CHAPTER II
BACKGROUND AND LITERATURE REVIEW

As we know that the energy from fossil fuels causes high environmental impacts. Especially, the accumulation of carbon dioxide or greenhouse gases in the atmosphere cause greenhouse effect and climate change. Furthermore, fluctuation of crude oil prices in the world market and the decrease of proved reserves of fossil fuels are the factors that stimulate the study and development of alternative energy in order to reduce reliance of fossil fuels. The important alternative fuels that the development countries are paying attention to are bio-hydrogen fuel, synthesis biofuels, bioethanol from lignocellulose, etc.

Biofuels have been proposed as an ecologically benign alternative to fossil fuels. They can be classified based on their production technologies: first generation biofuels; second generation biofuels; third generation biofuels; and fourth generation biofuels.

First generation biofuels refer to biofuels made from sugar, starch, vegetable oils, or animal fats using conventional technology. The first generation biofuels produced from food crops such as grains, sugar beet, and oil seeds are limited in their ability to achieve targets for oil-product substitution, climate change mitigation, and economic growth. A possible exception that appears to meet many of the acceptable criteria is bioethanol produced from sugar cane. The basic feedstocks for the production of first generation biofuels are often seeds or grains such as wheat, which yields starch that is fermented into bioethanol, or sunflower seeds, which are pressed to yield vegetable oil that can be used in biodiesel. The production of first generation biofuels such as sugarcane ethanol in Brazil, corn ethanol in US, oilseed rape biodiesel in Germany, and palm oil biodiesel in Malaysia is characterized by mature commercial markets and well understood technologies. Future targets and investment plans suggest strong growth will continue in the near future.

Second generation biofuels are studied in foreign countries widely. They are produced from larger feedstocks from lignocellulosic materials include cereal straw, forest residues, bagasse, and purpose-grown energy crops such as vegetative grasses and short rotation forests. The second generation biofuels could avoid many of the concerns facing first generation biofuels and potentially offer greater cost reduction potential in the longer term. Many of problems associated with first generation biofuels can be addressed by the production of biofuels manufactured from agricultural and forest residues and from non-food crop feedstocks. Low-cost crop and forest, wood process wastes, and the organic fraction of municipal solid wastes can all be used as lignocellulosic feedstocks. Second and third generation biofuels are also called advanced biofuels. Third generation biofuel, is a biofuel from algae. On the other hand, an appearing fourth generation is based in the conversion of vegetable oil and biodiesel into biogasoline using most advanced technology (Demirbas, 2009). Nowadays, there is the abundant development of second generation biofuel production technology.

2.1 Compressed Biomethane Gas (CBG)

Biomethane is renewable natural gas. It is made by upgrading biogas that is produced by the controlled decomposition of dairy manure or similar waste products. (Krich *et al.*, 2005)

2.1.1 Production of Biogas by Anaerobic Digestion

Anaerobic digestion is a naturally occurring process of decomposition of organic matter by microbes in an oxygen-free environment. Anaerobic digestion has been used throughout the globe for many years. However, it has not been applied widely for production of biomethane as a transportation fuel.

Anaerobic digestion is a complex process that involves two stages, as shown in the simplified schematic in Figure 2.1. In the first stage, decomposition is performed by fast-growing, acid-forming (*acidogenic*) bacteria. Protein, carbohydrate, cellulose, and hemicellulose in the manure are hydrolyzed and metabolized into mainly short-chain fatty acids—acetic, propionic, and butyric—along with CO_2 and hydrogen (H_2) gases. At this stage the decomposition products have noticeable, disagreeable, effusive odors from the organic acids, H_2S, and other metabolic products. In the second stage, most of the organic acids and all of the H_2

are metabolized by methanogenic bacteria, with the end result being production of a mixture of approximately 60% to 70% CH_4 and 30% to 40% CO_2, called biogas.

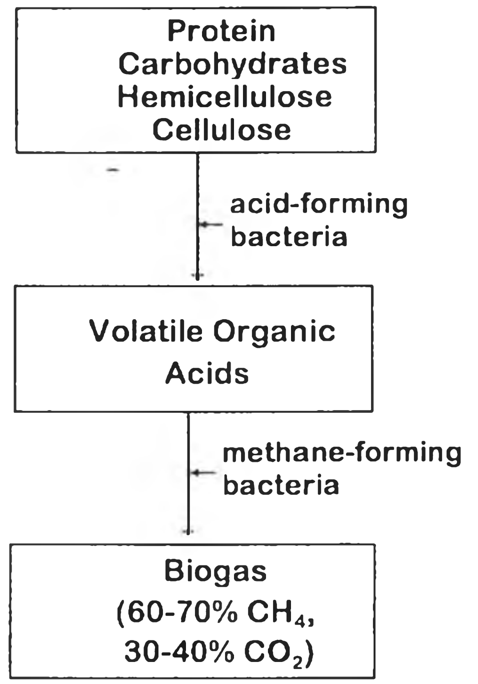

Figure 2.1 Simplified process of biogas production.

To augment methane production, manure from dairy cows can be co-digested with additional substrates such as agricultural residues and food-processing waste. Co-digestion is the simultaneous digestion of a mixture of two or more feedstocks. The most common situation is when a major amount of a main basic feedstock (e.g., manure or sewage sludge) is mixed and digested together with minor amounts of a single or a variety of additional feedstocks. The expression co-digestion is applied independently to the ratio of the respective substrates used simultaneously.

Chen *et al.* (2010) provided a high level economic analysis of biogas production from agricultural wastes and conversion to a usable transportation fuel, biomethane or renewable natural gas. They found that in terms of total production cost, biomethane is competitive with conventional natural gas. The scales of production are important and production is much more economical at larger facilities. Biomethane is a good alternative, especially when produced on a larger scale. The implementation of a biomethane facility is not currently cost effective for a private developer without public funding assistance, because of issues of scale and price volatility. If capital costs can be covered, however, the operations and maintenance costs are much smaller and allow reasonable production cost for pipeline quality biomethane.

The complete process of biogas production, cleanup to biomethane and usage is summarized as shown schematically in Figures 2.2 and 2.3.

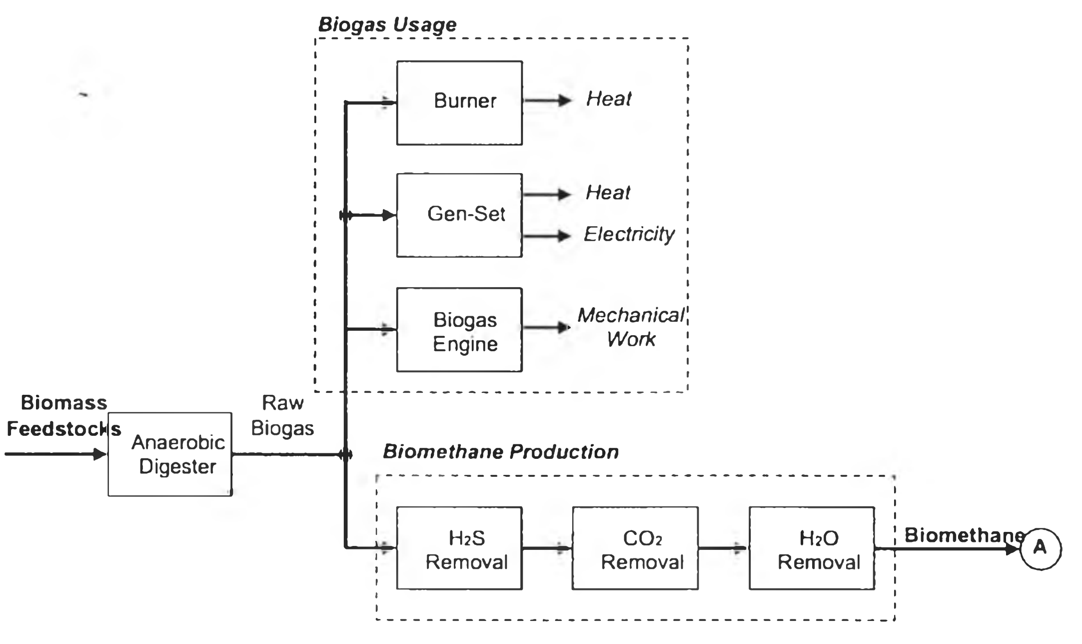

Figure 2.2 Schematic diagram of biogas and biomethane production and utilization (Part 1).

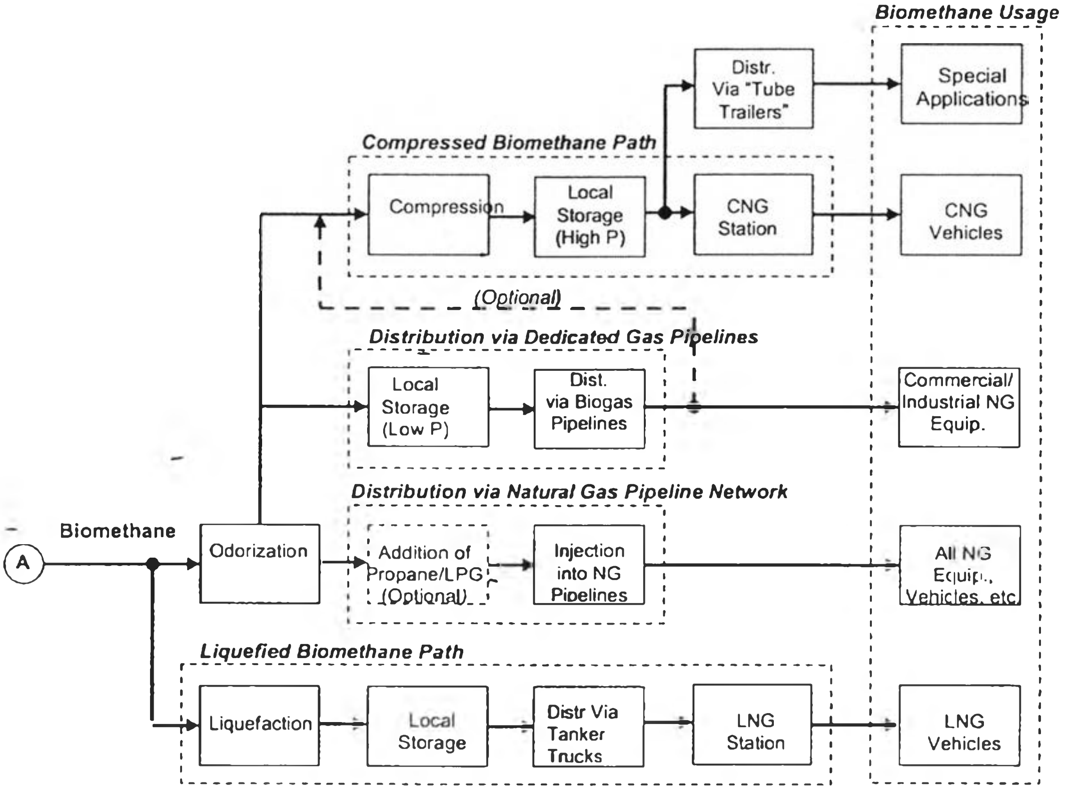

Figure 2.3 Schematic diagram of biomethane distribution and utilization (Part 2).

2.1.2 Upgrading Biogas to Biomethane

Biogas consists primarily of methane (CH_4) and carbon dioxide (CO_2), trace amounts of hydrogen sulfide (H_2S) and other components. The exact composition of biogas depends on the composition of the starting feedstock and digestion process. Biogas produced on agricultural facilities typically contains between 60 to 70% methane and 30 to 40% carbon dioxide by volume. In comparison, natural gas contains close to 90% methane and has a corresponding higher heating value.

By removing hydrogen sulfide, moisture, and carbon dioxide, dairy biogas can be upgraded to biomethane, a product equivalent to natural gas, which typically contains more than 95% methane. The process can be controlled to produce biomethane that meets a pre-determined standard of quality. Biomethane can be used interchangeably with natural gas, whether for electrical generation, heating, cooling, pumping, or as a vehicle fuel. Biomethane can also be pumped into the natural gas supply pipeline. High pressures can be used to store and transport biomethane as compressed biomethane, which is analogous to CNG, or very low temperatures can be used to produce liquefied biomethane, which is analogous to LNG.

Margareta (2003) studied four different techniques for upgrading of biogas. The techniques are water wash, Pressure Swing Adsorption (PSA), Selexol and absorption by chemical reaction. The aim was to evaluate the techniques in as many aspects as possible. They found that PSA, Selexol or absorption by chemical reaction have high security against sulphur and water content in the upgraded gas. An advantage with absorption with chemical reaction is that it has low methane losses and low use of electricity. A disadvantage is large demand of heat for regeneration. General conclusions for upgrading biogas are that the upgrading cost depends a great deal of the size of the plant. The upgrading cost for plants less than 100 m_n^3/h raw gas are 0.30-0.40 SEK/kWh upgraded gas, while its only 0.10-0.15 SEK/kWh upgraded gas for plants between 200 and 300 m_n^3/h raw gas. Another conclusion is that the electricity demand for upgrading gas corresponds to 3 to 6 % of the energy content in the upgraded gas.

Water scrubbing is a well-established and simple technology that can be used to remove both H_2S and CO_2 from biogas, because both of these gases are more soluble in water than methane is. Likewise, H_2S can be selectively removed by this process because it is more soluble in water than carbon dioxide. However, the H_2S desorbed after contacting can result in fugitive emissions and odor problems. Pre-removal of H_2S (e.g., using iron sponge technology) is a more practical and environmentally friendly approach.

Water scrubbing is the most applicable CO_2 scrubbing process because of its simplicity and low cost. Another advantage of water scrubbing over some other processes is that water is fairly easy to dispose of whereas the chemicals used in some of the other processes may require special handling and disposal when spent. The disadvantage of water scrubbing is that it is less efficient than other processes, both in terms of CH_4 loss and energy. However, some of the energy inefficiency of the process may be offset by the use of a single-pass water scrubbing system, since other processes require a regeneration stage (Krich *et al.*, 2005).

When water scrubbing is used for CO_2 removal, biogas is pressurized, typically to 150 to 300 pounds per square inch, gauge (psig) with a two-stage compressor, and then introduced into the bottom of a tall vertical column. The raw biogas is introduced at the bottom of the column and flows upward, while fresh water is introduced at the top of the column, flowing downward over a packed bed. The packed bed (typically a high-surface-area plastic media) allows for efficient contact between the water and gas phases in a countercurrent absorption regime. Water often pools at the bottom of the contact column and the biogas first passes through this water layer in the form of bubbles. The CO_2-saturated water is continuously withdrawn from the bottom of the column and the cleaned gas exits from the top.

A water scrubbing system preceded by H_2S removal would be a practical, low-cost process for upgrading dairy biogas to biomethane. It is important that the H_2S be removed prior to the removal of the CO_2, as H_2S is highly corrosive and would result in decreased life and higher maintenance of the subsequent compressors required in the CO_2-removal step.

In 2012, Scholz *et al.* studied membrane-based biogas upgrading processes. They investigated membrane materials, gas permeation modules and their respective operation as well as gas permeation processes for biogas upgrading. The biogas upgrading process as well as conventional upgrading processes and their characteristics are presented as a benchmark for the membrane process.

Their study presents the basic flowsheet of the membrane-based biogas upgrading process and the required unit operations when gas permeation membranes are applied (Figure 2.4). The raw gas is compressed to the required pipeline pressure. The pressure is slightly higher than the pipeline pressure in order to overcome the pressure losses in the upgrading equipment and the piping system. Subsequent to the compression, the gas flows to a heat exchanger to control the gas temperature and to avoid high temperatures in the membrane system. Then the gas enters the membrane system to purify the gas. A fine desulfurization unit lowers the hydrogen sulfide level when the membrane system is not able to achieve the required hydrogen sulfide level. Finally, the purified methane reaches the natural gas grid.

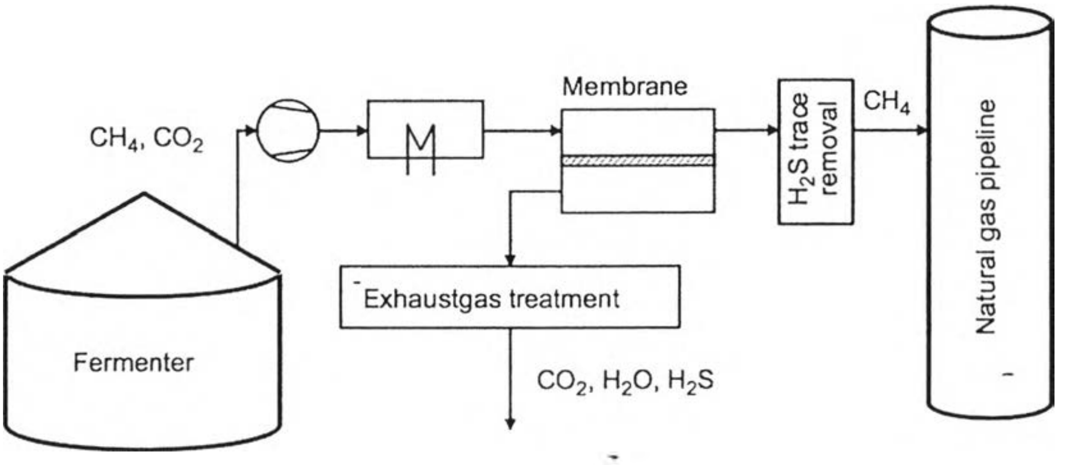

Figure 2.4 The process equipment for a membrane-based upgrading process.

They also found that gas permeation processes have outstanding properties which make them superior to conventional gas separation equipment in biogas upgrading. However, single stage membrane processes are not able to upgrade the raw biogas economically and limits of the gas permeation process are discussed in detail. Nevertheless, the typical tradeoff between product gas purity and methane recovery can be dismantled by applying multistage gas permeation networks.

2.2 Napier Grass

Pennisetum purpureum also known as Napier grass, elephant grass or Ugandan grass, is a monocot C4 perennial grass in the Poaceae family (Khan *et al.*, 2007). It is a species of perennial tropical grass native to the African grasslands (Farrell *et al.*, 2002). It has low water and nutrient requirements, and therefore can make use of otherwise uncultivated lands. Historically, this wild species has been used primarily for grazing; recently, however, it has been incorporated into a pest management strategy. This technique involves the desired crop being planted alongside a 'push' plan, which repels pests, in combination with a 'pull' crop around the perimeter of the plot, which draw insects out of the plot. Napier grass has shown potential at attracting stemborer moths (a main cause of yield loss in Africa) away from maize and hence is the "pull" crop. This strategy is much more sustainable, serves more purposes and is more affordable for farmers than insecticide use. In addition to this, Napier grasses improve soil fertility, and protect arid land from soil erosion. It is also utilized for firebreaks, windbreaks, in paper pulp production and most recently to produce bio-oil, biogas and charcoal.

It is a fast growing, deeply rooted grass growing up to 4 metres tall that can spread by underground stems to form thick ground cover. Napier is easy to establish and persistent; drought tolerant; suitable for cutting and very good for silage making. Napier grass is a high yielding fodder crop with good palatability, highly nutritious especially when young, dark green leaves and less than 1 metre tall.

It is also used as a soil stabilizer in soil conservation methods and can be intercropped with various forage legumes.

Napier grass is not suitable for direct grazing since stumping results in poor regeneration. It is vulnerable to disease and pest attacks. It takes up a lot of nutrients from the soils and is highly demanding on nutrient recycling/fertilizer application.
It can be grown at altitudes ranging from sea level to 2,000 m above sea level. When grown at altitudes above 2000 m, growth and regeneration after cutting is slow and it may die due to frost. It does best in high rainfall areas, over 1500 mm per year.

Napier grass can grow in almost any soils; but does best in deep, fertile, well draining soils. Napier can be propagated through seeds, however as seed production is inconsistent, collection is difficult. Alternatively, it can be planted through stem cuttings of the stolons. The cuttings can be planted by inserting them along furrows 75 cm apart, both along and between rows (Aminah *et al.*, 1997).

The Department of Livestock Development in Thailand calls a new hybrid Napier grass "Pakchong 1 Napier" but it might as well be called "Super Napier". It is very fast growing and high-yielding. One rai or 1,600 square meters can yield 20 tons of herbage per harvest. There is 60 tons per rai per year. The study of Energy Research and Development Institute (Nakornping), Chiangmai University and

Energy Policy and Planning Office, Thailand Ministry of Energy mentioned that Pakchong 1 Napier has highest yield comparing with other types of grasses (Department of Alternative Energy Development and Efficiency, Thailand Ministry of Energy, 2013).

Napier grass cultivation stage consists of:

2.2.1 Land Preparation

- Plough and harrow the field well before planting (i.e. seedbed should be as good as that for planting maize).

2.2.2 How To Plant

Two methods may be used, namely:

2.2.2.1 *Conventional Method*

- Dig up a width of 15-20 cm and a depth of 15-20 cm at a spacing of 3 feet (90 cm) between rows x 2 feet (60 cm) between plants.

- In each hole apply one or two handfuls of farm yard manure (10 tons/ha FYM) or (20 to 40 Kg $P_2 0_5$/ha)

- Place a 3-node cane at a slanting position in the soil, ensuring that two nodes are covered by the soil.

- Place the root splits into the planting holes and cover with soil.

2.2.2.2 *Tumbukiza Method*

This method gives higher herbage yields even during the dry season than the conventional method. There are two types of tumbukiza, namely the round pit type and the rectangular pit type.

For round pits:

- Dig up a diameter of 60 cm and a depth of 60 cm. The rows of pits should be 60 cm apart.

For rectangular pits:

- Dig pits 60 cm deep by 60-90 cm wide.

- The length of the pit can vary depending on available land.

- The pits should be 90 cm apart.

For both round and rectangular pit type:

- Separate top soil from sub soil.

- Mix 1 debe of top soil with 1 to 2 debes of farm yard manure and put into the pits.

- Leave about 15 cm unfilled space at the top of each pit.

- Plant 5-10 cane cuttings or single root splits in round pits.

- In rectangular pits, plant 5-10 cuttings or single root splits for every 90 cm length.

2.2.3 Management

- Hand weed after every cutting/harvesting if there are weeds.

- Apply farm yard manure at the rate of 5 to 10 ton ha^{-1} or slurry after every 4 to 6 harvests.

- Inorganic nitrogen fertilizers can also be used at the rate of 60-90 kg N ha^{-1} (5 to 8 bags of CAN fertilizer).

2.2.4 Harvesting

Napier grass is ready for harvesting 2-3 months after planting and harvesting can continue at an interval of 6-8 weeks.

2.3 Life Cycle Assessment (LCA)

Achieving sustainable development requires methods and tools to quantify and compare the environmental impacts of each product. Every product has a life, starting with design or development of the product, followed by production and consumption, and finally end-of-life activities including collection, waste disposal, reuse, and recycling (Rebitzer *et al.*, 2004). All of the processes throughout the product's life result in the environmental impacts due to consumption of resources, generation of wastes, and emissions of substances. Figure 2.5 shows a simplified scheme of the product life concept which is usually referred to as a "life cycle".

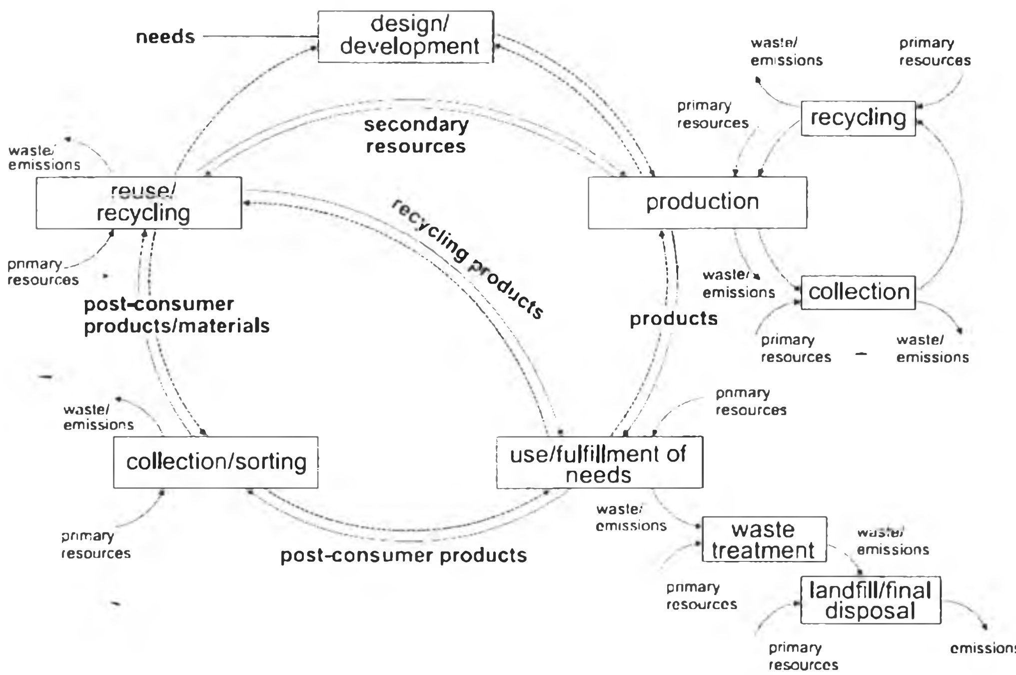

Figure 2.5 Schematic representation of a generic life cycle of a product (Rebitzer *et al.*, 2004).

2.3.1 History of LCA

Life cycle assessment (LCA) was developed around the late 1960s and early 1970s, a period in which oil crisis and environmental issue became a broadly public concern (Russell *et al.*, 2005). It became obvious that the petroleum resource will last forever and the exponential economic growth might result in both environmental and social disaster. Therefore, the concept of energy and environmental analysis, which had been conducted for several years, was later broadened to encompass resource requirement, waste generation, and emission loading.

- Decades of Conception (1970-1990)

Decades of conception are the beginning period of LCA with widely diverging approaches, terminologies, and results. LCA was performed by using different methods and without a common theoretical framework in this period. In 1969, the first LCA study was conducted by Midwest Research Institute (MRI) in the United States for the Coca Cola Company about different beverage containers

(Guinée *et al.*, 2010). In Europe, early LCA-like work started soon afterwards in Germany, England, Switzerland, and Sweden (Klöpffer, 1997). The main topic was the comparative analysis of packaging under environmental aspects, especially with regard to resource conservation and energy saving. The Swiss Federal Laboratories for Materials Testing and Research (EMPA) published a report that presented a comprehensive list of the data needed for LCA study in 1984 (Guinée *et al.*, 2010). In the late 1980s, not only packaging, but also many other systems were gradually studied and analyzed from "cradle to grave" (Klöpffer, 1997). Then a shift can be observed from comparative studies toward system optimization and benchmarking. It has been recognized that a large share of the environmental impacts of many products is not in the utilization of the product, but in its production, transportation, and disposal process.

- Decade of Standardization (1990-2000)

The number of LCA research works and handbooks has been produced since the beginning of the 1990s (Russell *et al.*, 2005). Many scientific journal papers have also been published. In the early 1990s, through its North American and European branches, the Society of Environmental Toxicology and Chemistry (SETAC) shaped the development of LCA in a series of important workshop resulting in the "Code of Practice" in 1993 (Perriman, 1993; Ekvall, 2005). This document describes a procedural framework for LCA and also includes some methodological recommendations. Next to SETAC, the International Organization of Standardization (ISO) has been involved in LCA since 1994 in order to start a standardizing process (Arvanitoyannis, 2008). Therefore, this period can be characterized as a period of convergence between SETAC's coordination and ISO's standardizing activity.

Nowadays, LCA becomes increasingly important due to awareness of the environmental impacts caused by products. Governments and corporations all over the world also encouraged the use of LCA (Reap *et al.*, 2008). As a result, LCA has become a core element in environmental policy as well as voluntary action.

2.3.2 <u>Definition of LCA</u>

Two of the most widely accepted definitions of LCA are presented below as they have been chronologically formulated to date.

- Definition of LCA by SETAC

"The life cycle assessment is an objective process to evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying energy and materials used and wastes released to the environment; to assess the impact of those energy and material uses and releases to the environment; and to identify and evaluate opportunities to effect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing extracting and processing raw materials; manufacturing; transportation and distribution; use, re-use, maintenance; recycling; and final disposal."

- Definition of LCA by ISO 14040

"LCA is a technique for assessing the environmental aspects and potential impacts associated with a product by:

1. Compiling an inventory of relevant inputs and outputs of a product system;

2. Evaluating the potential environmental impacts associated with those inputs and outputs;

3. Interpreting the results of the inventory analysis and impact assessment phases in relation to the objectives of the study.

LCA studies the environmental aspects and potential impacts throughout the product's life (i.e. cradle to grave) from raw materials acquisition through production, use and disposal. The general categories of environmental impacts needing consideration include resource use, human health, and ecological consequences".

2.3.3 LCA Methodology

The Society of Environmental Toxicology and Chemistry's (SETAC) ''Code of Practice'', which can be illustrated by the famous SETAC triangle as shown in Figure 2.6, originally distinguished four methodological components within LCA: goal and scope definition, inventory analysis, impact assessment, and improvement assessment (Rebitzer *et al.*, 2004).

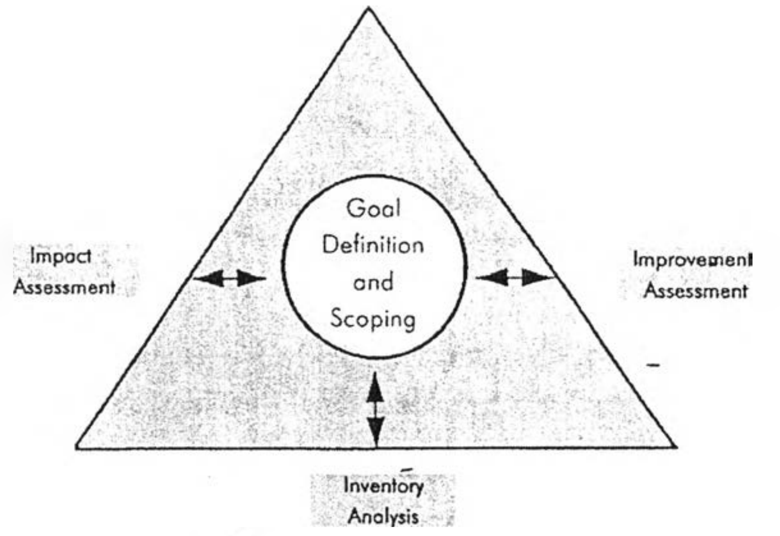

Figure 2.6 SETAC triangle (Klöpffer, 1997).

According to ISO 14040, improvement assessment is no longer regarded as a phase on its own, but rather as having an influence throughout the whole LCA methodology (Rebitzer *et al.*, 2004). Moreover, interpretation which is a phase that interacts with all other phases in the LCA has been introduced as illustrated in Figure 2.7. In practice, an LCA is often conducted iteratively, repeating some of the phases several times in order to eliminate uncertainties (Widheden and Ringström, 2007).

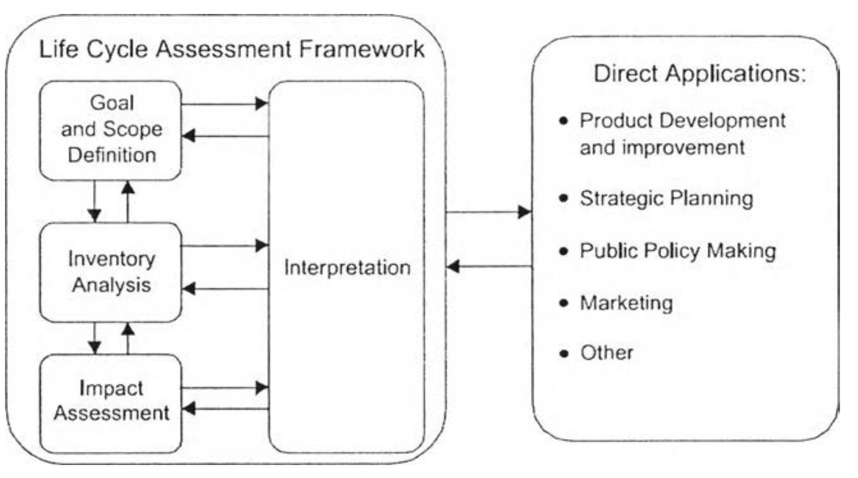

Figure 2.7 General methodological framework of LCA.

2.3.3.1 *Goal and Scope Definition*

The goal and scope definition phase in the LCA is the planning phase which attempts to set the extent of the inquiry and provides the following descriptions of the product system (Widheden and Ringström, 2007):

- **Objectives**

The ISO 14040 standard states that the goal definition "shall unambiguously state the intended application, the reason for carrying out the study and the intended audience".

- **System Boundaries**

The scope defines the boundaries of the study, including the products and unit processes for which data are to be collected, and the geographical locations and technological levels of these processes, resulting in a strategy for data collection.

- **Functional Unit**

The functional unit, which is the basis for the calculation, is a measure of the performance that the system delivers and also enables alternative products to be compared and analyzed.

- **Assumptions and Limitations**

The assumptions and limitations are very important to each LCA in case of the internal consistency of the study.

- **Allocation Methods**

The allocation methods are used to partition the environmental load of a process when several products or functions share the same process.

- **Impact Categories**

The impact categories represent environmental issues of concern to which LCI results may be assigned. The impact catagories which are selected in each LCA study have to be able to describe the impacts caused by the products being considered or the product system being analyzed.

2.3.3.2 *Inventory Analysis*

Life cycle inventory (LCI) is a methodology for quantifying the flow of material and energy attributable to a product's life cycle (Rebitzer *et al.*, 2004; Reap *et al.*, 2008). The implication of the inventory analysis is that all activities related to the production of one functional unit have to be analyzed concerning about raw material, intermediate, product, usage, and waste removal (Klöpffer, 1997). An LCI analysis includes (Widheden and Ringström, 2007):

- Construction of a flowchart representing the product system according to the system boundaries decided in the goal and scope definition.
  - All material flows are traced from the extraction of raw materials to their release into the environment.
  - All transport operations are also included.
- Data collection for all activities in the product system, followed by data quality-assessment and documentation of the collected data. Both numerical and qualitative/descriptive data need to be collected. The numerical data includes:
  - Inputs: raw materials, auxiliary inputs and other physical inputs.
  - Outputs: products and co-products.
  - Emissions to air and water and waste.

  The qualitative/descriptive data includes:
  - Descriptions of the technology of the process.
  - How and when emissions were measured and their uncertainty.
  - The geographical location of the process/activity.
  - Where inflows come from and outflows go to.
- Calculation of the environmental loads of the system in relation to the functional unit.
  - The numerical data for the activities have to be recalculated to fit the functional unit and summarized into a list of parameters representing the entire life cycle of the product.

- The result of the inventory analysis is the inventory table which is a list of all inputs and outputs per functional unit.

2.3.3.3 *Impact Assessment*

Since life cycle inventory (LCI) provides hundreds of parameters, it is difficult to draw any conclusions from LCI. Therefore, a formal impact assessment has to be performed. Life cycle impact assessment (LCIA) provides indicators and the basis for analyzing the potential contributions of the resource consumptions, waste generations, and emissions in an inventory analysis to a number of potential impacts (Rebitzer *et al.*, 2004). The result of the LCIA is an evaluation of a product life cycle, on a functional unit basis, in terms of several impact categories. According to the ISO 14040 standard for LCIA, the following steps have to be performed in order to convert the inventory data into the environmental impact estimates (Widheden and Ringström, 2007):

- **Impact Category Definition**

Some baseline examples of impact category considered in most of the LCA studies are illustrated in Table 2.1.

- **Classification**

Assignment of LCI result parameters to their respective impact categories, e.g., classifying CO_2 emission to global warming.

- **Characterization**

Modeling LCI impacts within impact categories using science-based conversion factors, e.g., modeling the potential impact of CO_2 and methane on global warming.

- **Normalization**

Relating the characterization results to a reference value in order to be compared, e.g. relating the impacts of the studied product to the impacts of the total amount of pollutants emitted in a region.

Table 2.1 Baseline examples of impact category (Iuga, 2009)

Impact category	Category indicator	Characterization model	Characterization factor
Abiotic depletion	Ultimate reserve, annual use	Guinee and Heijungs 95	ADP[9]
Climate change	Infrared radiative forcing	IPCC model[3]	GWP[10]
Stratospheric ozone depletion	Stratospheric ozone breakdown	WMO model[4]	ODP[11]
Human toxicity	PDI/ADI[1]	Multimedia model, e.g. EUSES[5], CalTox	HTP[12]
Ecotoxicity (aquatic, terrestrial, etc)	PEC/PNEC[2]	Multimedia model, e.g. EUSES, CalTox	AETP[13], TETP[14], etc
Photo-oxidant formation	Tropospheric ozone formation	UNECE[6] Trajectory model	POCP[15]
Acidification	Deposition critical load	RAINS[7]	AP[16]
Eutrophication	Nutrient enrichment	CARMEN[8]	EP[17]

[1] PDI/ADI Predicted daily intake/Aceptable daily intake

[2] PEC/PNEC Predicted environmental concentrations/Predicted no-effects concentrations

[3] IPCC Intergovernmental Panel on Climate Change

[4] WMO World Meteorological Organization

[5] EUSES European Union System for the Evaluation of Substances

[6] UNECE United Nations Economic Commission For Europe

[7] RAINS Regional Acidification Information and Simulation

[8] CARMEN Cause Effect Relation Model to Support Environmental Negotiations

[9] ADP Abiotic depletion potential

[10] GWP Global warming potential

[11] ODP Ozone depletion potential

[12] HTP Human toxicity potential

[13] AETP Aquatic ecotoxicity potential

[14] TETP Terrestrial ecotoxicity potential

[15] POCP Photochemical ozone creation potential

[16] AP Acidification potential

[17] EP Eutrophication potential

- **Grouping**

Sorting and possibly ranking of the indicators, e.g. sorting according to global, regional or local impact or sorting according to high, medium or low priority.

- **Weighting**

Aggregation of characterization results across impact categories into one total environmental impact value in order to generate a single score and also emphasizing the most important potential impact.

2.3.3.4 *Interpretation*

Life cycle interpretation, which occurs at every stage in an LCA, is a process of assessing results in order to draw conclusions. It is a critical evaluation of the whole LCA using mathematical tool such as sensitivity analysis and dominance analysis (Klöpffer, 1997). For example, if two product alternatives are compared and one alternative shows higher consumption of resource and emission of CO_2, an interpretation purely based on the LCI and LCIA data can be conclusive. In other word, the interpretation phase is desirable to prioritize areas of concern within a single life cycle study (Rebitzer *et al.*, 2004). Moreover, it also links the LCA with the applications which are not part of LCA. The International Organization for Standardization (ISO) has defined the following two objectives of life cycle interpretation (Widheden and Ringström, 2007):

- Analyze results, reach conclusions, explain limitations and provide recommendations based on the findings of the preceding phases of the LCA and then report the results of the life cycle interpretation in a transparent manner.
- Provide a readily understandable, complete, and consistent presentation of the results of an LCA study, in accordance with the goal and scope of the study.

The interpretation should include:

- Identification of significant issues based on the results of the LCI and LCIA of an LCA.
- Evaluation of the study considering completeness, sensitivity and consistency checking.

- Conclusions, limitations and recommendations.

2.3.4 Application of LCA

As mentioned, LCA is a method to help quantify and evaluate the potential environmental impacts of products. This implies that LCA can be applied to any applications where the environmental impacts of the complete or part of the product's life cycle are of interest. For instance, LCA can be used in order to identify significant environmental aspects and also provide a baseline for decisions about product improvements in product development projects.

Governmental organizations, non-governmental organizations, and industries have applied LCA in a wide variety of sectors, either autonomously or with the help of research institutes or consultants (Rebitzer *et al.*, 2004). For example, LCA can be used for identifying and improving waste treatment strategy in the nation level. Another application area is marketing. The LCA results can be used to communicate the environmental benefits of a product to customers, e.g., through the LCA-based communication tool environmental product declaration (EPD) (Widheden and Ringström, 2007).

While noting a great importance of LCA in many applications, activities in various industrial sectors and changes in consumer behavior are ultimately the most crucial factors for reducing the environmental impacts associated with products.

2.3.5 LCA and Related Studies on Biofuels

Hsu (2011) performed a LCA of the production of gasoline and diesel from forest residues via fast pyrolysis and hydroprocessing, from production of the feedstock to end use of the fuel in a vehicle. They concluded that although pyrolysis-derived gasoline and diesel have lower GHG emissions and higher NEV than conventional gasoline in 2005, they underperform ethanol produced via gasification from the same feedstock. GHG emissions for pyrolysis could be lowered further if electricity and hydrogen are produced from biomass instead of from fossil sources, as long as the fuel yield does not fall to offset the GHG savings. Based on a pyrolysis process using biomass-derived electricity, the GHG emissions are 62 g km^{-1} traveled for diesel and 74 g km^{-1} traveled for gasoline, and the NEV is 1.51 MJ km^{-1} for diesel and 1.80 MJ km^{-1} for gasoline.

Several LCA studies have also examined life cycle impacts on land use. Requena *et al.* (2010) performed the environmental impact from the production of biofuels whose origin is the oil obtained from sunflower, rapeseed and soybeans by applying the methodology of LCA. The comparison between production processes and waste treatment, They note that the impact produced by the production is always greater than the impact of waste treatment in each of the categories of impact. Note the category of climate change where both impacts are very similar. In this case, the impact of biofuels production is 53.72% and the waste treatment 46.27%. It also emphasizes the category of carcinogens, land use and acidification and eutrophication where the impact from the production is much higher, an order of 75%, than the impact for the treatment of waste.

From the LCA study, they also concluded that the process in which it should be made a greater effort in reducing the environmental impact of biofuel production is in the production of seeds. High impact values on land use observed in the results set the ground exploded during the agricultural process as the worst of the environmental factors analyzed. Minimizing the use of fertilizers and simplifying the labor it will be reduced the damage on the ground derived from the production of seeds.

Börjesson and Tufvesson (2010) analysed biofuels from agricultural crops in northern Europe regarding area and energy efficiency, greenhouse gases and eutrophication. They clearly show the importance of including direct land use changes in the LCAs of biofuels. Depending on whether traditional cropland or unfertilized grassland is used for the biofuel production, the GHG balance may vary by a factor of two, whereas the variation in the contribution to the eutrophication potential will be even larger. This is due to changes in the biogenic emissions of CO_2 and N_2O from the soils, and leakage of nitrate to water, respectively. If peat soils are utilized, the biogenic emissions of CO_2 may increase 10-20 times.

2.4 Land Use Change (LUC)

Biofuel lifecycle analyses traditionally assume that no land use change has occurred. Analyses have been extended to consider the impacts of land use change.

Depending on the location and type of land converted, significant GHG emissions may result. Land use change (LUC) occurs as lands are shifted from one use to another, for example, for urbanization or expansion of agriculture. LUC caused by biofuels can occur both directly, as land use is shifted into biofuel crop production, or indirectly through market responses to supply and demand changes of biofuel crops and other related agricultural commodities. Resulting price fluctuations may incentivize these indirect land use changes (ILUC) elsewhere. There is growing concern about the effects of ILUC on biofuel carbon intensity, due to the difficulties in measuring its direct relationship to increased biofuel production and use, its potentially significant impacts, and the uncertainties surrounding modeling practices (Broch *et al.*, 2013).

Several studies exclude biogenic, soil-derived emissions of nitrous gas (N_2O), induced by nitrogen fertilization. Depending on which land use reference system is chosen, the emission of biogenic CO_2 and N_2O will have a significant impact on the GHG balances of biofuels. The issue of nitrous oxide emissions during the cultivation of biofuel feedstock crops has been discussed intensively in the research community over recent years. The method developed by IPCC is one of the most utilized methods for calculating biogenic N_2O emissions. It is based on the assumption of a linear relationship between the input of nitrogen and N_2O emissions, also including mineralisation of crop residues and indirect emissions from nitrogen leaching and ammonia losses.

There is the importance of indirect land use change. Gallagher (2008) concluded that potential displacement of food and feed production may completely offset the potential reduction of GHG emissions of biofuels. However, assessment of potential indirect land use change and its GHG implications is a very complex and contentious issue.

The net GHG emission arising from LUC is caused by:

1) Change in aboveground vegetation and associated roots and dead organic matter (fallen leaves, branches, fruits, etcetera). The change in the aboveground vegetation can be deducted by comparing and subtracting the amount of carbon stored in the soy area from the amounts of carbon stored in the vegetation

originally present. Multiplying changes in carbon stocks with 44/12 gives the concurrent CO_2 emissions.

2) Change in carbon stocks in soil. Change in soil carbon stocks can be calculated by multiplying the carbon stock originally present with the three change factors:

$$SOC_2 = SOC_1 \cdot f_{land\ use} \cdot f_{tillage} \cdot f_{input}$$

These factors indicate the effects of different mechanisms on soil carbon content. Tillage and annual cropping mean disturbance of the soil structure and exposure of soil organic matter to oxygen and results in oxidation and degradation of the soil organic matter, releasing the carbon stored in the organic matter as CO_2.

On the other hand returning crop residues to the soil and application of manure and green manure all mean organic material is added to the soil organic matter. A small part of the organic material added to the soil will not be degraded and will instead accumulate in the soil, thus resulting in soil generation.

3) Change in N_2O emissions from soil. Changes in vegetation will also result in changes in N_2O emissions. Agricultural soils have a different soil hydrology, receive different amounts of organic and inorganic nitrogen and different amounts of carbon material. As a result the soil chemical reactions producing N_2O also occur with different reaction rate and volume of produced N_2O per unit of time.

The net GHG emissions due to LUC can be visualized as a function of the number of years of allocation - the period of time the once-only LUC related GHG emissions are divided by 20 years is the standard defined by IPCC (Croezen and Kampman, 2008).

Hoefnagels *et al.* (2010) studied the impact of different assumptions and methodological choices on the life-cycle GHG performance of biofuels by providing the results for different key parameters on a consistent basis. These include direct land-use change emissions. This study shows that a wide variation in performance can be found for the same biofuel type depending on reference land, location of crop cultivation and related yields and soil N_2O emissions and used allocation procedure for co-products.

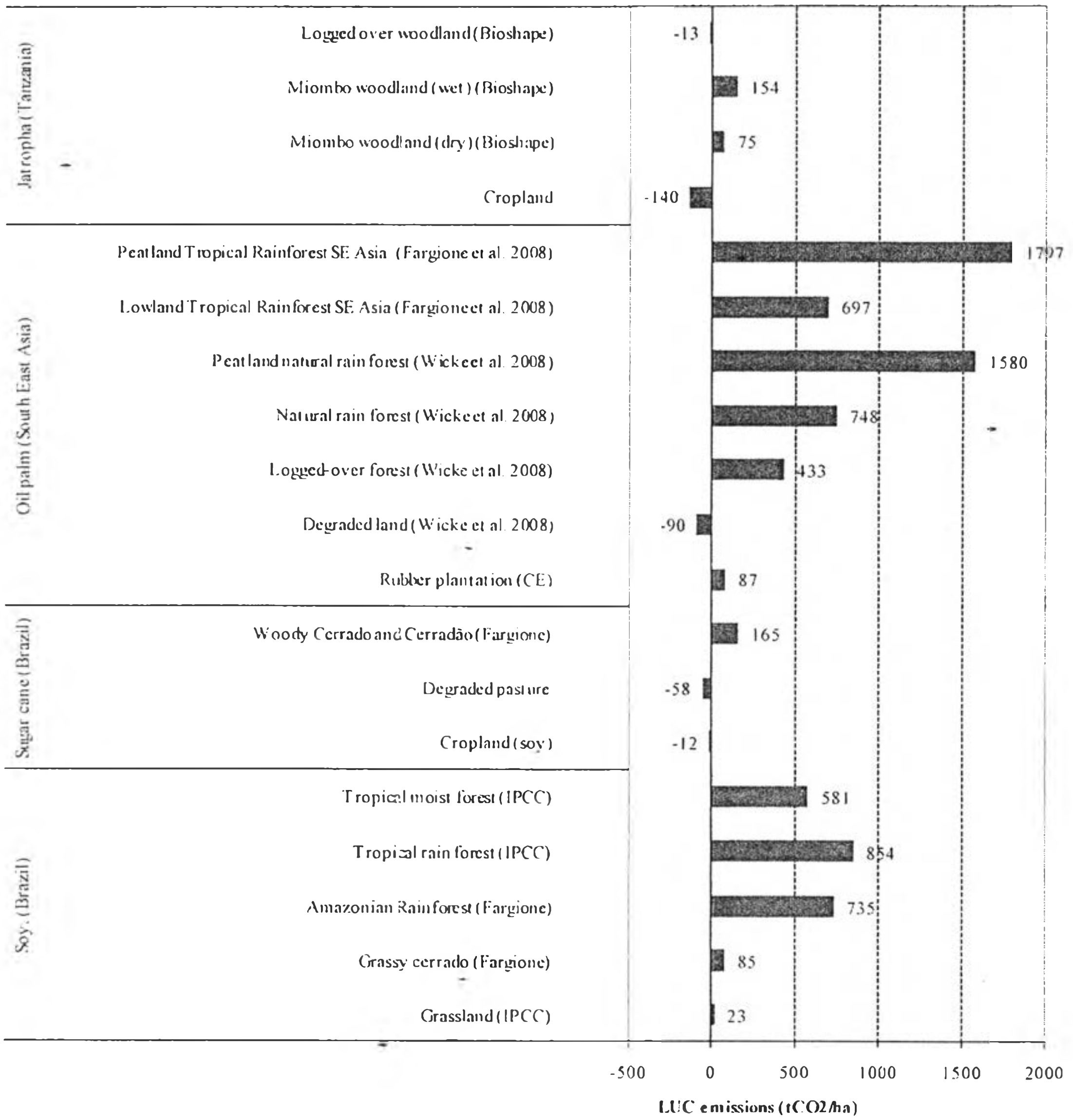

Figure 2.8 Net GHG emissions due to land-use change of various energy plants.

They indicated that those cases are extreme and illustrate the impact of selected land-use reference systems rather than realistic cases. Net GHG emissions due to land-use change of various energy plants and Greenhouse gas emissions from palm oil biodiesel production including land-use change emissions for different land types are shown in the Figure 2.8 and Figure 2.9 respectively.

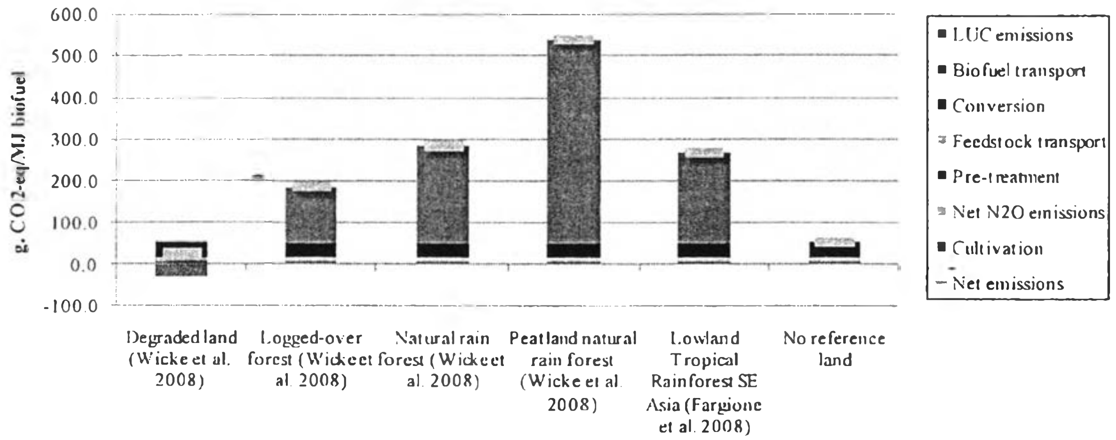

Figure 2.9 Greenhouse gas emissions from palm oil biodiesel production including land-use change emissions for different land types.

Requena *et al.* (2010) compared the environmental impact from the production of biofuels whose origin is the oil obtained from sunflower, rapeseed and soybeans. They find that the impact category most affected is the land use. In the comparative normalization between the production process and waste treatment (Figure 2.10), it is already clear that the two categories of impact where the impact is greater, land use and fossil fuels, again the result of greater impact is produced for production than for waste scenario. In the category of land use biofuel production represents a 75.07% of the total impact compared to 24.92% of the waste scenario, which is related with the greatest impact occurred in the category of ecosystem quality damage.

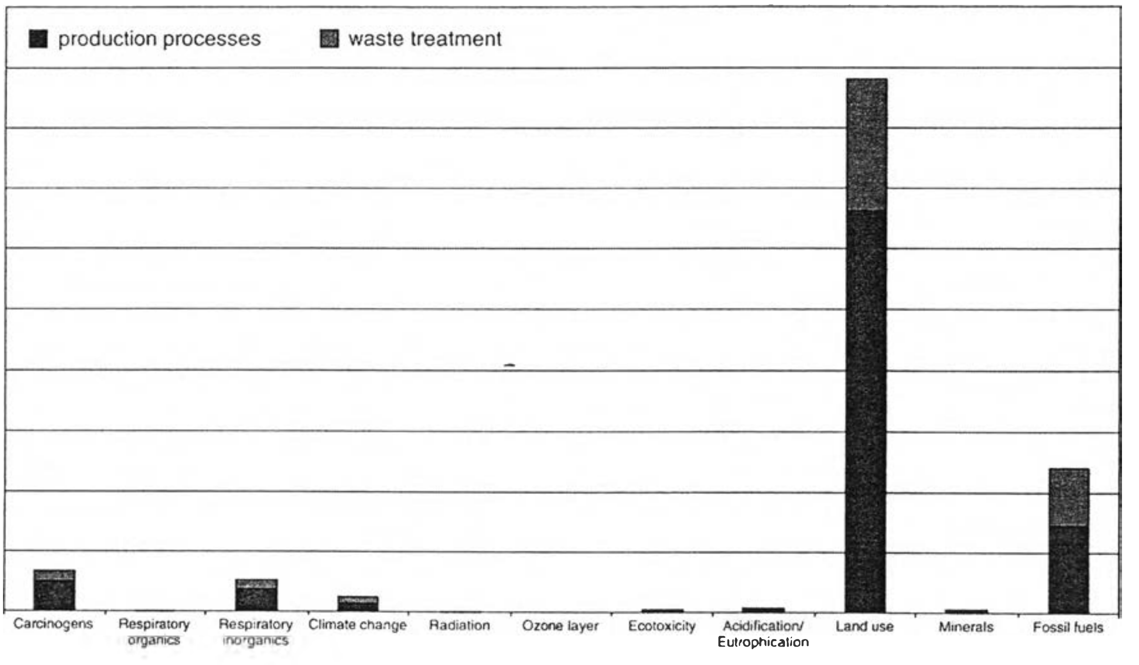

Figure 2.10 Normalization of the comparison between the production process and the waste treatment by impact category.