</sup>).

$$FI_{CO_2} = \frac{1 - F_{CO_2}}{S_{CO_2}} \times EqF_f \text{ (gha (t CO}_2\text{)}^{-1}\text{yr}^{-1}) \quad (7)$$

where  $F_{CO_2}$  is the fraction of CO<sub>2</sub> absorbed by oceans,  $S_{CO_2}$  is the sequestration rate of CO<sub>2</sub> by biomass (t CO<sub>2</sub> wha<sup>-1</sup> yr<sup>-1</sup>),  $EqF_f$  is the equivalence factor of forests (gha wha<sup>-1</sup>). Table 3 shows values of some identified parameters applied in EF estimation.

Biocapacity (BC) is a reverse form of ecological footprint that indicates the total bioproductive area gained of land-use type  $a$  ( $SA_a$ ) corresponding with either increasing primary productivity, or reducing CO<sub>2</sub> emission through the entire life cycle of a product during the project life time ( $n$ ) (Monfreda et al., 2004; Otsuka, 2011). Biocapacity is calculated as follows:

$$BC = \sum_{t=0}^n \sum_a SA_{ta} \times YF_a \times EqF_a \text{ (gha)} \quad (8)$$

where  $n$  is the project lifetime,  $SA_{ta}$  is the bioproductive area gained of land-use type  $a$  in year  $t$  (ha),  $EqF_a$  is the equivalence factor of land-use  $a$  (gha wha<sup>-1</sup>),  $YF_a$  is the yield factor of land-use  $a$  calculated by dividing the national average yield of land-use  $a$  ( $Y_{Na}$ ) by average world yield of land-use  $a$  ( $Y_{Wa}$ ) (Eq. (9) and (10)) (Ewing et al., 2010).

$$YF_{cr} = \frac{\sum_{i \in U} Y_{N,i}}{\sum_{i \in U} Y_{W,i}} \text{ (ha wha}^{-1}\text{)} \quad (9)$$

$$YF_a = \frac{Y_{Na}}{Y_{Wa}} \text{ (ha wha}^{-1}\text{)} \quad (10)$$

where  $i$  is the type of crop and  $U$  is the set of cultivation crops. Eq. (9) is used for cropland, since the cultivation normally includes

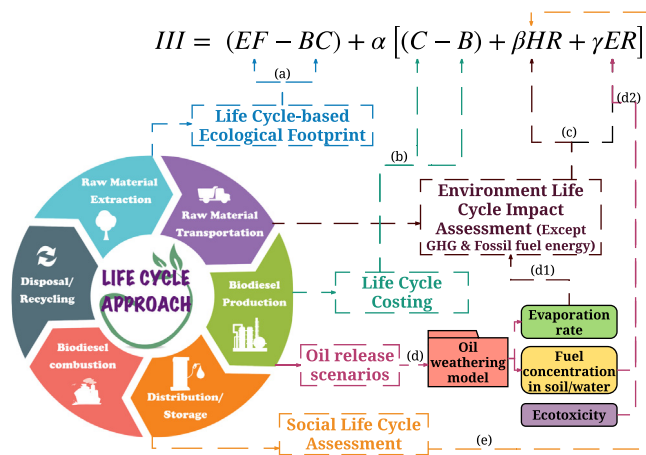


Fig. 1. Biodiesel sustainability assessment framework.

Table 3

Values of identified parameters for the ecological footprint estimation (Lin et al., 2016).

Parameter (unit) <sup>a</sup>	Abbreviation	Value
Equivalence factor forest (gha wha <sup>-1</sup> )	EqF <sub>f</sub>	1.28
Equivalence factor built-up land (gha wha <sup>-1</sup> )	EqF <sub>b</sub>	2.52
Equivalence factor cropland (gha wha <sup>-1</sup> )	EqF <sub>c</sub>	2.52
Equivalence factor pasture (gha wha <sup>-1</sup> )	EqF <sub>p</sub>	0.43
Equivalence factor marine area (gha wha <sup>-1</sup> )	EqF <sub>m</sub>	0.35
Fraction CO <sub>2</sub> absorbed by the ocean (–)	F <sub>CO2</sub>	0.281
Sequestration rate of CO <sub>2</sub> (t CO <sub>2</sub> wha <sup>-1</sup> yr <sup>-1</sup> )	S <sub>CO2</sub>	3.59
Footprint intensity of carbon (gha (t CO <sub>2</sub> ) <sup>-1</sup> yr <sup>-1</sup> )	FI <sub>CO2</sub>	0.256
Fossil fuel emission intensity of CO <sub>2</sub> (t CO <sub>2</sub> GJ <sup>-1</sup> )	I <sub>CO2</sub>	0.0573

<sup>a</sup> wha is world average hectares of a given land use type.

several types of crops and Eq. (10) is used for other land-use types, as they only have one primary product.

#### 4.3. Life cycle costing

Costs and benefits considered under Triple I are aggregated results from life cycle costing based on the money input-output flows. Consequently, the cost is the total money input/investment for the start-up, operation, maintenance and waste disposal of all processes in the product life cycle system (Huppes et al., 2004), and the benefit is the total monetary value of products, by-products, and co-products obtained from the system. Capital costs are one-time expenses (during a project lifetime) including, for example, payments for land use, biodiesel plant construction and facility set-up and preliminary cultivation of perennial biodiesel feedstock (seeds and seedlings) (Haas et al., 2006; Ong et al., 2012). Annual costs are costs of input materials for the biodiesel production, utilities, labor, maintenance, taxes and insurance, loan interest and depreciation, for instance (Haas et al., 2006; Ong et al., 2012). To properly estimate the time aggregate cost of the entire system, it is important to apply two cost aggregation methods, the NPV and the payback time.

##### 4.3.1. Net present value

NPV, the present value of cost, is a tool to compare the present monetary value of an investment to the dollar value of that investment in the future (Huppes et al., 2004). The computation of NPV is as follows:

$$NPV = \sum_{t=0}^n \frac{C_t}{(1+r)^t} \quad (\text{US \$}) \quad (11)$$

where  $n$  is the period of assessment (year),  $r$  is the discount rate,  $C_t$  is the estimated costs in year  $t$ . In Triple I, time-equivalent values of total one-time payment (TP) are considered under NPV, in which  $n$  is the project lifetime, and  $C_t$  is the average amount of TP over the project lifetime period. Discount rate ( $r$ ) is the key factor in the estimation of NPV mostly influenced by the inflation and interest rate (Eq. (12)) (Davis et al., 2005).

$$r = \frac{\text{Rate}_{\text{Interest}} - \text{Rate}_{\text{Inflation}}}{1 + \text{Rate}_{\text{Inflation}}} \quad (12)$$

##### 4.3.2. Payback time

The preparedness for installing a new technology or product requires an immense volume of investment, including natural capital, human capital, social capital, manufactured capital and financial capital. The recovery of those invested capitals entirely depends on annual revenue of the technology/product, and it may take a certain time. As an investor, whether a business individual or a policy maker, it is important to recognize when the investment can be totally recovered. Payback time is another important technique in economic life cycle assessment which denotes the possible period for recovering all the initial investment (Huppes et al., 2004). Payback period not only shows the economic potential of the entire product's life cycle but also projects years needed to consider interest rate as an annual cost. The payback time (PBT) is calculated as follows (Huppes et al., 2004):

$$PBT = \frac{C_0}{B} \quad (\text{year}) \quad (13)$$

where  $C_0$  is the total initial investment (US \$), and  $B$  is annual net

benefits (US \$ year<sup>-1</sup>).

#### 4.4. Estimation of ecological risk and human health risk

Fig. 1 (c, d, d1, d2) presents the pathway to assess total ecological and human health impacts of a product system from cradle-to-grave. Both direct emissions including ordinary and potential emissions from each stage within the biodiesel life cycle, and indirect emissions from the production of input materials of the system are considered. Ordinary emissions from biodiesel life cycle are those derived from, for example, fertilizers used in the cultivation stage, energy consumption, chemicals for oil extraction and biodiesel manufacturing, and biodiesel and conventional diesel combustion (Ginn et al., 2013). The occurrence of potential chemical releases possibly associated with fuel leakages from storage tanks (either above ground or underground), engine operation, fuel pipelines and transportation vehicles, or even more substantial releases due to traffic accidents, shipwrecks, and other coincidental incidents (Ginn et al., 2013).

It is worth noting that air emissions from biodiesel production are leakages from the storage of material and the operation of biodiesel manufacturing equipment. Several previous studies show that the total emissions are not high and their incidence is infrequent. Under thorough management, they can be controlled and do not cause risks to both human health and ecosystems (Charman et al., 2012; Sheehan et al., 1998). Therefore, the assessment of those emissions could be avoided. Other potential hazards would come from the transportation of input materials and substances due to, for example, transport accidents, fires, and leakage. In the industrial context, their incidence and impacts are common and perceptible, which are customarily controlled under risk response and mitigation schemes (Charman et al., 2012). Thus, they also can be omitted from the risk assessment.

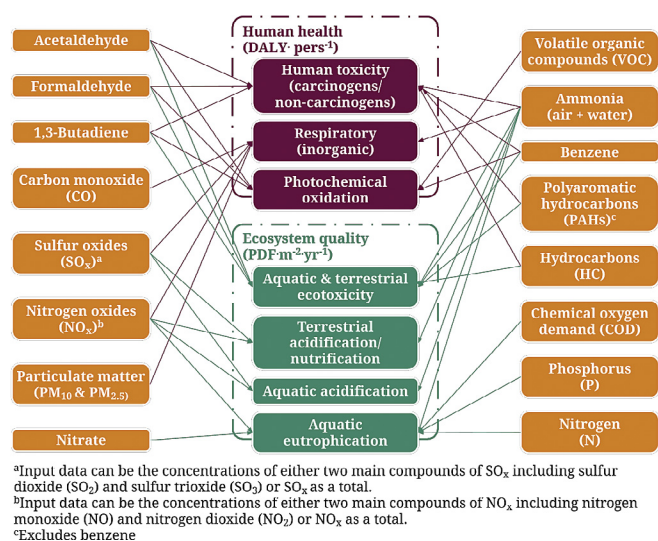
##### 4.4.1. Life cycle impact assessment

Current life cycle impact assessment methods can be applied in the assessment of potential ecological and human health impacts of those ordinary emissions from the system (Fig. 1 (c)). Several LCIA methods were briefly introduced in section 2.3. Since GHG emissions are examined under ecological footprint in Triple I, to avoid double counting, it is essential to check whether the chosen LCIA method can separate impacts of those emissions from impacts of other emissions. Being based on impact indicators, Impact 2002+ (Jolliet et al., 2003), which developed a framework to integrate the midpoint-oriented method with the damage-oriented method, is recommended. Impact 2002+ categorizes LCI results into 14 midpoint categories which then are combined into four damage categories, including human health, ecosystem quality, climate change and resources (Jolliet et al., 2003). Accordingly, two out of these categories, the human health and ecosystem quality impacts can be applied in Triple I as the human health risk (HR) and ecological risk (ER), respectively.

Fig. 2 illustrates the pathway between human health impacts and ecosystem quality impacts and their related midpoint categories covered by Impact 2002+.

##### 4.4.2. Oil weathering

Regarding environmental releases, biodiesel and petrodiesel possibly release into the air, water (groundwater included) and soil due to leakages from above/underground storage tanks, accidental spills from fuel tankers (both ship and lorry) (Charman et al., 2012), and releases from the operation of diesel engines. The releases of fuel into the environment obviously depend on different natural, economic, and social conditions of study areas. Therefore, the development of fuel release scenarios has to cover current and



**Fig. 2.** Human health and ecological impacts and pathway related to vegetable oil-based biodiesel under Impact 2002+ framework, adapted from Joliet et al. (2003).

historical data of the study area, for example, information about accidents (both man-made and extreme weather affected) related to tankers, current fuel storage and transportation technology, and status of diesel engines in operation (Charman et al., 2012; Ginn et al., 2013). After identifying release scenarios, behavior of spilled biodiesel and petrodiesel in the environment is analyzed under oil spill models (Fig. 1 (d)). Various oil weathering models were formed regarding spillages and leakages on the land surface, inland subsurface and in water environments, including river and marine environments. Data of common oil spill processes estimated under the oil weathering model are (Simmons and Keller, 2003; Vos, 2005):

- Both inland and off-land spills/leaks: area of surface spreading, evaporation rate of oil on the surface, (bio)degradation rate;
- In-land spills/leaks: infiltration rate and drainage rate into subsurface soil, an amount of oil enters the groundwater table;
- Spills/leaks in the water environment: formation of emulsification (water-in-oil emulsion) and dispersion (oil-in-water emulsion), dissolution of hydrocarbons from oil slicks, sedimentation, and shoreline stranding.

Although these models apply various calculation methods, they share an almost similar input data set including information on oil properties, for example, viscosity, emulsification, density, distillation cuts, surface tension, interfacial tension, solubility, pour point, flash point and (bio)degradation coefficient; and information on the natural conditions of the spill site, for example:

- Inland and off-land spills/leaks: weather, and wind speed and direction;
- In-land spills/leaks: soil properties, including soil type, mineral content, water retention and bulk density, and land surface properties, including, topography, roughness, and macropores (Simmons and Keller, 2003);
- Spills/leaks in the water environment: wave height and direction, and river/sea current (Vos, 2005).

Results from oil weathering models are used to identify the concentration of biodiesel and diesel in soil and water (water accommodated fraction (WAF)) based on the remaining amount on the land/water surface, amount of fuel entering the subsurface and

groundwater table, and the dissolved amount of spilled fuel into the water column and sediment; and leakage of biodiesel and diesel into air from natural evaporation. Then, current data about ecotoxicity of biodiesel and diesel in soil and water environments are applied to identify ecological impacts of the spill (Fig. 1 (d2)). Moreover, since the evaporation is responsible for more than 70% of petroleum mass loss (Transportation Research Board and National Research Council, 2003; U.S. National Research Council, 1975), it needs to be taken into account. Therefore, petroleum vapors are treated as another emission to air of the biodiesel and petrodiesel life cycle (Fig. 1 (d1)).

On the other hand, the application of an integrated model of oil weathering and environmental effects of spilled oil and fuel is another option. The Spill Impact Model Application Package (SIMAP) (French-McCay, 2004), for example, involves the use of the three-dimensional physical fate model to estimate behaviors of spilled oil at sea, and the biological effect model to evaluate adverse impacts on the mortality or decreased production of marine organisms due to the exposure to certain concentrations of spilled oil on the sea surface, in the water column and on sediment.

#### 4.5. Monetary evaluation of environmental impacts

Regarding environmental impacts, a study from SCORELCA in 2013 evaluated the possibility of available monetary valuation methods as a tool for the monetarization of environmental impacts in LCIA studies (Weidema et al., 2013). Several methods and their previous applications were reviewed and benchmarked, including market approach/observed preferences: market price method; revealed preferences: averting behavior, travel cost and hedonic pricing methods; stated preferences: contingent valuation and conjoint analysis methods; abatement cost method; budget constraint method; restoration cost method; and review/statistical method (NEEDS, 2006; Weidema et al., 2013). Among those methods, market price, contingent valuation, conjoint analysis: choice experiment, budget constraint, and restoration cost methods and their combination are high potential tools for monetarizing environmental impacts. Most of the previous LCA studies applied those tools, in which (Pizzol et al., 2015; Weidema et al., 2013):

- The market price method values a good and service based on its existing market price;
- The contingent valuation and conjoint analysis methods: choice experiment methods are monetarization tools for non-market goods and services which are based on the answers of respondents under a specific hypothetical scenario. The contingent valuation method applies direct questionnaires about respondents' willingness to pay/accept as compensation for an adverse impact on the availability of a product/service. The contingent choice method, meanwhile, requires respondents' trade-off choices among sets of goods/services having 'different availability of the same attributes and different total price';
- The budget constraint method is a particular tool for the monetarization of human well-being impacts. This method is based on data from estimated economic production per capita per year to value the economic implication of changes in well-being life years (in both additional or lost situations) (Dalal and Svanström, 2015; Weidema, 2009); and
- The restoration cost method is a monetarizing method referring to the total cost for the recovery of human-made damages to the environment as the monetary value of the affected ecosystem (NEEDS, 2006).



#### 4.5.1. Monetization of human health impacts ( $\beta$ )

Impact 2002+ adopts Disability Adjusted Life Years (DALY) as a damage index of human health, which is reported as DALY per person (DALY pers<sup>-1</sup>) after normalizing. DALY is an indicator that measures the burden of disease by incorporating total 'years of life lost (YLL) due to premature mortality' and total 'years lost due to disability (YLD)' (Eq. (14)) (Dalal and Svanström, 2015).

$$DALY = YLL + YLD \quad (14)$$

DALY quantitatively denotes the difference between a disease-affected population and a healthy population. A unit of DALY is equivalent to a one-year decrease in healthy life.

The budget constraint method (Dalal and Svanström, 2015; Weidema, 2009) could be used to value the DALY. Since a 'healthy' individual can contribute to a country's economy during that person's lifetime, the number of years lost due to death and disability means the non-economic-contributing period of that person. Therefore, the monetary value of DALY is computed by multiplying the DALY value by GDP per capita ( $\beta$ ) in the same year (Dalal and Svanström, 2015). The stated preferences approach is another option for monetary evaluation of DALY, including the contingent valuation method (Ahlooth and Finnveden, 2011; Desaiques et al., 2011) and the conjoint analysis method (Itsubo and Inaba, 2015). Under contingent valuation's questionnaire, respondents are asked to state how much they are willing to pay/accept for a one-year increase in well-being life year. Regarding conjoint analysis, respondents make their choices among various policies, in which a certain impact, for example, on human health (loss of life expectancy per person), on social assets (loss of social assets per person), on biodiversity (disappearance of species of organisms), and on primary production (inhibition of plant growth) are set followed by a particular tax increase (Itsubo and Inaba, 2015). Results from conjoint analysis questionnaires can be used for both human health and ecological impacts.

#### 4.5.2. Monetization of ecological impacts ( $\gamma$ )

Regarding ecosystem quality (ecological risk in Triple I), the damage index of ecosystem impacts is Potentially Disappeared Fraction of species on one m<sup>2</sup> of the earth's surface during one year (PDF m<sup>-2</sup> yr<sup>-1</sup>) under Impact 2002+ (Joliet et al., 2003), and Lethal/Effective Concentration (LC/EC) under ecotoxicity assessment of biodiesel and petrodiesel (section 5.3.2). In addition to the conjoint analysis method mentioned above, market price, contingent validation (Ahlooth and Finnveden, 2011) and restoration cost (NEEDS, 2006) are other monetization methods for ecological impacts used by several biological valuations and LCA studies. Since PDF directly relates to biodiversity, it can be valued under the contingent valuation, conjoint analysis, and restoration cost methods. Lethal concentration (LC<sub>a</sub>) (Hollebone et al., 2008) and effective concentration (EC<sub>a</sub>) (Leite et al., 2011) are the median concentrations of chemical in soil or water environments that lead to a degree of mortality or a certain level of effect (a%) in the test organism, respectively. Since the lethal concentration and effective concentration denote the effects of released biodiesel and petrodiesel on the available amounts of organisms in their habitat which are directly related to the production capacity of that ecosystem, they can be estimated through the market price method.

#### 4.6. Social life cycle assessment

The result from the social life cycle assessment is one of contributors to the human risk (HR) parameter of Triple I (Fig. 1 (e)). Social life cycle evaluation in Triple I deals with the impacts of the studied system on human well-being including, for example,

human rights, working conditions, health and safety, cultural heritage, governance and socio-economic repercussions (UNEP/SETAC, 2009). The pathway and indicators to integrate S-LCA with Triple I are still under development. Therefore, there is no further discussion about this issue in the current study.

#### 4.7. Adaptation and extension in the application of Triple I

As all the values of ecological risk and human health risk are supposed to be monetized (section 4.5), the current equation of Triple I can be reorganized as follows:

$$III = (EF - BC) + \alpha[(C - B) + \beta HR + \gamma ER] \quad (\text{gha}) \quad (15)$$

Installation of a new product system not only requires a considerable amount of monetary investment but also invites substantial burdens on the environment, human health and society. Similar to payback time in LCC (section 4.3.2), Triple I payback can be used to identify years required to recover those burdens. Consequently, Triple I can be developed into three types for more flexible and diverse applications:

- Triple I<sub>initial</sub> is Triple I assessing all the capital costs and emissions from the preparedness and start-up of a product life cycle system (EF<sub>initial</sub>, ER<sub>initial</sub>, HR<sub>initial</sub>);
- Triple I<sub>annual</sub> is Triple I considering annual costs (C<sub>annual</sub> and B<sub>annual</sub>) and emissions from the product life cycle system (EF<sub>annual</sub>, BC<sub>annual</sub>, ER<sub>annual</sub>, HR<sub>annual</sub>);
- Triple I<sub>total</sub> (normally, identified as Triple I) is average Triple I evaluating all costs and emissions from the entire life cycle of the product within a project lifetime.

Accordingly, if Triple I<sub>total</sub> has a minus value, which means that the studied system is sustainable, Triple I payback time (III<sub>payback</sub>) of a product is calculated as follows:

$$III_{\text{payback}} = \frac{III_{\text{initial}}}{|III_{\text{annual}}|} \quad (\text{year}) \quad (16)$$

It is important to note that the Triple I framework developed in this study can be applied in another type of research with the adaptation in the life cycle inventory.

#### 4.8. Data quality analysis in Triple I

Similar to other life cycle assessment studies, Triple I also has to face many uncertainties and variation in its input and output data. These issues are briefly classified by Björklund (2002) and Huijbregts (1998) as follows:

- Parameter uncertainty is due to the lack of data including data gaps and lack of representative data between the studied system and actual data, and empirical inaccuracy due to imprecise measurement;
- Model uncertainty caused by the ignorance of several complicated factors in life cycle inventory analysis, the assumption in impact assessment and the simplification of the assessment model;
- Uncertainty due to unavoidable choices in life cycle assessment including choices of, for example, system boundaries, functional unit, allocation rules, characterization methods, weighting methods, marginal or average data, and technology level;
- Spatial variability over geographical sites and temporal variability regarding short/long time scales in life cycle inventory and LCIA parameters and, respectively;



- Variability between sources of inventories (variations in comparable technical processes) and between objects determining impacts of the studied system (e.g. human characteristics);
- Epistemological uncertainty through lack of knowledge on system behavior.

Different uncertainties make divergent effects on the final result of a life cycle assessment as well as the outcome of Triple I. Therefore, the uncertainty in the final result of Triple I should be addressed and presented in a range of possible values (Tu and McDonnell, 2016). ISO 14040 and ISO 14044 require a sensitivity check of the final LCA results which can be performed under sensitivity analysis and/or uncertainty analysis. Sensitivity analysis aims to identify and evaluate key factors/parameters and methodological choices that possibly affect the outcome of a study (ISO, 2006b). Out of several tools which can be used in sensitivity analysis, scenario analysis is the most useful and common method which develops various scenarios based on variations of input data, specific future assumptions, different allocation methods, and other parameters affected by choices (Björklund, 2002). On the other hand, uncertainty analysis deals with the quantification of how uncertainties in life cycle inventory and impact assessment methods influence the reliability of the final results of life cycle assessment studies (ISO, 2006b). Among several uncertainty analysis methods, a probabilistic technique namely Monte Carlo analysis is widely recognized and highly recommended (Huijbregts et al., 2001; Sonnemann et al., 2003; Tu and McDonnell, 2016).

To increase the reliability of Triple I results, the assessment of data quality should be considered. However, identifying uncertainty distributions of an enormous number of parameters in life cycle studies to conduct Monte Carlo analysis is complicated and time-consuming (Huijbregts et al., 2001). Therefore, first and foremost, a sensitivity analysis needs to be conducted to detect factors/parameters that noticeably shape the outcome of Triple I. Then, Monte Carlo analysis, if possible, should be applied to those matters (Huijbregts et al., 2001; US EPA, 1997).

## 5. Life cycle inventory of vegetable oil-based biodiesel for Triple I

The lifetime of a project is identified following the lifetime of a biodiesel production plant and perennial biodiesel feedstock.

### 5.1. System boundaries and key assumptions

Fig. 3 shows general boundaries of a vegetable oil-based biodiesel system and diesel as a reference system. In general, key phases in the petrodiesel life cycle include extraction of crude oil from the earth and pretreatment, transport of crude oil to an oil refinery, refinement of crude oil to produce conventional diesel

fuel, distribution and storage of petrodiesel, and utilization in a diesel engine (Sheehan et al., 1998). With regard to biodiesel application, a 'cradle-to-grave' system of biodiesel comprises all the stages starting from feedstock cultivation, feedstock transportation, oil extraction, biodiesel production and blending, biodiesel distribution and use, to the practice of composting and application of compost from organic waste back into the cultivated area. Principal issues in the comparison and linkage between conventional diesel and biodiesel are also displayed in Fig. 3.

### 5.2. Summarization of common emissions from vegetable oil-based biodiesel systems

Main life cycle stages and associated sub-processes of a biodiesel system are as follows (Achten, 2010; Atabani et al., 2013; Sheehan et al., 1998; Whitaker and Heath, 2009):

- Feedstock agriculture: The cultivation of biodiesel feedstock includes seedling practice (not all crops need this step), planting operation and management, and harvest practice. Key input materials are as follows: CO<sub>2</sub> uptake by plants, plastic bags used in seedling, fertilizers and agricultural chemicals (pesticides and herbicides), energy for operating agricultural equipment and other supporting systems (for example, petrodiesel, electricity, and gasoline), and water used for irrigation and land area. In some cases, CO<sub>2</sub> uptake by plants is considered as zero because CO<sub>2</sub> is released back into the atmosphere due to the decay of plant residue after harvesting;
- Vegetable oil-based biodiesel production: The three main steps in this stage are kernel separation and oil extraction, biodiesel production through oil transesterification, and blending between biodiesel and petrodiesel. Various techniques were developed and applied for all these three steps. To extract oil from its kernel, three state-of-art approaches are mechanical extraction, solvent extraction (chemical extraction), and enzymatic oil extraction. Furthermore, the extraction can be supported by the ultrasonication technique to obtain higher oil yield and reduce time consumption (Thanh et al., 2010a, 2010b). Depending on feedstock oil properties (free fatty acid (FFA) value, for example) and producing techniques, biodiesel can be obtained via one-step/two-step transesterification, and with/without co-solvent and ultrasonic supporter (Luu et al., 2014; Thanh et al., 2010a). Input materials of this stage include, for example, chemicals for each process (hexane, ethyl acetate, sulfuric acid, methanol, and acetone), water for oil extraction and washing biodiesel, and energy for machinery and plant operation (electricity and petrodiesel);
- Biodiesel and its blends combustion: As an alternative fuel to mineral diesel, biodiesel and its blends are supposed to be used in current diesel engines. Engine performance and tailpipe

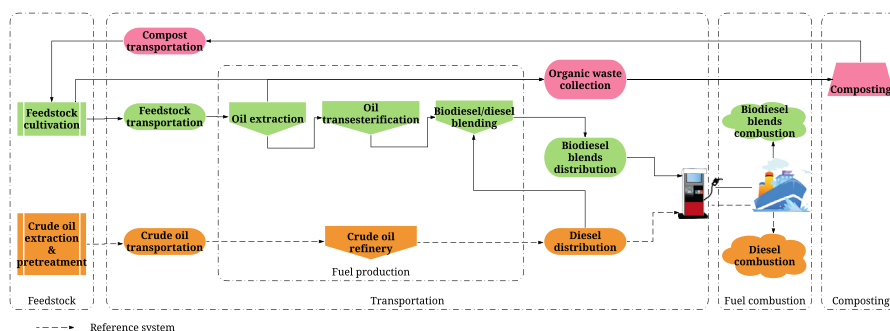


Fig. 3. System boundaries of vegetable oil-based biodiesel and petrodiesel systems, adapted from Achten (2010), Atabani et al. (2013), Nguyen et al. (2015) and Sheehan et al. (1998).

emission characteristics of biodiesel and its blends vary according to different biodiesel feedstocks and blended volumes. In general, the majority of scholars reported that the combustion of biodiesel and its blends decreases carbon monoxide (CO), particulate matter (PM), hydrocarbons (HC), and sulfur dioxide (SO<sub>2</sub>) but increase nitrogen oxides (NO<sub>x</sub>) emissions (Atabani et al., 2013; Jaichandar and Annamalai, 2011; Morris et al., 2003);

- Composting: Wooden stems and leaves from the field and fruit husk and oilseed cake from oil extraction can be gathered and composted;
- Compost used for the fields: The derived compost can be used as a substitute to chemical fertilizers;
- Transportation: The transportation of feedstock from cultivation area to biodiesel plant, biodiesel and its blends from the

production plant to a storage location, biomass waste to composting plant, and compost from the composting plant back to the biodiesel feedstock cultivation field are also included. The input material of this stage is petrodiesel for transport vehicle operation.

Expected direct emissions due to the use of input materials from each life cycle stage of the vegetable oil-derived biodiesel system are presented in Table 4. Moreover, as mentioned in section 4.4, other indirect emissions from the production and preparation of input materials for all life cycle stages of the biodiesel system including biodiesel plant construction and facility set-up are also analyzed. Fig. 2 provides some examples of the determination of environmental impacts of some emissions from a biodiesel life cycle system.

**Table 4**

Expected emissions from a vegetable oil-based biodiesel system by unit process, adapted from Cal EPA (2011), GenSolutions (2007) and Sheehan et al. (1998).

Stage of life cycle	Inputs	Emissions to air	Emissions to water <sup>a</sup>	Emissions to soil	Releases of products
Biodiesel feedstock agriculture	N:P:K Fertilizers Pesticides Composts Electricity Plastic bags Petrodiesel Water	CO <sub>2</sub> fossil, CO <sub>2</sub> biomass <sup>b</sup> , methane (CH <sub>4</sub> ), dinitrogen monoxide (N <sub>2</sub> O), carbon monoxide (CO), unburnt hydrocarbons (HC), volatile organic compounds (VOCs), particulate matter (PM), sulfur dioxide (SO <sub>2</sub> ), nitrogen oxides (NO <sub>x</sub> ), ammonia (NH <sub>3</sub> ), polyaromatic hydrocarbons (PAHs)	Agricultural chemicals, biochemical oxygen demand (BOD5), chemical oxygen demand (COD), metals, ammonia, nitrate	Solid wastes (hazardous & non-hazardous)	—
Feedstock transportation	Petrodiesel	CO <sub>2</sub> fossil, CH <sub>4</sub> , N <sub>2</sub> O, CO, HC, VOCs, PM, SO <sub>2</sub> , NO <sub>x</sub> , NH <sub>3</sub> , PAHs	BOD5, COD, metals, ammonia, nitrate	Solid wastes (hazardous & non-hazardous)	—
Dehusking & oil extraction	Fruits Hexane Ethyl acetate Water Electricity Natural gas	CO <sub>2</sub> fossil, CH <sub>4</sub> , N <sub>2</sub> O, CO, HC, VOCs, PM, SO <sub>2</sub> , NO <sub>x</sub> , NH <sub>3</sub> , PAHs	BOD5, COD, metals, ammonia, nitrate	Solid wastes (hazardous & non-hazardous)	—
Biodiesel production	Vegetable crude oil Methanol Sulfuric acid KOH Acetonitrile Acetone Water Electricity	[Total PM, SO <sub>2</sub> , NO <sub>x</sub> , VOCs, CO, lead (Pb), and other hazardous air pollutants (HAPs)] <sup>c</sup>	Waste water	Solid wastes (hazardous & non-hazardous)	Biodiesel leaks
Biodiesel & petrodiesel blending	Petrodiesel Electricity	n.incl. <sup>d</sup>	n.incl.	n.incl.	Biodiesel and petrodiesel leaks
Biodiesel and its blend distribution	Petrodiesel Electricity	CO <sub>2</sub> fossil, CH <sub>4</sub> , N <sub>2</sub> O, CO, HC, VOCs, PM, SO <sub>2</sub> , NO <sub>x</sub> , NH <sub>3</sub> , PAHs	BOD5, COD, metals, ammonia, nitrate	Solid wastes (hazardous & non-hazardous)	Biodiesel blend leaks & spills
Biodiesel and biodiesel blend combustion <sup>f</sup>	Biodiesel blends	+/-CO <sub>2</sub> , -PM, -CO, -VOCs, -HC, -SO <sub>2</sub> , +NO <sub>x</sub> , +CH <sub>4</sub> , PAHs	n.incl.	n.incl.	Biodiesel blend leaks & spills
Organic waste collection	Petrodiesel	CO <sub>2</sub> fossil, CH <sub>4</sub> , N <sub>2</sub> O, CO, HC, VOCs, PM, SO <sub>2</sub> , NO <sub>x</sub> , NH <sub>3</sub> , PAHs	BOD5, COD, metals, ammonia, nitrate	Solid wastes (hazardous & non-hazardous)	—
Composting	Organic wastes Water	CH <sub>4</sub> , N <sub>2</sub> O	n.incl.	n.incl.	—
Compost transportation and distribution	Petrodiesel	CO <sub>2</sub> fossil, CH <sub>4</sub> , N <sub>2</sub> O, CO, HC, VOCs, PM, SO <sub>2</sub> , NO <sub>x</sub> , NH <sub>3</sub> , PAHs	BOD5, COD, metals, ammonia, nitrate	n.incl.	—
Compost use as an alternative to chemical fertilizers	Composts	Benefits of applying compost include, for example: Increase soil carbon storage; Decrease water use; Decrease soil erosion; Decrease herbicide use.			

<sup>a</sup> Two forms of ammonia in water include unionized ammonia (NH<sub>3</sub>) and ammonium ion (NH<sub>4</sub><sup>+</sup>).

<sup>b</sup> CO<sub>2</sub> absorbed by plants.

<sup>c</sup> Emissions from biodiesel production facilities in biodiesel plants: leakages from the storage of input materials and the operation of biodiesel manufacturing equipment (GenSolutions, 2007). Since those emissions are not high and their incidence is infrequent, they can be controlled under thorough management and do not have much impact on human health and ecosystems (Charman et al., 2012; Sheehan et al., 1998). Those emissions are normally omitted in life cycle assessment studies.

<sup>d</sup> Data not included in this summary table.

<sup>f</sup> Differences in the amount of exhaust emissions from biodiesel blend-used engine in comparison with petrodiesel-used engine. (–) means probable decrease; and (+) means probable increase.

**Table 5**

Sample inventory of life cycle costs of vegetable oil-based biodiesel systems, adapted from Haas et al. (2006), Lisboa et al. (2014), Woodward (1997) and Yaakob et al. (2013).

Item	Description
<i>Initial capital costs:</i>	One-time investment
Preliminary cultivation costs	Payments for sowing seeds, seedling practices and post-cultivation soil preparation regarding perennial feedstocks
Fixed capital	Purchase costs of storage facilities, process equipment, and utility equipment
Installation costs	Costs of plant construction, machinery installation, and worker training
Indirect costs	Costs of licenses and engineering
<i>Operating and maintenance costs:</i>	Annual costs
Raw material costs	Costs of input materials for every stage including their transportation
Labor costs	Workforce for the operation, maintenance, supervisory, and fringe benefits
Utilities costs	Costs of electricity, cooling water, and steam
Waste treatment	Costs for wastewater and solid waste treatment
General works	Costs of administration, property taxes, and property insurance
Supply costs	Maintenance supplies and operating supplies
Depreciation	Discount rate of return
Payback period	Necessary period to recover initial investment costs
Benefits	Prices of biodiesel and co-products

### 5.3. Potential ecological impacts of biodiesel and its blends spill and leak

#### 5.3.1. Biodegradation of biodiesel

Previous related studies indicate that the biodegradation rate of biodiesel is considerably higher than petrodiesel. In aquatic environments, within 28 days, various feedstock-based biodiesel fuels degrade about 87% on average, which is three times faster than conventional diesel. Moreover, through co-metabolism, biodiesel in a mixture can boost the biodegradation of diesel and consequently, the biodegradation rates of biodiesel/petrodiesel mixtures are higher than petroleum diesel alone (Wedel, 1999; Zhang et al., 1998). In soil, within 28 days, biodiesel degrades about 88% on average - approximately 1.7 times higher than pure petrodiesel. Furthermore, an interesting result shows that the blend of 20% vegetable oil-based biodiesel has a higher biodegradable potential than that of pure vegetable oil-based biodiesel (100% biodiesel) (Ginn et al., 2013).

#### 5.3.2. Aquatic toxicity

Compared to petrodiesel, several studies observe significantly lower toxicity to the aquatic environment of biodiesel. According to CytoCulture, the concentration required to kill 50% of the population (LC50) for different species of larval fishes and shrimps exposed to biodiesel varies from 122 ppm to 736 ppm. Meanwhile, that of petrodiesel ranges from 2.9 ppm to 39 ppm (Wedel, 1999). This indicates that the potential for damage to an aquatic environment of biodiesel is from 19 to 42 times less than conventional diesel.

Another study from the Institute of Arable Crop Research also demonstrates a noticeable stress reduced from fuel spills of biodiesel compared to that of petrodiesel. For example, at the dose rate of 1.25 g/l, while the development of *Lemna minuta* (least duckweed) was completely stopped in diesel, it could remain at 60% in biodiesel. Regarding impacts on mortality rate and weight loss of aquatic species, biodiesel also presented remarkable improvement (Birchall et al., 1995).

#### 5.3.3. Toxicity in soil

Lapinskiene et al. (2006), in their study on assessing the difference in ecotoxicity potential between biodiesel and petrodiesel in aerated soil, found that up to 12% (by weight) of concentration, biodiesel has no toxic impacts, while petrodiesel is toxic at 3% of concentration (by weight).

### 5.4. Examples of life cycle cost inventory

Following the LCC approach, total life cycle costs of biodiesel

systems include one-time investment costs and annual costs. Table 5 shows the main common costs of biodiesel systems.

Capital costs of biodiesel production's life cycle is a one-time investment including costs of the land-use area, plant construction, storage and process facilities and equipment and utility equipment, installation costs, and other relevant costs. Operating costs are an annual payment for labor, utilities, required materials for biodiesel production, and other supplementation costs.

## 6. Concluding remarks

The purpose of this paper is to develop a precise methodological framework for the estimation of Triple I based on the current context of LCSA. In general, this framework can promote the application of Triple I in the biofuel field, as it provides several appropriate methods for the estimation and suggests various scenarios needed to be taken into account in case of biofuel use. Moreover, under this developed framework, the equation of Triple I was adapted to the new conversation factors where both human health and ecosystem quality impacts were monetarized first, then converted to global hectare together with the LCC data. On the other hand, this study also proposed a new application of Triple I, so-called Triple I payback which estimated time required for recovering all economic, human well-being, environmental and social burdens throughout a product's life cycle. It is noted that the Triple I framework developed in this study can be applied in other research fields with some adaptation; for instance, in life cycle inventory and product release scenarios. As mentioned in the introduction, the Triple I provides a complementary index for sustainability assessments which consider economic, environmental, human well-being and social issues. Therefore, its outcome is comprehensive, easy to understand and highly applicable, especially with respect to the policy-making process. Remarkably, due to the employment of average global hectare as its final unit, the Triple I also can become a transboundary tool to compare technologies and products among various nations. The limitation of this study was the absence of social issues in the method. This requires further research of social life cycle assessment in Triple I.

The demonstration for the application of this newly developed framework with a case study of an inedible vegetable oil-based biodiesel system in Ha Long Bay is shown in Part II of this study.

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