Final Report

Project Title “Development and Application of Consequential Life Cycle Assessment Method for Food and Fuel in Thailand and Asia”

By Asst. Prof. Dr. Trakarn Prapaspongsa

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Abstract

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Abstract: Currently, Thailand and many Asian countries have applied life cycle assessment (LCA) as a tool for impact assessment towards green economy. The main approach being applied is generally the “status-quo” or “attributional” approach, considering only average data and applying different allocation methods when dealing with co-products. The application of another approach which has been widely applied in Europe and many countries – so called consequential LCA (CLCA) is still rare in Thailand and Asia. This approach considers the actual impacts from change in demand of a product. It deals with cause-effect relationship by identifying what will happen if we increase the production of the product in question. The results of CLCA reflect reality addressing the consequences of changes. The main objective of this project was to develop the guideline for consequential life cycle assessment method with case studies on food and fuel in Thailand and Asia. Firstly, the CLCA method for assessing food and fuel crops was developed by providing clear modelling steps, important marginal suppliers and recommended applications. The guideline could be used for LCA researchers and practitioners in Thailand and other countries in Asia. Afterwards, case studies applying the CLCA method on biofuel in Thailand and food consumption in Asia were carried out. In the case studies, both CLCA and attributional LCA (ALCA) modelling approaches were used. Specific questions and conditions which could be more suitable for each modelling choice are addressed. The attributional modelling is more suitable for national environmental taxation and emission labelling/accounting for import-export while the consequential modelling is more appropriate for new production development and eco-design. Due to the potential environmental risks arising from the modelling limitations, the consideration of both the widely applied approaches could support decisions more comprehensively.

Keywords : Life cycle assessment method; LCI modelling; Cause-effect relationship; CLCA; ALCA.

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Executive summary

Introduction to the research problem and its significance

Currently, Thailand and many Asian countries have applied life cycle assessment (LCA) as a tool for impact assessment towards green economy. The main approach being applied is generally the “status-quo” or “attributional” approach, considering only average data and applying different allocation methods when dealing with co-products. The application of another approach which has been widely applied in Europe and many countries – so called consequential LCA (CLCA) is still very limited in Thailand and Asia. This approach considers the actual impacts from change in demand of a product. It deals with cause-effect relationship by identifying what will happen if we increase the production of the product in question. The results of CLCA will reflect reality to support future decisions better than the traditional approach which considers the average technologies or suppliers in the assessment. Therefore, it is important to identify how to apply such approach to improve the tool for supporting decisions towards the more sustainable society in the future.

LCA courses offered in Thailand do not typically include examples on CLCA with system expansion and focus mostly on the use of mass/energy/economic allocations and average technologies/suppliers. Furthermore, some LCA experts in Thailand indicated the difficulties in the application of CLCA method due to lack of understanding and supporting databases. In fact, the update version of the ecoinvent database (ecoinvent database 3) – the secondary LCA database widely applied worldwide has embedded the consequential and attributional LCI data for LCA practitioners. The applications of this approach are expected to be higher in the future due to the increase in supporting data availability. A guideline with clear modelling steps, recommended applications and illustrating case studies on food and fuel from this study will be useful for future decision supports in Thai and Asian context to minimize the risks of negative impacts while maximizing the potential benefits from improvements and consumption.

Literature review

Consequential and attributional LCA modelling approaches are described in **Table E1**. The aim of the CLCA methodology is to identify the impacts from an increase in demand of a product. The LCA shall make the critical issues arising from possible adverse impacts and alternatives to impact reduction visible to decision-makers.

Table E1: Main characteristics of consequential LCA (CLCA) and Attributional LCA (ALCA) (Guinée et al., 2002; Thomassen et al., 2008; Weidema et al., 2009; Weidema, 2003).

Characteristics	CLCA	ALCA
Synonym	Change-oriented	Status quo
Type of questions to be answered	Assessing "*consequences of changes*" or "*consequences of the decision or the impacts from change in demand*"	Accounting or assessing attribution or allocation of the impacts
Data	Marginal future (or including relevant unit processes and marginal (actual affected) suppliers/technologies	Average historical (or including average suppliers/technologies)
Knowledge required	Physical and market mechanisms	Physical mechanisms
Functional unit	Represents change in volume	Represents static situation
System boundaries	Affected processes by change in demand	Static processes
System expansion	Obligatory	Optional
Co-product allocation	Never used	Frequently used
Hotspot identification	System-dependent	System-dependent
Comprehensibility (by LCA practitioner)	Difficult of arbitrary allocation factors	Difficult inclusion of future processes
Quality	Higher sensitivity to uncertainties	Sensitive to uncertainties
Data availability	Similar	Similar

In the consequential LCA, it is important to identify marginal (or actually affected) suppliers with respect to relevant time and spatial aspects, affected processes/markets, market trends and constraints, and supplier flexibilities, as briefly described in **Table E2**.

Table E2: Short description of the approach to identifying marginal suppliers (Weidema et al., 2009).

Factors	Description
Scale	*Small scale:* If the change does not affect the overall market situation, the impacts will be linearly related to the size of the change. *Large scale:* If the change affects the overall market situation (e.g., brings new suppliers, new market, etc.), the impacts will not be linearly related to the size of change.
Time horizon	*Short-term change:* If the change affects only capacity utilization but does not affect capacity itself, capital goods in the life cycle inventories will be excluded. However, individual short-term purchase decisions will finally result in the accumulation of market trend and capital investment (long-term change). Long-term change: If the change affects capacity investment (e.g., installation of new capacity or phased-out machinery), capital goods in the life cycle inventories will be included.
Market delimitation	The identification of geographical, temporal and customer segmentation is required prior to the determination of market trends and change in supply and demand. The general assumption is that no market boundary exists.
Market trends	Increasing trend: New capacity must be installed. The modern or the most competitive supplier/technology will be affected. Decreasing trend: Existing capacity will be phased out. The least competitive supplier/technology (e.g., old technologies) will be affected.
Change in supply and demand	*Full elasticity of supply:* Increases in outputs of the upstream activities will require the same amount of increases in demands of the downstream activities. *Constraints:* Constrained suppliers are not affected by the change in demand and are therefore excluded from the assessment.

In order to identify the marginal suppliers or technologies and fulfil the factors in **Table E2**, the following questions need to be answered (Ekvall and Weidema, 2004; Weidema et al., 1999; 2009) as follows.

- What are the relevant time aspects (*Time horizon*)?
- Are specific processes or overall markets affected (*Scale and market delimitation*)?
- What is geographical, temporal and customer segmentation of the market (*Market delimitation*)?

- What is the trend in the market (*Market trend*)?
- What technologies are flexible (*Change in supply and demand*)?
- What technology is actually affected (*Marginal technology*?

Regarding the application area of CLCA, it is important to understand the questions to be answered. The aim of all LCAs is typically to support decisions on the replacement between two product systems. For examples, the studies on hot-spot identification are often aim to provide better alternatives for improving future production systems (after the hot-spot has been identified) and the studies of a single product are always later used for comparison. This can be argued that CLCA can be used in most LCA applications (Ekvall 1999; Weidema et al., 2009). The cases when CLCA are less relevant include studies at societal level with a focus on entire environmental impacts from all human activities, studies on environmental taxation and studies seeking to avoid criticism and to reward the past good behaviours (Weidema et al., 2009). The detailed methodology of this LCA approach was published in Ekvall and Weidema (2004) and Weidema et al. (2009). Apart from this modelling approach, other economic modelling approaches (such as multi-market, multi-regional partial equilibrium modeling; and computable general equilibrium modeling) have been applied in CLCA studies as well. **Table E3** demonstrates the CLCA applications reviewed by Earles and Halog (2011). It could be seen that the approach based on Weidema et al. (2009) which is the update version of Weidema et al. (1999) and Ekvall (2000) is the most widely used as also documented in Prox and Curran (2017). This approach is therefore selected as the main method in this research.

Table E3: Applications of CLCA (Obtained from Earles and Halog, 2011).

Reference	Topic	CvA	SWA	PEM	MMMR-PEM	CGE	ILUC	RE	Notes
Hofstetter and Norris (2003)	Occupational Health	Y	N	N	N	N	N	Y	Framework for CLCA w/r/t occupational health impacts; argues for inclusion of changes in unemployment and substitution effects in labor market
Weidema (2003)	Many topics	N	Y	N	N	N	N	N	Extension of Weidema et al. (1999)
Ekvall and Andrae (2006)	Electronics	Y	Y	Y	N	N	N	N	Extension of Ekvall (2000) to electronics industry
Thrane (2006)	Fishing	N	Y	N	N	N	N	N	Application of Weidema et al. (1999)
Lesage et al. (2007a)	Brownfields	N	N	Y	N	N	N	N	Extension of Ekvall (2000) to brownfields
Lesage et al. (2007b)	Brownfields	Y	N	Y	N	N	N	N	Extension of Ekvall (2000) to brownfields
Eriksson et al. (2007)	Heating	N	N	N	N	N	N	N	Optimization model of EU energy market used to determine affected technology
Sanden and Karlstrom (2007)	Renewable fuels	N	Y	N	N	N	N	N	Experience curves introduced into CLCA
Schmidt et al. (2007)	Paper	N	Y	N	N	N	N	N	Application of Weidema et al. (1999)
Spielmann et al. (2008)	Mass transit	N	N	N	N	N	N	Y	Rebound effects associated with changes in mobility patterns due to time savings

Reference	Topic	CvA	SWA	PEM	MMMR-PEM	CGE	ILUC	RE	Notes
Thiesen et al. (2008)	Consumer goods	N	N	N	N	N	N	Y	Rebound effect of price differences between products on consumption
Kløverpris et al. (2008)	Agriculture	N	N	N	N	Y	Y	N	Utilizes CGE model (GTAP) to estimate ILUC from agricultural expansion
Dalgaard et al. (2008)	Agriculture	N	N	Y	Y	N	N	N	Utilizes economic model ESMERALDA to identify affected technology
Schmidt and Weidema (2008)	Agriculture	N	Y	N	N	N	Y	N	Augments Weidema et al. (1999) and Ekvall (2000); agricultural statistics and outlooks to determine affected technologies
Schmidt (2008a)	Agriculture	N	Y	N	N	N	Y	N	Augments Weidema et al. (1999) and Ekvall (2000); agricultural statistics and outlooks to determine affected technologies
Schmidt (2008b)	Agriculture	N	Y	Y	N	N	Y	N	LCIA of land-use change with respect to biodiversity
Thomassen et al. (2008)	Agriculture	Y	Y	Y	N	N	Y	N	Application of Schmidt and Weidema (2008)
Frees (2008)	Metals	N	N	Y	N	N	N	N	Application of Ekvall (2000)
Pehnt et al. (2008)	Electricity	N	N	N	N	N	N	N	Coupling LCA and stochastic European electricity market model
Searchinger et al. (2008)	Biofuel	N	N	Y	Y	N	Y	N	Use of FAPRI PE model to estimate ILUC impacts with respect to US biofuels policy
Vieira and Horvath (2008)	Buildings	Y	Y	Y	N	N	N	N	Application of Weidema et al. (1999) and Ekvall (2000) to building end-of-life stage

Reference	Topic	CvA	SWA	PEM	MMMR-PEM	CGE	ILUC	RE	Notes
Kløverpris (2009)	Agriculture	N	N	N	N	Y	Y	N	Kløverpris et al. (2008) + biomass characterization
Reinhard and Zah (2009)	Biofuel	Y	Y	Y	N	N	Y	N	Application of Schmidt and Weidema (2008)
Silalertruksa et al. (2009)	Biofuel	N	Y	Y	N	N	Y	N	Application of Schmidt and Weidema (2008)
US EPA (US 2010)	Biofuel	N	N	Y	Y	Y	Y	N	Use of large FAPRI AND FASOM PE models, and GTAP CGE model to determine ILUC impacts with respect to US biofuels policy
Kløverpris et al. (2010)	Agriculture	N	N	N	N	Y	Y	N	Application of Kløverpris et al. (2008)

Y = Yes; *N* = No; *CvA* = Comparison of consequential and attributional LCA; *SWA* = use of step-wise approach to identifying affected technology (from Weidema et al. 1999); *PEM* = partial equilibrium modeling; *MMMR-PEM* = Multi-Market, Multi-Regional Partial Equilibrium Modeling; *CGE* = computable general equilibrium modeling; *ILUC* = indirect land-use change examined; *RE* = rebound effects examined.

Objectives

The main objective of this project is to develop the guideline for consequential life cycle assessment method with case studies on food and fuel in Thailand and Asia. The main question to be answered in the case studies is the assessment on "*the consequences of changes*" in food and fuel. Therefore, the application of CLCA could be more appropriate in order to support future decision how to minimize the risks of negative impacts while maximizing the potential benefits from the change in demand or increased consumption. The sub-objectives are:

- To develop CLCA method for assessing major food and fuel crops in Thailand and Asia by providing clear modelling steps, important marginal suppliers and recommended applications
- To carry out case studies applying the CLCA method on food and fuel in Thailand and Asia

Methodology

In this research, the CLCA method for food and fuel in Thailand and Asia is developed based upon existing LCA guidelines and new specific case studies for food and fuel. The main consequential LCA modelling guidelines used in the study are Weidema et al. (2009) and Ekvall and Weidema (2004) as well as other international LCA guidelines. The guideline includes the clear steps how to carry out CLCA method (i.e., how to avoid product allocation by system expansion and how to identify marginal technologies/suppliers), recommended applications, advantages and limitations. The research in this part is presented in **Chapter 1**. Afterwards, the CLCA method is applied on case studies by considering the four LCA phases according to the ISO 14040 and 14044 standards (ISO 14040, 2006; ISO 14044, 2006) - definition of goal and scope, life cycle inventory analysis, life cycle impact assessment and life cycle interpretation. The case study on LCA of biofuel in Thailand could be seen both in **Chapter 1** and **Chapter 2**. The case study on LCA of food consumption in Asia is documented in **Chapter 3**.

Chapter 1 Consequential Life Cycle Assessment (CLCA) Guideline

1.1 Why CLCA?

Various modelling approaches are available for life cycle assessment (LCA). Consequential and attributional LCA (CLCA and ALCA) are the modelling choices widely applied in LCA studies (Martin et al., 2015; Prapaspongsa and Gheewala, 2017; Schmidt and Brandão, 2013; Silalertruksa and Gheewala, 2013). UNEP/SETAC Life Cycle Initiative (2011) defined CLCA and ALCA modelling approaches as “system modelling approach in which activities in a product system, are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit” and “system modelling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule”, respectively. Both modelling choices have intensively been debated on vis-à-vis suitability, reliability and creditability for decision support (Anex and Lifset, 2014; Brandão et al., 2014; Dale and Kim, 2014; Hertwich, 2014; Plevin et al., 2014a, b, c; Suh and Yang, 2014).

Based on the stepwise allocation procedure from ISO 14044, the allocation should be avoided by unit process division or system expansion (Clause 4.3.4.2; ISO, 2006). ISO 14044 and ISO/TR 14049 (ISO, 2006; ISO, 2012) clearly stated that the unit process division and system expansion are not part of the allocation procedure. To avoid the allocation, wherever possible, the unit processes could be divided into sub-processes where specific input and output data of the sub-processes can be independently collected. ISO/TR 14049 (ISO, 2012) illustrated an example how to expand the boundaries for comparison of systems with different outputs (system expansion) by adding actually involved supplementary processes so that the comparative options will have the same final products. The actually involved unit processes or so-called marginal suppliers could be identified by market based relationship consideration (Weidema, 2003; Weidema et al., 2009). Consequential-LCA (2017) defined system expansion as “a procedure for eliminating by-products as activity outputs by including them instead as negative inputs, thereby including the additional functions related to the by-products and modelling the resulting changes (substitutions) in the product system, especially by including the reduction in supply of the same product from the marginal supplier to the market for the by-product”. System expansion is the main step to avoid allocation in CLCA whereas allocation is the key approach in ALCA (Brandão and Weidema, 2013; De Camillis et al., 2013). This may reflect the preference of CLCA applications based on the ISO 14040/44. Nevertheless, there have been different interpretations on the allocation hierarchy of the ISO standards for LCA (Weidema, 2014) resulting in the lack of consensus in the selection of modelling choices.

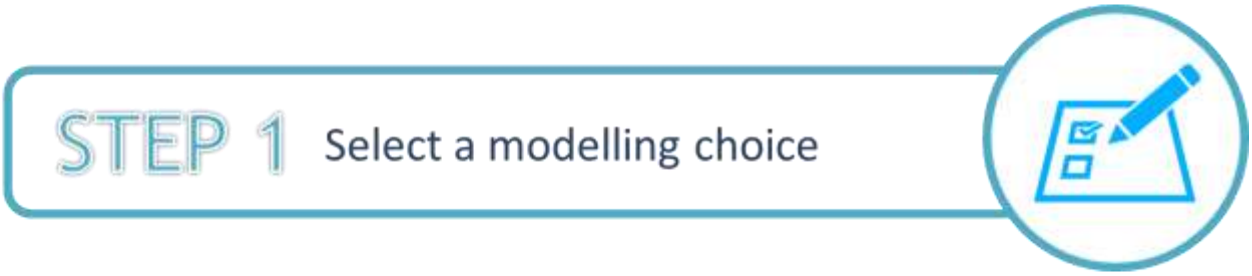
In order to provide a robust decision support, the inclusion of the two main modelling schools and the determination of specific questions which each modelling choice could potentially answer are crucial (Prapaspongsa and Gheewala, 2017). Most LCA studies in emerging economies and in Thailand are based on an attributional approach. The case studies and guideline for CLCA applications in Thailand are rare and needed. This English-Thai guideline is intended to provide a simplified operational guide for advanced LCA practitioners, researchers and graduate students to simply carry out CLCA based on existing CLCA methodology with illustrative cases on biofuels in Thailand (Prapaspongsa and Gheewala, 2017). The users of this guideline should have knowledge on basic LCA methodology and experiences in LCA research/applications. The detailed CLCA methodology could be furthered reviewed in Ekvall and Weidema (2004), Weidema et al. (2009) and Consequential-LCA (2017).

1.2 CLCA Step-by-Step

To carry out LCA under the consequential approach, the sequential procedure should be followed as illustrated in **Figure 1.1**. In this guideline, the CLCA step-by-step (Step 1 to 4) which are not context-specific are initially presented and followed by specific case studies on biofuels. Step 1 describes the differences in CLCA and ALCA and the relevant questions for the two modelling choices so that the users are able to select the relevant modelling choice according to the goal and scope of the study. If CLCA is chosen, Step 2 to 4 will be followed. The detailed procedure is documented in next section.

Figure 1.1: CLCA step–by-step

1.3 CLCA Procedure



STEP 1 describes the different characteristics of CLCA and ALCA and the specific questions which are relevant for the two modelling choices. By considering the characteristics and relevant questions, the users are able to select a modelling choice in accordance with the objectives defined in the goal and scope of the study. If CLCA is chosen, the next 3 steps in this guideline should be followed.

DIFFERENCES BETWEEN CLCA AND ALCA

CLCA and ALCA are different by definition as previously mention in **Section 1.1**. The main characteristics and the key differences between CLCA and ALCA are summarised in **Table 1.1** and **Figure 1.2**.

Table 1.1: Main characteristics of CLCA and ALCA (Guinée et al., 2002; Thomassen et al., 2008; Weidema et al., 2009; Weidema, 2003).

Characteristics	CLCA	ALCA
Synonym	Change-oriented	Status quo
Type of questions to be answered	Assessing "*consequences of changes*" or "*consequences of the decision or the impacts from change in demand*"	Accounting or assessing attribution or allocation of the impacts
Data	Marginal future (or including relevant unit processes and marginal (actual affected) suppliers/technologies	Average historical (or including average suppliers/technologies)
Knowledge required	Physical and market mechanisms	Physical mechanisms
Functional unit	Represents change in volume	Represents static situation

Characteristics	CLCA	ALCA
System boundaries	Affected processes by change in demand	Static processes
System expansion	Obligatory	Optional
Co-product allocation	Never used	Frequently used
Hotspot identification	System-dependent	System-dependent
Comprehensibility (by LCA practitioners)	It is difficult for CLCA practitioners to include arbitrary allocation factors which have been used in ALCA.	It is difficult for ALCA practitioners to include marginal future data which have been considered in CLCA.
Quality	Higher sensitivity to uncertainties	Sensitive to uncertainties
Data availability	Similar	Similar

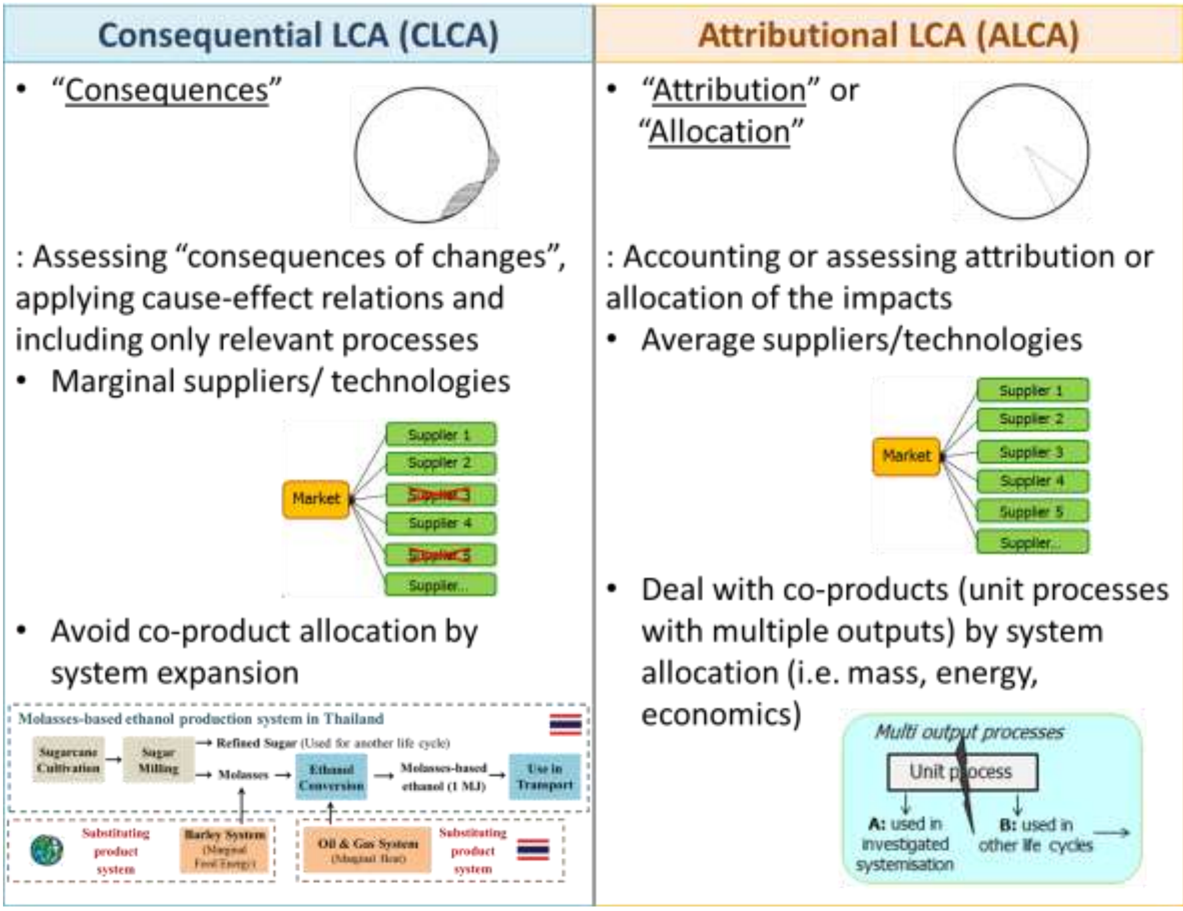


Figure 1.2: Key differences between CLCA and ALCA (Based on Weidema, 2003; Weidema et al., 2009; Prapaspongsa, 2017; Prapaspongsa and Gheewala, 2017)

In order to provide case studies on biofuels in this guideline, the overall system boundaries of Thailand's palm biodiesel, molasses ethanol and rice straw electricity production systems are illustrated in **Figures 1.3 to 1.5**.

Each figure demonstrates the investigated life cycle stages under CLCA and ALCA modelling approaches.

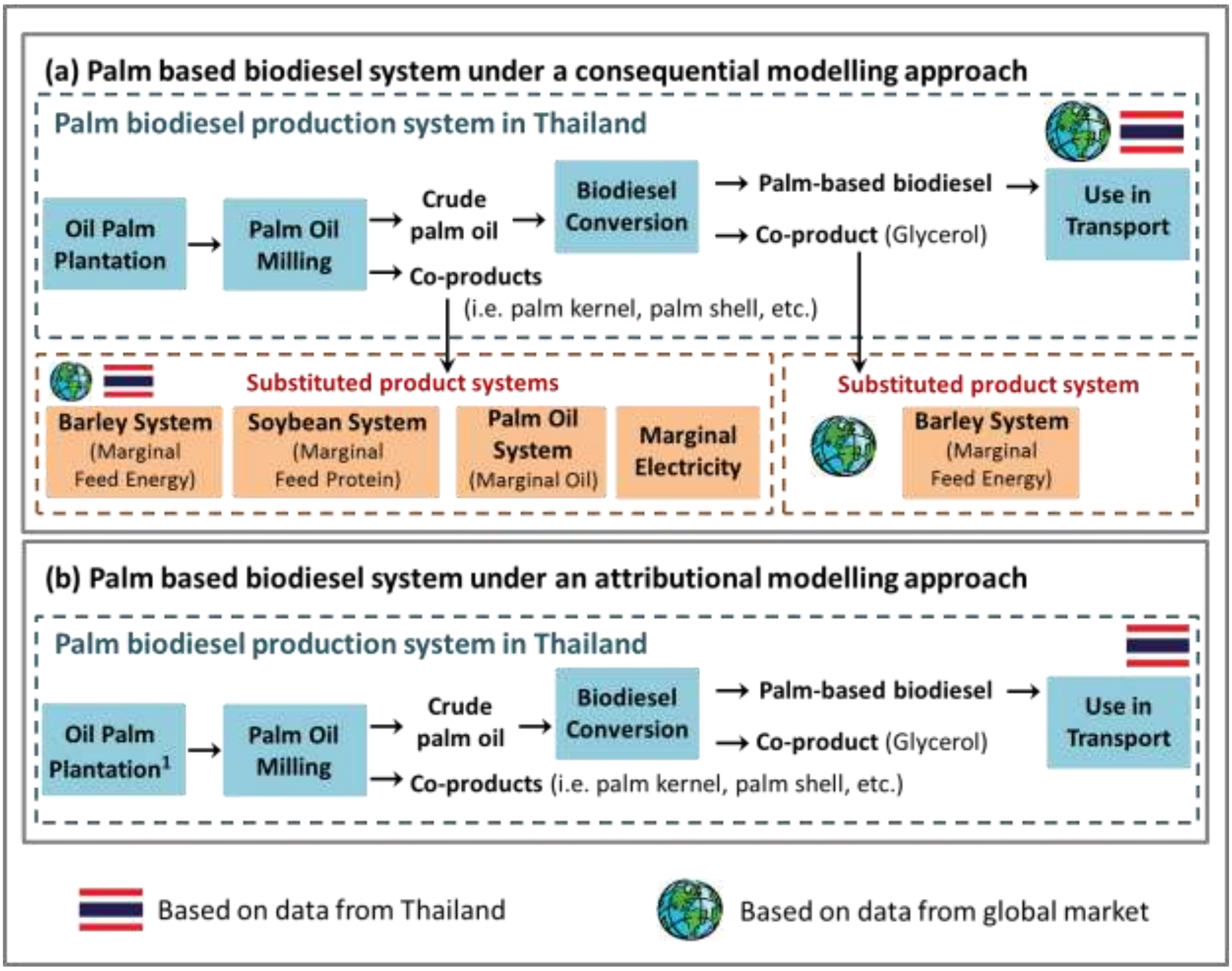


Figure 1.3: Overall system boundaries of Thailand's palm biodiesel production (Prapaspongsa and Gheewala, 2017). (a) Palm based biodiesel system under a consequential modelling approach. Electricity production from palm kernel and palm shell substitutes the marginal electricity system. Palm kernel oil substitutes the marginal oil system. Palm kernel meal substitutes the marginal feed protein and feed energy. Glycerol substitutes the marginal feed energy. (b) Palm based biodiesel system under an attributional modelling approach.

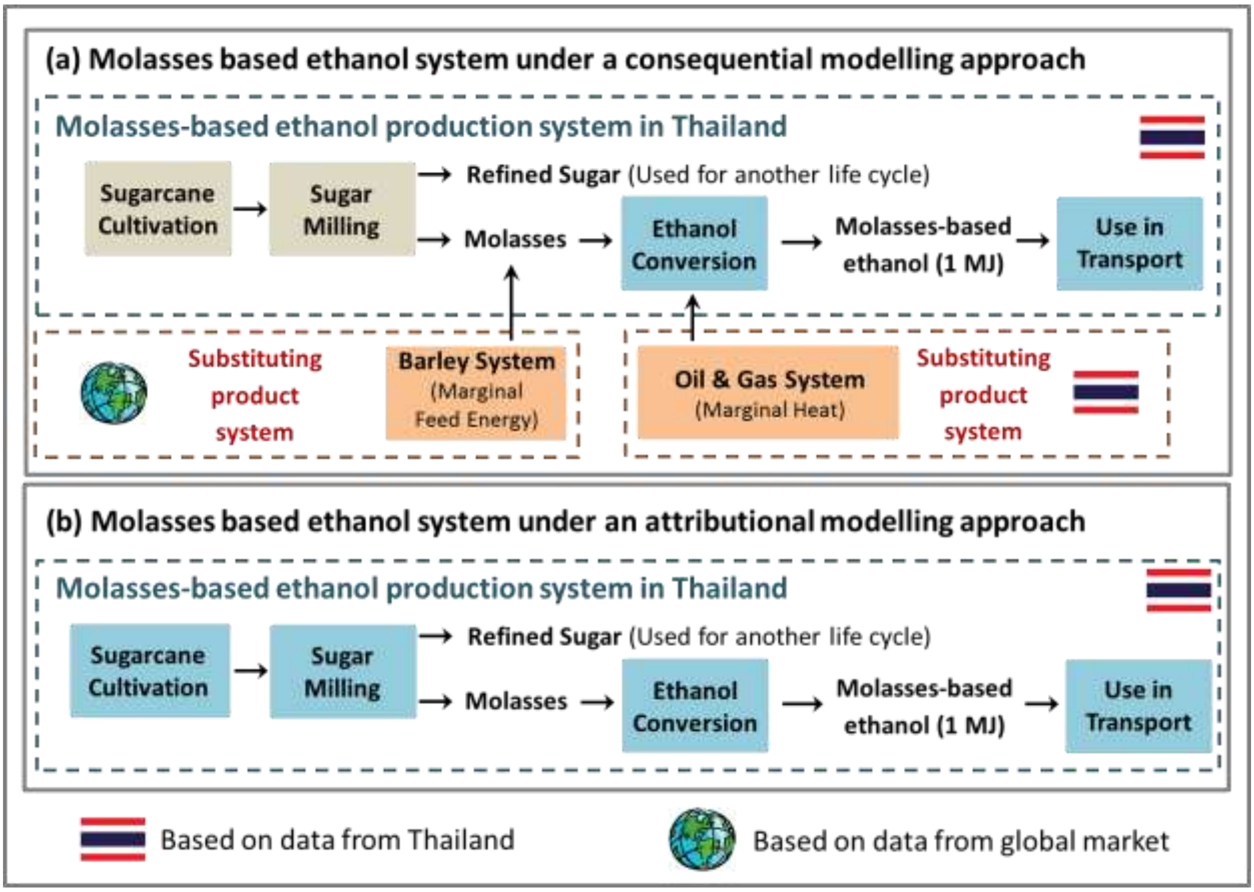


Figure 1.4: Overall system boundaries of Thailand's molasses ethanol production (Prapaspongsa and Gheewala, 2017). (a) Molasses based ethanol system under a consequential modelling approach. The grey boxes represent the excluded processes because they are not affected by the additional molasses ethanol production. (b) Molasses based ethanol system under an attributional modelling approach.

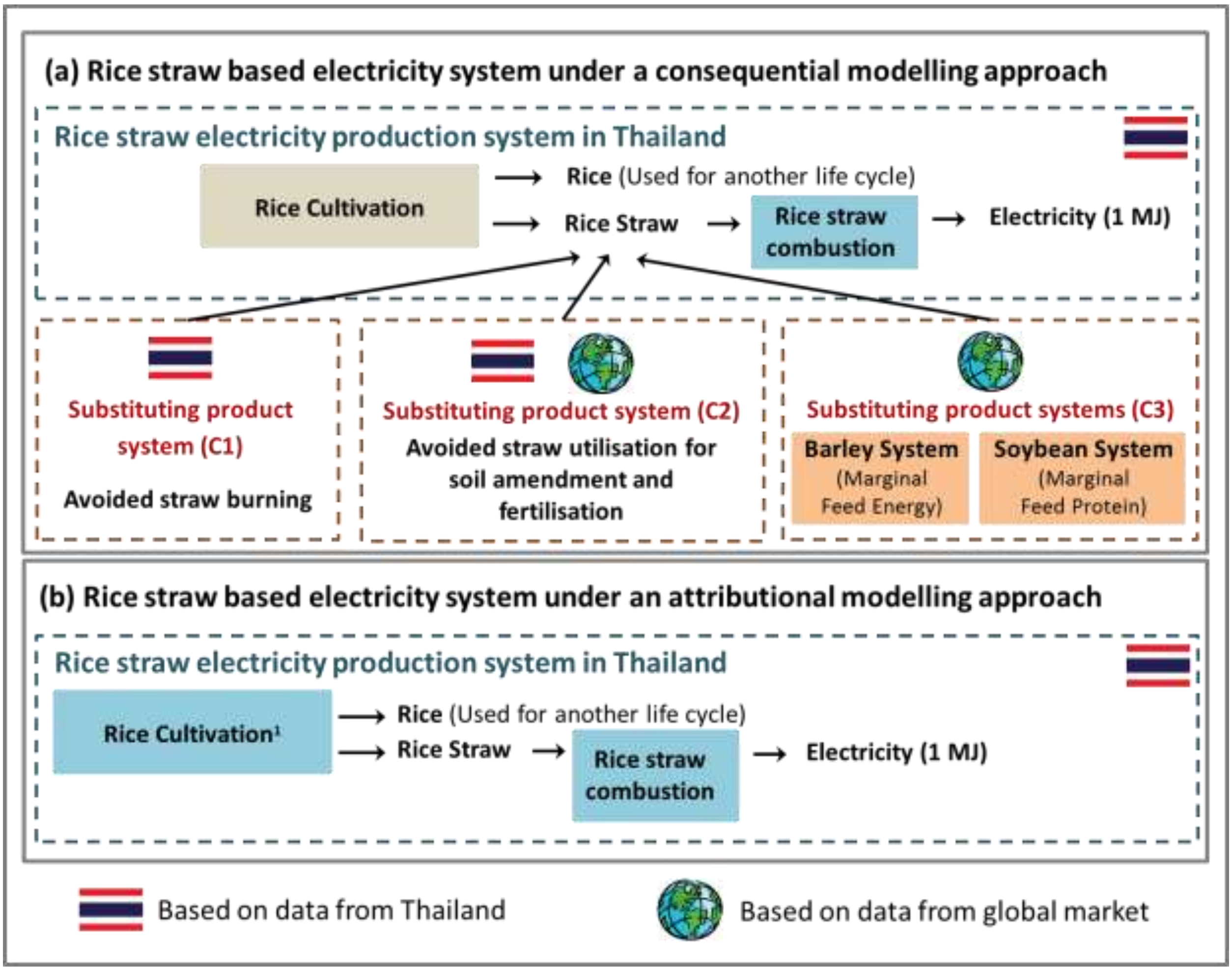


Figure 1.5: Overall system boundaries of Thailand's rice straw electricity production (Prapaspongsa and Gheewala, 2017). (a) Rice straw based electricity system under a consequential modelling approach. The grey boxes represent the excluded processes because they are not affected by the additional rice straw electricity production. (b) Rice straw based electricity system under an attributional modelling approach.

How to identify marginal suppliers and avoid co-product allocation by system expansion in the CLCA case studies (**Figure 1.3a, 1.4a, 1.5a**) will be explained later in **Step 3 and 4**. In the ALCA case studies, the environmental impacts from relevant life cycle stages are allocated to palm biodiesel and other co-products (**Figure 1.3b**); to refined sugar and molasses (**Figure 1.4b**); and to rice and rice straw (**Figure 1.5b**) by using economic allocation factors (Prapaspongsa and Gheewala, 2017).

QUESTIONS FOR CLCA AND ALCA

The relevant questions for LCA in general, CLCA and ALCA presented in this section have been published in Prapaspongsa and Gheewala (2017). "*To assess climate change mitigation potential by considering GHG emissions of biofuels in comparison with fossil energy under consequential and attributional modelling approaches*" with the functional unit (FU) as 1 MJ of biomass-based fuels and fossil energy could be set as the objective for both approaches. The problem is that the defined objective is too broad and does not distinguish the specific situations each approach could be suitable for. This objective is intentionally set to imitate a typical condition when an LCA study is carried out to support general policies without specific users which could lead to different interpretations and conclusions due to its generality. *What could then be the specific objectives?*

The CLCA modelling approach determines the impacts from a change in demand and to include only the systems which are expected to change due to the additional product. Brandão and Weidema (2013) identified the questions which the CLCA approach attempts to answer are "*What are the net impacts associated to a change (in a product system) relative to the baseline scenario, where that change does not take place?*" and "*What are the consequences of a decision relative to the "no action" baseline?*". The specific question in this study could be "*What would actually happen or what are the potential consequences if we additionally produce 1 MJ of the biofuels in Thailand?*" The keywords are "*additionality*" and "potential affected processes linked via market mechanisms". The functional unit is rather the additional production of 1 MJ biofuels and fossil fuels in this case. Based on Brandão and Weidema (2013), this model looks at the baseline system as the situation without any action and considers only the additional product of concern without the assessment of the baseline system itself. With the specific question a CLCA study could properly provide the answer, it is therefore crucial for the policy makers/researchers to explicitly state the question during the goal and scope definition phase. To answer such a question, the expectedly actually affected processes linked via market mechanism or the marginal technologies/suppliers are identified by considering market information such as market delimitation, trends and constraints. This will later be explained in the next step.

The ALCA modelling approach partitions the total impacts to the functional unit of a product system by using a normative rule (i.e. economic allocation technique in this study). De Camillis et al. (2013) determined the relevant questions for this approach as "*What is the environmental impact of a certain product system at a given time (when baseline scenario is assessed)?*" and

"*What is the environmental impact of a certain product system in a given future scenario if the product were designed or/and produced or/and consumed or/and managed differently at the end of its life?*". For this study, the specific question could be "*What are the specific GHG emissions of 1 MJ biofuel production in Thailand allocated by economic values of all related co-products?*". The keywords are "*specification*" and "*allocation/attribution*". The functional unit as of 1 MJ biofuels and fossil fuels might still be valid in this case. This model firstly looks at the total emissions from specific production systems by including all product outputs in the systems which are directly linked throughout the supply-chains (i.e. the direct supply-chains for molasses-based ethanol, palm-based biodiesel and rice straw-based electricity in Thailand as illustrated in **Figures 1.3b, 1.4b and 1.5b**) and, secondly, seeks for a specific allocated share of the relevant impacts for the specific product being investigated.

STEP 2 Define a functional unit

STEP 2 explains how to define a functional unit for CLCA studies.

The functions and functional unit of product systems should be properly defined in the goal and scope definition phase of LCA. For comparative LCA studies, the studied products have to provide comparable or similar functions/outputs while the functional unit is defined as a reference unit reflecting a quantified description of the performance requirements (i.e. quality, quality and duration/time period) of the studied product systems (see **Figure 1.6**). For CLCA, the extent of the consequences of studied product systems (or decisions) studied should be reflected in the functional unit (Weidema et al., 2009). As mentioned earlier, in order to address the consequences of the decision, "additionality" could be specifically addressed. The functional unit of the CLCA case study could be "the additional production of 1 MJ biofuels and fossil fuels".

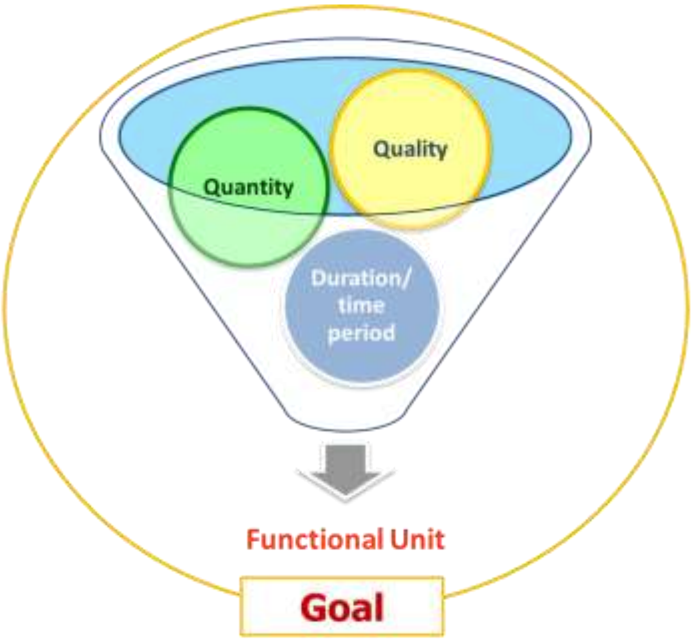


Figure 1.6: Functional Unit Identification

STEP 3 Identify marginal suppliers

STEP 3 explains how to identify marginal suppliers or suppliers/producers that will change production capacity in response to a change in demand for a product (Consequential-LCA, 2017). The identification of marginal suppliers (or the affected unit processes) in CLCA can be done by considering the scale and time horizon of the changes being assessed, the market delimitation, the market trends and the changes in supply and demand (Weidema et al., 2009). It the market trends are increasing, the most competitive suppliers/technologies will be affected, and vice versa. For system expansion, only the unconstrained systems driven the change in demand are included. The approach for marginal supplier identification is briefly described in **Table 1.2**.

Table 1.2: Short description of the approach to identifying marginal suppliers (Weidema et al., 2009).

Factors	Description
Scale	***Small scale:*** If the change does not affect the overall market situation, the impacts will be linearly related to the size of the change. ***Large scale:*** If the change affects the overall market situation (e.g., brings new suppliers, new market, etc.), the impacts will not be linearly related to the size of change.
Time horizon	***Short-term change:*** If the change affects only capacity utilization but does not affect capacity itself, capital goods in the life cycle inventories will be excluded. However, individual short-term purchase decisions will finally result in the accumulation of market trend and capital investment (long-term change). ***Long-term change:*** If the change affects capacity investment (e.g., installation of new capacity or phased-out machinery), capital goods in the life cycle inventories will be included.
Market delimitation	The identification of geographical, temporal and customer segmentation is required prior to the determination of market trends and change in supply and demand. The general assumption is that no market boundary exists.
Market trends	***Increasing trend:*** New capacity must be installed. The modern or the most competitive supplier/technology will be affected.

Factors	Description
	Decreasing trend: Existing capacity will be phased out. The least competitive supplier/technology (e.g., old technologies) will be affected.
Change in supply and demand	***Full elasticity of supply:*** Increases in outputs of the upstream activities will require the same amount of increases in demands of the downstream activities. ***Constraints:*** Constrained suppliers are not affected by the change in demand and are therefore excluded from the assessment.

In order to identify the marginal suppliers or technologies and fulfil the factors in **Table 1.2**, the following questions need to be answered (Ekvall and Weidema, 2004; Weidema et al., 1999; 2009) as follows.

- What are the relevant time aspects (*Time horizon*)?
- Are specific processes or overall markets affected (*Scale and market delimitation*)?
- What is geographical, temporal and customer segmentation of the market (*Market delimitation*)?
- What is the trend in the market (*Market trend*)?
- What technologies are flexible (*Change in supply and demand*)?
- What technology is actually affected (*Marginal technology)*?

For the CLCA case studies on biofuels demonstrated previously, the specific questions could be answered below and further explanation is provided after the list of these questions.

- What are the relevant time aspects?
  ⇨ Long-term change.
- Are specific processes or overall markets affected?
  ⇨ It is a small scale and there are both national and global markets for different systems.
- What is geographical, temporal and customer segmentation of the market?
  ⇨ There are both national and global markets for different systems.
- What is the trend in the market?
  ⇨ It is an increasing trend for all systems considered.
- What technologies/unit processes are flexible?
  ⇨ Some are flexible and some are constrained.
- What technology/process is actually affected?
  ⇨ It depends on each system (as described below).

As presented in **Figures 1.3a, 1.4a and 1.5a**, the palm biodiesel, molasses ethanol and rice straw electricity production systems in Thailand influence the substituted/substituting product systems via national and international market linkages. Molasses and rice straw are dependent co-products (so called by-products) from sugar and rice production systems whereas palm oil is the main product from the palm oil production system producing various co-products throughout the life cycle stages. The additional demand for molasses and rice straw will not increase sugarcane and rice production and result in additional production of substitutable products.

According to Prapaspongsa and Gheewala (2016), the use of molasses (as the main raw material) and bagasse (as heat energy source in the ethanol conversion stage) will contribute to additional feed energy and marginal heat energy source production. For rice straw utilisation, various utilisation pathways including field burning, soil fertilising and animal feeding are common in Thailand (Sansiribhan et al., 2014). The additional demand for palm diesel will increase the production of co-products during palm oil milling and biodiesel conversion stages and result in displaced production of substitutable products. Under consequential modelling, the co-products will substitute marginal electricity production (for palm kernel and shell) in Thailand and refined oil (for palm kernel oil), feed energy (for palm kernel meal and glycerol) and feed protein (for palm kernel meal) in the global market based on Schmidt (2007, 2015) and Schmidt and Brandão (2013). The marginal feed energy (or feeds which are the energy source for animals) system is from Lechón (2011) and Lechón et al. (2011) as identified in Prapaspongsa and Gheewala (2015) whereas the marginal feed protein (or feeds which are the protein source for animals) and refined oil systems are defined as palm oil based on Schmidt (2015) and Schmidt and Brandão (2013). Only the markets for marginal electricity and heat are delimited at a national level (in Thailand) whereas the markets for marginal oil, feed energy and feed protein are considered at a global level due to their current import-exports (Prapaspongsa and Gheewala, 2016; Prapaspongsa et al., 2017). Based on Thailand’s power development plan: 2012 – 2030 (PDP2010) and AEDP: 2012-2021, the marginal sources for electricity in Thailand could be identified according to the additional installed capacity from 2012 to 2021 as illustrated in **Figure 1.7** (Prapaspongsa and Gheewala, 2016). Biogas and municipal solid waste are not included because they are driven by the demand of waste and wastewater management.

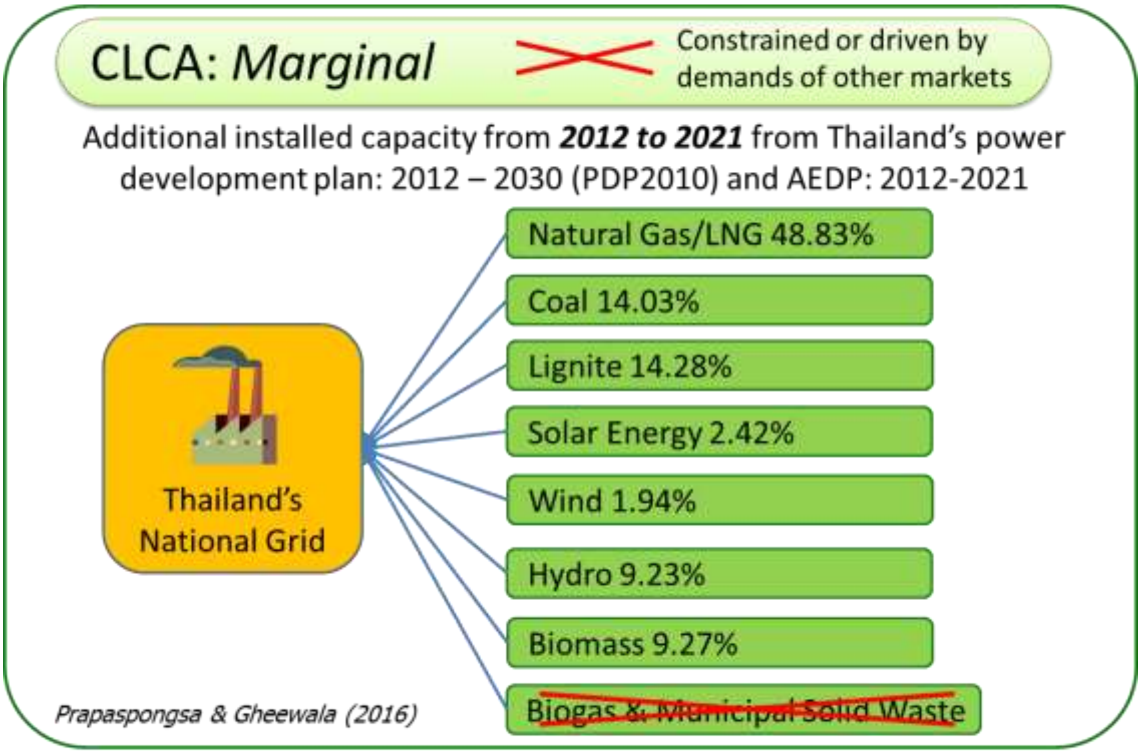


Figure 1.7: Marginal electricity in Thailand (Prapaspongsa and Gheewala, 2016).

It should be mentioned that **Figures 1.3 to 1.5** only illustrate the main and co-products of palm-based biodiesel, molasses-based ethanol and rice straw-based electricity systems. Wastewater treatment processes are not illustrated in the figures but their relating emissions are taken into account. In CLCA modelling, the wastewater treatment systems with biogas recovery with electricity generation and land application will displace marginal electricity and fertiliser production.

STEP 4 Deal with multi-output activities

STEP 4 demonstrates how to deal with multi-output activities by physical causality or system expansion under the consequential approach. Two production types - combined and joint production systems – are distinguished and explained.

COMBINED PRODUCTION

In the combined production systems, the relative amounts of co-products can be varied independently. The impacts of a co-product could be modelled by physical causality. Since there is no specific example from the CLCA case studies on biofuels in Thailand, a case study in Denmark is illustrated in **Figure 1.8**.

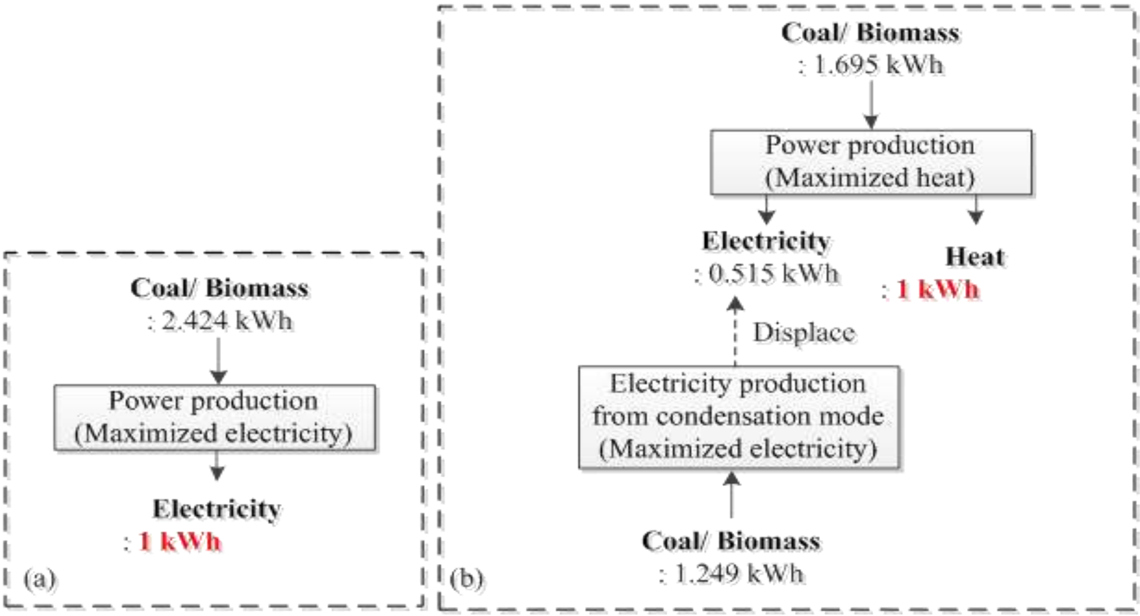


Figure 1.8: The diagrams of energy flow in the district heating system (Prapaspongsa et al., 2011). (a) Functional unit as the net production of 1 kWh of electricity when the electricity is maximized (b) Functional unit as the net production of 1 kWh of heat when the heat is maximized. The reference units are presented in red figures.

The diagrams illustrate the energy flow in the district heating system which is an extraction plant where the electricity and heat outputs could be varied (Prapaspongsa et al., 2011). Under normal operations, the plant co-produces heat and electricity. In order to assess the impacts from electricity production, the power production could be adjusted to maximise the production of electricity without heat (condensation mode). Afterwards, the heat production could be assessed by considered the maximised heat mode and the additional electricity will displace the electricity production from the condensation mode.

JOINT PRODUCTION

In the joint production systems, the relative amounts of co-products (or the output volumes) are fixed and cannot be varied independently. There are 3 types of joint production which are a determining co-product, a dependant co-product and multiple determining co-products. The determining co-product is the main product which will drive the additional production in respond to an increase in demand. The dependant co-product is the by-product which cannot drive the additional production in respond to an increase in demand. The product type could be identified by considering the alternative production route, revenues and costs. There is only one determining product if other co-products have an alternative production route (stand alone, competitive, flexible). The co-products with an alternative production route are often dependant. Another consideration to be determining is that a co-product shall provide economic revenue that exceeds the net marginal cost of changing the production volume and have a larger market trend than any other joint products.

If the studied/used product is the determining co-product, CLCA will consider the environmental interventions of the multi-output activities, the intermediate treatment activities and the avoided activities (multiple-output activity + intermediate treatment activities - avoided activities). Based on the biofuel case studies (see **Figure 1.5**), palm biodiesel and oil palm are the determining co-products (the main products) from the palm oil production system producing various co-products throughout the life cycle stages. Therefore, CLCA of the palm biodiesel will include the oil palm cultivation and the palm oil refinery (the multiple-output activities). The additional demand for palm diesel will increase the production of co-products during palm oil milling and biodiesel conversion stages and result in displaced production of substitutable products. The substitution of the co-products will also be included in the CLCA of the palm biodiesel. The substitution pathways (*the avoided activities*) are that energy production from palm kernel and palm shell substitutes the marginal electricity production; palm kernel oil substitutes the marginal oil; glycerol substitutes feed energy; and palm kernel meal substitutes the marginal feed energy and feed protein.

For the dependant co-product, the demand for dependant co-product (or by-product) will drive the additional production of marginal suppliers which could provide the same functions of the by-product studied. Therefore, CLCA will exclude the multiple-output activities and consider only the environmental interventions of the marginal supplies. Based on the biofuel case studies (see **Figures 1.4 and 1.5**), molasses and rice straw are dependent co-products (so called by-products) from sugar and rice production systems. The additional demand for molasses and rice straw will not increase sugarcane and rice production and result in additional production of substitutable products. According to Prapaspongsa and Gheewala (2016), the use of molasses (as the main raw material) and bagasse (as heat energy source in the ethanol conversion stage) will contribute to additional feed energy and marginal heat energy source production. For rice straw utilisation, various utilisation pathways including field burning, soil fertilising and animal feeding are common in Thailand (Sansiribhan et al., 2014). In case the used product is the determining co-product and there are more than one determining co-products, the economical based system expansion and correction in use stage which most often leads to same results as economic allocation.

Chapter 2 Consequential and Attributional Environmental Assessment of Biofuels

The research presented in this chapter was published in Prapaspongsa and Gheewala (2017). Full details including the supplementary material could be seen in **Appendix A**. The case studies presented here are also used for demonstrating CLCA method documented in **Chapter 1**.

2.1 Introduction

Biofuels for power production and transport have been promoted nationally and globally as an essential element of climate change mitigation strategies (OECD/IEA, 2014; 2015; ONEP, 2015; DEDE, 2015). The world energy outlook 2014 (OECD/IEA, 2014, p.255) stated that “renewable energy technologies emit no greenhouse gases as they produce electricity, making them an essential element of a strategy to mitigate climate change”. According to the outlook, the climate change impacts from biofuels – an important renewable energy source - were considered as avoided CO_2 emissions. These values were directly calculated from the emissions which other generation technologies could produce if the renewable energy technologies have not been employed. In 2040, bioenergy and biofuels might avoid 760 and 450 Mt CO_2 emissions, respectively (OECD/IEA, 2014). Not only have the avoided CO_2 emissions been reported at a global level (OECD/IEA, 2014; 2015), but it was also documented in Thailand’s Alternative Energy Development Plan: AEDP2015 (DEDE, 2015, p.20) that alternative energy sources including biomass and biofuels could potentially reduce greenhouse gas (GHG) emissions due to the decrease in fossil fuel combustion.

In fact, there are emissions and environmental impacts associated with the acquisition of renewable energy sources, production, transportation, use, and end-of-life treatment. Previous studies have shown that some renewable energy technologies in certain conditions may possibly lead to additional GHG emissions as well as other environmental impacts when compared with fossil energy (Pehnt, 2006; Plevin et al., 2014a; Prapaspongsa et al., 2017; Searchinger et al., 2008). For bioenergy, some GHG emissions such as biogenic CO2 emissions during combustion process might be omitted but there are other GHG emissions from the whole life cycle to be considered. In order to develop strategies which actually lead to climate change mitigation, it is therefore crucial to carry out an environmental assessment of biofuels in term of GHG emissions from raw material acquisition to end-of-life treatment in comparison with conventional fossil fuels.

Various life cycle thinking approaches are available for assessing GHG emission reduction. Consequential and attributional LCA (CLCA and ALCA) modelling approaches are chosen in this research because they have been widely applied in LCA studies and in the estimation of climate change mitigation benefits of biofuels (Martin et al., 2015; Prapaspongsa and Gheewala, 2016; Schmidt and Brandão, 2013; Silalertruksa and Gheewala, 2013). UNEP/SETAC Life Cycle Initiative (2011) defined CLCA and ALCA modelling approaches as “system modelling approach in which activities in a product system, are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit” and “system modelling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule”, respectively. Both modelling choices have recently and intensively been debated on vis-à-vis suitability, reliability and creditability for decision support (Anex and Lifset, 2014; Brandão et al., 2014; Dale and Kim, 2014; Hertwich, 2014; Plevin et al., 2014a, b, c; Suh and Yang, 2014). In order to provide a robust decision support, the inclusion of the two main modelling schools and the determination of specific questions which each modelling choice could potentially answer are crucial. Previous CLCA and ALCA studies on biofuels have focused on methodological discussions at a conceptual level or have used case studies with a preference of a certain modelling approach (Martin et al., 2015; Plevin et al., 2014a; Schmidt and Brandão, 2013; Silalertruksa and Gheewala, 2013). Furthermore, the case studies considering both modelling choices in emerging economies, where there are very high potentials in actual climate change mitigation, are rare. This study targets at presenting various illustrative cases in assessing climate change mitigation potential of biofuel production and consumption in Thailand for fossil energy replacement under consequential and attributional modelling approaches. The biofuels include molasses-based ethanol, palm-based biodiesel and rice straw (for electricity production) in Thailand which could be useful for supporting the decisions on other environmental strategies in Thailand, and other emerging and developed economies.

2.2 Methodology

2.2.1 Goal and scope

The main goals of this environmental assessment study are (1) to assess climate change mitigation potential by considering GHG emissions of biofuels in comparison with fossil energy under CLCA and ALCA modelling approaches, (2) to determine the implications of each modelling choice for policy support, and (3) to recommend the specific contexts in which each approach may be more appropriate. It should be mentioned that this study has focused only on a single issue which is climate change mitigation potential. In fact, an LCA study shall consider potential environmental impacts. Nonetheless, with the aim to assess the implications of modelling choices in decision making, a single impact assessment could be applied for illustrating various cases and conditions more effectively since a multi-impact assessment with various environmental impact categories might potentially shift the research focuses. The functional unit (FU) is defined as 1 MJ of biomass-based fuels and fossil energy. The fossil comparators are gasoline for molasses based ethanol, conventional diesel for palm based biodiesel, and electricity from natural gas, coal and Thailand's grid mix for rice straw based electricity. The overall system boundaries of Thailand's molasses ethanol, palm biodiesel and rice straw electricity production systems are illustrated in **Figures 1.3 to 1.5** (in Chapter 2). Each figure demonstrates the investigated life cycle stages under CLCA and ALCA data modelling approaches.

The CLCA methodology according to Weidema (2003) and Weidema et al. (2009), widely applied/considered in LCA research and methodology (Earles and Halog, 2011; Prox and Curran, 2017; Schmidt, 2015), is applied in this study. The CLCA approach attempts to assess consequences from a change in demand and includes only finally affected product systems under market-based causal relationships. By using this approach, marginal suppliers/technologies/unit processes (or the suppliers/technologies/unit processes which are subjected to change according to the additional demand in the study) are taken into account whereas the co-product allocation is avoided by using system expansion or substitution (Weidema et al., 2009; Pelletier et al., 2015). The identification of marginal suppliers (or the affected unit processes) in CLCA can be done by considering the scale and time horizon of the changes being assessed, the market delimitation, the market trends and the changes in supply and demand (Weidema et al., 2009). It the market trends are increasing, the most competitive suppliers/technologies will be affected, and vice versa. For system expansion, only the unconstrained systems driven the change in demand are included. Further detailed CLCA methodology can be seen in Weidema et al. (2009).

As presented in **Figures 1.3a, 1.4a and 1.5a**, the molasses ethanol, palm biodiesel and rice straw electricity production systems in Thailand influence the substituted/substituting product systems via national and international market linkages. Molasses and rice straw are dependent co-products (so called by-products) from sugar and rice production systems whereas palm oil is the main product from the palm oil production system producing various co-products throughout the life cycle stages. The additional demand for molasses and rice straw will not increase sugarcane and rice production and result in additional production of substitutable products. According to

Prapaspongsa and Gheewala (2016), the use of molasses (as the main raw material) and bagasse (as heat energy source in the ethanol conversion stage) will contribute to additional feed energy and marginal heat energy source production. For rice straw utilisation, various utilisation pathways including field burning, soil fertilising and animal feeding are common in Thailand (Sansiribhan et al., 2014). The additional demand for palm diesel will increase the production of co-products during palm oil milling and biodiesel conversion stages and result in displaced production of substitutable products. The substitution pathways from Prapaspongsa et al. (2017) are followed in this work (energy production from palm kernel and palm shell substitutes the marginal electricity production; palm kernel oil substitutes the marginal oil; glycerol substitutes feed energy; and palm kernel meal substitutes the marginal feed energy and feed protein). Only the markets for marginal electricity and heat are delimited at a national level (in Thailand) whereas the markets for marginal oil, feed energy and feed protein are considered at a global level (Prapaspongsa and Gheewala, 2016; Prapaspongsa et al., 2017). It should be mentioned that the figures only illustrate the main and co-products of molasses-based ethanol, palm-based biodiesel and rice straw-based electricity systems. Wastewater treatment processes are not illustrated in the figures but their relating emissions are taken into account. In CLCA modelling, the wastewater treatment systems with biogas recovery with electricity generation and land application will displace marginal electricity and fertiliser production. In order to capture the potential substitution pathways and conditions, CLCA scenarios (M-ethanol_C1 and C2, P-biodiesel_C1 and C2, and R-electricity_C1 to C3) are developed as described in **Table 2.1**.

Table 2.1: Scenario description of three major biofuel systems in Thailand.

Scenarios	**Description**
Molasses based ethanol system (M-ethanol) ***General Description for the system:*** Existing cane molasses ethanol system in Thailand where the molasses ethanol plant uses steam and electricity from bagasse supplied from a sugar mill. Wastewater and vinasse treatment processes in the sugar milling and ethanol production plants include oxidation and stabilisation ponds system, anaerobic digestion system with biogas recovery and land application. The surplus energy at sugar mills and ethanol plants is sold to the national grid system. [a] ***Fossil Comparator [b]:*** Gasoline	
M-ethanol_C1	The consequential modelling approach is applied under the fully utilised condition of cane molasses and bagasse.

Scenarios	Description
M-ethanol_C2	The consequential modelling approach is applied under the non-fully utilised condition of cane molasses and bagasse. It is assumed that available molasses and bagasse are not fully utilised and will be kept in stock without being thrown away due to their economic value.
M-ethanol_A	The attributional modelling approach is applied by using average suppliers and economic allocation factors.
Palm oil based biodiesel system (P-biodiesel) ***General Description for the system:*** Palm biodiesel production under average condition in Thailand with approximately 83% biogas capture in palm oil mill effluent treatment during the oil milling stage (weight average from 6 palm oil mills). The surplus energy at oil mills is sold to the national grid system. [c] ***Fossil Comparator [b]:*** Diesel	
P-biodiesel_C1	The consequential modelling approach is applied under the fully utilised condition of all co-products.
P-biodiesel_C2	The consequential modelling approach is applied under the situation that the co-products during the palm oil milling stage (palm kernel, palm shell, empty fruit bunch, and palm kernel meal) are not fully utilised. 50% of the co-products are left as waste.
P-biodiesel_A	The attributional modelling approach is applied by using average suppliers and economic allocation factors.
Rice straw based electricity system (R-electricity) ***General Description for the system:*** Rice straw based electricity production system based on the 10-MWe straw based power plant from Delivand et al. (2011). The surplus energy is sold to the national grid system. [d] ***Fossil Comparator [b]:*** Electricity from coal, natural gas and Thailand's national grid mix	
R-electricity_C1	The consequential modelling approach is applied under the situation that the rice straw was previously burned in the field to hasten the planting process for the next crop.
R-electricity_C2	The consequential modelling approach is applied under the situation that the rice straw was previously chopped and used as fertiliser in the field.

Scenarios	Description
R-electricity_C3	The consequential modelling approach is applied under the situation that the rice straw was previously used as an animal feed.
R-electricity_A	The attributional modelling approach is applied by using average suppliers and economic allocation factors.

[a] The system and life cycle inventory data are based on Silalertruksa et al. (2017) and Gheewala et al. (2017), Prapaspongsa and Gheewala (2016), and Yuttitham et al., (2011)

[b] The life cycle inventory data are from TGO (2016; 2017), Phumpradab et al. (2009) and supplemented with ecoinvent database.

[c] The system and life cycle inventory data are based on Prapaspongsa et al. (2017).

[d] The system and life cycle inventory data are based on Silalertruksa and Gheewala (2013) except the consequential modelling approach and the rice straw utilisation pathway as an animal feed.

In the ALCA approach, average suppliers/technologies and economic allocation of the molasses ethanol, palm biodiesel and rice straw electricity production systems are taken into consideration (See the system boundaries in **Figures 1.3b, 1.4b and 1.5b**). The general description of all ALCA scenarios (M-ethanol_A, P-biodiesel_A, and R-electricity_A) is explained in **Table 2.1**.

Direct and indirect land use changes (dLUC and iLUC, respectively) are taken into account by considering the GHG emissions from direct land transformation (for dLUC) and the upstream impacts of the land tenure (for iLUC). The dLUC calculation is performed according to IPCC (2007) and Silalertruksa and Gheewala (2012b) taking into account GHG emissions from the direct changes in biomass carbon stocks, dead organic matter and soil carbon stocks. Land use changes for agricultural systems in Thailand are dynamic. Farmers often change their agricultural products (e.g. rice, sugarcane, and other crops) depending on market prices without considering land suitability. Oil palm has been planted across Thailand (66 out of 77 provinces in Thailand; OAE, 2015) although not all areas are suitable for palm cultivation. Saswattecha et al. (2016) presented significantly different land use baselines for oil palm plantation during two periods of time (between 2000 and 2009; and between 2009 to 2012). New oil palm plantation during the latter period of time mostly occurred in the established cropland. Due to the data limitation, the previous land use type (or the land use baseline) of the sugarcane, oil palm and rice cultivation in Thailand in the dLUC analysis is assumed to be the current share of paddy fields, para rubber plantation, oil palm plantation, and crop cultivation areas in Thailand in 2013 (as 59.5%, 10%, 4% and 26.5%, respectively) (OAE, 2015). The same land use type for the agricultural systems outside Thailand is assumed. Hence, there is no dLUC for such case.

For iLUC, a biophysical indirect land use change model developed by Schmidt and colleagues (2015) considering the upstream consequences from land being in use, land expansion and intensification is applied for both CLCA and ALCA modelling approaches. This model has been applied in recent LCA studies (Dalgaard et al., 2014; Flysjö et al., 2012; Prapaspongsa and Gheewala, 2016; Schmidt, 2015). Specific data of GHG emissions from iLUC due to land use in Thailand are based on Prapaspongsa and Gheewala (2016). The iLUC emissions are also considered in the agricultural systems outside Thailand.

The life cycle impact assessment method in this study is IPCC 2007 GWP 100a (IPCC, 2007) and the calculations are carried out by using SimaPro 8.0.3 (PRé Consultants bv, Amersfoort, the Netherlands).

2.2.2 Data collection

The foreground data were mainly obtained from existing studies with field data collection in Thailand (Gheewala et al., 2017; Kaewmai et al., 2013, 2016; OAE and GIZ, 2012; Silalertruksa and Gheewala, 2009, 2011, 2012a, 2013; Silalertruksa et al., 2017; Suttayakul et al., 2016). The marginal electricity production in CLCA modelling only considers the additional installed capacity/generation and excludes the constrained energy sources. The average electricity production in ALCA modelling considers the total production from the national grid mix. The marginal electricity sources in Thailand identified by Prapaspongsa and Gheewala (2016) consist of 49% gas, 28% coal and lignite, 2% solar energy, 2% wind, 9% hydropower and 9% biomass. Due to the lack of data, only specific share of marginal electricity sources in Thailand is used; and the background data are from the ecoinvent database. For the marginal biomass energy, it was determined as wood pellets from Brazil because of its highest cost competitiveness (Prapaspongsa and Kørnøv, 2012; Prapaspongsa et al., 2016). The marginal heat production in Thailand is from oil and gas (Prapaspongsa and Gheewala, 2016). Marginal feed energy was defined as barley in Spain due to its cost competitiveness and largest incremental production volume (Prapaspongsa and Gheewala, 2016). The foreground data on barley production were obtained from Lechón (2011) and Lechón et al. (2011). According to Schmidt (2007, 2015) and Schmidt and Brandão (2013), marginal feed protein and oil were considered as soybean meal and palm oil, respectively. The data for soybean meal and palm oil were obtained from Schmidt and Brandão (2013) and Prapaspongsa et al. (2017). The data for fossil comparators (gasoline, diesel, and electricity from hard coal, natural gas and national grid) are from TGO (2016; 2017), Phumpradab et al. (2009), Thai national life cycle inventory (LCI) database and supplemented with Ecoinvent database. The background data were obtained from the existing database in SimaPro, such as Ecoinvent data v2.2 (ecoinvent Centre, 2010). The LCI data and important assumptions are documented in Supplementary Material.

2.3 Results and Discussion

Potential GHG emissions of 1 MJ molasses based ethanol, palm based biodiesel and rice straw based electricity scenarios in comparison with fossil energy are shown in **Figure 2.1**. Total climate change potential values of all scenarios are documented in the Supplemental Material. At the beginning, the assessment results by using the consequential and attributional approaches will be discussed separately. Later, the implications on policy recommendations for climate change mitigation strategies under various contexts in Thailand are pointed out and discussed.

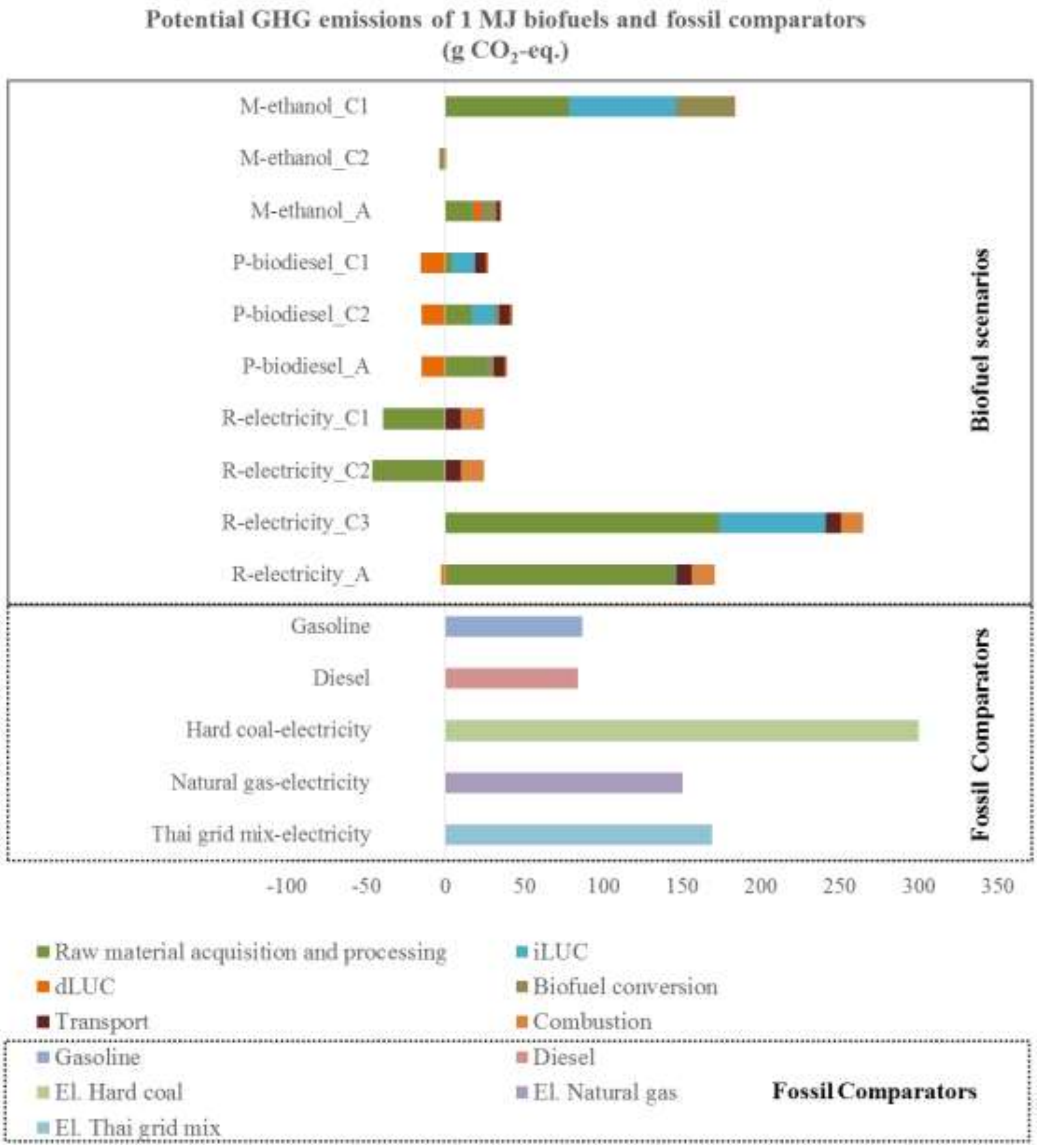


Figure 2.1: Potential GHG emissions of 1 MJ biofuels and fossil comparators with process contribution. The biofuel systems include molasses-based ethanol scenarios (M-ethanol), palm-based biodiesel scenarios (P-biodiesel) and rice straw-based electricity scenarios (R-electricity) under consequential (C) and attributional (A) modelling approaches. The fossil comparators are gasoline, diesel, and electricity production from hard coal, natural gas and Thai grid mix.

2.3.1 Consequential environmental assessment of biofuels

This section will demonstrate the climate change mitigation potentials from the fossil fuel replacement by biofuels with the use of the CLCA approach without questioning the methodological choices. Specifically, the policy implications of this modelling choice application will later be discussed in **Section 2.3.3**.

The palm based biodiesel scenarios (P-Biodiesel_C1 and C2) potentially mitigate climate change by the fossil energy (conventional diesel) displacement. The molasses based ethanol scenarios (M-ethanol_C1 and C2) and the rice straw based electricity scenarios (R-electricity_C1 to C3) may or may not be able to yield lower GHG emissions in comparison with their fossil comparators (Gasoline, Hard coal-electricity, Natural gas-electricity, Thai grid mix-electricity). The assessment can be categorised into 3 specific cases – palm oil as the main product (or determining co-product) during the oil palm plantation and biodiesel conversion stage, molasses as the high value by-product (or dependant co-products) during the sugarcane cultivation stage and rice straw as the low value by-product during the rice cultivation stage.

As the main product under the effective by-product utilisation system, the palm biodiesel could reduce GHG emissions from the conventional diesel consumption up to 73 g CO_2-eq./MJ. During the raw material acquisition and processing stage with fully utilised co-product condition (P-Biodiesel_C1), the total emissions from oil palm cultivation (excluding dLUC and iLUC) and palm oil mill processing are 29 g CO_2-eq./MJ with avoided emissions from the co-product substitutions of 25 g CO2-eq./MJ resulting in the total emissions during the stage of 4 g CO_2-eq./MJ (See **Figure 2.1**). Since palm cultivation in Thailand comparatively has higher carbon stocks than the other land use types (except natural forest), the dLUC from oil palm cultivation also contribute to GHG saving for the biofuel scenarios (P-Biodiesel_C1 and C2; See **Figure 2.1**). It should be mentioned that the land baseline mix in Thailand for oil palm plantation is different from the ones in other countries (i.e. Indonesia and Malaysia). The direct land use change from wetlands and peatlands to be oil palm represented 0.2% of total land use change from 2009 to 2012 (Saswattecha et al., 2016). If illegal deforestation (i.e. converting forest/wetlands to be oil palm) occurs, the GHG emissions from palm-based biodiesel will be around 2.5 times of those from the conventional diesel (Prapaspongsa et al, 2017). If the co-products are only 50% utilised, the GHG emissions from the palm biodiesel will increase with a factor of 2.4 (See P-Biodiesel_C1 and C2 in **Figure 1.3**). Nevertheless, the by-products from palm oil industry such as palm kernel, palm shell, palm kernel oil, and palm kernel meal could be easily used or sold for energy recovery and/or animal feed resulting in economic benefits of the owners. In a well-established market, the default condition for the by-products with high economic values is rather full utilisation.

The molasses based ethanol illustrates a case of the high value by-product. In case the molasses is fully utilised, the ethanol production from molasses (M-ethanol_C1) will contribute to additional marginal feed production and higher GHG emissions around 2 times that of gasoline. Part of the high emissions is also associated with the use of bagasse for heat energy in the biofuel conversion stage. Bagasse has also been fully utilised for energy production and the affected system from the additional bagasse consumption will be the marginal heat production (from oil and gas) in Thailand (Prapaspongsa and Gheewala, 2016). Under the non-fully utilised condition, molasses and bagasse could be seen as waste and freely available without environmental impacts. The molasses based ethanol under such condition (M-ethanol_C2) will yield overall GHG saving due to the electricity and fertiliser replacement from vinasse treatment system (51% biogas capture and 49% land application as fertiliser); as well as be able to reduce GHG emissions from gasoline displacement of around 90 g CO_2-eq./MJ. The important question is whether molasses and bagasse in Thailand have generally been fully utilised. Prapaspongsa and Gheewala (2016) reviewed the market situations in Thailand and the world (Esther, 2013; FAOSTAT, 2014; UN data, 2014; UM Trading, 2014) and concluded that the molasses from Thai sugar industry has been used in the local market, traded in the global market, and therefore fully utilised locally and globally. Bagasse has also been fully utilised nationally (Prapaspongsa and Gheewala, 2016). Consequently, the molasses based ethanol might not mitigate climate change potential from the fossil energy displacement in this first conclusion from the consequential environmental assessment.

The rice straw based electricity production represents the low value by-product case. Three common utilisation pathways are direct burning in the field burning, using for soil fertilisation and amendment and selling for animal feed (R-electricity_C1 to C3, respectively) which highly affect the mitigation potential. If the rice straw is directly burnt in the field or used for fertiliser displacement and soil amendment (R-electricity_C1 and C2), GHG savings from rice straw when comparing with fossil energy yield the highest among the three biofuels (up to 322 g CO_2-eq./MJ). The main contribution for the GHG savings is from the avoided emissions if the rice straw is burnt or used in the field (See **Figure 2.1**). In case the rice straw is used as animal feed (R-electricity_C3), it will increase marginal feed production and its related GHG emissions (including iLUC emissions) will be higher than those of the electricity production from natural gas and Thai grid mix (See **Figure 2.1**). Which situation is likely to be the most representative for rice straw utilisation? Due to the fact that rice straw has low value (moderate energy content) with high operation costs for the handling and transport processes, field burning is still the most practical option for farmers. As a result, the rice straw based electricity seems to be the most promising biofuel for mitigating climate change under the investigated systems in this research.

2.3.2 Attributional environmental assessment of biofuels

This section will discuss the attributional assessment of the potential GHG emission reduction when replacing fossil fuels with the biofuels without being in doubt on the methodological choice itself. The comparison with the CLCA methodology is also discussed briefly here. Nonetheless, the specific context and questions which this approach might be or might not be used for decision support will later be discussed in **Section 2.3.3**.

In contrast with the previous section, under the ALCA approach the molasses based ethanol (M-ethanol_A) potentially contributes to climate mitigation whereas the rice straw based electricity (R-electricity_A) yields the GHG emissions almost equal to electricity production from the Thai grid mix. Similarly, the palm based biodiesel could reduce the GHG emissions from conventional diesel substitution. **Figures 1.3 to 1.5** have clearly illustrated that the investigated systems of each product under the CLCA and ALCA approaches are not the same. As a result, their climate change mitigation potentials are different. Based on the set of scenarios in the assessment, the CLCA scenarios may vary to a larger extent while the ALCA scenarios seem to be static. However, the previous results only include economic allocation factors to partition total GHG emissions of all life cycle stages to different co-products. Other allocation factors (i.e. mass and energy) may not be adequate for all cases. For instance, it will be less meaningful to use an energy allocation factor for the rice straw based electricity system because the main function of rice is for food. In order to determine the potential variability in ALCA, economic, energy and mass allocation factors during cultivation (10%, 24% and 26%, respectively) are applied for allocating GHG emissions to the molasses in the molasses based ethanol cases as can be seen in **Figure 2.2**.

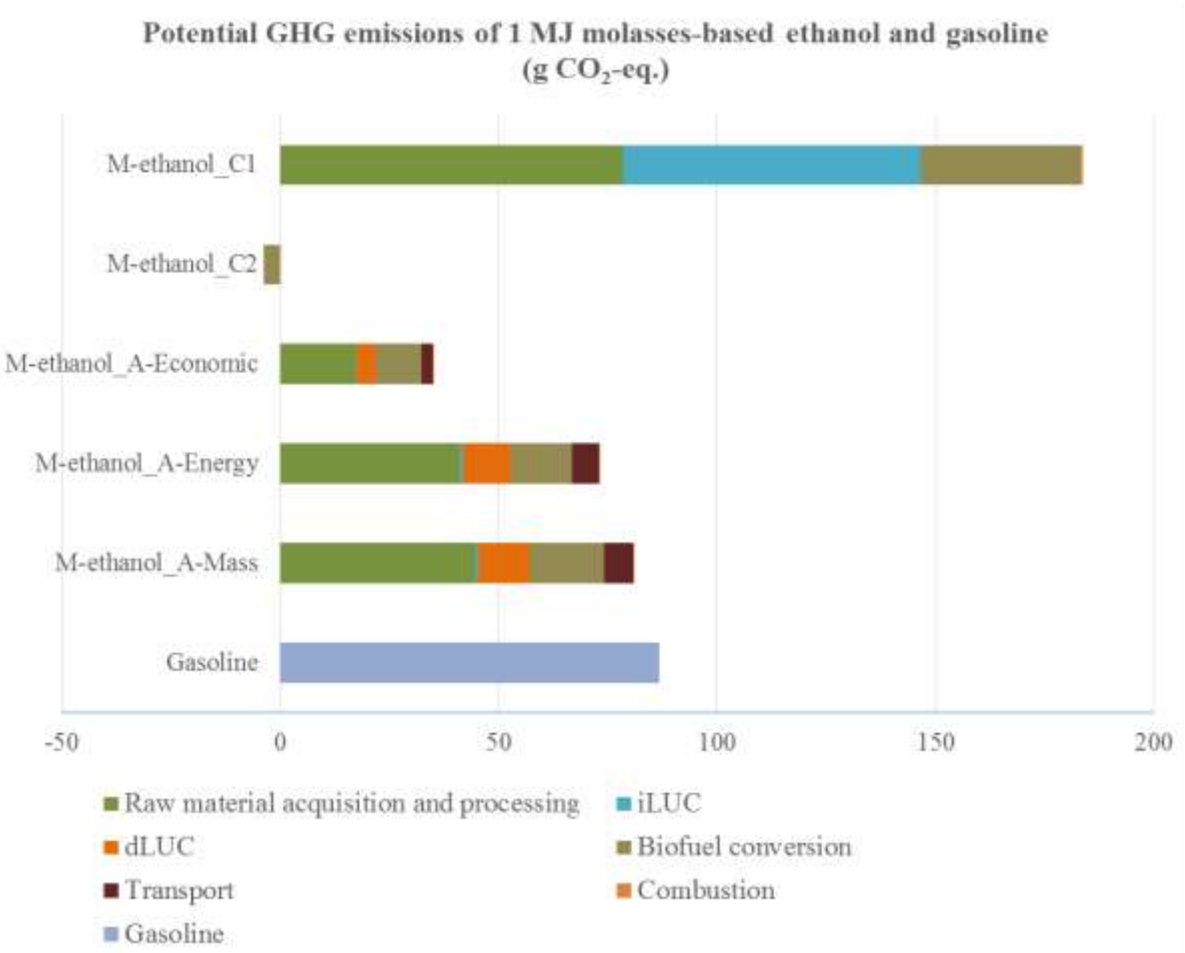


Figure 2.2: Potential GHG emissions of 1 MJ molasses-based ethanol scenarios and gasoline with process contribution. The molasses-based ethanol scenarios under an attributional modelling approach apply economic, energy and mass allocation factors (M-ethanol_A-Economic, M-ethanol_A-Energy and M-ethanol_A-Mass, respectively).

From economic to mass allocation, the GHG emissions from the molasses based ethanol could change by a factor of 2 (See M-ethanol_A-Economic and A-Mass in **Figure 2.2**). For the rice straw electricity production, if other allocation factors are higher than the economic one, this biofuel will contribute to GHG emissions at a higher rate than the fossil energy (except the electricity from hard coal). The selection of the allocation factors therefore plays an important role in ALCA.

2.3.3 Implications on policy recommendations for establishing future climate change mitigation strategies

The climate change mitigation potentials of the fossil energy replacement by the three biofuels taking into account the CLCA and ALCA approaches have been discussed in the previous parts. This section aims at assessing the policy implications of the methodological choices and to suggest the specific contexts and questions which each modelling approach may support or provide the answers for. It should be addressed that the main purpose of this paper is not to determine the superiority or to judge each approach. The authors would suggest that an LCA study is highly recommended to apply both modelling choices to identify possible risks of a decision since a single and standalone approach may impossibly capture all potential consequences. Nevertheless, under the situations when different models provide conflicting conclusions, the main choice to be held upon is crucial for decision support. The specific objectives and recommended applications of the two modelling choices are discussed as follows.

2.3.3.1 Specific questions for CLCA and ALCA approaches

"To assess climate change mitigation potential by considering GHG emissions of biofuels in comparison with fossil energy under consequential and attributional modelling approaches" with the functional unit (FU) as 1 MJ of biomass-based fuels and fossil energy could be set as the objective for both approaches. The problem is that the defined objective is too broad and does not distinguish the specific situations each approach could be suitable for. This objective is intentionally set to imitate a typical condition when an LCA study is carried out to support general policies without specific users which could lead to different interpretations and conclusions due to its generality. What could then be the specific objectives?

The CLCA modelling approach determines the impacts from a change in demand and to include only the systems which are expected to change due to the additional product. Brandão and Weidema (2013) identified the questions which the CLCA approach attempts to answer are "What are the net impacts associated to a change (in a product system) relative to the baseline scenario, where that change does not take place?" and "What are the consequences of a decision relative to the "no action" baseline?". The specific question in this study could be "What would actually happen or what are the potential consequences if we additionally produce 1 MJ of the biofuels in Thailand?" The keywords are "additionality" and "potential affected processes linked via market mechanisms". The functional unit is rather the additional production of 1 MJ biofuels and fossil fuels in this case. Based on Brandão and Weidema (2013), this model looks at the baseline system as the situation without any action and considers only the additional product of concern without the assessment of the baseline system itself. With the specific question a CLCA study could properly provide the answer, it is therefore crucial for the policy makers/researchers to explicitly state the question during the goal and scope definition phase. To answer such a question, the expectedly actually affected processes linked via market mechanism or the marginal technologies/suppliers are identified by considering market information such as market delimitation, trends and constraints. As previously mentioned in Section 2.1, the market in study is delimited at the national level for electricity and heat and at global level for other expanded systems. With the current stable and/or increasing market trends, only the most competitive suppliers responding to the change in demand are included (i.e. for palm based biodiesel and rice straw based electricity systems, marginal feed protein and energy for co-product displacement being considered as soybean meal and barley, respectively; See Figures 2a and 3a). The constrained suppliers/processes will not be considered (i.e. for the molasses based ethanol and rice straw based electricity systems, sugar cane cultivation, sugar milling and rice cultivation processes are excluded; See Figures 1a and 3a).

The ALCA modelling approach partitions the total impacts to the functional unit of a product system by using a normative rule (i.e. economic allocation technique in this study). De Camillis et al. (2013) determined the relevant questions for this approach as “What is the environmental impact of a certain product system at a given time (when baseline scenario is assessed)?” and “What is the environmental impact of a certain product system in a given future scenario if the product were designed or/and produced or/and consumed or/and managed differently at the end of its life?”. For this study, the specific question could be “What are the specific GHG emissions of 1 MJ biofuel production in Thailand allocated by economic values of all related co-products?”. The keywords are “specification” and “allocation/attribution”. The functional unit as of 1 MJ biofuels and fossil fuels might still be valid in this case. This model firstly looks at the total emissions from specific production systems by including all product outputs in the systems which are directly linked throughout the supply-chains (i.e. the direct supply-chains for molasses-based ethanol, palm-based biodiesel and rice straw-based electricity in Thailand as illustrated in Figures 1b, 2b and 3b) and, secondly, seeks for a specific allocated share of the relevant impacts for the specific product being investigated.

2.3.3.2 Potential limitations and risks of CLCA and ALCA approaches

As explained before, both CLCA and ALCA approaches are able to support decisions for climate change mitigation strategies in different contexts; however, an important concern would be “What are potential limitations and risks of the choices made?”. Ekvall et al. (2005) indicated that the risks from the ALCA approach (or the retrospective LCA) are unaccounted and undesirable consequences and those from the CLCA approach (or the prospective LCA) are unfairness and sub-optimised systems. The potential limitations in this analysis refer to what cannot be assessed or captured by each modelling choice; as well as are certain conditions which may make the approaches incorrectly assess the environmental impacts associated with the products in question. The potential risks are consequences from certain limitations of the two approaches.

Specification and allocation/attribution could be seen as main limitations of the ALCA approach. The ALCA approach covers the allocated impacts from processes directly linked with the specific supply-chain. The specific processes are not necessarily affected by the additional production of the investigated products and the assessment may not reflect what would happen in reality. For example, the sugarcane and rice cultivation will not potentially be driven or affected by the demand for molasses and rice straw in reality because molasses and rice straw are dependent by-products. Since the ALCA approach do not include the consequences of individual actions (a marginal change in demand), this could eventually lead to the unaccounted harmful environmental impacts. For instance, Thai policy makers may support the molasses based ethanol for gasoline replacement since it could reduce GHG emissions around 61 g CO_2-eq./MJ under an economic allocation rule and the inclusion of sugarcane cultivation and sugar milling processes in Thailand. Nonetheless, supporting the increased use of molasses for ethanol production may lead to an increase in the animal feed production somewhere else in the world (i.e., barley in Spain) and its emissions may far outweigh the fossil comparator (i.e., 142 g CO_2-eq./MJ additional emissions). With this illustrative case, the use of the ALCA approach might be questionable for supporting a decision. However, UNEP/SETAC Life Cycle Initiative (2011) described that the ALCA model aims at identifying the portion of global burdens which a product could potentially contribute to throughout its life cycle. It was also mention that “In theory, if one were to conduct LCAs of all final products with attributional modelling, one would end up with the total observed environmental burdens worldwide” (UNEP/SETAC Life Cycle Initiative, 2011). At a given time of the study, the background situation will be static. If all products are being assessed under the same allocation rule, it will therefore be able to capture the overall environmental impacts and the allocated emissions to the molasses based ethanol would be reasonable and not lead to unintended consequences.

With respect to the CLCA modelling, the main features of affected processes linked via market mechanisms and additionality could also lead to certain limitations. Firstly, the limitation is on the market based considerations: producers/suppliers are price-takers and those systems with lowest production costs will likely be the most competitive (if the market trend is stable or increasing); and elasticity of supply and demand will lead to linear and direct relationships, etc. Since market failures are very common in reality, the modelling choice based on market information may not capture the real consequences as expected. Secondly, with the additionality viewpoint CLCA only considers additional production for an additional demand and excludes existing production for an existing demand. It may be unfair for new suppliers who demand for the fully utilised dependent co-products such as molasses (See **Section 2.3.1**). If a sugar factory produces 100 tonnes of molasses a year and has already used internally and sold in the market for all, the additional demand of 1 tonne of molasses will affect the marginal supplier for the molasses consumption which is Barley in this study. It is questionable who will get the credits of the existing 100 tonnes of molasses co-produced with sugar production. Furthermore, in case it is a CLCA study of refined sugar for sugar industry, the additional sugar production will lead to the additional molasses production and environmental impact reduction from the co-product substitution. If it is a CLCA study of molasses-based ethanol for energy industry, the use of molasses will lead to increased production of the substituted product and additional environmental impacts. The limitation potentially leads to unfairness and risks of sub-optimised situation. This could once again be demonstrated by the molasses based ethanol case. Throughout the continuous efforts, initiatives and investments, the sugar industry and its complex (i.e. energy industry) have successfully utilised all their products, co-products and wastes. It would be unfair to account the associated emissions from barley production (marginal feed energy) and oil and gas (marginal heat) for the use of molasses and bagasse in the ethanol production system in Thailand. Moreover, some stakeholders (i.e. energy industry) in the whole supply chain will get credits from impact reduction whereas some (i.e. sugar industry) will be responsible for negative environmental impacts. In case the ethanol business owners considered the by-product usage as drawbacks since the associated emissions are high they may stop using the by-products and leave them wasted. Nevertheless, such situations might not easily happen for the high value by-products which are fully utilised by nature due to their economic values. If the by-products have low value and high operational costs (i.e. the rice straw), the non-fully utilised and freely available condition could be assumed.

2.3.3.3 Policy recommendations on potential specific applications for CLCA and ALCA

With regard to policy recommendations, ones of the main environmental assessment applications are for environmental taxation, accounting and labelling for trade, and new production development and/or eco-design. In contrast with previous studies considering retrospective and prospective applications for ALCA and CLCA, respectively, this study argues that both modelling choices could be used for investigating future impacts since future situations could be modelled by various conditions. The future could be formed by what will additionally occur in the future under market-based causal relationships; and what happened in the past on average (e.g. there may also be potentials that the likely to be constrained suppliers will be exercise flexibility so will not actually be constrained). Brandão and Weidema (2013) and De Camillis et al. (2013) also addressed that CLCA and ALCA approaches can model future scenarios. Hence, the focuses in this research for each modelling choice are rather on the main characteristics of ALCA as "specification" and "allocation/attribution" and those of CLCA as "additionality" and "expectedly actually affected processes linked via market mechanisms".

Environmental taxation is often a legal requirement while environmental accounting and labelling for biofuel trade may be perceived as a non-tariff barrier. Consequently, a platform of specific modelling choices where all stakeholders are fairly judged is crucial for such a case. Specific and direct linkages in ALCA are more reasonable. Businesses would be reluctant to accept tax charges on the actions they have not directly done. The taxes under the market based cause-effect relationship might be accepted by the businesses who gain the benefits for this modelling choice (i.e. the palm biodiesel using palm oil and producing as well as utilising various high-value by-products). Same reasons could be applicable for environmental accounting and/or labelling for import-export since it could lead to the barriers affecting the businesses financially. The methodology of Product Environmental Footprint (PEF) - a recent harmonised environmental assessment approach recommended by European Commission, also applies the attributional approach with some elements inspired by consequential thinking (but still handled in an attributional context) (EC, 2013; Schau et al., 2013). The mixed approaches imply that decision makers, LCA practitioners and researchers have realised the existence and importance of both methodological choices.

New production development and/or eco-design have intrinsically addressed additionality in additional production. In order to design new products aiming at environmental improvements while providing economic profits, the CLCA model which concerns the consequences of an increase in demand and covers the likely to be affected systems under market based relationships is therefore more appropriate. Moreover, according to the case of accounting and labelling for trade, if the main target is to declare environmental information to the consumers as a support for their decisions to buy more products leading to additional production, the CLCA approach also plays an important role. In reality, a specific environmental assessment application always has multi-purposes. It will not be realistic to set a single target and to apply a single approach for actual applications. Under multi-purpose applications, it is crucial to clarify and prioritise objectives as well as to clearly justify the selected or preferred choices being made during the goal and scope definition phase. Under the analysis in this study, both modelling approaches are highly recommended to be applied for LCA studies but preferences on a specific approach for specific applications are suggested. ALCA considering total global burdens of all products directly and attributing a specific portion of impacts to the product in question at a given point of time is recommended for taxation and labelling studies whereas CLCA taking additionality and market-based cause-effect relationships into account is more preferred for new product development and eco-design.

2.4 Conclusions

Climate change mitigation potentials of biofuels in term of GHG emission reduction from fossil energy replacement have been assessed by using the consequential and attributional modelling approaches. Under the CLCA approach considering consequences of an additional biofuel production, GHG emissions could be reduced by using palm biodiesel to replace conventional diesel while it is unclear whether the molasses based ethanol and the rice straw based electricity can be used for climate change mitigation. The molasses based ethanol production with the use of bagasse in ethanol conversion process tends to increase GHG emissions since the consumed by-products have high value and are fully utilised. The electricity production from rice straw with low value and not fully utilised (i.e., the background situations are rice straw burning or rice straw use as fertiliser) is able to reduce GHG emissions. With the ALCA approach using average data and economic allocation, palm biodiesel yields a similar conclusion being able to reduce GHG emissions but the other two biofuels have different performance. The GHG emissions of electricity production from rice straw are comparable to the ones from the national grid mix whereas the molasses based ethanol potentially contributes to climate mitigation. The different results are derived from the differences in the systems being assessed.

To ensure the consideration of possible limitations and risks prior to making decisions, the application of both environmental assessment approaches is recommended. However, when dealing with the situations under conflicting conclusions from the choices being made, the main modelling approach to be considered is crucial for an effective decision support. The specific questions, applications and limitations of each approach are therefore addressed in this research. The specific questions for the consequential and attributional approaches suggested in this work are “What would likely happen or what are the consequences if we additionally produce 1 MJ of the biofuels in Thailand?” and “What are the specific GHG emissions of 1 MJ biofuel production in Thailand allocated by economic values of all related co-products”, respectively. Due to the fact that the CLCA modelling choice includes only affected processes indirectly linked via market mechanisms and excludes the direct actions (if it is not influenced by the additional demand) in the investigated systems, it could lead to the risks for unfairness and sub-optimised systems. The limited CLCA application is also derived from the fact that CLCA considers only changes or additionality and excludes total and existing production. In addition, with the market-based cause-effect modelling in CLCA, the typical market failures in reality result in the conclusions because the assessment cannot capture the actual impacts under such conditions. The risks of the ALCA approach are unaccounted and undesirable consequences since this model does not capture the impacts from individual actions. In environmental taxation, accounting for import-export and labelling studies, the ALCA approach may be more recommended since it considers the inputs, outputs and associated emissions of the whole value chains directly. The indirect linkages via global market might not be acceptable for setting up legal requirements via environmental taxation or trade barriers via environmental accounting and labelling. For new product development and eco-design, the CLCA approach concerning consequences of additional demand and including the potential affected processes is more appropriate. Future studies need to consider other environmental impacts and to determine the implications and recommendations for applying different approaches on other applications and cases to support individual, national regional and global policies.

Chapter 3 Environmental Sustainability of Food Consumption in Asia

The research presented in this chapter will be improved and later submitted to Journal of Cleaner Production (*Adhikari, B., Prapaspongsa, T., 2018. Environmental Sustainability of Food Consumption in Asia. Manuscript. To be submitted to Journal of Cleaner Production*). The preliminary work will be presented in the international conference in October 2018 (*Adhikari, B., Prapaspongsa, T., 2018. Life Cycle Assessment of Food Consumption in Asia towards Sustainable Consumption and Production. Submitted to the 13th Biennial International Conference on EcoBalance (EcoBalance 2018), 9-12 October 2018, Tokyo, Japan*). The supplementary material of the draft manuscript and the submitted conference abstract could be seen in **Appendix A**.

3.1 Introduction

The growing human population, expanding urbanization and increasing environmental concerns stress the need for sustainable production and consumption more than ever before (Akenji and Bengtsson, 2014). Globally, food systems have seen to contribute to 15-20% of greenhouse emissions on average (Benders et al., 2012; Berners-Lee et al., 2012; Hertwich et al., 2009), implying that food systems are one of the major contributors to environmental impacts. Therefore, it is imperative to comprehensively analyse them in greater depth and learn how their impacts can be mitigated. As such, numerous authors have attempted to quantify the impacts associated with country level average diets rather than individual food items through methodologies like Life Cycle Assessment (LCA) and Input Output Models. For example, Carlsson-Kanyama et al. (2003) studied how the Swedish diet could be modified to make it more energy efficient. Similarly, Baroni et al. (2007) analysed the life cycle environmental impacts of three diet patterns (omnivorous, vegetarian and vegan) combined with two agricultural practices (conventional and organic) to know the environmental implications of different meals with the same nutritional levels. Muñoz et al. (2010) analysed the relevance of human excretion in the system boundary of the Spanish food system LCA study. Also, Galli et al. (2017), through an Input Output methodology analysed the ecological footprint of Mediterranean diets and the implications of shift in diets to the ecological footprint. There are various other studies related to food consumption which determine hotspots in the supply chain, the food groups that are the most environmentally burdensome and diets that are environmentally friendly. But most studies account for European Diets, and there hasn't been much attempt to apply similar methodologies to Asian diets yet. Since considerable differences in diet patterns can have significant implications in environmental impacts, specific studies for Asian countries are required in order to come up with sustainable consumption solutions that apply in the Asian context. Therefore, this study analyses the environmental impacts of food consumption of five Asian countries to 1) identify the major food groups responsible for environmental impacts and 2) compare the impacts of consumption between the countries analysed. The results of the study will provide insights on the major stressors to the environment related to food consumption on each country. Together with further studies related to diet and nutrition, the results can be used to provide recommendations for mitigation of impacts through shifts to healthier and environment friendly diets, specifically with regards to the countries that have been analysed.

3.2 Methodology

The five countries selected for the study are Thailand, India, China, Japan and Saudi Arabia. Countries were selected based on wide ranges of demographic, economic and food consumption characteristics. The characteristics of each country in terms of their population, surface area, urbanized population, population density, Gross Domestic Production (GDP) and GDP per capita are given in **Table 3.1.**

Table 3.1: Various parameters for country chosen for the study (The World Bank, 2013)

SN	Parameters	Countries				
		Thailand	India	China	Japan	Saudi Arabia
1	Population (thousands)	68,143	1,279,000	1,357,000	127,445	29,944
2	Surface Area (sq.km)	513,120	3,287,260	9,562,950	377,962	2,149,690
3	Urbanized Population (% of total)	47.94%	32%	53.17%	92.49%	82.72%
4	Population Density (people per sq. km)	133.38	430.03	144.58	349.59	13.93
5	GDP (Billion US$)	420.53	1,857	9,607	5,156	746.65
6	GDP per capita (US$)	15,293.26	5,250.5	12,368	38,974.08	51,264.87

Besides demographic and economic characteristics, the chosen countries also show differences in food consumption patterns, which is evident in **Table 3.2**. For example, according to the data collected, the total demand of Sugar and Confectionary ranges from 1% to 17% and the demand for Vegetables ranges from 39% to 9% in China and Thailand respectively (Food and Agriculture Organization; FAO, 2018). This allows the study to get insights on the differences in environmental impacts produced by varied consumption patterns, which can be used for policies and planning related to sustainable food consumption.

Table 3.2: Per capita consumption of twelve food groups for each country (FAO, 2018)

		Per Capita Consumption (kg/capita-year)				
S.N.	Food Group	Thailand	India	China	Japan	Saudi Arabia
1	Cereals	136	148	150	115	154
2	Root Vegetables	23	31	68	31	24
3	Legumes, nuts and oil-seeds	17	24	12	12	11
4	Oils	8	9	7	15	20
5	Vegetables	52	89	348	102	105
6	Fruits	103	56	94	53	92
7	Coffee and tea	2	1	1	6	6
8	Meat	30	4	65	52	68
9	Fish and Seafood	26	5	45	50	13
10	Animal Products	42	90	54	93	94
11	Sugar and Confectionery	101	33	7	27	32
12	Alcoholic Beverages	41	2	45	47	0
TOTAL		**581**	**492**	**896**	**603**	**618**
Total Calorie Intake (kcal/capita-day)		**2785**	**2454**	**3112**	**2747**	**3255**

The study uses LCA in compliance to ISO 14040 and ISO 14044 to assess the environmental impacts of the five countries under study. In order to capture overall environmental impacts from food consumption under different modelling chices, both attributional and consequential LCA (ALCA and CLCA) are applied in this work. The goal of the study was set according to the objectives stated in the previous section. The system boundary was set from cradle (raw material acquisition) to consumption. For this, the functional unit for the study has been defined as food consumption in kilograms per capita per year for each country. In order to model the food consumption patterns, foreground data was obtained from food balance sheets provided by the FAOSTAT database for the most recent fiscal year (2013; FAO, 2018). The food balance sheets classify food into a total of ninety-four "items" which are categorized into several "item groups". For each of these food items, data related to domestic supply quantity, export quantity, fat supply quantity, feed quantity and food quantity are available in the balance sheets. For this study, data related to food quantity is taken, and is defined as the total amount of food available for human consumption which includes the food item in question as well as any other commodity derived from it (FAO, 2018). Background data in the study is the life cycle inventory data for each food item. Most of the background data were taken from the Ecoinvent v3.4 and Agri-footprint v4 databases. Some of the background data were taken from literature on existing LCA. For ALCA, the background data under the system model of the Allocation at the point of Substitution (APOS) were used. APOS is "*the allocation approach that uses expansion of product systems to avoid allocating within treatment systems"* (ecoinvent, 2018). For CLCA modelling, the background dataset under the system model of "*substitution, consequential, long-term*" were taken into consideration.

Each food item from the database was classified primarily according to the same classification used by FAO and was modified as described in supplementary information. Based on this, the study has grouped food items into twelve food groups: 1. Cereals, 2. Root Vegetables, 3. Legumes, nuts and oil-seeds, 4. Oils, 5. Vegetables, 6. Fruits, 7. Coffee and Tea, 8. Meat, 9. Fish and Seafood, 10. Animal products, 11. Sugar and Confectionary, and 12. Alcoholic Beverages. **Table 3.2** provides a summary of the twelve food groups and the per capita consumption of each group in terms of kilograms per year. A more comprehensive table that details the amounts of consumption of each food item (foreground data) and the datasets used to model their life cycle (background data) is provided in supplementary information. For the selection of impact categories, studies related to life cycle assessment of food were taken and their impact categories with relation to the scope of study was noted down. The table considered for the selection of impact categories is available in supplementary information. Based on the analysis, six impact categories were considered relevant to the study: global warming, terrestrial acidification, marine eutrophication, ecotoxicity, human toxicity and fossil resource scarcity. The computer software used in this LCA is Simapro v8.5. The life cycle impact assessment (LCIA) method chosen is the ReCiPe 2016 v1.01 method (Huijbregts et al., 2017).

3.3 Results and Discussion

3.3.1 Results of Consumption Patterns of each country

Table 3.3 shows the results of Attributional LCA (ALCA) and Consequential LCA (CLCA) for each country. According to the analysis, the environmental impacts of China were the highest in all the selected impact categories except marine eutrophication, in which Saudi Arabia had the highest value. This could be due to higher consumption of cereals, oils, coffee and tea, meat and animal products - five significant contributors to marine eutrophication, in Saudi Arabia (a total of 341 kg per capita) in comparison to China (a total of 278 kg per capita). Similarly, India showed the lowest impacts in all categories except human toxicity, in which Thailand had the lowest value. The reason for this is due to higher consumption of legumes, nuts and oil seeds, the major contributor to human toxicity, in India (24 kg per capita) in comparison to Thailand (17 kg per capita).

Table 3.4 provides a basis for comparison of values for one of the impact categories, i.e. Global Warming, with regards to other studies conducted in different countries and regions. In comparison to these values, the results of this study show that the global warming potential is comparatively lower than existing literature, especially in Thailand and India. The lower values for impact categories in these two countries could be primarily because of low consumption of food per capita. For example, the total consumption per capita in Europe is taken as 933.2 kg (Notarnicola et al., 2017), which is comparable to the consumption per capita in China (896 kg), and significantly higher than consumption in Thailand and India. This could explain relatively high impacts of China and Europe in comparison to Thailand (581 kg) and India (492 kg). Also, the differences in system boundaries used in the study (as it can be seen in **Table 3.4**) and methodologies adopted can explain differences in results in this study as compared to others. Particularly, authors highlight the importance of end-of-life stage in modelling the impacts of food consumption, as they are found to be important contributors to total impacts (Muñoz et al., 2010; Notarnicola et al., 2017). This implies that the values obtained in this study might be underestimated due to the exclusion of post-consumption phase and waste scenarios. None-the-less, impacts would depend on the waste handling scenarios of each country, and thereby it provides a prospect for future research to increase the accuracy of the results.

Table 3.3: Results of ALCA and CLCA on food consumption patterns of each country

Attributional Life Cycle Assessment						
Impact category	Unit	Thailand	India	China	Japan	Saudi Arabia
Global warming	kg CO2 eq	811.44	675.96	1420.55	1109.13	1122.47
Terrestrial acidification	kg SO2 eq	4.96	3.79	9.27	7.21	7.78
Marine eutrophication	kg N eq	0.91	0.87	1.41	1.35	1.44
Ecotoxicity[1]	kg 1,4-DCB	1417.67	1147.77	2651.19	1987.24	1850.08
Human toxicity[2]	kg 1,4-DCB	419.20	471.71	935.56	708.82	709.11
Fossil resource scarcity	kg oil eq	109.78	82.07	212.14	145.22	126.52
Consequential Life Cycle Assessment						
Impact Category	Unit	Thailand	India	China	Japan	Saudi Arabia
Global warming	kg CO2 eq	685.14	552.60	1032.11	884.10	814.53
Terrestrial acidification	kg SO2 eq	3.72	3.40	5.89	5.58	5.24
Marine eutrophication	kg N eq	0.89	0.98	1.23	1.23	1.38
Ecotoxicity	kg 1,4-DCB	1690.26	1373.50	2928.53	2221.32	1908.94
Human toxicity	kg 1,4-DCB	145.46	442.45	622.33	500.53	696.92
Fossil resource scarcity	kg oil eq	93.01	56.85	170.15	130.74	92.90

Table 3.4: Values of kg CO_2 eq. per capita from food systems in various studies

Diet	System Boundary	Annual kg CO_2 eq. per capita	References
European	Agricultural production to treatment of waste	1400	Notarnicola et al. (2017)
Spanish	Agricultural production to treatment of waste	2100	Muñoz et al. (2010)
Mediterranean	Agricultural production to consumption	1870	Pairotti et al. (2015)
German	Agricultural production to consumption (with consideration to food waste)	2700	Eberle and Fels (2016)

[1] Ecotoxicity is taken as a sum of terrestrial ecotoxicity, marine ecotoxicity and freshwater ecotoxicity. Values of these impact categories can be found in supplementary information

[2] Human toxicity is taken as the sum of human non-carcinogenic toxicity and human carcinogenic toxicity. Vales of these impact categories can be found in supplementary information

3.3.2 Major Contributors to Environmental Impacts of each country

In order to determine the food groups with the highest environmental burdens, a hypothetical consumption pattern was modelled, wherein each food group was given a value of 1 kilogram. This was done to evenly distribute the composition of food groups in the diet to see which food item had the highest environmental impact per kilogram. **Figure 3.1** shows the contributions to various impact categories by the twelve food groups. The impact assessment of this model showed that the biggest contributors to most of the impact categories were coffee and tea (due to the transformation of forests into arable land for cultivation, and high emissions from the coffee production process) and meat (also due to transformation of forest to arable land for the cultivation of feed to raise cattle, and the high amount of biogenic methane produced during the rearing of cows, sheep and pigs). Other major impacts were from animal products; particularly from milk and milk-derived animal products, and oils. For human toxicity, legumes, nuts and oilseeds had higher impacts from the almond production process. Since the impacts of coffee was found to be much higher than anticipated, it was compared with an existing study on European diets. The study showed that Global Warming Potential caused by 3.5 kg of roasted coffee consumption was 40 kg CO_2 eq., i.e. 11.42 kg CO_2 eq. per kilogram of coffee consumed (Notarnicola et al., 2017). This value is comparable to the value obtained in this study – 12.22 kg CO_2 eq. per kg.

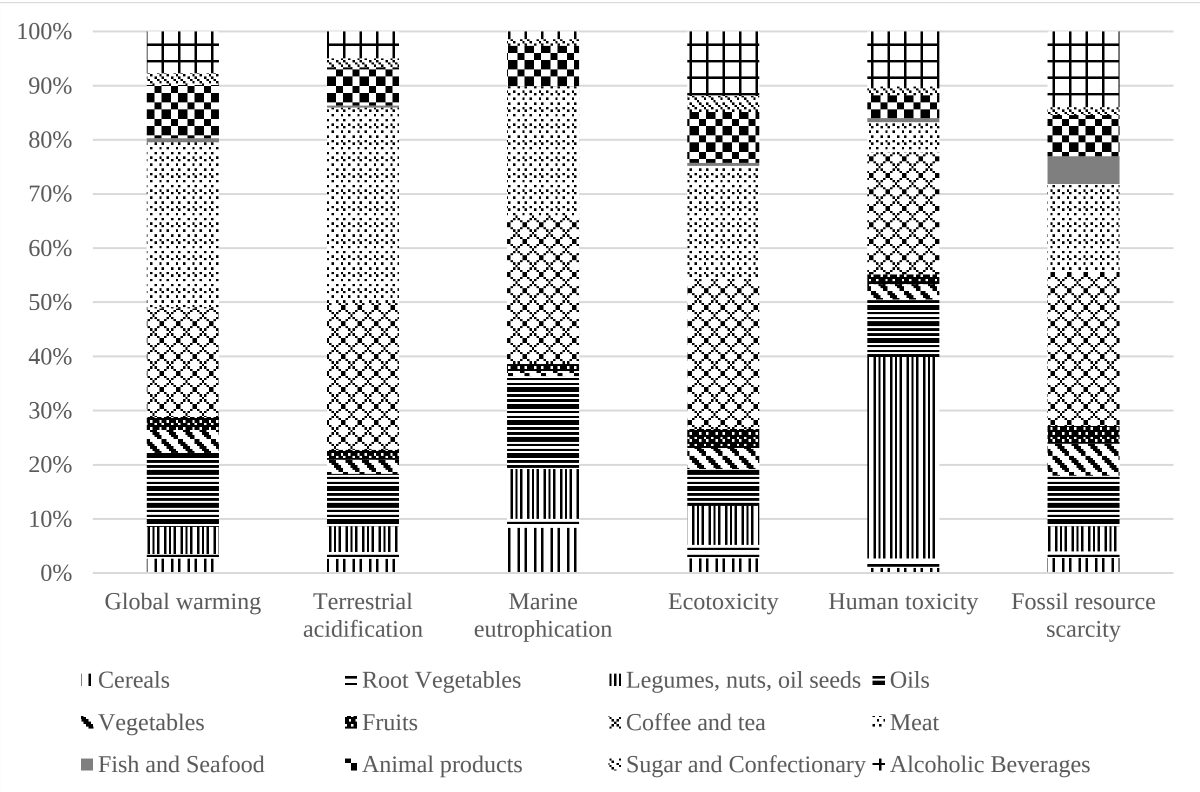


Figure 3.1: Relative impacts of food groups for the hypothetical diet of 1 kilogram per each food group

Upon analysis of food consumption in the selected countries, results showed that the highest contributors to most of the impact categories were cereal, animal products and alcoholic beverages, as can be seen in **Figure 3.2**. Meat was found to be the chief contributor to the impacts in all countries except India, due to India's low consumption of meat. This result is consistent with numerous LCA studies on food that states that meat and animal products are the highest contributors to environmental impacts (Foster et al., 2006; Muñoz et al., 2010; Notarnicola et al., 2017). Vegetables also seemed to contribute significantly in case of China, due to its high consumption of vegetables per capita (348 kg, or 39% of total consumption).

In comparison of these results with the hypothetical diet of 1 kilogram per category explained above, a few differences were found regarding the values of impact categories due to the differences in amount of food groups in the diet. For example, for the hypothetical diet of 1 kilogram per food group, even though oils and coffee and tea showed very high contributions to all the studied impact categories, they were not seen significantly contributing to impacts in Asian diets because this food group didn't have a big proportion in Asian diets (maximum of 1%). Conversely, although the impacts of cereals were not significant in the 1 kilogram per food group model, cereals were found to be one of the major contributors to the impact categories because a large portion of the diets were composed of cereals (17% in China to 30% in India). Therefore, these results showed that for national diets, the environmental impacts primarily depended on the total amount of food consumed, and the portions of food groups that has high contributions to environmental impacts.

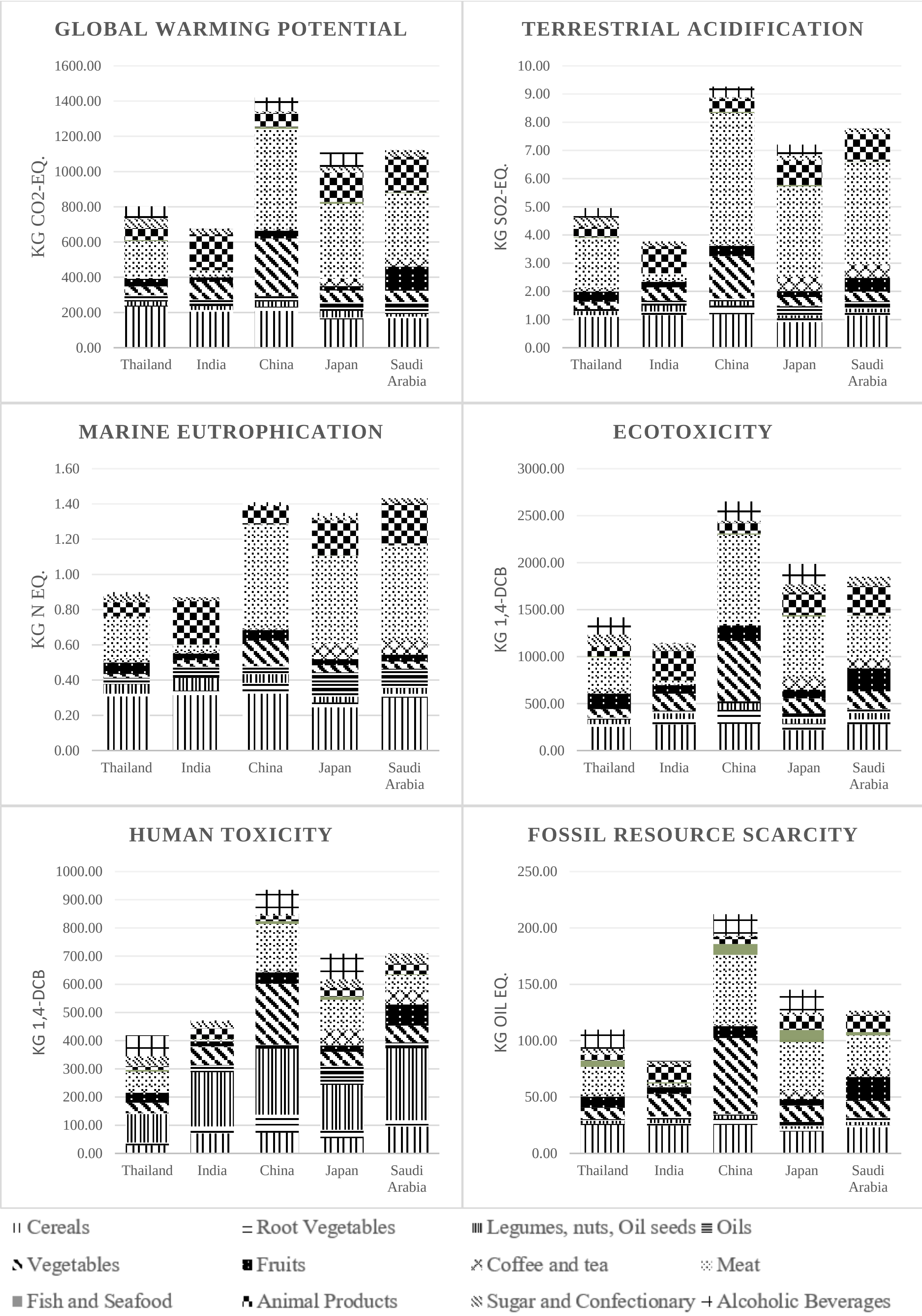


Figure 3.2: Life cycle environmental impacts of food consumption and contribution by twelve food groups.

3.3.3 Comparison of Impacts with Calorie Intake per capita per day

Referring to **Table 3.2** and comparing the per capita calorie intake per day and food intake per year, it shows that the average calorie intake per person significantly exceeds the Food and Agriculture Organization (FAO) benchmark adequate calorie intake of 2500 kcal/capita-day (Galli et al., 2017) for China and Saudi Arabia, which shows opportunities for improvement towards environmental impacts by reducing consumption to a calorie-adequate diet. Further, looking at the impacts of food consumption in each country in relation to calorie intake and kg consumption, it shows that although Thailand has higher calorie intake than Japan, the amount of food consumed per capita per year and the environmental impacts produced by it are lower, which signifies a calorie-rich diet that is also environmentally friendly. But analysing the calorie intake per capita in the Thai diet, it shows that out of 2,785 kcal, 439 kcal (or 16%)is obtained from Sugar and Confectionaries per person in one day in Thailand, as compared to only 255 kcal (or 9%) in Japan (FAO, 2018). Therefore, although the Thai diet is rich in calorie and has low impacts, the high amount of sugar signifies a diet with less nutritional value, prompting negative impacts to human health (Schulze et al., 2004). Furthermore, Drewnowski (2017) stated that healthy diets do not always ensure lower environmental impacts than unhealthy diets, affirming the need for another criterion, i.e. human health to identify a sustainable diet. Therefore, further analysis on the nutritional composition such as protein, lipids, carbohydrate and dietary fibres for each food item in the diet is necessary.

3.4 Conclusions

This study shows that the environmental impacts of food consumption primarily depends on two characteristics of national diets – the total amount of food consumed and the proportions of different food groups in the diet. The highest environmental impacts were found in China and the lowest in India, and the primary sources of environmental impacts from Asian diets were cereals, meat, animal products and alcoholic beverages. Relating the environmental impacts with major contributors, and with calorie intake per capita per day with each country, the study recognizes two viable options regarding the reduction of impacts from the consumption side – shifting to a calorie-adequate diet, i.e. reducing the calorie intake to 2500 kcal/capita-day, and by shifting towards diets that are less environmentally damaging, i.e., in this case, substituting the nutrition obtained from environmentally harmful food groups such as animal products, alcoholic beverages and meat. A study of the ecological footprint of food consumption of Mediterranean countries found that the shifting of consumption towards a calorie-adequate diet and a less ecological footprint intensive diet would lead to the decrease in ecological footprint by 28% and

30% respectively (Galli et al., 2017). Although this study didn’t use the LCA methodology, is specific to Mediterranean countries, and only assesses the ecological footprint, it provides a good incentive to further this study by considering these diet shift scenarios in the LCA to find out the implications to environmental impacts by dietary shifts.

While valuable insights have been drawn from the study, necessary assumptions that were made limits the accuracy of results obtained, and therefore opens opportunity for improvement of the study. For example, most of the background life cycle inventory in the research uses general global data and are therefore not country-specific. Moreover, while the study focussed on the composition of food in Asian diets and their impacts on the environment, it didn’t address the impacts of individual life cycle stages such as agricultural production, transport and storage to find out the hotspots of environmental burdens on the supply chain. Therefore, modelling the food consumption patterns by considering country specific life cycle data would be a step further to improve the accuracy of the results obtained and to ensure the completeness of the life cycle research by also focussing on the impacts of each life cycle stage.

Downs et al. (2017), in their study developed an analysis framework to analyse sustainable diet policies through five chief constructs in their study: (i) nutrition and health; (ii) agriculture and food security; (iii) environment and ecosystems; (iv) markets, trade and value chains for economic growth; and (v) socio-cultural and political factors, which shows that sustainable policies for food consumption should be based on a variety of other factors. Similarly, FAO (2012) defined sustainable diets as not only protective and respectful of biodiversity and ecosystems, but also culturally acceptable, accessible, economically fair, affordable, nutritionally adequate, safe and healthy. Therefore, while this study provides valuable insights that relate food consumption patterns with environmental impacts, it can only address diets that are less environmentally demanding and it only falls under a small domain that can contribute to planning and policy related to sustainable food consumption. Together with this study, further research in the fields mentioned above is required, to develop a holistic tool that can guide policy makers in directing national diets towards being more sustainable.

Chapter 4 Conclusions

This project developed the CLCA method for assessing major food and fuel crops was developed by providing clear modelling steps, important marginal suppliers and recommended applications. The guideline could be used for LCA researchers and practitioners in Thailand and other countries in Asia. Afterwards, case studies applying the CLCA as well as ALCA methods on food and fuel crops in Thailand and Asia were carried out.

The first case study aims to assess climate change mitigation potentials when using biomass-based fuels to replace fossil energy under consequential and attributional modelling approaches in Thailand. The objectives are also to determine policy implications and to recommend the specific contexts suitable for each modelling choice by using specific illustrative cases on biofuels. Both consequential and attributional modelling approaches are chosen for life cycle greenhouse gas emission assessment of several bioenergy options. The assessed functional unit is 1 MJ of energy from molasses-based ethanol, palm-based biodiesel and electricity production from rice straw. The fossil fuel comparators are gasoline (for molasses-based ethanol), diesel (for palm-based biodiesel) and coal and gas (for rice straw). The substituted and substituting product systems are modelled under the global and national markets depending on the market delimitation of each product. The climate change mitigation potentials when using different approaches are dissimilar because the affected product systems being included in the analysis are not the same. The palm biodiesel could reduce greenhouse gas emissions. The molasses-based ethanol and rice straw-based electricity may or may not mitigate the climate change since it depends on the methodological choices as well as the baseline situations of the product systems being investigated. The main characteristics of consequential modelling as additionality and the inclusion of only actually affected processes under market-based mechanisms while those of attributional modelling as specification and attribution/allocation have limitations. The limitations lead to potential risks on unintended and undesirable consequences (for the attributional model), unfairness and sub-optimisation (for the consequential model) in policy recommendations. This research clearly illustrates how certain modelling choices affect the climate change mitigation potentials of biomass-based fuels in comparison with fossil energy. Specific questions and conditions which could be more suitable for each modelling choice are addressed. The attributional modelling is more suitable for national environmental taxation and emission labelling/accounting for import-export while the consequential modelling is more appropriate for new production development and eco-design. Due to the potential environmental risks arising from the modelling limitations, the consideration of both the widely applied approaches could support decisions more comprehensively.

The second case study aims to quantify the life cycle environmental impacts of food consumption in five Asian Countries by using ALCA and CLCA modelling approaches under the LCA framework from ISO 14044. Based on surface area, population density and GDP per capita, the countries chosen are Thailand, India, China, Japan and Saudi Arabia. Food consumption data of each country in 2013 are from FAOSTAT food balance sheets. Background data are from peer-reviewed scientific publications, Ecoinvent 3.4 and Agri-footprint databases. The functional unit used in the study is kilograms of food consumption per capita per year. The ReCiPe2006 v1.1 method was used for impact assessment. The use of ALCA and CLCA modellind methods does not significantly affect the main findings. The assessment shows that food consumption in China yields the highest impacts for most of the impact categories being considered (global warming, terrestrial acidification, eutrophication, eco-toxicity, human toxicity and resource scarcity), followed by the consumption in Japan, Saudi Arabia, Thailand and India. The major contributors to the environmental impacts are Cereals, Animal Products and Alcoholic Beverages. Meat is the key contributor to most impact categories in all countries except India, due to India's low consumption of that food group. Also, comparison of kilocalories (kcal) intake per capita per year of each country with a recommended intake provided by FAO showed opportunities for reduction of impacts through a shift towards calorie-adequate diets, especially in China and Saudi Arabia. Building on the insights of this study, more accurate results could be achieved by the acquisition of country-specific data on food production systems. Linking each food item with energy and nutrient content could further help identify healthier diets and provide recommendations for sustainable consumption.

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Appendix A International Journal Publication

This Appendix presents the international journal publication from this project "*Prapaspongsa, T., Gheewala, S.H., 2017. Consequential and attributional environmental assessment of biofuels: Implications of modelling choices on climate change mitigation strategies. Int. J. Life Cycle Assess. 22(11) 1644–1657*.".

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PROMOTING SUSTAINABILITY IN EMERGING ECONOMIES VIA LIFE CYCLE THINKING

Consequential and attributional environmental assessment of biofuels: implications of modelling choices on climate change mitigation strategies

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Abstract
Purpose The study aims to assess climate change mitigation potentials when using biomass-based fuels to replace fossil energy under consequential and attributional modelling approaches. The objectives are also to determine policy implications and to recommend the specific contexts suitable for each modelling choice by using specific illustrative cases on biofuels.
Methods Consequential and attributional modelling approaches are chosen for life cycle greenhouse gas emission assessment of several bioenergy options. The assessed functional unit is 1 MJ of energy from molasses-based ethanol, palm-based biodiesel and electricity production from rice straw. The fossil fuel comparators are gasoline (for molasses-based ethanol), diesel (for palm-based biodiesel) and coal and gas (for rice straw). The substituted and substituting product systems are modelled under the global and national markets depending on the market delimitation of each product.
Results and discussion The climate change mitigation potentials when using different approaches are dissimilar, because the affected product systems being included in the analysis are not the same. The palm biodiesel could reduce greenhouse gas emissions. The molasses-based ethanol and rice straw-based electricity may or may not mitigate the climate change, since it depends on the methodological choices as well as the baseline situations of the product systems being investigated. The main characteristics of consequential modelling as additionality and the inclusion of only actually affected processes under market-based mechanisms while those of attributional modelling as specification and attribution/allocation have limitations. The limitations lead to potential risks on unintended and undesirable consequences (for the attributional model), unfairness and sub-optimisation (for the consequential model) in policy recommendations.
Conclusions This research clearly illustrates how certain modelling choices affect the climate change mitigation potentials of biomass-based fuels in comparison with fossil energy. Specific questions and conditions which could be more suitable for each modelling choice are addressed. The attributional modelling is more suitable for national environmental taxation and emission labelling/accounting for import-export, while the consequential modelling is more appropriate for new production development and eco-design. Due to the potential environmental risks arising from the modelling limitations, the consideration of both the widely applied approaches could support decisions more comprehensively.

Keywords Consequential modelling approach · Attributional modelling approach · Biofuel · Direct land use change · Indirect land use change

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1 Introduction

Biofuels for power production and transport have been promoted nationally and globally as an essential element of climate change mitigation strategies (OECD/IEA 2014, 2015; ONEP 2015; DEDE 2015). The world energy outlook 2014 (OECD/IEA 2014, p.255) stated that "renewable energy technologies emit no greenhouse gases as they produce electricity, making them an essential element of a strategy to mitigate climate change". According to the outlook, the climate change impacts from biofuels—an important renewable energy source—were considered as avoided CO_2 emissions. These values were directly calculated from the emissions which other generation technologies could produce if the renewable energy technologies have not been employed. In 2040, bioenergy and biofuels might avoid 760 and 450 Mt. CO_2 emissions, respectively (OECD/IEA 2014). Not only have the avoided CO_2 emissions been reported at a global level (OECD/IEA 2014, 2015), but it was also documented in Thailand's Alternative Energy Development Plan: AEDP2015 (DEDE 2015, p.20) that alternative energy sources including biomass and biofuels could potentially reduce greenhouse gas (GHG) emissions due to the decrease in fossil fuel combustion.

In fact, there are emissions and environmental impacts associated with the acquisition of renewable energy sources, production, transportation, use and end-of-life treatment. Previous studies have shown that some renewable energy technologies in certain conditions may possibly lead to additional GHG emissions and other environmental impacts when compared with fossil energy (Pehnt 2006; Plevin et al. 2014a; Prapaspongsa et al. 2017; Searchinger et al. 2008). For bioenergy, some GHG emissions such as biogenic CO_2 emissions during combustion process might be omitted, but there are other GHG emissions from the whole life cycle to be considered. In order to develop strategies which actually lead to climate change mitigation, it is therefore crucial to carry out an environmental assessment of biofuels in terms of GHG emissions from raw material acquisition to end-of-life treatment in comparison with conventional fossil fuels.

Various life cycle thinking approaches are available for assessing GHG emission reduction. Consequential and attributional LCA (CLCA and ALCA) modelling approaches are chosen in this research, because they have been widely applied in LCA studies and in the estimation of climate change mitigation benefits of biofuels (Martin et al. 2015; Prapaspongsa and Gheewala 2016; Schmidt and Brandão 2013; Silalertruksa and Gheewala 2013). UNEP/SETAC Life Cycle Initiative (2011) defined CLCA and ALCA modelling approaches as "system modelling approach in which activities in a product system, are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit" and "system modelling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule", respectively. Both modelling choices have recently and intensively been debated on vis-à-vis suitability, reliability and creditability for decision support (Anex and Lifset 2014; Brandão et al. 2014; Dale and Kim 2014; Hertwich 2014; Plevin et al. 2014a, b, c; Suh and Yang 2014). In order to provide a robust decision support, the inclusion of the two main modelling schools and the determination of specific questions which each modelling choice could potentially answer are crucial. Previous CLCA and ALCA studies on biofuels have focused on methodological discussions at a conceptual level or have used case studies with a preference of a certain modelling approach (Martin et al. 2015; Plevin et al. 2014a; Schmidt and Brandão 2013; Silalertruksa and Gheewala 2013). Furthermore, the case studies considering both modelling choices in emerging economies, where there are very high potentials in actual climate change mitigation, are rare. This study targets at presenting various illustrative cases in assessing climate change mitigation potential of biofuel production and consumption in Thailand for fossil energy replacement under consequential and attributional modelling approaches. The biofuels include molasses-based ethanol, palm-based biodiesel and rice straw (for electricity production) in Thailand which could be useful for supporting the decisions on other environmental strategies in Thailand, and other emerging and developed economies.

2 Methodology

2.1 Goal and scope

The main goals of this environmental assessment study are (1) to assess climate change mitigation potential by considering GHG emissions of biofuels in comparison with fossil energy under CLCA and ALCA modelling approaches, (2) to determine the implications of each modelling choice for policy support and (3) to recommend the specific contexts in which each approach may be more appropriate. It should be mentioned that this study has focused only on a single issue which is climate change mitigation potential. In fact, an LCA study shall consider potential environmental impacts. Nonetheless, with the aim to assess the implications of modelling choices in decision-making, a single impact assessment could be applied for illustrating various cases and conditions more effectively, since a multi-impact assessment with various environmental impact categories might potentially shift the research focuses. The functional unit (FU) is defined as 1 MJ of biomass-based fuels and fossil energy. The fossil comparators are gasoline for molasses-based ethanol, conventional diesel for palm-based biodiesel and electricity from natural gas, coal and Thailand's grid mix for rice straw-based electricity. The overall system boundaries of Thailand's molasses ethanol, palm biodiesel and rice straw electricity production systems are illustrated in Figs. 1, 2 and 3. Each figure demonstrates the investigated life cycle stages under CLCA and ALCA data modelling approaches.

The CLCA methodology according to Weidema (2003) and Weidema et al. (2009), widely applied/considered in LCA research and methodology (Earles and Halog 2011; Prox and Curran 2017; Schmidt 2015), is applied in this study. The CLCA approach attempts to assess consequences from a change in demand and includes only finally affected product systems under market-based causal relationships. By using this approach, marginal suppliers/technologies/unit processes (or the suppliers/technologies/unit processes which are subjected to change according to the additional demand in the study) are taken into account, whereas the co-product allocation is avoided by using system expansion or substitution (Weidema et al. 2009; Pelletier et al. 2015). The identification of marginal suppliers (or the affected unit processes) in CLCA can be done by considering the scale and time horizon of the changes being assessed, the market delimitation, the market trends and the changes in supply and demand (Weidema et al. 2009). If the market trends are increasing, the most competitive suppliers/technologies will be affected, and vice versa. For system expansion, only the unconstrained systems driven the change in demand are included. Further detailed CLCA methodology can be seen in Weidema et al. (2009).

As presented in Figs. 1a, 2a and 3a, the molasses ethanol, palm biodiesel and rice straw electricity production systems in Thailand influence the substituted/substituting product systems via national and international market linkages. Molasses and rice straw are dependent co-products (so called by-products) from sugar and rice production systems, whereas palm oil is the main product from the palm oil production system producing various co-products throughout the life cycle stages. The additional demand for molasses and rice straw will not increase sugarcane and rice production and result in additional production of substitutable products. According to Prapaspongsa and Gheewala (2016), the use of molasses (as the main raw material) and bagasse (as heat energy source in the ethanol conversion stage) will contribute to additional feed energy and marginal heat energy source production. For rice straw utilisation, various utilisation pathways including field burning, soil fertilising and animal feeding are common in Thailand (Sansiribhan et al. 2014). The additional demand for palm diesel will increase the production of co-products during palm oil milling and biodiesel conversion stages and result in displaced production of substitutable products. The substitution pathways from Prapaspongsa et al. (2017) are followed in this work (energy production from palm kernel and palm shell substitutes the marginal electricity production; palm kernel oil substitutes the marginal oil;

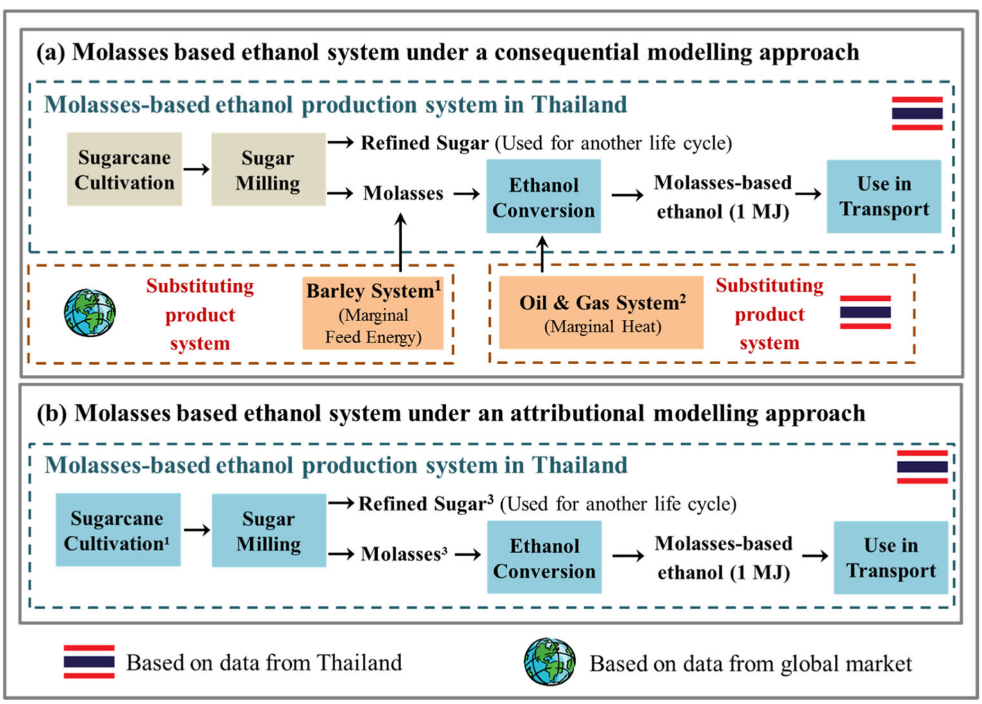

Fig. 1 Overall system boundaries of Thailand's molasses ethanol production (modified from Prapaspongsa and Gheewala 2016). **a** Molasses-based ethanol system under a consequential modelling approach. The *grey boxes* represent the excluded processes, because they are not affected by the additional molasses ethanol production. **b** Molasses-based ethanol system under an attributional modelling approach. (*1*) dLUC and iLUC are included in this life cycle stage/product system. (*2*) The use of bagasse for heat energy (steam) in the ethanol conversion stage will affect the marginal heat system (natural gas and fuel oil), since bagasse is fully utilised in Thailand. (*3*) The environmental impacts from relevant life cycle stages are allocated to refined sugar and molasses by using an economic allocation factor

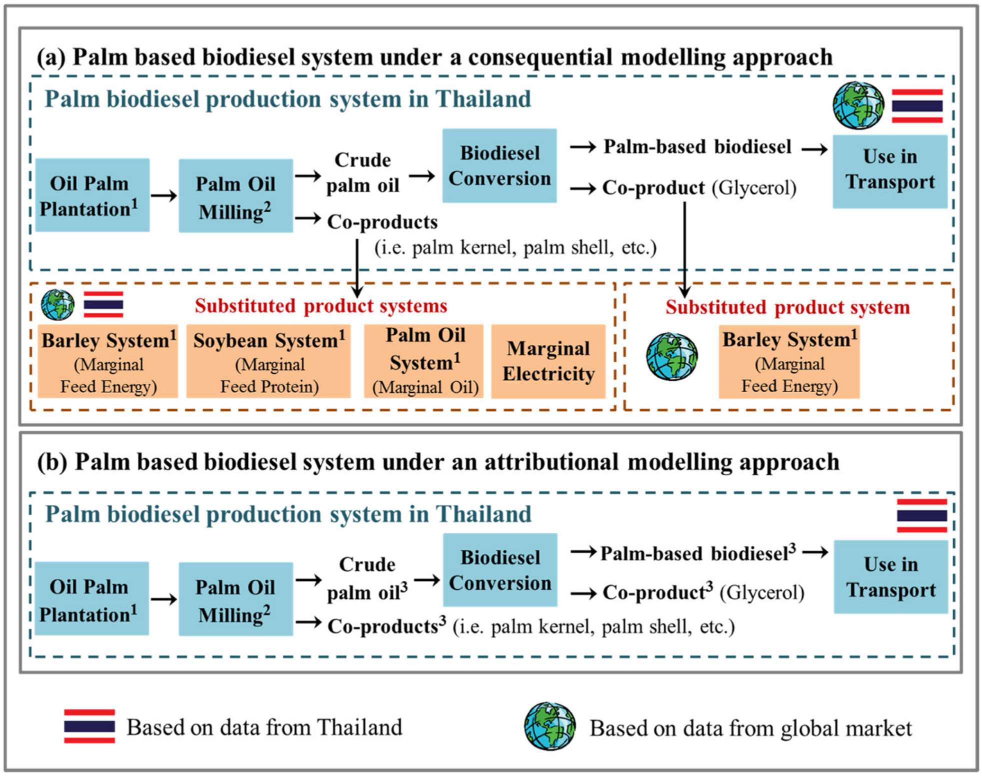

Fig. 2 Overall system boundaries of Thailand's palm biodiesel production (modified from Prapaspongsa et al. 2017). **a** Palm-based biodiesel system under a consequential modelling approach. Electricity production from palm kernel and palm shell substitutes the marginal electricity system. Palm kernel oil substitutes the marginal oil system. Palm kernel meal substitutes the marginal feed protein and feed energy. Glycerol substitutes the marginal feed energy. **b** Palm-based biodiesel system under an attributional modelling approach. (*1*) dLUC and iLUC are included in this life cycle stage/product system. (*2*) Average technologies for palm oil mill effluent treatment with 83% biogas capture (weight average from 6 palm oil mills) according to Kaewmai et al. (2013) are considered. (*3*) The environmental impacts from relevant life cycle stages are allocated to crude palm oil, palm biodiesel and other co-products by using economic allocation factors

glycerol substitutes feed energy; and palm kernel meal substitutes the marginal feed energy and feed protein). Only the markets for marginal electricity and heat are delimited at a national level (in Thailand), whereas the markets for marginal oil, feed energy and feed protein are considered at a global level (Prapaspongsa and Gheewala 2016; Prapaspongsa et al. 2017). It should be mentioned that the figures only illustrate the main and co-products of molasses-based ethanol, palm-based biodiesel and rice straw-based electricity systems. Wastewater treatment processes are not illustrated in the figures, but their relating emissions are taken into account. In CLCA modelling, the wastewater treatment systems with biogas recovery with electricity generation and land application will displace marginal electricity and fertiliser production. In order to capture the potential substitution pathways and conditions, CLCA scenarios (M-ethanol_C1 and C2, P-biodiesel_C1 and C2, and R-electricity_C1 to C3) are developed as described in Table 1.

In the ALCA approach, average suppliers/technologies and economic allocation of the molasses ethanol, palm biodiesel and rice straw electricity production systems are taken into consideration (see the system boundaries in Figs. 1b, 2b and 3b). The general description of all ALCA scenarios (M-ethanol_A, P-biodiesel_A, and R-electricity_A) is explained in Table 1.

Direct and indirect land use changes (dLUC and iLUC, respectively) are taken into account by considering the GHG emissions from direct land transformation (for dLUC) and the upstream impacts of the land tenure (for iLUC). The dLUC calculation is performed according to IPCC (2007) and Silalertruksa and Gheewala (2012b) taking into account GHG emissions from the direct changes in biomass carbon stocks, dead organic matter and soil carbon stocks. Land use changes for agricultural systems in Thailand are dynamic. Farmers often change their agricultural products (e.g. rice, sugarcane and other crops) depending on market prices without considering land suitability. Oil palm has been planted across Thailand (66 out of 77 provinces in Thailand; OAE 2015), although not all areas are suitable for palm cultivation. Saswattecha et al. (2016) presented significantly different land use baselines for oil palm plantation during two periods of time (between 2000 and 2009; and between 2009 and 2012). New oil palm plantation during the latter period of time mostly occurred in the established cropland. Due to the data limitation, the previous land use type (or the land use baseline) of

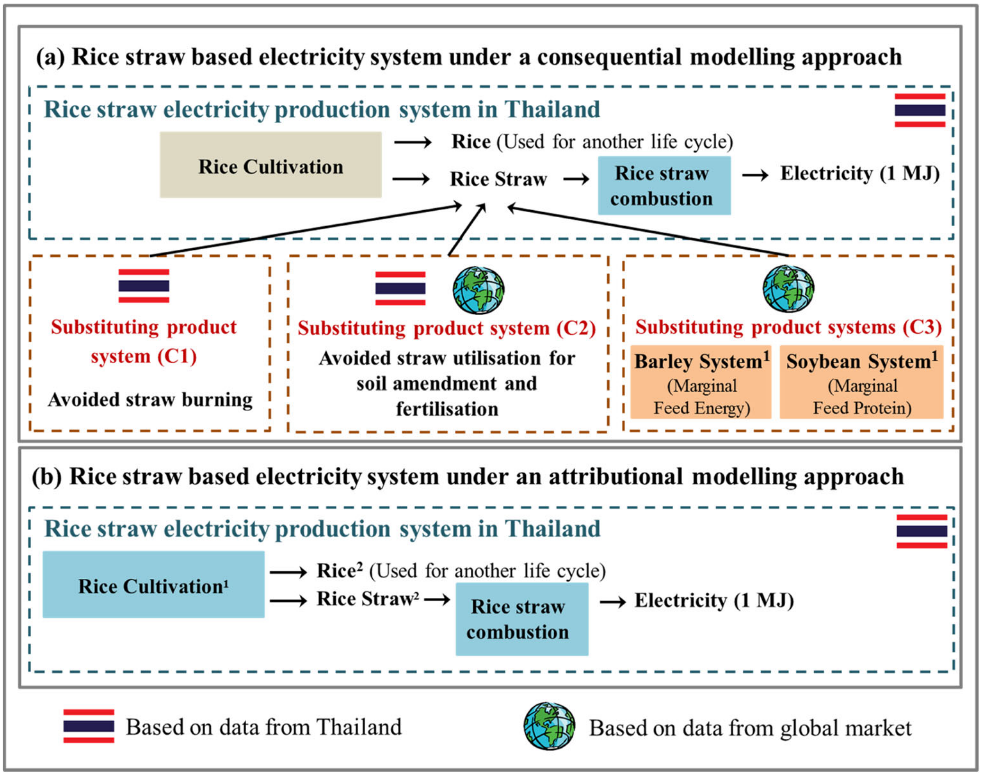

Fig. 3 Overall system boundaries of Thailand's rice straw electricity production. **a** Rice straw-based electricity system under a consequential modelling approach. The *grey boxes* represent the excluded processes, because they are not affected by the additional rice straw electricity production. **b** Rice straw-based electricity system under an attributional modelling approach. (*1*) dLUC and iLUC are included in this life cycle stage/product system. (*2*) The environmental impacts from relevant life cycle stages are allocated to rice and rice straw by using an economic allocation factor

the sugarcane, oil palm and rice cultivation in Thailand in the dLUC analysis is assumed to be the current share of paddy fields, para rubber plantation, oil palm plantation and crop cultivation areas in Thailand in 2013 (as 59.5, 10, 4 and 26.5%, respectively) (OAE 2015). The same land use type for the agricultural systems outside Thailand is assumed. Hence, there is no dLUC for such case.

For iLUC, a biophysical indirect land use change model developed by Schmidt et al. (2015) considering the upstream consequences from land being in use, land expansion and intensification is applied for both CLCA and ALCA modelling approaches. This model has been applied in recent LCA studies (Dalgaard et al. 2014; Flysjö et al. 2012; Prapaspongsa and Gheewala 2016; Schmidt 2015). Specific data of GHG emissions from iLUC due to land use in Thailand are based on Prapaspongsa and Gheewala (2016). The iLUC emissions are also considered in the agricultural systems outside Thailand.

The life cycle impact assessment method in this study is IPCC (2007) GWP 100a, and the calculations are carried out by using SimaPro 8.0.3 (PRé Consultants bv, Amersfoort, the Netherlands).

2.2 Data collection

The foreground data were mainly obtained from existing studies with field data collection in Thailand (Gheewala et al. 2017; Kaewmai et al. 2013; OAE and GIZ 2012; Silalertruksa and Gheewala 2009, 2011, 2012a, 2013; Silalertruksa et al. 2015, 2017; Suttayakul et al. 2016). The marginal electricity production in CLCA modelling only considers the additional installed capacity/generation and excludes the constrained energy sources. The average electricity production in ALCA modelling considers the total production from the national grid mix. The marginal electricity sources in Thailand identified by Prapaspongsa and Gheewala (2016) consist of 49% gas, 28% coal and lignite, 2% solar energy, 2% wind, 9% hydropower and 9% biomass. Due to the lack of data, only specific share of marginal electricity sources in Thailand is used; and the background data are from the ecoinvent database. For the marginal biomass energy, it was determined as wood pellets from Brazil because of its highest cost competitiveness (Prapaspongsa and Kørnøv 2012; Prapaspongsa and Gheewala 2016). The marginal heat production in Thailand is from oil and gas (Prapaspongsa and Gheewala 2016). Marginal feed energy was defined as barley in Spain due to its cost competitiveness and largest incremental production volume (Prapaspongsa and Gheewala 2016). The foreground data on barley production were obtained from Lechón (2011) and Lechón et al. (2011). According to Schmidt (2007, 2015) and Schmidt and Brandão (2013), marginal feed protein and oil were considered as soybean meal and palm oil, respectively. The data for soybean meal and palm oil were obtained from Schmidt and Brandão (2013) and Prapaspongsa et al. (2017). The data for fossil comparators (gasoline, diesel, and electricity from hard coal, natural gas and national grid) are from TGO (2016, 2017), Phumpradab et al. (2009), Thai national life cycle inventory (LCI) database and supplemented with

Table 1 Scenario description of three major biofuel systems in Thailand

Scenarios	Description
Molasses-based ethanol system (M-ethanol)	
General description for the system: existing cane molasses ethanol system in Thailand where the molasses ethanol plant uses steam and electricity from bagasse supplied from a sugar mill. Wastewater and vinasse treatment processes in the sugar milling and ethanol production plants include oxidation and stabilisation ponds system, anaerobic digestion system with biogas recovery and land application. The surplus energy at sugar mills and ethanol plants is sold to the national grid system.[a]	
Fossil comparator[b]: gasoline	
M-ethanol_C1	The consequential modelling approach is applied under the fully utilised condition of cane molasses and bagasse.
M-ethanol_C2	The consequential modelling approach is applied under the non-fully utilised condition of cane molasses and bagasse. It is assumed that available molasses and bagasse are not fully utilised and will be kept in stock without being thrown away due to their economic value.
M-ethanol_A	The attributional modelling approach is applied by using average suppliers and economic allocation factors.
Palm oil-based biodiesel system (P-biodiesel)	
General description for the system: palm biodiesel production under average condition in Thailand with approximately 83% biogas capture in palm oil mill effluent treatment during the oil milling stage (weight average from 6 palm oil mills). The surplus energy at oil mills is sold to the national grid system.[c]	
Fossil comparator[b]: diesel	
P-biodiesel_C1	The consequential modelling approach is applied under the fully utilised condition of all co-products.
P-biodiesel_C2	The consequential modelling approach is applied under the situation that the co-products during the palm oil milling stage (palm kernel, palm shell, empty fruit bunch and palm kernel meal) are not fully utilised. 50% of the co-products are left as waste.
P-biodiesel_A	The attributional modelling approach is applied by using average suppliers and economic allocation factors.
Rice straw-based electricity system (R-electricity)	
General description for the system: rice straw-based electricity production system based on the 10-MWe straw-based power plant from Delivand et al. (2011, 2012). The surplus energy is sold to the national grid system.[d]	
Fossil comparator[b]: electricity from coal, natural gas and Thailand's national grid mix	
R-electricity_C1	The consequential modelling approach is applied under the situation that the rice straw was previously burned in the field to hasten the planting process for the next crop.
R-electricity_C2	The consequential modelling approach is applied under the situation that the rice straw was previously chopped and used as fertiliser in the field.
R-electricity_C3	The consequential modelling approach is applied under the situation that the rice straw was previously used as an animal feed.
R-electricity_A	The attributional modelling approach is applied by using average suppliers and economic allocation factors.

[a] The system and life cycle inventory data are based on Silalertruksa et al. (2015, 2017), Gheewala et al. (2017), Prapaspongsa and Gheewala (2016), and Yuttitham et al. (2011)

[b] The life cycle inventory data are from TGO (2016, 2017) and Phumpradab et al. (2009) supplemented with ecoinvent database

[c] The system and life cycle inventory data are based on Prapaspongsa et al. (2017)

[d] The system and life cycle inventory data are based on Silalertruksa and Gheewala (2013) except the consequential modelling approach and the rice straw utilisation pathway as an animal feed

Ecoinvent database. The background data were obtained from the existing database in SimaPro, such as Ecoinvent data v2.2 (ecoinvent Centre 2010). The LCI data and important assumptions are documented in the Electronic Supplementary Material.

3 Results and discussion

Potential GHG emissions of 1 MJ molasses-based ethanol, palm-based biodiesel and rice straw-based electricity scenarios in comparison with fossil energy are shown in Fig. 4. Total climate change potential values of all scenarios are documented in the Electronic Supplementary Material. At the beginning, the assessment results by using the consequential and attributional approaches will be discussed separately. Later, the implications on policy recommendations for climate change mitigation strategies under various contexts in Thailand are pointed out and discussed.

3.1 Consequential environmental assessment of biofuels

This section will demonstrate the climate change mitigation potentials from the fossil fuel replacement by biofuels with the use

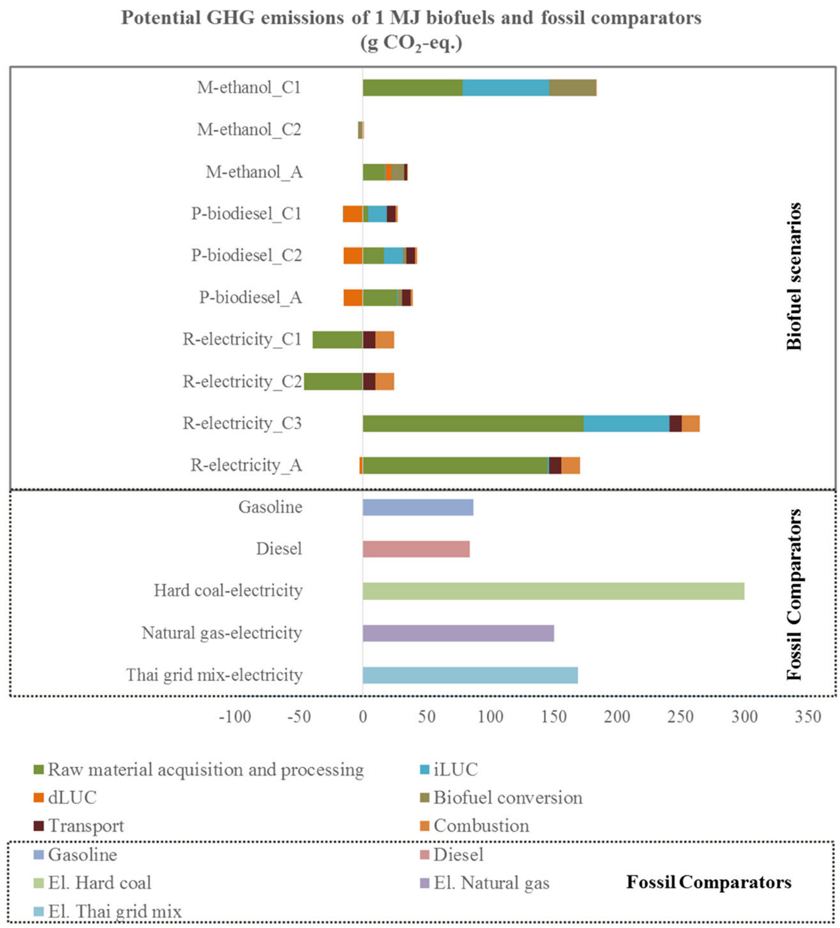

Fig. 4 Potential GHG emissions of 1 MJ biofuels and fossil comparators with process contribution. The biofuel systems include molasses-based ethanol scenarios (M-ethanol), palm-based biodiesel scenarios (P-biodiesel) and rice straw-based electricity scenarios (R-electricity) under consequential (C) and attributional (A) modelling approaches. The fossil comparators are gasoline, diesel and electricity production from hard coal, natural gas and Thai grid mix

of the CLCA approach without questioning the methodological choices. Specifically, the policy implications of this modelling choice application will later be discussed in Section 3.3.

The palm-based biodiesel scenarios (P-Biodiesel_C1 and C2) potentially mitigate climate change by the fossil energy (conventional diesel) displacement. The molasses-based ethanol scenarios (M-ethanol_C1 and C2) and the rice straw-based electricity scenarios (R-electricity_C1 to C3) may or may not be able to yield lower GHG emissions in comparison with their fossil comparators (gasoline, hard coal-electricity, natural gas-electricity, Thai grid mix-electricity). The assessment can be categorised into three specific cases—palm oil as the main product (or determining co-product) during the oil palm plantation and biodiesel conversion stage, molasses as the high value by-product (or dependant co-products) during the sugarcane cultivation stage and rice straw as the low value by-product during the rice cultivation stage.

As the main product under the effective by-product utilisation system, the palm biodiesel could reduce GHG emissions from the conventional diesel consumption up to 73 g CO_2-eq./MJ. During the raw material acquisition and processing stage with fully utilised co-product condition (P-Biodiesel_C1), the total emissions from oil palm cultivation (excluding dLUC and iLUC) and palm oil mill processing are 29 g CO_2-eq./MJ with avoided emissions from the co-product substitutions of 25 g CO_2-eq./MJ resulting in the total emissions during the stage of 4 g CO_2-eq./MJ (see Fig. 4). Since palm cultivation in Thailand comparatively has higher carbon stocks than the other land use types (except natural forest), the dLUC from oil palm cultivation also contribute to GHG saving for the biofuel scenarios (P-Biodiesel_C1 and C2; see Fig. 4). It should be mentioned that the land baseline mix in Thailand for oil palm plantation is different from the ones in other countries (i.e. Indonesia and Malaysia). The direct land use change from wetlands and peatlands to be oil palm represented 0.2% of total land use change from 2009 to 2012 (Saswattecha et al. 2016). If illegal deforestation (i.e. converting forest/wetlands to be oil palm) occurs, the GHG emissions from palm-based biodiesel will be around 2.5 times of those from the conventional diesel (Prapaspongsa et al. 2017). If the co-products are only 50% utilised, the GHG emissions from the palm biodiesel will increase with a factor of 2.4 (see P-Biodiesel_C1 and C2 in Fig. 4). Nevertheless, the by-products from palm oil industry such as palm kernel, palm shell, palm kernel oil and palm kernel meal could be easily used or sold for energy recovery and/or animal feed resulting in economic benefits of the owners. In a well-established market, the default condition for the by-products with high economic values is rather full utilisation.

The molasses-based ethanol illustrates a case of the high value by-product. In case the molasses is fully utilised, the ethanol production from molasses (M-ethanol_C1) will contribute to additional marginal feed production and higher GHG emissions around two times that of gasoline. Part of the high emissions is also associated with the use of bagasse for heat energy in the biofuel conversion stage. Bagasse has also been fully utilised for energy production, and the affected system from the additional bagasse consumption will be the marginal heat production (from oil and gas) in Thailand (Prapaspongsa and Gheewala 2016). Under the non-fully utilised condition, molasses and bagasse could be seen as waste and freely available without environmental impacts. The molasses-based ethanol under such condition (M-ethanol_C2) will yield overall GHG saving due to the electricity and fertiliser replacement from vinasse treatment system (51% biogas capture and 49% land application as fertiliser) and be able to reduce GHG emissions from gasoline displacement of around 90 g CO_2-eq./MJ. The important question is whether molasses and bagasse in Thailand have generally been fully utilised. Prapaspongsa and Gheewala (2016) reviewed the market situations in Thailand and the world (Esther 2013; FAOSTAT 2014; UN data 2014; UM Trading 2014) and concluded that the molasses from Thai sugar industry has been used in the local market, traded in the global market, and therefore fully utilised locally and globally. Bagasse has also been fully utilised nationally (Prapaspongsa and Gheewala 2016). Consequently, the molasses-based ethanol might not mitigate climate change potential from the fossil energy displacement in this first conclusion from the consequential environmental assessment.

The rice straw-based electricity production represents the low value by-product case. Three common utilisation pathways are direct burning in the field burning, using for soil fertilisation and amendment and selling for animal feed (R-electricity_C1 to C3, respectively) which highly affect the mitigation potential. If the rice straw is directly burnt in the field or used for fertiliser displacement and soil amendment (R-electricity_C1 and C2), GHG savings from rice straw when comparing with fossil energy yield the highest among the three biofuels (up to 322 g CO_2-eq./MJ). The main contribution for the GHG savings is from the avoided emissions if the rice straw is burnt or used in the field (see Fig. 4). In case the rice straw is used as animal feed (R-electricity_C3), it will increase marginal feed production, and its related GHG emissions (including iLUC emissions) will be higher than those of the electricity production from natural gas and Thai grid mix (see Fig. 4). *Which situation is likely to be the most representative for rice straw utilisation?* Due to the fact that rice straw has low value (moderate energy content) with high operation costs for the handling and transport processes, field burning is still the most practical option for farmers. As a result, the rice straw-based electricity seems to be the most promising biofuel for mitigating climate change under the investigated systems in this research.

3.2 Attributional environmental assessment of biofuels

This section will discuss the attributional assessment of the potential GHG emission reduction when replacing fossil fuels with the biofuels without being in doubt on the methodological choice itself. The comparison with the CLCA methodology is also discussed briefly here. Nonetheless, the specific context and questions which this approach might be or might not be used for decision support will later be discussed in Section 3.3.

In contrast with the previous section, under the ALCA approach, the molasses-based ethanol (M-ethanol_A) potentially contributes to climate mitigation, whereas the rice straw-based electricity (R-electricity_A) yields the GHG emissions almost equal to electricity production from the Thai grid mix. Similarly, the palm-based biodiesel could reduce the GHG emissions from conventional diesel substitution. Figures 1, 2 and 3 have clearly illustrated that the investigated systems of each product under the CLCA and ALCA approaches are not the same. As a result, their climate change mitigation potentials are different. Based on the set of scenarios in the assessment, the CLCA scenarios may vary to a larger extent, while the ALCA scenarios seem to be static. However, the previous results only include economic allocation factors to partition total GHG emissions of all life cycle stages to different co-products. Other allocation factors (i.e. mass and energy) may not be adequate for all cases. For instance, it will be less meaningful to use an energy allocation factor for the rice straw-based electricity system, because the main function of rice is for food. In order to determine the potential variability in ALCA, economic, energy and mass allocation factors during cultivation (10, 24 and 26%, respectively) are applied for allocating GHG emissions to the molasses in the molasses-based ethanol cases as can be seen in Fig. 5.

From economic to mass allocation, the GHG emissions from the molasses-based ethanol could change by a factor of 2 (see M-ethanol_A-Economic and A-Mass in Fig. 5). For the rice straw electricity production, if other allocation factors are higher than the economic one, this biofuel will contribute to GHG emissions at a higher rate than the fossil energy (except the electricity from hard coal). The selection of the allocation factors therefore plays an important role in ALCA.

3.3 Implications on policy recommendations for establishing future climate change mitigation strategies

The climate change mitigation potentials of the fossil energy replacement by the three biofuels taking into account the

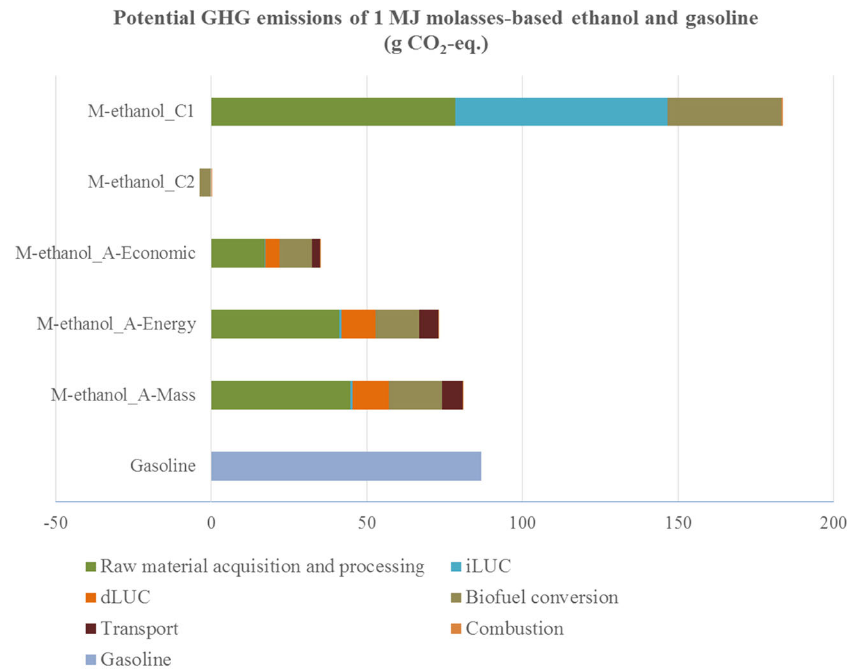

Fig. 5 Potential GHG emissions of 1 MJ molasses-based ethanol scenarios and gasoline with process contribution. The molasses-based ethanol scenarios under an attributional modelling approach apply economic, energy and mass allocation factors (M-ethanol_A-Economic, M-ethanol_A-Energy and M-ethanol_A-Mass, respectively)

CLCA and ALCA approaches have been discussed in the previous parts. This section aims at assessing the policy implications of the methodological choices and to suggest the specific contexts and questions which each modelling approach may support or provide the answers for. It should be addressed that the main purpose of this paper is not to determine the superiority or to judge each approach. The authors would suggest that an LCA study is highly recommended to apply both modelling choices to identify possible risks of a decision, since a single and standalone approach may impossibly capture all potential consequences. Nevertheless, under the situations when different models provide conflicting conclusions, the main choice to be held upon is crucial for decision support. The specific objectives and recommended applications of the two modelling choices are discussed as follows.

3.3.1 Specific questions for CLCA and ALCA approaches

"To assess climate change mitigation potential by considering GHG emissions of biofuels in comparison with fossil energy under consequential and attributional modelling approaches" with the FU as 1 MJ of biomass-in fuels and fossil energy could be set as the objective for both approaches. The problem is that the defined objective is too broad and does not distinguish the specific situations each approach could be suitable for. This objective is intentionally set to imitate a typical condition when an LCA study is carried out to support general policies without specific users which could lead to different interpretations and conclusions due to its generality. *What could then be the specific objectives?*

The CLCA modelling approach determines the impacts from a change in demand and to include only the systems which are expected to change due to the additional product. Brandão and Weidema (2013) identified the questions which the CLCA approach attempts to answer are "What are the net impacts associated to a change (in a product system) relative to the baseline scenario, where that change does not take place?" and "What are the consequences of a decision relative to the 'no action' baseline?". The specific question in this study could be "What would actually happen or what are the potential consequences if we additionally produce 1 MJ of the biofuels in Thailand?" The keywords are "additionality" and "potential affected processes linked via market mechanisms". The functional unit is rather the additional production of 1 MJ biofuels and fossil fuels in this case. Based on Brandão and Weidema (2013), this model looks at the baseline system as the situation without any action and considers only the additional product of concern without the assessment of the baseline system itself. With the specific question a CLCA study could properly provide the answer, it is therefore crucial for the policy makers/researchers to explicitly state the question during the goal and scope definition phase. To answer such a question, the expectedly actually affected processes linked via market mechanism or the marginal technologies/suppliers are identified by considering market information such as market delimitation, trends and constraints. As previously mentioned in Section 2.1, the market in study is delimited at the national level for electricity and heat and at global level for other expanded systems. With the current stable and/or increasing market trends, only the most competitive suppliers responding to the change in demand are included (i.e. for palm-based biodiesel and rice straw-based electricity systems, marginal feed protein and energy for co-product displacement being considered as soybean meal and barley, respectively; see Figs. 2a and 3a). The constrained suppliers/processes will not be considered (i.e. for the molasses-based ethanol and rice straw-based electricity systems, sugar cane cultivation, sugar milling and rice cultivation processes are excluded; see Figs. 1a and 3a).

The ALCA modelling approach partitions the total impacts to the functional unit of a product system by using a normative rule (i.e. economic allocation technique in this study). De Camillis et al. (2013) determined the relevant questions for this approach as "What is the environmental impact of a certain product system at a given time (when baseline scenario is assessed)?" and "What is the environmental impact of a certain product system in a given future scenario if the product were designed or/and produced or/and consumed or/and managed differently at the end of its life?". For this study, the specific question could be "What are the specific GHG emissions of 1 MJ biofuel production in Thailand allocated by economic values of all related co-products?". The keywords are "specification" and "allocation/attribution". The functional unit as of 1 MJ biofuels and fossil fuels might still be valid in this case. This model firstly looks at the total emissions from specific production systems by including all product outputs in the systems which are directly linked throughout the supply-chains (i.e. the direct supply-chains for molasses-based ethanol, palm-based biodiesel and rice straw-based electricity in Thailand as illustrated in Figs. 1b, 2b and 3b) and, secondly, seeks for a specific allocated share of the relevant impacts for the specific product being investigated.

3.3.2 Potential limitations and risks of CLCA and ALCA approaches

As explained before, both CLCA and ALCA approaches are able to support decisions for climate change mitigation strategies in different contexts; however, an important concern would be "What are potential limitations and risks of the choices made?". Ekvall et al. (2005) indicated that the risks from the ALCA approach (or the retrospective LCA) are unaccounted and undesirable consequences, and those from the CLCA approach (or the prospective LCA) are unfairness and sub-optimised systems. The potential limitations in this analysis refer to what cannot be assessed or captured by each modelling choice and are certain conditions which may make the approaches incorrectly assess the environmental impacts associated with the products in question. The potential risks are consequences from certain limitations of the two approaches.

Specification and allocation/attribution could be seen as main limitations of the ALCA approach. The ALCA approach covers the allocated impacts from processes directly linked with the specific supply-chain. The specific processes are not necessarily affected by the additional production of the investigated products, and the assessment may not reflect what would happen in reality. For example, the sugarcane and rice cultivation will not potentially be driven or affected by the demand for molasses and rice straw in reality, because molasses and rice straw are dependent by-products. Since the ALCA approach do not include the consequences of individual actions (a marginal change in demand), this could eventually lead to the unaccounted harmful environmental impacts. For instance, Thai policy makers may support the molasses-based ethanol for gasoline replacement, since it could reduce GHG emissions around 61 g CO_2-eq./MJ under an economic allocation rule and the inclusion of sugarcane cultivation and sugar milling processes in Thailand. Nonetheless, supporting the increased use of molasses for ethanol production may lead to an increase in the animal feed production somewhere else in the world (i.e. barley in Spain), and its emissions may far outweigh the fossil comparator (i.e. 142 g CO_2-eq./MJ additional emissions). With this illustrative case, the use of the ALCA approach might be questionable for supporting a decision. However, UNEP/SETAC Life Cycle Initiative (2011) described that the ALCA model aims at identifying the portion of global burdens which a product could potentially contribute throughout its life cycle. It was also mentioned that "In theory, if one were to conduct LCAs of all final products with attributional modelling, one would end up with the total observed environmental burdens worldwide" (UNEP/SETAC Life Cycle Initiative 2011). At a given time of the study, the background situation will be static. If all products are being assessed under the same allocation rule, it will therefore be able to capture the overall environmental impacts, and the allocated emissions to the molasses-based ethanol would be reasonable and not lead to unintended consequences.

With respect to the CLCA modelling, the main features of affected processes linked via market mechanisms and additionality could also lead to certain limitations. Firstly, the limitation is on the market-based considerations: producers/suppliers are price-takers, and those systems with lowest production costs will likely be the most competitive (if the market trend is stable or increasing); and elasticity of supply and demand will lead to linear and direct relationships, etc. Since market failures are very common in reality, the modelling choice based on market information may not capture the real consequences as expected. Secondly, with the additionality viewpoint, CLCA only considers additional production for an additional demand and excludes existing production for an existing demand. It may be unfair for new suppliers who demand for the fully utilised dependent co-products such as molasses (see Section 3.1). If a sugar factory produces 100 t of molasses a year and has already used internally and sold in the market for all, the additional demand of 1 t of molasses will affect the marginal supplier for the molasses consumption which is Barley in this study. It is questionable who will get the credits of the existing 100 t of molasses co-produced with sugar production. Furthermore, in case it is a CLCA study of refined sugar for sugar industry, the additional sugar production will lead to the additional molasses production and environmental impact reduction from the co-product substitution. If it is a CLCA study of molasses-based ethanol for energy industry, the use of molasses will lead to increased production of the substituted product and additional environmental impacts. The limitation potentially leads to unfairness and risks of sub-optimised situation. This could once again be demonstrated by the molasses-based ethanol case. Throughout the continuous efforts, initiatives and investments, the sugar industry and its complex (i.e. energy industry) have successfully utilised all their products, co-products and wastes. It would be unfair to account the associated emissions from barley production (marginal feed energy) and oil and gas (marginal heat) for the use of molasses and bagasse in the ethanol production system in Thailand. Moreover, some stakeholders (i.e. energy industry) in the whole supply chain will get credits from impact reduction, whereas some (i.e. sugar industry) will be responsible for negative environmental impacts. In case the ethanol business owners considered the by-product usage as drawbacks since the associated emissions are high, they may stop using the by-products and leave them wasted. Nevertheless, such situations might not easily happen for the high value by-products which are fully utilised by nature due to their economic values. If the by-products have low value and high operational costs (i.e. the rice straw), the non-fully utilised and freely available condition could be assumed.

3.3.3 Policy recommendations on potential specific applications for CLCA and ALCA

With regard to policy recommendations, one of the main environmental assessment applications are for environmental taxation, accounting and labelling for trade, and new production development and/or eco-design. In contrast with previous studies considering retrospective and prospective applications for ALCA and CLCA, respectively, this study argues that both modelling choices could be used for investigating future impacts, since future situations could be modelled by various conditions. The future could be formed by what will additionally occur in the future under market-based causal relationships and what happened in the past on average (e.g. there may also be potentials that the likely to be constrained suppliers will be exercise flexibility so will not actually be constrained). Brandão and Weidema (2013) and De Camillis et al. (2013) also addressed that CLCA and ALCA approaches can model future scenarios. Hence, the focuses in this research for each modelling choice are rather on the main characteristics of ALCA as "specification" and "allocation/attribution" and those of CLCA as "additionality" and "expectedly actually affected processes linked via market mechanisms".

Environmental taxation is often a legal requirement, while environmental accounting and labelling for biofuel trade may be perceived as a non-tariff barrier. Consequently, a platform of specific modelling choices where all stakeholders are fairly judged is crucial for such a case. Specific and direct linkages in ALCA are more reasonable. Businesses would be reluctant to accept tax charges on the actions they have not directly done. The taxes under the market-based cause-effect relationship might be accepted by the businesses who gain the benefits for this modelling choice (i.e. the palm biodiesel using palm oil and producing as well as utilising various high value by-products). Same reasons could be applicable for environmental accounting and/or labelling for import-export, since it could lead to the barriers affecting the businesses financially. The methodology of Product Environmental Footprint (PEF)—a recent harmonised environmental assessment approach recommended by European Commission—also applies the attributional approach with some elements inspired by consequential thinking (but still handled in an attributional context) (EC 2013; Schau et al. 2013). The mixed approaches imply that decision-makers, LCA practitioners and researchers have realised the existence and importance of both methodological choices.

New production development and/or eco-design have intrinsically addressed additionality in additional production. In order to design new products aiming at environmental improvements while providing economic profits, the CLCA model which concerns the consequences of an increase in demand and covers the likely to be affected systems under market-based relationships is therefore more appropriate. Moreover, according to the case of accounting and labelling for trade, if the main target is to declare environmental information to the consumers as a support for their decisions to buy more products leading to additional production, the CLCA approach also plays an important role. In reality, a specific environmental assessment application always has multi-purposes. It will not be realistic to set a single target and to apply a single approach for actual applications. Under multi-purpose applications, it is crucial to clarify and prioritise objectives as well as to clearly justify the selected or preferred choices being made during the goal and scope definition phase. Under the analysis in this study, both modelling approaches are highly recommended to be applied for LCA studies, but preferences on a specific approach for specific applications are suggested. ALCA considering total global burdens of all products directly and attributing a specific portion of impacts to the product in question at a given point of time is recommended for taxation and labelling studies, whereas CLCA taking additionality and market-based cause-effect relationships into account is more preferred for new product development and eco-design.

4 Conclusions

Climate change mitigation potentials of biofuels in terms of GHG emission reduction from fossil energy replacement have been assessed by using the consequential and attributional modelling approaches. Under the CLCA approach considering consequences of an additional biofuel production, GHG emissions could be reduced by using palm biodiesel to replace conventional diesel, while it is unclear whether the molasses-based ethanol and the rice straw-based electricity can be used for climate change mitigation. The molasses-based ethanol production with the use of bagasse in ethanol conversion process tends to increase GHG emissions, since the consumed by-products have high value and are fully utilised. The electricity production from rice straw with low value and not fully utilised (i.e. the background situations are rice straw burning or rice straw use as fertiliser) is able to reduce GHG emissions. With the ALCA approach using average data and economic allocation, palm biodiesel yields a similar conclusion being able to reduce GHG emissions, but the other two biofuels have different performance. The GHG emissions of electricity production from rice straw are comparable to the ones from the national grid mix, whereas the molasses-based ethanol potentially contributes to climate mitigation. The different results are derived from the differences in the systems being assessed.

To ensure the consideration of possible limitations and risks prior to making decisions, the application of both environmental assessment approaches is recommended. However, when dealing with the situations under conflicting conclusions from the choices being made, the main modelling approach to be considered is crucial for an effective decision support. The specific questions, applications and limitations of each approach are therefore addressed in this research. The specific questions for the consequential and attributional approaches suggested in this work are "What would likely happen or what are the consequences if we additionally produce 1 MJ of the biofuels in Thailand?" and "What are the specific GHG emissions of 1 MJ biofuel production in Thailand allocated by economic values of all related co-products", respectively. Due to the fact that the CLCA modelling choice includes only affected processes indirectly linked via market mechanisms and excludes the direct actions (if it is not influenced by the additional demand) in the investigated systems, it could lead to the risks for unfairness and sub-optimised systems. The limited CLCA application is also derived from the fact that CLCA considers only changes or additionality and excludes total and existing production. In addition, with the market-based cause-effect modelling in CLCA, the typical market failures in reality result in the conclusions, because the assessment cannot capture the actual impacts under such conditions. The risks of the ALCA approach are unaccounted and undesirable consequences, since this model does not capture the impacts from individual actions. In environmental taxation, accounting for import-export and labelling studies, the ALCA approach may be more recommended, since it considers the inputs, outputs and associated emissions of the whole value chains directly. The indirect linkages via global market might not be acceptable for setting up legal requirements via environmental taxation or trade barriers via environmental accounting and labelling. For new product development and eco-design, the CLCA approach concerning consequences of additional demand and including the potential affected processes is more appropriate. Future studies need to consider other environmental impacts and to determine the implications and recommendations for applying different approaches on other applications and cases to support individual, national, regional and global policies.

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ELECTRONIC SUPPLEMENTARY MATERIAL

PROMOTING SUSTAINABILITY IN EMERGING ECONOMIES VIA LIFE CYCLE THINKING

Consequential and attributional environmental assessment of biofuels: Implications of modelling choices on climate change mitigation strategies

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Total GHG emissions of 1 MJ of biofuels and fossil comparators are presented in **Table S1**. Life cycle inventories of the molasses based ethanol system are presented in **Tables S2 to S4, S10 and S11**. Life cycle inventories of the palm based biodiesel system are presented in **Tables S5 to S12**. Life cycle inventories of the rice straw based electricity system are presented in **Tables S13 to S16**. Life cycle inventories of barley and soybean production systems are presented in **Tables S17 to S18**.

Table S1. Total GHG emissions of 1 MJ of biofuels and fossil comparators.

Scenarios	Total GHG emissions (g CO_2–eq.)
Molasses based ethanol system (M-ethanol) [a] and its fossil comparator	
M-ethanol_C1	184
M-ethanol_C2	-3
M-ethanol_A	35
Gasoline [b]	87
Palm oil based biodiesel system (P-biodiesel) [c] and its fossil comparator	
P-biodiesel_C1	11
P-biodiesel_C2	27
P-biodiesel_A	24
Diesel [b]	84
Rice straw based electricity system (R-electricity) [d] and their fossil comparators	
R-electricity_C1	-16
R-electricity_C2	-22
R-electricity_C3	265
R-electricity_A	168
Hard coal-electricity [e]	300
Natural gas-electricity [f]	150
Thai grid mix-electricity [g]	169

[a] Life cycle inventories of the molasses based ethanol system are presented in **Tables S2 to S4, S10 and S11**.

[b] Emission factors are from TGO (2016; 2017). The net calorific values (42.5 MJ kg^{-1} gasoline and 42.8 MJ kg^{-1} diesel) and the fuel density (0.75 kg L^{-1} gasoline and 0.84 kg L^{-1} diesel) according to Jungbluth (2007) are applied.

[c] Life cycle inventories of the palm based biodiesel system are presented in **Tables S5 to S12**.

[d] Life cycle inventories of the rice straw based electricity system are presented in **Tables S13 to S16**.

[e] Ecoinvent database.

[f] Phumpradab et al. (2009)

[g] Thai national LCI database (National Grid Mix in 2009).

Table S2. Life cycle inventory of sugarcane plantation in Thailand. The data were primarily collected from field survey in central Thailand in 2013/2014 (Silalertruksa et al. 2017) except the emission factors of cane trash burning which were obtained from Silalertruksa and Gheewala (2011).

Item	Unit	Amount
Inputs		
Fertilizer usage		
- Ammonium sulphate ($(NH_4)_2SO_4$)	kg ha^{-1} yr^{-1}	88
- Phosphate fertilizer (P_2O_5)	kg ha^{-1} yr^{-1}	88
- Potassium chloride (KCl)	kg ha^{-1} yr^{-1}	225
- Urea	kg ha^{-1} yr^{-1}	71
Chemical usage		
- Glyphosate	kg ha^{-1} yr^{-1}	8
- Paraquat	kg ha^{-1} yr^{-1}	5
- Atrazine	kg ha^{-1} yr^{-1}	6
Diesel consumption		
- Manual planting with burnt cane harvesting (65%)	L ha^{-1} yr^{-1}	117
- Mechanized farming with green cane harvesting (35%)	L ha^{-1} yr^{-1}	157
Transport by truck		
- Average distance from farm to mill	km	30
Outputs		
Cane	t ha^{-1} yr^{-1}	75
Cane trash	kg DM t^{-1} cane	140
Emissions from cane trash burning		
- CH_4 emission	kg kg^{-1} DM	0.0027
- N_2O emission	kg N_2O-N kg^{-1} DM	0.00007

Table S3. Life cycle inventory of sugar milling and molasses ethanol conversion process.

Item	Unit	Amount
Inputs		
Cane [a]	t	1
Energy usage in sugar milling process [a]		
- Steam (from bagasse combustion)	kg t^{-1} cane	479
- Electricity (from bagasse combustion)	kWh t^{-1} cane	13
Energy usage in ethanol conversion process [a]		
- Steam (from bagasse combustion)	kg t^{-1} cane	27
- Electricity (from bagasse combustion)	kWh t^{-1} cane	3
Outputs		
Raw sugar [a, b]	kg t^{-1} cane	53
Refined sugar [a, b]	kg t^{-1} cane	56
Molasses ethanol [a, b, c, d]	L t^{-1} cane	10.2
Electricity sold to the grid (from surplus bagasse) [a]	kWh t^{-1} cane	3.5
Wastewater from sugar milling process [e]		
- Wastewater generation	L t^{-1} cane	260
- COD value	mg L^{-1}	2,300-4,700
- CH_4 emission factor (Oxidation and stabilisation ponds system)	kg CH_4 t^{-1} cane	0.25
Wastewater (vinasse) from ethanol conversion process		
- Wastewater generation [f]	L vinasse L^{-1} ethanol	11
- COD value (before evaporation) [f]	mg L^{-1} vinasse	136,000
- N content [g]	mg L^{-1} vinasse	1,052
- P content [g]	mg L^{-1} vinasse	89
- K content [g]	mg L^{-1} vinasse	3,093

[a] The data were primarily collected from plants located in the Northeast Thailand in 2013/2014 (Silalertruksa et al., 2017; Gheewala et al., 2017).

[b] For attributional LCA modelling, the economic, energy and mass allocation factors for molasses (0.1, 0.24, 0.26, respectively) based on Silalertruksa et al. (2017) and Silalertruksa and Gheewala (2009; 2011) are applied.

[c] The energy content of molasses ethanol is 21.2 MJ L^{-1} ethanol (Silalertruksa and Gheewala, 2011)

[d] For consequential LCA modelling, the molasses displacement ratio is 0.764 kg barley kg^{-1} molasses. The value was estimated from molasses (70% DM) and barley (85% DM) energy content of 8.05 and 8.68 MJ kg^{-1} DM, respectively (Schmidt and Brandão, 2013). The ethanol conversion efficiency is 4.8 kg molasses L^{-1} ethanol (Silalertruksa et al., 2017). Hence, the production of 1 L ethanol will require and an input of 3.667 kg barley.

[e] Yuttitham et al. (2011)

[f] World Bank (2009)

[g] Due to the lack of data, the N-P-K nutrients contained in the wastewater were estimated by using mass balance approach based on total nutrient contents in vinasse from Silalertruksa et al. (2017) and the vinasse generation rate from World Bank (2009).

Table S4. Wastewater management in the ethanol conversion process

Item	Unit	Amount
Wastewater treatment systems [a]		
- Anaerobic digestion with biogas recovery	%	41
- Land application (as fertiliser)	%	59
Anaerobic digestion with biogas recovery [b, c]		
- COD removal efficiency	%	80
- Methane yield (55% CH_4 in biogas)	m^3 CH_4 kg^{-1} COD removed	0.35

[a] The share of wastewater treatment systems were estimated based on official data published in annual and/or sustainability reports/websites of 11 ethanol plants (using molasses as feedstock only) in Thailand with total production capacity of 2.3 million L ethanol day^{-1}

[b] World Bank (2009)

[c] Other data including emission factors are documented in **Tables S10 and S11**.

Table S5. Life cycle inventory of oil palm plantation in Thailand (Weight average data in 2011 from small and large oil palm cultivators in Krabi, Chumphon, Suratthani, and Chonburi; Suttayakul et al., 2016).

Item	Unit	Amount (per t of FFBs)
Inputs		
Seedling	seed	at least 1
Fertilizer usage		
Nitrogen fertilizer (N)	kg	4.40
Phosphorus fertilizer (P_2O_5)	kg	2.42
Potassium fertilizer (K_2O)	kg	22.4
Plant growing media		
Agriculture bag	kg	3.44E-03
Soil	kg	1.08
Coconut dust	kg	0.54
Chemical usage		
Sevin-85	kg	4.24E-04
Scaptan	kg	2.99-E03
Thiram	kg	1.02-E04
Glyphosate	kg	0.35
Paraquat	kg	0.07
Grammoxone	kg	0.04
Carbendazim	kg	7.79-E05
Mancozeb	kg	4.51-E05
Water consumption		
Freshwater	L	57.5
Transport		
Diesel fuel	L	10.1
Gasoline	L	0.42
Electricity consumption		
Electricity	kWh	0.18
Output		
Fresh fruit bunches (FFB)	t	1

Table S6. Life cycle inventory of CPO production in Thailand (Kaewmai et al., 2013). The data were collected for a period of one year (2010) from six commercial oil mills in Krabi, Chonburi, Suratthani and Chonburi. The palm oil milling processes are sterilisation, palm fruit separation, digestion, oil extraction (wet extraction process), and oil purification.

Parameter	**Unit**	**Weight Average (6 mills)**
Inputs		
Fresh fruit bunches (FFBs)	t	5.88
Water consumption in factories	m^3	4.59
Electricity consumption from grid	kWh	14.4
Diesel oil consumption	L	3.71
Chemical usage		
- Kaolin	kg	12.8
- Polyaluminium chloride	kg	0.53
- Chloride	kg	0.08
- Sodium chloride	kg	0.81
- Sodium sulphite	kg	0.05
- Magnesium	kg	0.76
- Soda ash	kg	0.10
- Phosphate	kg	0.08
Outputs		
Main product		
- CPO	t	1.00
Co-products		
- PKO	t	0.13
- PKs	t	0.32
- Fibers [a]	t	0.37
- Shells	t	0.30
- Palm kernel meal	t	0.16
Solid waste		
- EFBs [b]	t	0.96
- Decanter cake [b]	t	0.17

[a] Surplus amount of used as biomass fuel for boiler.

[b] Surplus amount of used from bio-fertilizer plant.

Table S7 Characteristics of co-products and wastes from palm biodiesel production (modified from Prapaspongsa et al., 2017).

Parameter	Unit	Palm shell [a]	Fibres [a]	Palm kernel [a]	Palm kernel meal [a]	Empty fruit bunches [a]	Decanter cake [a]	Glycerol [a]
DM	%	78.6 [b]	62.8 [b]	95.3 [c]	92.1 [c]	42.8 [b]	23.6 [d]	40 [e]
N	% of DM	1.01 [b]	1.12 [b]	1.61 [c]	2.86 [c]	0.45 [b]	2.38 [d]	0 [e]
P	% of DM	-	-	-	-	-	0.17 [d]	0 [e]
C	% of DM	52.4 [b]	46.92 [b]	67.49 [c]	52.53 [c]	45.53 [b]	51.70 [d]	39.1 [e]
VS	% of DM	-	-	86.50 [c]	71.84 [c]	-	-	78 [e]
Methane yield	m^3 kg^{-1} VS	-	-	-	-	-	-	0.75 [f, h]
Theoretical methane yield	m^3 kg^{-1} total	-	-	-	-	-	-	0.43 [g, h]
Energy content	MJ kg^{-1}	18.46 [i]	11.8 [i]	17 [i]	-	17.86 [i]	-	18.05 [i]

[a] For consequential LCA modelling, palm shell, palm fibres and palm kernel will displace marginal electricity production in Thailand with the ratio of 2.05, 1.31, and 1.9 kWh kg^{-1}, respectively based on their energy contents and the electricity conversion efficiency of 40%. The supply of 1 t palm kernel meal will displace 0.34 t soybean meal (marginal feed protein) and require additional production of 0.083 t palm oil (marginal oil) and 43 kg barley (marginal feed energy) based on their energy and protein contents. One tonne of glycerol will displace 1.77 kg barley (marginal feed energy) based on their energy contents.

[b] Uemura et al. (2011)

[c] Razuan et al. (2010)

[d] Yahya et al. (2010)

[e] Prapaspongsa et al. (2010)

[f] The methane yield was measured in batch experiments at 38 to 40 ºC by Amon et al., (2006). The mesophilic condition is also common in biogas plants in Thailand.

[g] The theoretical methane yield from Viana et al. (2012)

[h] Methane density is 0.7167 kg m^{-3} at standard conditions (0˚C, 1 atm) (Crittenden et al., 2012) and 0.6777 kg m^{-3} at 20˚C, atmospheric pressure.

[i] Pleanjai and Gheewala (2009); Silalertruksa and Gheewala (2012).

Table S8 Wastewater (Palm Oil Mill Effluent; POME) generation rate and characteristics of raw and treated wastewater with and without biogas capture (Prapaspongsa et al., 2017).

Parameter	Value	Unit
Wastewater generation rate[a]	2.426	m^3 t^{-1} CPO
Wastewater characteristics		
(1) COD [a] (*100% biogas capture*)		
: COD of influent into an open pond for cooling	93,044	mg L^{-1}
: COD of influent into a biogas system	73,027	mg L^{-1}
: COD of effluent from a biogas system	16,085	mg L^{-1}
: COD of effluent from a stabilisation pond	4,694	mg L^{-1}
(2) COD [a] (*Without biogas capture*)		
: COD of influent into anaerobic ponds without biogas capture	93,044	mg L^{-1}
: COD of effluent from final system	4,694	mg L^{-1}
(3) Nutrients[b]		
: Total Nitrogen	880	mg L^{-1}
: Total Phosphorus	111	mg L^{-1}

[a] Modified from Kaewmai et al. (2013).

[b] O-Thong et al. (2008). Raw POME was collected from the receiving tank of Trang Palm Oil Co., Ltd in Trang province, Thailand.

Table S9. N-related emissions from non-fully utilised co-products (palm shell, fibres, and palm kernel) and solid waste (empty fruit bunches and decanter cake) at the oil milling stage. 50% of the co-products which are not commercially utilised (for energy, animal feed, and oil substitution) and 100% of solid waste are used as organic fertilisers and soil conditioners during oil palm plantation (Prapaspongsa et al., 2017).

Parameter	Unit	Palm shell	Fibres	Palm kernel	Palm kernel meal	Total (Co-products)	Empty fruit bunches	Decanter cake	Total (Waste)
The amount applied to soils	t t^{-1} CPO	0.15	0.19	0.16	0.08		0.96	0.17	
N content of the co-products [a]	kg N kg^{-1} total	0.008	0.007	0.015	0.026		0.002	0.006	
Total N applied to soils [b]	kg N t^{-1} CPO	1.19	1.30	2.45	2.11	***7.05***	1.85	0.95	***2.80***
N fertiliser substitution [c]	kg N t^{-1} CPO	0.91	0.99	1.87	1.60	***5.36***	1.41	0.73	***2.13***
NH_3 emissions [d]	kg NH_3-N t^{-1} CPO	0.12	0.13	0.25	0.21	***0.71***	0.18	0.10	***0.28***
NO_x emissions [d]	Kg NOx-N t^{-1} CPO	0.12	0.13	0.25	0.21	***0.71***	0.18	0.10	***0.28***
Direct N_2O emissions [d]	kg N_2O-N t^{-1} CPO	0.01	0.01	0.02	0.02	***0.07***	0.02	0.01	***0.03***
Indirect N_2O emissions [d]	kg N_2O-N t^{-1} CPO	0.002	0.003	0.005	0.004	***0.014***	0.004	0.002	***0.006***
N_2 emissions [d]	kg N_2-N t^{-1} CPO	0.04	0.04	0.07	0.06	***0.21***	0.055	0.029	***0.08***

[a] N amounts are calculated from the characteristics of co-products in **Table S7**.

[b] Refer to the N amount in the unused co-products (50% of total co-products) and 100% solid waste which are applied to soils as organic fertilisers.

[c] The remaining N content will substitute synthetic N fertiliser production. Due to the lack of data, the P content of co-products is not considered (See **Table S7**).

[d] Apply the N-related emission factors in **Table S10**.

Table S10. N-related emission factors for unused co-products, POME and wastewater from molasses ethanol production (Prapaspongsa et al., 2017).

	Value	Unit
NH_3 emission factor [a]	0.1	kg NH_3-N kg^{-1} N applied
NO_x emission factor [a]	0.1	kg NO_x-N kg^{-1} N applied
Direct N_2O emission factor [b]	0.01	kg N_2O-N kg^{-1} N applied
Indirect N_2O emission factor [c]	0.002	kg N_2O-N kg^{-1} N applied
N_2 emission factor [d]	0.03	kg N_2-N kg^{-1} N applied

[a] IPCC (2006) Table 11.3. Volatilisation from all organic N fertilisers applied. Equal distribution between NH_3 and NO_x emissions is assumed.

[b] IPCC (2006) Table 11.1. Direct N_2O emissions refer to the amount of N_2O emitted from the various synthetic and organic N applications to soils, including crop residues.

[c] IPCC (2006) Table 11.3. In this study, indirect N_2O emissions include only atmospheric deposition of N volatilised from managed soils due to organic N additions applied to soils. It is estimated from the emission factors of [1] 0.20 kg (NH_3-N + NO_x-N) kg^{-1} N applied and [2] 0.01 kg N_2O-N kg^{-1} (NH_3-N + NO_x-N).

[d] The values are derived from the presented calculation method as 3xN_2O-N (direct emissions) based on Payraudeau et al. (2007).

Table S11. Equations for estimating CH_4 emissions from POME and wastewater treatment systems (Prapaspongsa et al., 2017).

	Description
Methane emissions from the baseline wastewater treatment systems (UNFCC, 2013). (*Note: the baseline wastewater treatment systems are referred to all treatment systems without biogas recovery in this study*)	$BE_{ww,treatment} = \sum_i \left(Q_{ww,i} \times COD_{inflow,i} \times \eta_{COD,BL,i} \times MCF_{ww,treatment,BL,i}\right) \times B_{o,ww} \times UF_{BL} \times GWP_{CH4}$ *Where:* $Q_{ww,i,y}$ = Volume of wastewater treated in baseline wastewater treatment system i (m^3). $COD_{inflow,i,y}$ = Chemical oxygen demand of the wastewater inflow to the baseline treatment system i (t m^{-3}). $\eta_{COD,BL,i}$ = COD removal efficiency of the baseline treatment system i. $MCF_{ww,treatment,BL,i}$ = Methane correction factor for baseline wastewater treatment systems i. i = Index for baseline wastewater treatment system $B_{o,ww}$ = Methane producing capacity of the wastewater (IPCC value of 0.25 kg CH_4 kg^{-1} COD) UF_{BL} = Model correction factor to account for model uncertainties (0.89) GWP_{CH4} = Global Warming Potential for methane
Fugitive emissions through capture inefficiencies in the anaerobic wastewater treatment systems (t CO_{2e}) (UNFCC, 2013).	$PE_{fugitive,ww} = (1 - CFE_{ww}) \times MEP_{ww,treatment} \times GWP_{CH4}$ *Where:* CFE_{ww} = Capture efficiency of the biogas recovery equipment in the wastewater treatment systems (a default value of 0.9 shall be used) $MEP_{ww,treatment}$ = Methane emission potential of wastewater treatment systems equipped with biogas recovery system (t)
Methane emission potential of wastewater treatment systems equipped with biogas recovery system (UNFCC, 2013)	$MEP_{ww,treatment} = Q_{ww} \times B_{o,ww} \times UF_{PJ} \times \sum_k COD_{removed,PJ,k} \times MCF_{ww,treatment,PJ,k}$ *Where:* $B_{o,ww}$ = Methane producing capacity of the wastewater (IPCC value of 0.25 kg CH_4 kg^{-1} COD) $COD_{removed,PJ,k}$ = The chemical oxygen demand removed by the treatment system k of the project activity equipped with biogas recovery (t m^{-3}). $MCF_{ww,treatment,PJ,k}$ = Methane correction factor for the project wastewater treatment system k equipped with biogas recovery equipment UF_{PJ} = Model correction factor to account for model uncertainties (1.12)

Table S12. CH_4 emission factor for unused and untreated glycerol (industrial wastewater).

	Description
Equation from IPCC (2006)	$EF_j = B_0 \times MCF_j$ EF_j = CH_4 emission factor for each treatment/ discharge pathway or system, kg CH_4 kg^{-1} COD j = Each treatment/discharge pathway or system B_0 = Maximum producing capacity, kg CH_4 kg^{-1} COD MCF_j = Methane correction factor (fraction)
CH_4 emission factor (this study) [a]	$EF_{glycerol, untreated}$ = 0.29 x 0.1 = 0.029 kg CH_4 kg^{-1} glycerol *Where:* a) B_0 = Theoretical methane yield (kg CH_4 kg^{-1} Glycerol) (See **Table S10**) = 0.43 m^3 CH_4 kg^{-1} glycerol x 0.6777 kg CH_4 m^{-3} CH_4 = 0.29 kg CH_4 kg^{-1} glycerol b) MCF_j = 0.1 (Untreated industrial wastewater) [b]
CH_4 emission factor (based on IPCC, 2006)	$EF_{glycerol, untreated}$ = 0.25 x 0.1 = 0.025 kg CH_4 kg^{-1} COD = 0.025 kg CH_4 kg^{-1} COD x 1 kg COD kg^{-1} Glycerol = 0.025 kg CH_4 kg^{-1} glycerol *Where:* a) B_0 = 0.25 kg CH_4 kg^{-1} COD (the IPCC COD-Default factor) b) MCF_j = 0.1 (Untreated industrial wastewater) [b] c) Density of glycerol[c] = 1,250 kg Glycerol m^{-3} d) COD value of glycerol[c] = 1,250 kg COD m^{-3} = 1,250 kg COD m^{-3} / 1,250 kg Glycerol m^{-3} = 1 kg COD kg^{-1} Glycerol

[a] CH_4 emission factor being used in this study considers theoretical methane yield as maximum methane producing capacity. Using this specific value for glycerol is more accurate than using the IPCC COD-Default factor (0.25 kg CH_4 kg^{-1} COD) for all types of wastewater. Moreover, the value is also comparable with the one estimated from the COD value of glycerol.

[b] IPCC (2006) Table 6.8. Untreated wastewater: Sea, river and lake discharge.

[c] Viana et al. (2012)

Table S13. Life cycle inventory of rice cultivation in Thailand (Silalertruksa and Gheewala, 2013).

Item	Unit	Amount (per ha)	Data sources
Inputs			
Seed rate	kg ha^{-1}	37.25	ACFS (2008)
Diesel fuel used for seedling	L ha^{-1}	42	ACFS (2008)
Diesel fuel used for other processes	L ha^{-1}	78	Suramaythangkoor and Gheewala (2008)
Fertilizer usage			
- Nitrogen fertilizer	kg ha^{-1}	25	ACFS (2008)
- Phosphorus fertilizer	kg ha^{-1}	25	ACFS (2008)
- Potassium fertilizer	kg ha^{-1}	12.5	ACFS (2008)
- Urea	kg ha^{-1}	130	ACFS (2008)
- Manure	kg ha^{-1}	3,750	ACFS (2008)
Outputs			
Rice grain	t ha^{-1}	3.66	OAE (2012)
Rice straw [a]	t ha^{-1}	2.745	Estimated by using straw-to-grain ratio of 0.75 from Gadde et al. (2009)
Emissions from rice field			
- CH_4 emission	kg	240	Towprayoon et al. (2005)
- N_2O emission	kg	0.33	Towprayoon et al. (2005)
Emissions from rice straw burning			
- GHG emissions	kg CO_2 eq. t^{-1} DM	92	Silalertruksa and Gheewala (2013)

[a] For R-electricity_C1 (the rice straw was previously burned in the field), GHG emissions in this table are applied. For R-electricity_C2 (the rice straw was previously chopped and used as fertiliser in the field), life cycle inventory data of rice straw based fertiliser in **Table S16** are applied. For R-electricity_C3 (the rice straw was previously used as an animal feed), the displacement ratio was estimated by using the methodology in Schmidt (2007) and Schmidt and Weidema (2008), the straw characteristics in **Table S16** and animal properties in Schmidt and Brandão (2013); that the demand of 1 t of dry rice straw will need additional production of 0.25 t soybean meal (marginal feed protein) and substitute the production of 0.02075 t palm oil (marginal oil) and 79 kg barley (marginal feed energy).

Table S14. Life cycle inventory of straw baling and transport (Silalertruksa and Gheewala, 2013)

Item	Unit	Amount (per ton straw)	Data sources
Inputs			
Diesel fuel used: baling	L	1.2	Delivand et al. (2011)
Truck 15 ton and distance round trip	km	90	Suramaythangkoor and Gheewala (2008)

Table S15. Life cycle inventory of rice straw based electricity production (Silalertruksa and Gheewala, 2013)

Item	Unit	Amount	Data sources
Inputs			
Rice straw based electricity			
Rice straw power plant capacity	MWe	10	Delivand et al. (2011)
Rice straw feedstock used	t yr^{-1}	116,000	Delivand et al. (2011)
Diesel fuel used		8	Delivand et al. (2011)
Outputs			
Electricity	MWh yr^{-1}	65,700	Delivand et al. (2011)
Electricity sold to the grid	kWh t^{-1} straw	613	Delivand et al. (2011)
Emissions from co-generation system			
CH_4 emission	kg t^{-1} dry straw	0.006	Delivand et al. (2012)
N_2O emission	kg t^{-1} dry straw	0.0173	Delivand et al. (2012)
CO emission	kg t^{-1} dry straw	3.72	USEPA (1993)
NO_x emission	kg t^{-1} dry straw	0.544	USEPA (1993)
SO_2 emission	kg t^{-1} dry straw	0.036	USEPA (1993)
Particulate	kg t^{-1} dry straw	0.403	USEPA (1993)

Table S16. Life cycle inventory of rice straw based fertiliser (Silalertruksa and Gheewala, 2013)

Item	Unit	Amount	Data sources
Inputs			
Organic material and nutrients:			
C	kg t^{-1} rice straw	383	Bhattacharyya et al. (2012)
N	kg t^{-1} rice straw	6	Bhattacharyya et al. (2012)
P	kg t^{-1} rice straw	1	Bhattacharyya et al. (2012)
K	kg t^{-1} rice straw	19	Bhattacharyya et al. (2012)
Rice straw	t ha^{-1}	5	Bhattacharyya et al. (2012)
Manure	t ha^{-1}	1.2	Bhattacharyya et al. (2012)
Diesel fuel required for chopping rice straw	L ha^{-1}	78	
Green manure used	t t^{-1} incorporated straw	0.24	Bhattacharyya et al. (2012)
Diesel fuel used	L t^{-1} incorporated straw	16	Blank et al. (1993)
Outputs			
CH_4 emission due to organic amendments	kg t^{-1} incorporated straw	43	Bhattacharyya et al. (2012)

Table S17. Life cycle inventory of barley production in Spain.

Parameter	Unit	Amount	Data sources
Inputs			
Nitrogen fertiliser	kg ha^{-1} yr^{-1}	56.4	Lechón (2011) and Lechón et al. (2011)
Phosphorus fertiliser	kg ha^{-1} yr^{-1}	56.4	Lechón (2011) and Lechón et al. (2011)
Potassium fertiliser	kg ha^{-1} yr^{-1}	56.4	Lechón (2011) and Lechón et al. (2011)
Pesticides	kg ha^{-1} yr^{-1}	0.509	Dalgaard and Schmidt (2012)[a]
Transport, lorry	tkm	83.1	Dalgaard and Schmidt (2012)[a]
Diesel	MJ ha^{-1} yr^{-1}	3,046	Dalgaard and Schmidt (2012)[a]
Light fuel oil	MJ ha^{-1} yr^{-1}	1.1	Dalgaard and Schmidt (2012)[a]
Outputs			
Barley	t ha^{-1} yr^{-1}	2.676	Lechón (2011) and Lechón et al. (2011)
Emissions			
- Dinitrogen monoxide	kg ha^{-1} yr^{-1}	0.0676	Lechón (2011) and Lechón et al. (2011)
- Ammonia	kg ha^{-1} yr^{-1}	1.4061	Lechón (2011) and Lechón et al. (2011)

[a] European average for barley production

Table S18. Life cycle inventory of soybean production in Brazil (Schmidt, 2007; 2015; Schmidt and Brandão, 2013).

Parameter	Unit	Amount
Inputs		
Seed	kg ha^{-1} yr^{-1}	138
P fertiliser (as P_2O_5)	kg ha^{-1} yr^{-1}	46
Herbicide, glyphosate	kg ha^{-1} yr^{-1}	2.3
Insecticide, pyrethroid, cypermethrin	kg ha^{-1} yr^{-1}	0.013
Insecticide, chlorpyrifos	kg ha^{-1} yr^{-1}	0.20
Transport, lorry 40t	tkm	1,701
Transport, transoceanic tanker	tkm	811
Traction, burned diesel	MJ ha^{-1} yr^{-1}	1,911
Grain drying	kg ha^{-1} yr^{-1}	68
Outputs		
Soybean	t ha^{-1} yr^{-1}	3.212
Emissions to air, water and soil		
- Dinitrogen monoxide (air)	kg ha^{-1} yr^{-1}	4.7
- Nitric oxide (air)	kg ha^{-1} yr^{-1}	1.2
- Phosphorus (water)	kg ha^{-1} yr^{-1}	0.069
- Glyphosate (air/water/soil)	kg ha^{-1} yr^{-1}	0.77/0.77/0.77
- Cypermethrin (air/water/soil)	kg ha^{-1} yr^{-1}	0.0043/0.0043/0.0043
- Chlorpyrifos (air/water/soil)	kg ha^{-1} yr^{-1}	0.067/0.067/0.067
- Cadmium (water/soil)	g ha^{-1} yr^{-1}	0.022/1.2
- Chromium (water/soil)	g ha^{-1} yr^{-1}	0.19/28
- Cobalt (water/soil)	g ha^{-1} yr^{-1}	0.029/-0.016
- Copper (water/soil)	g ha^{-1} yr^{-1}	5.7/1.5
- Mercury (water/soil)	g ha^{-1} yr^{-1}	0/0.00046
- Nickel (water/soil)	g ha^{-1} yr^{-1}	1.9/0.58
- Lead (water/soil)	g ha^{-1} yr^{-1}	0.026/0.53
- Selenium (water/soil)	g ha^{-1} yr^{-1}	0.036/0.044
- Zink (water/soil)	g ha^{-1} yr^{-1}	17/9.6

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Appendix B Research Utilization and Application

This Appendix describes research utilization and application of this project. It is mainly for educational and academic purposes. The guideline and case studies from this project have been embedded and used as the teaching material in the course "*EGEW 539 Life Cycle Assessment*" of International Master and PhD programs in Environmental and Water Resources Engineering in Department of Civil and Environmental Engineering, Faculty of Engineering, Mahidol University since the academic year 2016. Furthermore, the contents in this guideline were used in 5 seminars/guest lectures/workshop as listed below.

- **31 July 2014** [National Metal and Materials Technology Center (MTEC), National Science and Technology Development Agency (NSTDA)]: Seminar entitled "Consequential LCA" for researchers and engineers in MTEC, NSTDA.
- **21 April 2015** and **3 May 2016** [Kasetsart University]: Guest lecture entitled "Consequential LCA" as a part of the course "202563 Cleaner Technology and EcoDesign" for graudate students in Thai and interational graduate programs in Department of Chemical Engineering, Faculty of Engineering, Kasetsart University.
- **10-16 February 2017** [Chiang Mai University]: Workshop for developing and demonstrating CLCA guideline entitiled as "Development and Application of Consequential Life Cycle Assessment Method for Food and Fuel in Thailand and Asia" at Consequential LCA Workshop at Center of Excellence on Energy, Economic & Ecological Management Science and Technology, Chiang Mai University
- **10 March 2017** [MTEC, NSTDA]: Seminar entitled "Consequential LCA" for researchers and engineers in MTEC, NSTDA as well as for lecturers and students from different universities.

Appendix C Others

This Appendix presents other publications from this project as listed below.

- *Prapaspongsa, T., Gheewala S. H., 2015. Policy implications from consequential LCA applications for future food and fuel policies: Gainers and losers in a world with constrained materials. Proceedings of the 7th International Conference on Life Cycle Management (LCM 2015), 30 August - 2 September 2015, Bordeaux, France.*
- *Prapaspongsa, T., Gheewala S. H., 2015. System expansion and allocation in life cycle assessment: Implications of modelling choices for biofuel on climate change mitigation. Proceedings of the 5th International Conference on Green and Sustainable Innovation (ICGSI 2015), 8-10 November 2015, Pattaya, Thailand*.
- *Adhikari, B., Prapaspongsa, T., 2018. Life Cycle Assessment of Food Consumption in Asia towards Sustainable Consumption and Production. Submitted to the 13th Biennial International Conference on EcoBalance (EcoBalance 2018), 9-12 October 2018, Tokyo, Japan.*
- *Adhikari, B., Prapaspongsa, T., 2018. Environmental Sustainability of Food Consumption in Asia. Supplementary Information for the Draft Manuscript (**Chapter 3**). To be submitted to Journal of Cleaner Production.*

Apart from the conference proceedings and the supplementary information, due to the expertise gained throughout this project, the researcher was invited as a guest editor in International Journal Life Cycle Assessment of a Special Issue after the international conference - the 5th International Conference on Green and Sustainable Innovation (ICGSI 2015). The Preface of this special issue lised below which the researcher co-authors with the mentor who is the first and corresponding author for this publication is also documented in this Appendix.

- *Gheewala, S.H., Silalertruksa, T., Malakul, P., Prapaspongsa, T., 2017. Preface. International Journal of Life Cycle Assessment. 22, 1641 – 1643.*

Policy implications from consequential LCA applications for future food and fuel policies: Gainers and losers in a world with constrained materials

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Abstract

Life cycle assessment (LCA) has been applied for supporting decisions towards sustainable consumption and production at product, policy, national and international levels. Two LCA modelling approaches widely applied are consequential LCA (CLCA) and attributional LCA (ALCA). CLCA aims to assess the environmental consequences caused by the decisions with the use of marginal data whereas ALCA aims to assess the environmental impacts of an existing product system with the use of average or supplier-specific data. In this study, the main objective is to identify major ethical implications of using CLCA to support future food and fuel policies. CLCA is chosen since the policies in question focus on the impacts from the increase in food and fuel demand. Previous studies have addressed ethical limitations of both CLCA and ALCA but solutions on how to overcome the constraints remain in question. A case on bagasse utilization for energy production in bioethanol production plants in Thailand is demonstrated. According to annual balances of production and consumption volumes for the past ten years, bagasse has clearly been fully utilized in Thailand for energy production. The bagasse used for steam production in bioethanol plants are generally supplied by sugar mills. If the LCA is done for bioethanol industry, the increased use of constrained bagasse in the bioethanol system (loser) will affect marginal heat production (i.e. natural gas and fuel oil). If the LCA is done for sugar industry (gainer), the increased sugar production will provide additional bagasse to substitute the marginal heat production. The ethical limitations could be on the risks of unfair LCA results among the ones with good actions (i.e. bioethanol and sugar industries) as well as in comparison with the ones with business-as-usual actions (i.e. bioethanol plants using natural gas and fuel oil). In case that bioethanol plants cannot gain environmental benefits from using the constrained bagasse, it may lead to sub-optimized situation with the direct additional use of conventional heat sources on-site. It is recommended that to support decisions for food and fuel policies it is crucial to present the potential consequences of the additional demand but the share of responsibilities among different stakeholders needs to be specifically discussed and agreed upon. Future studies on how to fairly allocate the environmental consequences could play an important role to overcome these ethical limitations.

System expansion and allocation in life cycle assessment: Implications of modelling choices for biofuel on climate change mitigation

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ABSTRACT

The study assesses climate change mitigation potentials when using biomass-based fuels to replace fossil energy under various choices in co-product handling (system expansion and allocation). With different modelling choices, the same biomass-based fuels might mitigate or contribute to climate change potentials when comparing with fossil energy.

Introduction

Life cycle assessment (LCA) has been widely used as a tool to support decisions for climate change mitigation and sustainable consumption and production. Nonetheless, LCA modelling choices have significant influence on conclusions. To illustrate the consequences of important modelling choices in LCA studies, this research evaluates climate change mitigation potentials from the replacement of fossil energy by biomass-based fuels under various modelling choices in co-product handling including system expansion (typically used in consequential LCA[1]) and system allocation (often applied in attributional LCA[1]). The important biomass-based fuels in Thailand including molasses-based ethanol, palm-based biodiesel and rice straw (being used for direct combustion and electricity generation) are chosen for illustration.

Methods

The modelling framework for system expansion and allocation based on literature[1] is applied in this work. The assessed functional unit is 1 MJ of energy from biomass-based fuels. The fossil fuel comparators are gasoline (for molasses-based ethanol), diesel (for palm-based biodiesel) and coal and gas (for rice straw). The boundaries of all systems excluding fossil energy are illustrated in Fig.1. The substituted and substituting product systems are modelled under the global and national markets depending on the market delimitation of each product.

Results and Discussion

The climate change mitigation potentials when using system expansion and co-product allocation of all systems are significantly different because the affected product systems being included in the analysis are not the same. To assess climate change potentials of the co-products, the boxes b, d, h1-3 in Fig.1 are the relevant systems under system expansion whereas the boxes a, e, g in Fig.1 are the investigated systems for allocation. The molasses-based bioethanol under system expansion yields higher climate change potential than the fossil energy while the one with co-product allocation has an opposite result. When using both modelling choices, the palm biodiesel can reduce the climate change impacts from the fossil comparator but the climate change potential is lower if

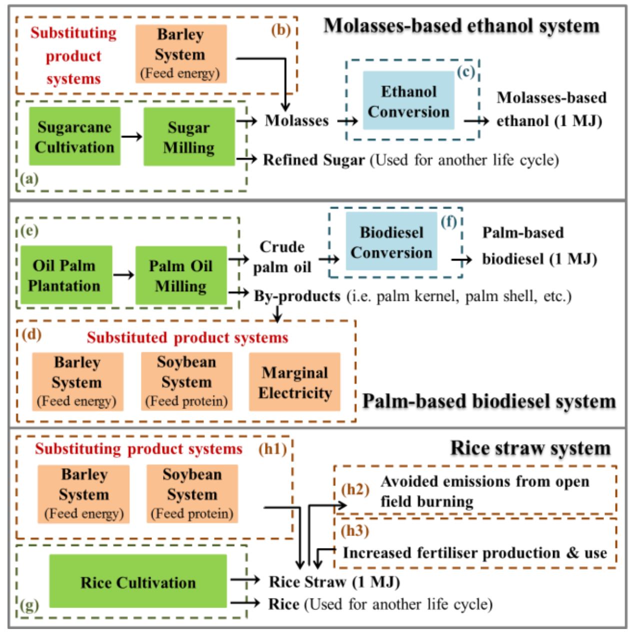

Fig. 1 The palm-based biodiesel, molasses-based ethanol and rice straw systems. The illustrated systems are simplified and only important affected products/systems are presented. The use phase is included in the analysis but omitted from this figure.

considering the affect expanded product systems. The climate change potentials of 1 MJ rice straw range from higher to lower than the fossil energy depending on the substitution pathways in system expansion (h1-3, Fig.1) and the fossil energy sources. For the context of decision making, the system expansion approach is recommended to assess potential risks of climate change potentials; and the co-product allocation with attributed emissions is suggested if the aim is to account the emissions for national environmental taxation and international trade.

Conclusion

This research clearly illustrates how certain modelling choices affect the climate change mitigation potentials of biomass-based fuels in comparison with fossil energy. System expansion is recommended to be used for assessing the potential climate change risks while allocation is more suitable for national environmental taxation and emission accounting for import-export.

Acknowledgments

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Life Cycle Assessment of Food Consumption in Asia towards Sustainable Consumption and Production

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The growing food demand of the increasing human population presents a major challenge in global sustainability, which enforces the need to comprehensively analyse the environmental impacts of food systems and determine how to mitigate them. As such, this study aims to quantify the life cycle environmental impacts of food consumption in five Asian Countries by using the framework from ISO 14044. Based on surface area, population density and GDP per capita, the countries chosen were Thailand, India, China, Japan and Saudi Arabia. Food consumption data of each country in 2013 were gathered from FAOSTAT food balance sheets. Background data were obtained from peer-reviewed scientific publications, Ecoinvent 3 and Agri-footprint databases. The functional unit used in the study is kilograms of food consumption per capita per year. The ReCiPe2006 v1.1 method was used for impact assessment. The assessment showed that the major contributors to most impact categories were Cereals, Meat, Animal Products and Alcoholic Beverages. The food consumption in China yielded the highest impacts for most of the impact categories being considered (global warming, terrestrial acidification, eutrophication, and eco-toxicity), followed by the consumption in Japan, Saudi Arabia, Thailand and India. However, the impact on land use was the highest in Saudi Arabia owed to the country’s high cumulative consumption of cereals, meat and animal products – the major contributors for impacts on land use. Although the total food consumption per capita in Japan was less than Thailand, most of Japan’s impact categories exceeded that of Thailand’s due to its higher consumption of meat and animal products. Building on the insights of this study, more accurate results could be achieved by the acquisition of country-specific data on food production systems. Linking each food item with energy and nutrient content could further help identify healthier diets and provide recommendations for sustainable consumption.

Supplementary Information

Environmental Sustainability of Food Consumption in Asia

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Table S1: Data of per capita food consumption for each country and the dataset used

SN		Item/Type	Per Capita Consumption (kg/person-yr)					Dataset Used
			Thailand	India	China	Japan	Saudi Arabia	
TOTAL			**572**	**477**	**937**	**601**	**594**	
	1	**Cereals**	**133**	**141**	**157**	**115**	**148**	
a.		Wheat and Products	10.76	62.79	67.54	45.60	89.47	Wheat grain {GLO}| market for | APOS, U
b.		Rice (Milled Equivalent) and other cereals[1]	112.66	68.03	80.86	59.70	34.70	Rice {GLO}| market for | APOS, U
c.		Barley and Products	0.00	0.65	0.17	0.54	0.00	Barley grain {GLO}| market for | APOS, U
d.		Maize and Products	9.99	6.16	7.09	9.19	20.00	Maize grain {GLO}| market for | APOS, U
e.		Rye and Products	0.00	0.00	0.09	0.01	0.00	Rye grain {GLO}| market for | APOS, U
f.		Oats	0.01	0.04	0.16	0.00	0.60	Oat grain {GLO}| market for | APOS, U
g.		Sorghum and Products	0.00	3.50	1.37	0.00	3.47	Sweet sorghum stem {GLO}| market for | APOS, U
	2	**Root Vegetables**	**23**	**30**	**71**	**31**	**23**	
a.		Cassava and products	12.78	5.42	1.96	0.09	0.00	Cassava, at farm/TH Economic
b.		Potatoes and other root vegetables[2]	10.05	24.72	68.59	30.63	22.74	Potato {GLO}| market for | APOS, U
	3	**Legumes, nuts and oil-seeds**	**17**	**23**	**13**	**12**	**11**	
a.		Beans, pulses and other products[3]	3.32	12.91	1.14	1.80	5.78	Broad bean, at farm/DE Economic
b.		Peas	0.13	1.27	0.63	0.12	0.33	Pea, meal, at plant/RER Economic
c.		Nuts and products	0.90	1.53	2.78	1.88	2.64	Almond {GLO}| market for almond | APOS, U
d.		Soybeans	2.10	0.38	4.04	7.32	0.03	Soybean {GLO}| market for | APOS, U
e.		Ground-nuts	0.57	0.35	3.98	0.71	0.43	Groundnuts, seed, with shell, at farm/CN Economic
f.		Sunflower seed	0.00	0.00	0.00	0.00	0.13	Sunflower seed {GLO}| market for | APOS, U
g.		Rapeseed and Mustard seed	0.03	0.59	0.00	0.09	0.07	Rape seed {GLO}| market for | APOS, U
h.		Coconuts	10.08	6.24	0.36	0.09	0.77	Coconut, dehusked {GLO}| market for coconut, dehusked | APOS, U
i.		Olives	0.01	0.00	0.00	0.03	0.57	Olive {GLO}| market for olive | APOS, U
	4	**Oils**	**7**	**8**	**8**	**15**	**18**	

[1] Other Cereals include Millet, Buckwheat, Canary Seed, Bran, etc.

[2] Other root vegetables include Sweet Potatoes, Yams and other tubers.

[3] Pulses and other products include Broad beans, horse beans, Chick peas, Cow peas, Lentils, Pigeon peas, Bambara beans, Vetches, Lupins, Pulses, other Oilcrops and Sesame seeds.

SN		Item/Type	Per Capita Consumption (kg/person-yr)					Dataset Used
			Thailand	India	China	Japan	Saudi Arabia	
a.		Soybean Oil and others[4]	3.32	1.86	2.32	3.63	1.90	Soybean oil, crude {GLO}| market for | APOS, U
b.		Groundnut Oil	0.35	0.97	0.65	0.01	0.03	Crude peanut oil, from crushing at plant/AR Economic
c.		Sunflowerseed Oil	0.28	0.90	0.16	0.14	1.74	Crude sunflower oil, from crushing (solvent), at plant/CN Economic
d.		Rape and Mustard Oil	0.00	1.79	1.55	8.34	0.10	Rape oil, crude {CH}| market for | APOS, U
e.		Cottonseed Oil	0.01	0.81	0.58	0.05	0.43	Cottonseed oil, crude {GLO}| market for | APOS, U
f.		PalmKernel Oil	0.07	0.00	0.07	0.63	0.20	Palm kernel oil, crude {GLO}| market for | APOS, U
g.		Palm Oil	2.74	1.28	1.97	0.86	11.82	Palm oil, crude {GLO}| market for | APOS, U
h.		Coconut Oil	0.43	0.32	0.10	0.27	0.07	Coconut oil, crude {PH}| production | APOS, U
i.		Rice bran Oil	0.23	0.44	0.08	0.53	0.00	Crude rice bran oil, from rice bran oil production, at plant/CN Economic
j.		Maize Germ Oil	0.00	0.00	0.20	0.66	1.60	Crude maize germ oil, from wet milling (germ oil production, pressing), at plant/DE Economic
	5	**Vegetables**[5]	**51**	**87**	**363**	**102**	**101**	
a.		Tomatoes and Products	8.26	22.80	37.92	13.34	58.27	Tomato, fresh grade {GLO}| market for tomato, fresh grade | APOS, U
b.		Onions[6]	4.81	23.76	17.04	13.56	25.55	Onion {CN}| onion production | APOS, U
c.		Aubergine	7.08	14.43	44.23	5.79	3.81	Aubergine {GLO}| market for | APOS, U
d.		Cabbage/lettuce/cauliflower/spinach	12.39	14.86	202.94	57.99	12.49	Cabbage white {GLO}| market for | APOS, U
e.		Carrots	18.14	10.99	61.01	11.37	1.34	Carrot {CN}| carrot production | APOS, U
	6	**Fruits**[7]	**101**	**55**	**98**	**53**	**88**	
a.		Oranges, Mandarins	23.39	8.71	25.92	14.85	12.08	Mandarin {GLO}| market for mandarin | APOS, U; Orange, processing grade {GLO}| market for orange, processing grade | APOS, U
b.		Lemons, Limes and Products[8]	13.29	4.72	11.49	5.26	5.61	Lemon {GLO}| market for lemon | APOS, U
c.		Bananas and Plantains	33.62	33.06	12.41	7.86	11.47	Banana {GLO}| market for | APOS, U

[4] Others include Sesame-seed oil, Olive oil and oil derived from other sources.

[5] The category "Vegetables, others" in the FAO food balance sheets has been proportionally distributed to the five vegetable items.

[6] This category also includes Pepper and other spices.

[7] The category "Fruits, others" in the FAO food balance sheets has been proportionally distributed to the seven fruit items.

[8] This category also includes grapefruit and other citrus fruits.

SN		Item/Type	Per Capita Consumption (kg/person-yr)					Dataset Used
			Thailand	India	China	Japan	Saudi Arabia	
d.		Apples and products	3.06	2.85	35.36	19.97	11.66	Apple {GLO}| market for | APOS, U
e.		Pineapples and Products	22.29	2.36	2.06	2.24	2.43	Pineapple {GLO}| market for | APOS, U
f.		Dates	0.00	0.47	0.19	0.01	38.14	Palm date {GLO}| market for palm date | APOS, U
g.		Grapes and Products (excl. wine)	5.75	3.00	10.90	2.52	6.76	Grape {GLO}| market for | APOS, U
	7	**Coffee and Tea**	**2**	**1**	**1**	**6**	**6**	
a.		Coffee and Products[9]	0.92	0.04	0.19	5.2	4.54	Coffee, green bean {IN}| coffee green bean production, arabica | APOS, U
c.		Tea	1.14	0.75	1.19	0.95	1.14	Tea, dried {CN}| tea production, dried | APOS, U
	8	**Meat**	**30**	**4**	**68**	**52**	**65**	
a.		Bovine Meat[10]	3.65	1.30	10.39	11.98	11.49	Cattle for slaughtering, live weight {GLO}| market for | APOS, U
b.		Mutton and Goat meat	0.04	0.57	3.22	0.14	6.31	Sheep for slaughtering, live weight {GLO}| market for | APOS, U
c.		Pig meat	12.78	0.28	40.30	20.57	0.00	Swine for slaughtering, live weight {GLO}| market for | APOS, U
d.		Poultry meat	13.46	1.84	14.33	19.37	47.22	Chicken for slaughtering, live weight {GLO}| market for | APOS, U
	9	**Fish and seafood**[11]	**26**	**5**	**47**	**50**	**13**	**(Verones et al., 2017)**
	10	**Animal Products**	**41**	**88**	**56**	**92**	**90**	
a.		Butter, Ghee	0.16	2.96	0.13	0.57	1.77	Butter, from cow milk {GLO}| butter production, from cream, from cow milk | APOS, U
b.		Cream	0.01	0.00	0.01	0.00	2.30	Cream, from cow milk {GLO}| market for | APOS, U
c.		Raw Animal Fats[12]	0.22	0.04	2.00	0.77	0.57	Fat from animals, consumption mix, at feed compound plant/NL Economic
d.		Eggs	12.14	2.53	19.47	19.11	5.18	Consumption eggs, broiler parents >20 weeks, at farm/NL Economic
e.		Milk - Excluding Butter	28.87	82.73	34.64	71.89	80.52	Cow milk {GLO}| market for | APOS, U
	11	**Sugar and Confectionery**	**99**	**32**	**8**	**27**	**31**	
a.		Sugarcane	60.09	9.34	0.02	0.00	0.30	Sugarcane {IN}| sugarcane production | APOS, U

[9] This category also includes Coco and Products

[10] This category also includes Offals and meat from horse, ass, mule, camel, snail, rabbit and other rodents.

[11] This category includes Freshwater fish, Demersal fish, Pelagic fish, Marine fish, Crustaceans, Cephalopods, Mollusca, other aquatic animals and aquatic plants.

[12] This category also includes Fish body oil and Fish liver oil.

SN		Item/Type	Per Capita Consumption (kg/person-yr)					Dataset Used
			Thailand	India	China	Japan	Saudi Arabia	
b.		Sugar from Sugarcane[13]	39.27	22.94	7.71	27.01	30.52	Sugar, from sugarcane {GLO}| market for | APOS, U
	12	**Alcoholic Beverages**	**40**	**2**	**47**	**46**	**0**	
a.		Wine	0.12	0.00	1.57	2.71	0.00	(Ardente et al., 2006)
b.		Beer[14]	40.19	1.97	45.18	43.78	0.00	(Amienyo & Azapagic, 2016)

Remarks: Data refers to the total amount of the commodity available as human food during the reference period. This does not include the food available for feed, food losses or the commodity being used for other purposes (such as oil for soap). Data include the commodity in question, as well as any commodity derived from it by processing, unless stated otherwise (FAO 2018). Classification of each commodity into aggregated food items were done primarily on the basis of the FAO classification, but modified in accordance to existing articles regarding food consumption (Muñoz *et al.*, 2010). To model the food items into SimaPro, several food items whose life cycle inventory data were not available were merged into other food items. These food items have been mentioned in the food notes for **Table S1**.

Regarding the vegetables, FAO had data for only Tomatoes and Onions while majority of the vegetables were classified under "Others". Therefore, the agricultural census of each country was taken, and three other items were added: Aubergine, Cabbage/lettuce/cauliflower/spinach and Carrots (General Authority of Statistics 2013; Ministry of Statistics and Programme Implementation 2017; National Statistical Office 2013; Statistics of Japan 2015).

[13] This category combines Sugar (non-centrifugal) and Sugar (raw equivalent) and includes sweeteners and honey.

[14] This category also includes other fermented and alcoholic beverages.

Table S2: Life cycle inventory used for one kilogram of fish and seafood *(Verones et al. 2017)*

Materials/fuels/electricity	**Amount**	**Unit**
Roundwood, eucalyptus ssp. from sustainable forest management, under bark {GLO}| market for | APOS, U	0.0000103	m3
Acrylic varnish, without water, in 87.5% solution state {RER}| acrylic varnish production, product in 87.5% solution state | APOS, U	0.000226	kg
Alkyd paint, white, without solvent, in 60% solution state {GLO}| market for | APOS, U	0.000102	kg
Steel, low-alloyed {RER}| steel production, converter, low-alloyed | APOS, U	0.000104	kg
Cast iron {RER}| production | APOS, U	0.000432	kg
Aluminium, primary, ingot {CN}| production | APOS, U	0.0000299	kg
Aluminium alloy, AlMg3 {RER}| production | APOS, U	0.0000246	kg
Synthetic rubber {RER}| production | APOS, U	0.00000627	kg
Nylon 6-6 {GLO}| market for | APOS, U	0.00877	kg
Lead {GLO}| primary lead production from concentrate | APOS, U	0.00702	kg
Polyethylene, LDPE, granulate, at plant/RER	0.0439	kg
Polypropylene, granulate {RER}| production | APOS, U	0.00439	kg
Diesel {CH}| market for | APOS, U	0.102	kg

Table S3: Life cycle inventory used for one litre of Wine *(Ardente et al. 2006)*

Materials/fuels/electricity	**Amount**	**Unit**
Grape {GLO}| market for | APOS, U	1.33	kg
Compost {CH}| treatment of biowaste, composting | APOS, S	0.19	kg
Potassium sulphate (NPK 0-0-50), at regional storehouse/RER Economic	0.057	kg
[sulfonyl]urea-compound {GLO}| market for | APOS, U	0.038	kg
Phosphate fertiliser, as P2O5 {RER}| monoammonium phosphate production | APOS, U	0.038	kg
Ammonium nitrate, as 100% (NH4)(NO3) (NPK 35-0-0), at plant/RER Economic	0.016	kg
Expanded perlite {CH}| production | APOS, U	0.00133	kg
Sodium sulfite {GLO}| market for | APOS, U	0.000233	kg
Fodder yeast {CH}| ethanol production from whey | Cut-off, U	0.102	kg
Packaging glass, brown {GLO}| market for | APOS, U	275.95	g
Carton board box production, with gravure printing {CA-QC}| carton board box production service, with gravure printing | APOS, U	20.13	g
Wood pellet, measured as dry mass {RER}| market for wood pellet | APOS, U	11.33	g
Tap water {CA-QC}| market for | APOS, U	105.31	kg
Calcium chloride {RER}| soda production, solvay process | APOS, U	2.63	g
Aluminium hydroxide {GLO}| market for | APOS, U	0.396	g
Pesticide, unspecified {GLO}| market for | APOS, U	0.0041	kg
Diesel, burned in agricultural machinery {GLO}| market for diesel, burned in agricultural machinery | APOS, U	4.51	MJ
Electricity, medium voltage {CH}| market for | APOS, U	6.48	MJ
Heat, central or small-scale, natural gas {GLO}| propane extraction, from liquefied petroleum gas | APOS, U	0.25	MJ

Table S4: Life cycle inventory used for one litre of Beer *(Amienyo and Azapagic 2016)*

Materials/fuels/electricity	Amount	Unit
Barley grain {GLO}| market for | APOS, U	74.3	g
Tap water {CA-QC}| market for | APOS, U	8.43	kg
Fodder yeast {GLO}| market for | Cut-off, U	21	g
Clay {CH}| market for clay | APOS, U	1.7	g
Sodium hydroxide (50% NaOH), production mix/RER Economic	9	g
Phosphoric acid, fertiliser grade, without water, in 70% solution state {GLO}| market for | APOS, U	2	g
Sulfuric acid {GLO}| market for | APOS, U	2.5	g
Carbon dioxide, liquid {RER}| market for | APOS, U	30	g
Light fuel oil {CH}| market for | APOS, U	0.04	kg
Container glass (delivered to the end user of the contained product, reuse rate: 7%), technology mix, production mix at plant RER S	691	g
Aluminium removed by drilling, computer numerical controlled {GLO}| market for | APOS, U	36	g
Aluminium alloy, AlLi {GLO}| market for | APOS, U	76	g
Transport, truck >20t, EURO3, 50%LF, default/GLO Economic	0.6	tkm
Electricity grid mix 1kV-60kV, AC, consumption mix, at consumer, 1kV - 60kV AT S	0.236	kWh
Process steam from light fuel oil, heat plant, consumption mix, at plant, MJ CH S	0.006	MJ
Compressed air, 1000 kPa gauge {GLO}| market for | APOS, U	0.01	m3

Table S5: Identified Literature that uses the LCA approach to evaluate environmental impacts of food consumption (GWP = Global Warming Potential, ODP = Ozone Depletion Potential, POCP = Photochemical Ozone Creation Potential, AP = Acidification Potentia

S.N.	Aim of the study	Region/ Country (and City)	Impact Assessment Methodology	Assessed Impact Category	Reference
1	Hotspot analysis for environmental impact, choice of functional unit and affects in conclusion, analysis of prospects for adjustments in emission levels from food systems	Sweden	Not mentioned	GWP	(Carlsson-Kanyama 1998)
2	Assistance to consumers on buying environmentally friendly food products through LCA study	Switzerland	Eco-indicator 95	Every Impact category from Eco-Indicator 95	(Jungbluth, Tietje, and Scholz 2000)
3	Analysis of how energy efficient meals and diets can be composed through Swedish food system study	Sweden	Not mentioned	CED	(Carlsson-Kanyama, Ekstrom, and Shanahan 2003)
4	Comparison of environmental impact of current Swedish diet with a sustainable diet	Sweden	Not mentioned	GHGs	(Wallén, Brandt, and Wennersten 2004)
5	Comparison of three meal preparation methods: Homemade, semi-prepared and ready to eat	Sweden	Not mentioned	CED, GWP, EP, AP, POCP	(Sonesson et al. 2005)
6	Comparison of environmental impacts from 3 diet patterns (omnivorous, vegetarian, vegan) and 2 agricultural practices (conventional, organic)	Italy	Ecoindicator 99 W	GWP, ODP, AP, EP, LU Carcinogens, Respiratory Organics, Respiratory Inorganics, Radiation	(Baroni et al. 2007)
7	Comparison of environmental impacts of two different chicken meals (home-made and semi-prepared)	Sweden	Not mentioned	CED, EP, AP, POCP	(Davis and Sonesson 2008)
8	Comparison of impacts on the environment from four meals with different protein sources	Sweden, Spain	Not mentioned	CED, GWP, EP, AP, POCP, ODP	(Davis et al. 2010)
9	Analysis of the relevance to consider human excretion into the system boundary of a food system life cycle study	Spain	CML 2001	GWP, EP, AP, CED	(Muñoz, Milà I Canals, and Fernández-Alba 2010)
10	Analysis of the environmental effects due to changes to recommended diets in Austria	Austria	Not mentioned	LU, CED, GWP	(Fazeni et al. 2011)
11	Comparison of environmental impacts of three alternative “healthy” diet scenarios	27 Countries in the EU	CMI 2002	GWP, ODP, AP, EP, Human Toxicity, POCP, Ecotoxicity,	(Tukker et al. 2011)

S.N.	Aim of the study	Region/ Country (and City)	Impact Assessment Methodology	Assessed Impact Category	Reference
				Abiotic Resource Depletion	
12	Analyses the reduction in GHGs due to shifts in realistic dietary choices	UK	Not mentioned	GHGs	(Berners-Lee et al. 2012)
13	Comparison of the environmental impacts of Nordic Nutritional Recommendations (NNR) and New Nordic Diet (NND) with the Average Danish Diet	Denmark	Stepwise 2006	GHGs	(Saxe, Larsen, and Mogensen 2013)
14	Comparison of environmental impacts of recommended diets with Average German diet	Germany	Not mentioned	GHGs, NH3, LU Blue water use, phosphorus use, primary energy use	(Meier and Christen 2013)
15	Comparison of environmental impacts of ready-made dish v/s home cooked dish for roast dinner	UK	CML 2011	GWP, AP, EP, ODP, POCP Abiotic Depletion Potential, Human Toxicity, Ecotoxicity	(Schmidt Rivera, Espinoza Orias, and Azapagic 2014)
16	Assessment of environmental impacts due to food consumption and food losses in Germany along the whole life cycle	Germany	ReCiPe	GWP, EP, ODP, Particulate Matter Formation, POCP, AP, LU, Resource Depletion	(Eberle and Fels 2016)
17	Evaluation of the environmental impact of food consumption of an average EU-27 citizen in one year	27 EU countries	ILCD version 1.04	GWP, ODP, human toxicity, Particulate Matter Formation, Ionizing Radiation HH, POCP, AP, EP, ecotoxicity, LU, resource depletion	(Notarnicola et al. 2017)
18	Analysis of contribution of Urban and Peri-urban agriculture to mitigate the environmental impacts of urban food systems	Portugal (Lisbon)	ReCiPe Midpoint	GWP, LU	(Benis and Ferrão 2017)

Table S6: Result of ALCA for the diet patterns of each country

Impact category	Unit	Thailand	India	China	Japan	Saudi Arabia
Global warming	kg CO2 eq	811.44	675.96	1420.55	1109.13	1122.47
Terrestrial acidification	kg SO2 eq	4.96	3.79	9.27	7.21	7.78
Marine eutrophication	kg N eq	0.91	0.87	1.41	1.35	1.44
Terrestrial ecotoxicity	kg 1,4-DCB	1378.79	1120.25	2579.92	1933.19	1805.74
Freshwater ecotoxicity	kg 1,4-DCB	18.15	12.85	32.47	25.25	20.87
Marine ecotoxicity	kg 1,4-DCB	20.73	14.67	38.81	28.79	23.47
Human carcinogenic toxicity	kg 1,4-DCB	23.83	14.69	45.25	31.14	23.97
Human non-carcinogenic toxicity	kg 1,4-DCB	395.37	457.02	890.32	677.67	685.14
Fossil resource scarcity	kg oil eq	109.78	82.07	212.14	145.22	126.52

Table S7: Result of CLCA for the diet patterns of each country

Impact category	Unit	Thailand	India	China	Japan	Saudi Arabia
Global warming	kg CO2 eq	685.14	552.60	1032.11	884.10	814.53
Terrestrial acidification	kg SO2 eq	3.72	3.40	5.89	5.58	5.24
Marine eutrophication	kg N eq	0.89	0.98	1.23	1.23	1.38
Terrestrial ecotoxicity	kg 1,4-DCB	1663.51	1349.80	2875.29	2178.94	1870.70
Freshwater ecotoxicity	kg 1,4-DCB	8.87	10.95	20.39	15.79	18.17
Marine ecotoxicity	kg 1,4-DCB	17.87	12.75	32.84	26.59	20.08
Human carcinogenic toxicity	kg 1,4-DCB	16.76	5.15	31.66	25.12	12.31
Human non-carcinogenic toxicity	kg 1,4-DCB	128.70	437.30	590.68	475.42	684.61
Fossil resource scarcity	kg oil eq	93.01	56.85	170.15	130.74	92.90

Table S8: Result of LCA for a diet with 1kg of each food Item

Impact category	Unit	Total	Cereals	Root Vegetables	Legumes, nuts, Oil seeds	Oils	Vegetables	Fruits	Coffee, and tea	Meat	Fish and Seafood	Animal Products	Sugar and Confectionary	Alcoholic Beverages
Global warming	kg CO2 eq	30.47	0.82	0.24	1.54	4.16	1.27	0.74	6.11	9.36	0.22	3.00	0.65	2.36
Terrestrial acidification	kg SO2 eq	0.23	0.01	0.00	0.01	0.02	0.01	0.00	0.06	0.08	0.00	0.02	0.00	0.01
Marine eutrophication	kg N eq	0.04	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00
Terrestrial ecotoxicity	kg 1,4-DCB	52.86	1.45	1.33	3.84	3.46	2.07	1.90	14.44	11.15	0.29	5.01	1.60	6.32
Freshwater ecotoxicity	kg 1,4-DCB	0.96	0.02	0.02	0.04	0.14	0.03	0.03	0.41	0.12	0.00	0.05	0.01	0.08
Marine ecotoxicity	kg 1,4-DCB	0.88	0.02	0.01	0.05	0.10	0.03	0.02	0.30	0.15	0.01	0.07	0.01	0.11
Human carcinogenic toxicity	kg 1,4-DCB	0.83	0.02	0.01	0.03	0.03	0.04	0.03	0.28	0.18	0.01	0.05	0.01	0.13
Human non-carcinogenic toxicity	kg 1,4-DCB	28.18	0.26	0.50	10.76	3.04	0.78	0.50	6.25	1.44	0.22	1.17	0.46	2.81
Fossil resource scarcity	kg oil eq	4.25	0.12	0.05	0.20	0.40	0.25	0.14	1.21	0.69	0.22	0.32	0.06	0.59

Table S9: Result of LCA for food consumption of Thailand

Impact category	Unit	Total	Cereals	Root Vegetables	Legumes, nuts, Oil seeds	Oils	Vegetables	Fruits	Coffee, and tea	Meat	Fish and Seafood	Animal Products	Sugar and Confectionary	Alcoholic Beverages
Global warming	kg CO2 eq	811.44	233.60	5.35	23.57	35.56	50.11	42.93	12.67	210.36	5.82	67.43	53.30	70.74
Terrestrial acidification	kg SO2 eq	4.96	1.10	0.06	0.14	0.08	0.25	0.36	0.12	1.78	0.02	0.37	0.33	0.34
Marine eutrophication	kg N eq	0.91	0.31	0.02	0.05	0.04	0.02	0.06	0.02	0.24	0.00	0.08	0.05	0.01
Terrestrial ecotoxicity	kg 1,4-DCB	1378.79	246.83	28.49	45.47	21.31	87.40	159.43	29.35	349.63	7.69	96.40	130.71	176.08
Freshwater ecotoxicity	kg 1,4-DCB	18.15	2.99	0.34	1.01	0.57	1.11	2.07	0.84	3.96	0.13	0.88	0.91	3.34
Marine ecotoxicity	kg 1,4-DCB	20.73	3.86	0.22	0.78	0.39	1.44	1.70	0.60	4.94	0.18	1.06	1.05	4.50
Human carcinogenic toxicity	kg 1,4-DCB	23.83	4.92	0.14	0.54	0.21	1.64	1.65	0.60	7.06	0.14	0.87	1.26	4.81
Human non-carcinogenic toxicity	kg 1,4-DCB	395.37	23.53	10.26	99.89	2.81	36.05	34.03	12.31	54.66	5.79	12.84	32.83	70.37
Fossil resource scarcity	kg oil eq	109.78	25.21	1.14	2.55	1.55	10.00	9.79	2.54	23.90	5.68	5.13	5.21	17.08

Table S10: Result of LCA for food consumption of India

Impact category	Unit	Total	Cereals	Root Vegetables 1 kg	Legumes, nuts, Oil seeds	Oils	Vegetables	Fruits	Coffee, and tea	Meat	Fish and Seafood	Animal Products	Sugar and Confectionary	Alcoholic Beverages
Global warming	kg CO2 eq	675.96	204.87	9.95	21.84	37.80	98.91	26.30	4.21	34.06	1.13	204.24	29.17	3.48
Terrestrial acidification	kg SO2 eq	3.79	1.16	0.13	0.22	0.17	0.47	0.20	0.03	0.27	0.00	0.96	0.16	0.02
Marine eutrophication	kg N eq	0.87	0.31	0.03	0.07	0.06	0.04	0.04	0.01	0.04	0.00	0.25	0.02	0.00
Terrestrial ecotoxicity	kg 1,4-DCB	1120.25	270.14	58.53	56.68	28.91	174.78	89.24	8.13	38.36	1.49	312.72	72.61	8.66
Freshwater ecotoxicity	kg 1,4-DCB	12.85	2.98	0.73	0.95	0.82	2.15	1.06	0.25	0.41	0.02	2.88	0.43	0.16
Marine ecotoxicity	kg 1,4-DCB	14.67	3.89	0.53	0.81	0.59	2.83	1.01	0.16	0.48	0.03	3.57	0.53	0.22
Human carcinogenic toxicity	kg 1,4-DCB	14.69	4.57	0.34	0.52	0.26	3.40	1.08	0.22	0.50	0.03	2.88	0.66	0.24
Human non-carcinogenic toxicity	kg 1,4-DCB	457.02	66.63	24.45	191.29	25.64	62.85	16.54	2.30	4.06	1.12	35.01	23.66	3.46
Fossil resource scarcity	kg oil eq	82.07	24.69	2.20	3.21	2.71	19.90	5.95	0.94	2.36	1.10	15.54	2.64	0.84

Table S11: Result of LCA for food consumption of China

Impact category	Unit	Total	Cereals	Root Vegetables 1 kg	Legumes, nuts, Oil seeds	Oils	Vegetables	Fruits	Coffee, and tea	Meat	Fish and Seafood	Animal Products	Sugar and Confectionary	Alcoholic Beverages
Global warming	kg CO2 eq	1420.55	209.13	24.33	28.88	31.14	326.74	44.73	7.09	570.71	9.94	80.22	8.60	79.03
Terrestrial acidification	kg SO2 eq	9.27	1.18	0.32	0.14	0.10	1.51	0.37	0.05	4.66	0.04	0.47	0.05	0.38
Marine eutrophication	kg N eq	1.41	0.32	0.06	0.05	0.05	0.14	0.06	0.01	0.59	0.00	0.10	0.01	0.02
Terrestrial ecotoxicity	kg 1,4-DCB	2579.92	276.87	146.66	67.14	21.90	624.86	160.72	14.37	926.00	13.14	108.74	21.54	197.98
Freshwater ecotoxicity	kg 1,4-DCB	32.47	3.05	1.86	0.68	0.40	7.54	2.81	0.44	10.69	0.22	1.01	0.12	3.66
Marine ecotoxicity	kg 1,4-DCB	38.81	3.99	1.37	0.79	0.31	9.96	2.06	0.29	13.46	0.31	1.20	0.15	4.93
Human carcinogenic toxicity	kg 1,4-DCB	45.25	4.67	0.87	0.51	0.20	10.39	1.98	0.36	19.57	0.24	0.98	0.19	5.30
Human non-carcinogenic toxicity	kg 1,4-DCB	890.32	68.93	63.17	233.59	13.62	205.42	38.47	4.62	147.65	9.89	17.39	7.64	79.92
Fossil resource scarcity	kg oil eq	212.14	25.26	5.43	2.81	1.63	67.18	10.60	1.54	61.64	9.70	6.45	0.75	19.15

Table S12: Result of LCA for food consumption of Japan

Impact category	Unit	Total	Cereals	Root Vegetables 1 kg	Legumes, nuts, Oil seeds	Oils	Vegetables	Fruits	Coffee, and tea	Meat	Fish and Seafood	Animal Products	Sugar and Confectionary	Alcoholic Beverages
Global warming	kg CO2 eq	1109.13	161.10	11.27	36.38	49.48	66.50	24.95	42.01	425.07	11.10	164.71	32.85	83.72
Terrestrial acidification	kg SO2 eq	7.21	0.91	0.15	0.09	0.33	0.33	0.21	0.51	3.21	0.05	0.86	0.17	0.40
Marine eutrophication	kg N eq	1.35	0.25	0.03	0.03	0.14	0.04	0.04	0.09	0.50	0.00	0.20	0.02	0.02
Terrestrial ecotoxicity	kg 1,4-DCB	1933.19	211.88	68.17	51.71	54.89	154.73	90.25	109.74	646.45	14.66	237.70	82.26	210.74
Freshwater ecotoxicity	kg 1,4-DCB	25.25	2.35	0.87	0.57	1.03	1.82	1.59	3.01	7.35	0.24	2.17	0.45	3.81
Marine ecotoxicity	kg 1,4-DCB	28.79	3.06	0.64	0.63	0.86	2.37	1.16	2.29	9.10	0.34	2.62	0.58	5.13
Human carcinogenic toxicity	kg 1,4-DCB	31.14	3.61	0.41	0.40	0.51	2.34	1.10	1.81	12.25	0.26	2.16	0.73	5.56
Human non-carcinogenic toxicity	kg 1,4-DCB	677.67	50.65	29.49	158.80	67.12	47.20	21.46	54.57	93.24	11.04	29.16	29.21	85.72
Fossil resource scarcity	kg oil eq	145.22	19.49	2.52	2.06	3.98	14.41	5.91	7.69	42.65	10.83	12.45	2.88	20.34

Table S13: Result of LCA for food consumption of Saudi Arabia

Impact category	Unit	Total	Cereals	Root Vegetables 1 kg	Legumes, nuts, Oil seeds	Oils	Vegetables	Fruits	Coffee, and tea	Meat	Fish and Seafood	Animal Products	Sugar and Confectionary	Alcoholic Beverages
Global warming	kg CO2 eq	1122.47	168.19	8.65	14.78	68.55	63.66	136.59	39.76	377.38	3.00	203.44	38.47	0.00
Terrestrial acidification	kg SO2 eq	7.78	1.14	0.11	0.13	0.25	0.35	0.49	0.48	3.63	0.01	0.98	0.20	0.00
Marine eutrophication	kg N eq	1.44	0.30	0.02	0.03	0.11	0.04	0.04	0.08	0.54	0.00	0.24	0.03	0.00
Terrestrial ecotoxicity	kg 1,4-DCB	1805.74	272.48	52.36	65.26	42.95	182.21	237.26	102.67	443.23	3.97	307.02	96.32	0.00
Freshwater ecotoxicity	kg 1,4-DCB	20.87	2.85	0.67	0.59	0.75	2.12	3.26	2.83	4.41	0.07	2.81	0.53	0.00
Marine ecotoxicity	kg 1,4-DCB	23.47	3.69	0.49	0.79	0.56	2.79	3.63	2.14	5.12	0.09	3.47	0.68	0.00
Human carcinogenic toxicity	kg 1,4-DCB	23.97	4.05	0.31	0.51	0.33	3.38	4.77	1.73	5.13	0.07	2.83	0.86	0.00
Human non-carcinogenic toxicity	kg 1,4-DCB	685.14	90.78	22.66	254.52	21.00	55.69	70.34	50.33	47.49	2.99	35.21	34.13	0.00
Fossil resource scarcity	kg oil eq	126.52	23.15	1.93	2.69	4.06	14.76	21.16	7.34	29.56	2.93	15.54	3.38	0.00

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PROMOTING SUSTAINABILITY IN EMERGING ECONOMIES VIA LIFE CYCLE THINKING

Preface

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Emerging economies have played an important role in driving global sustainability due to their high share of the world population, production, as well as consumption. Even so, emerging markets lag behind developed countries, and all countries face common challenges including energy security, environmental protection, and affordable energy for fulfilling the rapidly growing demand for energy (IEA 2013). Despite the age of science, technology, and innovations, the current development trends in developing regions may still pose threats to the environment, human health, and natural resources. According to the latest regional assessments in global environmental outlook (UNEP 2016a, b, c), major concerns include changes in demography and lifestyle, inequality, increased resource use with decreased efficiency and productivity, increased environmentally related health risks, increased vulnerability to natural hazards and extreme events, land degradation, resource scarcity, biodiversity and habitat losses, and widened gaps between policy and implementation. In order to leapfrog these problems in developing countries and avoid the environmentally harmful stages of development, it is necessary to have measures as well as tools for supporting both policy decision makers and producers for strategic planning.

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Green growth, low-carbon society, and cleaner production are ideas that are often discussed in environmental circles. Life cycle thinking is a key concept that binds these ideas. It is a useful analytical tool, not only as environmental LCA but also life cycle costing (LCC), social LCA (SLCA), and other related tools such as material flow analysis (MFA). This special issue aims to bring together experiences from emerging economies related to the application of life cycle assessment-based tools via case studies, industrial applications, policy applications, and even new methods or adaptation of existing ones to the local context.

One of the indicators of the development and mainstreaming of LCA is the development of the national life cycle inventory (LCI) database. One country that stands out in this context is Thailand where the national LCI database was started to be developed more than a decade ago. It currently has more than 700 datasets comprising ten sectors including natural gas, refinery, petrochemical products, infrastructure and transportation, construction materials, agriculture and agro-products, basic chemicals, waste management, and textiles (Chomkhamsri et al. 2017). This has helped the government in promoting LCA-based policy and also encouraged industry to be more receptive to conducting LCAs and related tools (Gheewala and Mungcharoen 2017). In fact, now the readiness of the national LCI is being used for testing the Product Environmental Footprint (PEF) which may need to be conducted by Thai companies producing parts and products for export to the European Union (Poolsawad et al. 2017). Thailand has been quite advanced in the development and implementation of LCA in Southeast Asia and has been working with countries in the region to promote the application of LCA, especially in the agrifood sector through the LCA Agrifood Asia network (Gheewala 2012).

Development of national databases, however, is not enough as LCAs by nature are global. They need to consider the entire supply chain which is commonly spread over many countries. This brings forth the need for interoperability of databases which is confounded by the issue of confidentiality. However, the aspects of harmonized review criteria and review process are important and must be carried out while keeping in mind that there are high stakes vis-à-vis competition and trade (Vigon et al. 2017).

The development of national LCI databases is the first important step in supporting the applications of life cycle thinking and LCA tools to enhance sustainability in emerging economies. Life cycle thinking can be used as the logical conceptual approach for strategic environmental planning and management for the industrial sector such as a case of mining sector in the Philippines (Balanay and Halog 2017). Another aspect from a practical point of view when implementing LCAs for the industry is the need for sector/product-based standardization via environmental product declarations (EPD). This may pose certain challenges in the emerging economy context; thus, research is needed on whether it is appropriate to directly implement the EPDs formulated by developed countries or if there could be context-specific variations. These discussions are also sector-specific as shown by the study on Mexico's building sector (Arvizu-Piña and Burgos 2017).

The development of national databases and promotion of life cycle thinking in many of the emerging economies has led to the proliferation of LCA studies. Many traditional LCA studies have been conducted to assess the environmental performance of products. This special issue presents some of the studies that have focused on using LCA as a rigorous method for testing the perceived environmental advantages of products such as biodiesel from beef tallow (Magalini et al. 2017), hotspots analysis of office furniture (Medeiros et al. 2017), or environmental comparison of products such as bricks and palm oil (Prateep Na Talang et al. 2017; Bunchai et al. 2017) and processes such as waste treatment and construction (Hassanain et al. 2017; Sedpho et al. 2017; Salzer et al. 2017). In addition to conventional LCAs, allocation methods have also been considered in complex multiproduct systems such as oil refineries using multi-scale modeling (Silveira et al. 2017). The development and use of the LCA-based methods for water assessment such as water stress index and water deprivation have been tested for policy applications (Pingmuanglek et al. 2017; Nilsalab et al. 2017). New methods on valuation of environmental impacts have also been adopted to country-specific contexts and tested for policy application (Kaenchan and Gheewala 2017). The application of consequential LCA, which is relatively new to emerging economies, has been attempted in analyzing policy decisions showing the coming-of-age of LCA in these countries (Prapaspongsa and Gheewala 2017; Prateep Na Talang et al. 2017). The combination of life cycle -based environmental and economic tools is a first step towards life cycle sustainability assessment (LCSA); studies have been done using tools such as LCA, LCC, and economically extended material flow analysis (Bhochhibhoya et al. 2017; Khonpikul et al. 2017). Finally, a full LCSA has also been attempted considering LCA, LCC, and SLCA (van Kempen et al. 2017). Thus, it can be seen that the whole gamut of LCA and life cycle thinking-based tools have been used in several emerging economies across the world, in Asia, Africa, and America.

Under challenging conditions in emerging economies (e.g., needs for economic growth and industrial development and low environmental awareness), lessons learned from this special issue will be beneficial for both developed and emerging economies.

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