Towards Sustainability: An Optimisation Framework for the Australian Hardwood Plantation Mid-Thinning Management Using Life Cycle Approach

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ABSTRACT

Australian hardwood plantations cover more than 1 million hectare, which is almost half of the total national plantations area. Hardwood plantations are commonly slow to mature, with a long rotation time potentially reaching 35 years. Pruning and thinning processes are required during the early stage of the plantation to produce high-quality logs for commercial purpose. Approximately, 50% of the trees are typically cut from the third year during the first thinning, with further 30% removed during the second thinning (10 to 15 years). It is estimated that more than 2 million cubic metres of logs are generated annually during the plantation mid-rotation thinning operation in Australia. However, these logs are considered as low commercial value product because of high defect ratio and poor mechanical quality. In order to ensure the continued expansion of the Australian hardwood plantation sector, higher value products need to be developed to maximise the utility of the available resources and to reduce wastage. The use of the low commercial value thinned logs may offer an opportunity to improve the environmental and economic performance of the hardwood forestry sector. Therefore, this study focused on the potential utilisation pathways for logs produced during the second thinning operation.

The study used the case of South-east Queensland to demonstrate the framework, and it was conducted following the standardised methods for Life Cycle Assessment (LCA) (ISO14040:2006) and Life Cycle Costing (LCC) (AS/NZ4536:1999 R2014). To provide an equitable comparison among the different alternatives assessed in this study, the functional unit was defined as the treatment of 1 Mg of green timber logs from the second thinning at the plantation floor. Both the LCA and LCC analysis were conducted based on a 60-year timeframe. The OpenLCA 1.4.1 and SimaPro v.8.0.4.30 software were used in the simulation. Primary data were used whenever possible to calculate the life cycle inventory. Eco-invent 3.3 and AusLCI databases were used to model the background processes and in cases when the primary data were missing. The lifecycle impact assessment was processed followed the best practice guide for conducting LCA studies in Australia and included five different environmental impact categories: Global Warming (GWP); Eutrophication (EP); Acidification (AP); Fossil depletion (FDP) and Human Toxicity (HTP). Excel spreadsheet was used to conduct the LCC analysis and calculate the present value of the future costs. A constrained stepwise, multi-objective linear programming (LP) model was then constructed to identify the optimal solution for the plantation thinned logs utilisation. The LP problem was solved using LINDO software package over a 60-year period. The time intervals used was ten years. The constraints of the model were classified into three groups: the total feedstocks availability; final product requirements; and environmental and economic targets.

A number of pathways may be followed to valorise the thinning logs including the production of engineered wood; solid fuel and biomass to liquid fuels (BTL). This study examined several products under each utilisation pathway in an attempt to identify the optimal combination to maximise the environmental and economic benefits from a lifecycle perspective. Under the engineered wood options, two possible utilisations were assessed: (1) veneer based composite (VBC) utility poles and (2) structural building frame using laminated veneer lumber (LVL). The LCA results indicated that engineered wood products manufactured from the second thinning could offer tangible environmental benefits compared to the conventional construction materials such as concrete and steel. The study highlighted that resin consumptions during engineered wood manufacturing, and use of preventatives for wood treatment were the major contributors to the environmental impacts of engineered wood. Furthermore, for the end of life treatment, incineration with energy recovery rather than landfilling was identified as the most favourable waste management option to reduce the environmental impacts. Additionally, using LVL in multi-level building structural frame presents greater environmental benefits than the VBC utility poles case. The longer durability and higher material efficiency of the LVL option compared to the VBC option contributed significantly to the better environmental performance.

Under the energy utilisation pathway, six options were assessed: woodchip gasification in combined heat and power plant (WCG); wood pellets gasification in a combined heat and power plant (WPG); wood pellet combustion for domestic water and space heating (WPC); pyrolysis for power generation (PyEl); pyrolysis with bio-oil upgrading to transportation fuels (PyLT) and ethanol production for transportation fuel mix (EthP). All the bioenergy conversion options had noticeable environmental benefits, particularly on the GWP impact. The WCG option was identified as the best performer followed by the WPG and then WPC. The carbon offsets due to the displaced fossil fuel from electricity and heat generation were the main reason for lowering the GWP impact of these options. Although wood pellets have higher energy density than woodchips, they required additional manufacturing processes, extra energy consumption and additional transportation leading to higher environmental impacts which could not be offset by the improved energy density. The study highlighted that use of the biomass as solid fuel with the least processing requirement had higher environmental benefits than BTL options. Lastly, the environmental benefits gained are dependent on the energy being displaced by the final product.

From a life-cycle cost perspective, the results showed that per functional unit, utilising thinned logs to produce engineered wood products was more likely to result in higher LCC than energy production. Overall, the VBC utility pole option had the highest LCC followed by the LVL building frame option. The shorter lifespan and lower material efficiency of the VBC compared to LVL option was a significant contributor to the higher LCC of the VBC option. In regard to the energy conversion, the WCG had the least LCC per functional unit among all options, followed by the EthP and PyEl options. However, it is important to note the shortcoming of the functional unit considered here was based on input (treatment of 1 Mg of thinned logs). To overcome this limitation, the output should be considered. For the energy pathway, the levelised cost per megajoule energy was calculated, which showed that the WCG had the best performance followed by the WPG and then WPC. This is in line with the LCA results. Nevertheless, the energy output does not offer a valid comparison for the engineered wood pathway. Therefore, the costs of the displaced products by the output were considered, in line with the LCA analysis, to provide a more equitable comparison. The concept of 'substituted values' (SV) was then introduced in this study to overcome this limitation. The 'SV' borrows the concept of 'displaced emissions' from the LCA. The SV was calculated as the difference between the LCC of the final product (alternative) and the displaced (substituted) product. When the substituted materials were considered, the engineered wood products presented the highest economic savings while the energy options continued to pose a cost. Nonetheless, the WCG option continued to have the best economic performance among the energy options.

The LCA and LCC analysis showed conflicting results regarding the best option to follow in order to maximise the economic and environmental utility of the plantation thinning. Based on equal weighting of the economic and environmental objectives, the multi-objective optimisation (MOO) program solution indicated that the LVL and WCG were the dominating options. However, the percentage allocated to each option varied in different periods. For the first 30 years, the solution favoured energy production with 85% allocation to the WCG in the first 10 years period. Nevertheless, the percentage allocated to energy production declined progressively over the subsequent periods. The LVL option became more dominant starting from the 30th year of the simulation. The share allocated to the LVL option reached 100% of the biomass during the last 10 years period. However, when the environmental weight exceeded 70%, energy production (particularly the WCG option) became the dominant solution while the LVL option became less favourable with only 2% allocation of the biomass. On the other hand, the VBC option did not feature in the solution until the weight assigned to the economic objective exceeded 80%.

This research study is relevant nationally and internationally as it presents a novel method to integrate LCA with LCC analysis in a multi-objective optimisation framework to identify the optimal utilisation of forestry products including multiple utilisation pathways. This study also presents a novel method to simplify the complex optimisation problem by converting the MOO question to a single objective optimisation (SOO) problem. The framework introduced three novelties: (a) introduce normalisation factors for better representation of the Australian situation; (b) incorporate the potential economic credits from alternative substitutions into the LCC; this is in line with the accepted accounting methods of LCA to allow credits for offset emissions, and (c) formulate and solve the problem as a stepwise constraint LP to avoid over/under allocation issues resulting from averaging over the timescale.

Although the study focused on the South-east Queensland case, the developed method can also be applied in different industry sectors and locations. The outcomes of this research have major implications on Australian forestry sector, particularly on hardwood plantation management. The developed optimisation management strategy can enhance the current economic profitability of the forestry sector by developing new markets for the low value thinned logs from timber plantation while increasing the utilisation rate of wood waste hence to satisfy the global rising timber demand and accumulating global carbon. The results of the study are also relevant to other timber residues and low-grade products from the softwood and pulp-wood plantations. In addition, results of this study can be potentially used by decision-makers to make informed choices to lower the environmental impacts and lifecycle cost of utility infrastructures systems and buildings. Furthermore, this study has implications for the bio-energy generation sector. This study confirms the feasibility of using forestry residue as feedstock to substitute fossil fuel energy while mitigating negative environmental impacts and achieving sustainable development strategy. Globally, this study is a small yet significant contribution towards the achievement of the United Nations Sustainable Development Goals, specifically under the Affordable and Clean Energy; Climate Action and Responsible Consumption and Production.

STATEMENT OF ORIGINALITY

This work has not previously been submitted for a degree or diploma in any university. To the best of my knowledge and belief, the thesis contains no material previously published or written by another person except where due reference is made in the thesis itself.

(Signed)______________________________

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This thesis is conducted by a series of papers. Chapter 3 to Chapter 7 of this thesis that have been published, or submitted for publication are listed below:

Journal Papers

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- Lu, HR. & El Hanandeh, A. (2016). Life Cycle Assessment of ACQ-treated veneer composite (VBC) Hollow Utility Poles from Hardwood Plantation Mid-thinning. *Sustainable Production and Consumption*, 5, 36-50. doi:10.1016/j.spc.2015.11.002. (2016 Cite Score: 2.73)

Conference Papers

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- Lu, HR., El Hanandeh, A., Gilbert, BP. (2015). ACQ-Treated Veneer Based Composite (VBC) Hardwood Hollow Utility Poles from Mid-rotation Plantation Thinned Trees: Life Cycle GHG Emissions. *Second International Conference on Performance-based and Life cycle Structural Engineering (PLSE 2015)*, Brisbane, QLD (Australia), 9-11 December 2015.

Publication Award

The paper entitled: "Environmental and economic assessment of utility poles using life cycle approach" (published in October 2016), has received the **Outstanding Paper Award** for 2016 in Journal Clean Technologies and Environmental Policy.

ABBREVIATIONS

ABS: Australian Bureau of Statistics

AP: Acidification Potential

AusLCI: Australian Life Cycle Inventory

CH_4: Methane

CO: Carbon Monoxide

$CO_{2\text{-eq}}$: Carbon Dioxide Equivalent

EP: Eutrophication Potential

FDP: Fossil Depletion Potential

GHG: Greenhouse Gas

GJ: GigaJoule

GWP: Global Warming Potential

HTP: Human-toxicity Potential

ISO: International Organisation for Standardisation

KWh: Kilowatt Hours

LCA: Life Cycle Assessment

LCC: Life Cycle Costing

LCI: Life Cycle Inventory

LCIA: Life Cycle Impact Assessment

LVL: Laminated Veneer Lumber

MJ: Mega Joule

MOO: Multi-Objective Optimisation

N_2O: Nitrous Oxide

NPV: Net present value

SOO: Single-Objective Optimisation

VBC: Veneer Based Composite

CHAPTER 1 - INTRODUCTION

This chapter provides a general overview of the research and describes the existing issues faced by the Australian hardwood plantations with a special focus on the mid-rotation thinning. Different existing or potential valuable applications for the utilisation of low quality thinned logs are identified and discussed. The research aims and objectives are also included, followed by the research motivation and thesis layout.

1.1 Wood Plantation

The Australian plantations area increased rapidly between the 1950s and the 1980s due to government investment in the domestic softwood resources. Since the 1990s, the plantation estate has increased by over 50% (500,000 ha) and reached 1.5 million hectares by 2008. Australian plantation estate saw further expansion phase between 2008 and 2010 when it reached approximately 2.0 million ha. Successive Australian governments and the forestry industry set a target to double the existing area by next decade (Australian Government Department of Agriculture Fisheries and Forestry, 2010).

Timber plantations are established mainly in response to the rise of wood consumption and concern over losing natural forests (Sedjo and Lyon, 2015). Population growth increases demand of all products and services from forests, including wood and wood fibre, which results in a dramatic decrease of natural forest area. Plantations provide an opportunity to protect natural forests by providing alternative sources of timber (Sedjo, 1999). In many regions, the loss of natural forests has been offset somewhat by a rapid increase in the amount of forest land being allocated to plantations (Paquette and Messier, 2010).

Nowadays, more than 15% of the global timber products come from plantations (Paquette and Messier, 2010). Although the existing plantation area is relatively small compared to natural forests, the industrial plantations are major suppliers for global timber demand (e.g. 17% of the total pulpwood and 4% of the sawlog volume) (Cossalter and Pye-Smith, 2003). Timber plantations are more productive than natural forests (Cossalter and Pye-Smith, 2003). The annual productivities are reported to be between 10 m^3/ha and 25 m^3/ha of logs (Cossalter and Pye-Smith, 2003). Sedjo (1999) noted that some hardwood plantation could even achieve a much higher production rate of 60 m^3/ha per year.

Timber plantations are also seen as carbon sinks due to their role in the carbon cycle (Mitchell et al., 2012). Like native forests, plantations contribute significantly to carbon sequestration (Nijnik et al., 2013). Dry wood contains approximately 50% carbon by weight (Chazdon et al., 2016).

Hence, theoretically, 3.7 kg of CO_2 are sequestrated from the atmosphere for each kilogram of carbon captured in the forest biomass. Thus, forest plantations are substantial carbon stores (England et al., 2013). The stored carbon is released after trees harvesting and decaying. Periodic harvesting for long-lived timber applications can significantly increase the carbon storage (Laird et al., 2009, Fan et al., 2011, Thakur et al., 2014, Shen et al., 2015). Carbon stored by the Australian plantation forests has increased during the last decade from 137 Mt to 171 Mt in 2010 (Gavran, 2013). Nunery and Keeton (2010) emphasised that most absorbed carbon would be stored in the wood after harvesting, while only a short-term loss of stored carbon might be experienced after converting timber to wood products. Nevertheless, much of the carbon will be transferred to the wood products pool (Ximenes et al., 2015). The stored carbon from wood products will slowly return to the atmosphere during the wood decay in a landfill over several decades (Ximenes et al., 2015).

Timber plantation is a source of minimal impact renewable material. Utilisation of wood to substitute high energy intensive materials such as steel, concrete, and plastic have been considered high energy efficient and help to avoid releasing GHG emissions from energy consumption associated with alternative materials (Gustavsson et al., 2006). Bolin and Smith (2011) stated that from a life-cycle perspective, less energy is required during manufacturing of wood products than alternatives. Replacing fossil energy by woody biomass is another potential way to reduce GHG emissions and other environmental impacts such as acidification and eutrophication (Bolin and Smith, 2011). Raymer (2006) estimated the GHG emissions avoided per cubic metre of wood used as a woody fuel compared to fossil fuel to be between 0.21 to 0.64 tonne CO_2-equivalents.

In addition to the marketable timber outputs and environmental savings, plantations could provide a wide range of social benefits to the local society. For instance, establishing new plantation could result in providing substantial levels of employment through the development of new industry sectors for the local community. Schirmer et al. (2005) estimated that for each million Australian dollar spent by the plantation sector; approximately 20 job positions could be created. Another study undertaken by The State of Queensland (2010) noted that the softwood plantation timber processing sector in Queensland directly employed almost 1800 people and a further 670 employees in contracting businesses. Meanwhile, utilisation of locally produced raw materials could enhance forestry-related sectors by reducing dependence on imports (Gan and Smith, 2007). The non-market benefits from expansion of forestry also include the protection of sites and landscapes of high cultural, spiritual or recreational value (Willis et al., 2003). Also, there has been an increasing recognition that the forestry could be an important sink for pollutants. Plants facilitate the uptake, transport and assimilation or decomposition of many gaseous and particulate pollutants (Willis et al., 2003). Willis et al. (2003) also estimated that the net pollution absorption by woodland could result in reducing the number of deaths brought forward by air pollution by between 59-88 deaths and between 40-62 hospital omissions based on 1 km^2 scale.

Furthermore, forestry has potential to improve biodiversity (particularly plantations comprising native species) when compared to some other commercial land uses (The State of Queensland, 2010). Abundant evidence shows that forests plantation can contribute to the conservation of biodiversity and help restore the degraded land (Brockerhoff et al., 2008). Large areas of land are degraded annually due to human activities, causing significant concern for biodiversity (Brockerhoff et al., 2008). Loss of biodiversity, soil erosion, and the destruction of natural forests are among the major ecological impacts the world is facing now. Recently, forest plantation has been recognised as a way to provide habitat for a wide range of animals and fungi (Carnus et al., 2006). Establishing plantation on an impoverished site could result in increasing biodiversity by receiving new vegetation and return of wildlife (Brockerhoff et al., 2008). In Australia, a range of forest plantations have been established on degraded rainforest landscapes with the purpose of restoring the degraded land and biodiversity (Kanowski and Catterall, 2010). Maintaining and enhancing these functions is an integral part of sustainable forest management.

1.2 Australian Hardwood Plantations Thinning

Approximately 50% of the Australian plantations are hardwood species (991,000 hectares) (May et al., 2012). Hardwood plantations usually have a long-rotation. Hardwood trees are valuable species with special technical properties, such as natural strength, durability, and hardness, which make them appropriate for high-value applications (Ferguson, 2014). Wood plantation involves a sequence of management decisions regarding the timing and cost of forest operations. The plantation operations include site preparation, establishment, early age silviculture, thinning and final harvesting (May et al., 2012). Plantation thinning is a process to remove poorly formed trees to allow the remaining trees to steadily grow (McKenna and Woeste, 2004). The first thinning usually takes place around the third year after the hardwood plantation is established based on the growing conditions. Approximately 50% of trees would be removed to increase the amount of light, moisture, and nutrients available to improve the growing conditions for the remaining trees. The second thinning usually takes place when the trees are around 10 to 15 years old. The aim of the second thinning is to remove substandard or poor quality trees and reduce the stock rate by 15% to 20% of the total. The remaining trees are allowed to reach maturity and harvested when they are between 30 and 35 years old (Underhill et al., 2014). The removed trees during the second thinning typically have a Diameter at Breast Height Over Bark (DBHOB) of 150 mm to 300 mm. These trees usually have high defect rate, poor mechanical quality and low commercial value (Underhill et al., 2014).

Hardwood plantations are commonly known as slow to mature with low financial returns in the early stage. The slow return may be attributed to the handling of these small trees, which have relatively low financial value but have a relatively high operational cost per unit harvested (Spinelli and Magagnotti, 2010). The thinning process also requires significant investment while contributing negligible financial return to plantation operators (Paul et al., 2013b). Thus, valuable products are required to be developed to satisfy the current market to ensure the continued expansion of Australian hardwood plantation industries. The use of the low commercial value logs from hardwood plantation mid-rotation thinning is an opportunity to improve the Australian forestry profitability and win new markets (Underhill et al., 2014).

1.3 Valorisation Options for Hardwood Plantation Thinning

The utilisation of small-diameter trees could alleviate the shortage of high-quality timber such as sawn timber by providing alternatives and displacing the usage of high-quality timber for low-value applications. Traditionally, to keep the logs in their natural round-form is the most cost-effective way to manage these thinned logs. An earlier study showed that the utilisation of debarked thinned logs as vineyard posts is one of the most popular applications (Nolan et al., 2005). Other applications such as timber piles or posts for the building and landscaping industries were also mentioned by Nolan et al. (2005). However, these markets are usually dominated by softwood. Although the hardwood logs from mid-rotation thinning have advantages on their durability and mechanical properties compared to softwood, the problems of excessive splitting and checking need to be addressed for these logs to compete favourably (McGavin et al., 2013). Underhill et al. (2014) suggested that hardwood plantation thinned logs might be used for solid wood applications in the fabrication of furniture. Other industrial applications include sawn timber roof trusses and pallets, which were successfully manufactured from mid-rotation hardwood thinning (McGavin et al., 2006b). However, McGavin et al. (2006b) reported that more than 60% rejection rate occurred due to defects in the material or subsequent material processing. Although timber is still extensively used in its solid form, the use of solid timber has obvious limitations. For example, the availability and the length of natural timber sections that cannot meet the spans required for modern building components (Barbu et al., 2017). Hence, there is a strong need for developing viable products from mid-rotation thinning, which if successfully achieved could re-invigorate an entire industry (McGavin et al., 2006b).

1.3.1 Engineered Wood Products

Modern technology has managed to remove many of the quality limitations present in the natural wood. Through lamination and other technologies, wood durability and mechanical qualities can be improved (Gilbert et al., 2017). The development of new fasteners also contributed to better load carrying capacities of engineered wood, as well as the more consistent performance of wooden sections (Issa and Kmeid, 2005). Engineered wood products are relatively new structural wood materials. During the past decade, engineered wood products have experienced rapid growth and acceptance in the marketplace (LeVan-Green and Livingston, 2001). This kind of wood-based product is manufactured by bonding wood boards, veneers, strands or flakes using adhesives to form panels or other shaped structural products (Barbu et al., 2013). Engineered wood products enhance the quality by converting logs with defects to more uniform structural products, resulting in higher strength and less variability in mechanical properties compared to sawn timber (Barbu et al., 2013). Nowadays, engineered wood products are increasingly being accepted as a cost-competitive building material (Bribián et al., 2011, Kasal et al., 2015). The examples of engineered wood products include Laminated Veneer Lumber (LVL), plywood, oriented strand board (OSB), as well as Veneer Based Composite (VBC) products.

1.3.2 Woody Biomass Energy Applications

The energy generation sector in Australia is highly reliant on cheap fossil fuels which are a major contributor to GHG emissions. In order to achieve an effective reduction of GHG emissions, the Australian government mandated using renewable energy (Byrnes et al., 2013). Renewable energy sources are increasingly becoming important in the global energy supply (Chu and Majumdar, 2012). The International Energy Agency predicts that renewable energy will increase to supply 15% of the global energy demand by 2030, and at the same time, biomass will be the most important renewable energy source, accounting for more than 70% of the total renewable energy supply (Moriarty and Honnery, 2012). Forestry biomass is seen as an effective renewable, low-carbon energy source because it is sourced from harvested vegetation that sequestered carbon from the atmosphere during growth (Cherubini et al., 2009). It can be derived from silvicultural thinning operations, residues from merchantable trees, mill wastes, and forestry operations (Esteban et al., 2008). Woody biomass can be converted into many forms, such as solid, liquid, and gaseous fuels, which make it suitable for a wide range of applications (Picchio et al., 2012). In comparison with other biomass sources, woody biomass is available throughout the year from various sources, which could provide a more reliable feedstock and eliminates the long-term storage issue associated with agricultural residues (Castellano et al., 2009, Zhu and Pan, 2010). Additionally, utilising woody biomass sourced from short rotation forestry systems has considerable benefits, since wood can be continually replenished, which has the potential to provide a cost-effective and sustainable supply of energy (Whittaker and Shield, 2016). Using forest biomass could also improving financial returns through timber plantation management (Munsell and Fox, 2010). As a result, the bioenergy market may help to improve the financial state of silvicultural products and improve the value of the remaining growing stock in commercial forests.

In Australia, the Renewable Energy Target (RET) is currently implemented, and the proposed Clean Energy policy (CEP) provides a rich platform to increase the utilisation of biomass as an energy source (Australian Government Department of the Environment and Energy, 2015b). Significant expansion of biomass-based power-generation especially from short rotation plantation is projected for Australia (Sochacki et al., 2007). There is a potentially large supply of low-cost residual woody biomass from sawlog-driven harvesting, thinning, and wood processing. The logs generated from wood plantation represent a potential raw material for the production of bioenergy.

The existing technologies allow woody biomass to be either directly used through combustion and gasification, or indirectly by upgrading to other forms, such as wood pellets, bio-oil, or liquid biofuels to substitute fossil fuel (Searcy et al., 2007). However, due to the high moisture content, irregular shape and size, as well as low bulk density; woody biomass is usually difficult to handle, transport, store, and utilise in its original form (Picchio et al., 2012). For effective use, the large size woody biomass materials are required to be converted into smaller size products with higher energy density such as woodchips, pellets, or briquettes (Picchio et al., 2012). Wood chipping process is an essential procedure of all modern wood energy conversion chains because most of the boilers only accept homogeneous fuel particles within specified size limits (Picchio et al., 2012). Wood chipping also helps to increase load density and improve handling quality (Spinelli et al., 2011a). The energy conversion efficiency of woodchips depends on the moisture content, ash content, and particle-size distribution (Spinelli et al., 2011b). The green woodchips moisture content is between 50% and 60%. Therefore, drying is required to reduce transportation cost and improve energy density. Wood pellets can be manufactured from the chips to further increase the energy density, uniformity, reduce moisture content, improve storage and handling and reduce transportation cost (Goh et al., 2013, Tarasov et al., 2013).

Another energy utilisation of the biomass is to produce liquid fuels. Bio-oil produced from the pyrolysis of woody biomass is a promising alternative to fossil crude oil (Xiu and Shahbazi, 2012). Fast pyrolysis is commercially available, and it has been promoted as an effective technology for converting biomass into bio-oil, char, and gases (Zhong et al., 2010). The produced char and gases are separated and usually burned for process heating. The crude bio-oil can be directly used through co-combustion in conventional fossil fuel power plants or existing industrial boilers for electricity and heat production (Fan et al., 2011). However, Chang et al. (2013) noted that the chemical composition of the pyrolytic bio-oil is different from that of petroleum-derived oils due to the high moisture content and oxygenated organic compounds content. Therefore, hydro-treatment is usually applied to upgrade the low-quality crude bio-oil to petroleum and petroleum-like products, which can be directly used as a substitute for gasoline or diesel (Xiu and Shahbazi, 2012).

Bio-ethanol produced through fermentation of sugars and starches, also known as first generation biofuel, is the most widely used bio-fuel for substituting gasoline in transportation sector so far (Demirbas, 2011). It is continuing to be developed as a transport fuel produced in several countries and traded internationally, for use primarily as a gasoline additive (Balat and Balat, 2009). However, bio-ethanol can also be produced from lignocellulosic materials and is known as second-generation bioethanol. The second generation bioethanol presents economic and environmental advantages over first generation bioethanol (Alvira et al., 2010). The second generation bioethanol also overcomes the moral argument over food for fuel because it uses residues and non-edible products. During the fermentation process, biomass is ground down and the starch converted by enzymes to sugars, with yeast then converting the sugars to ethanol (Naik et al., 2010). Nowadays, the production and use of bio-ethanol fuel from fast-growing wood crops via fermentation as alternatives to conventional gasoline are seen as an opportunity to reduce GHG emissions and fossil fuels consumption (de Paula Gomes and de Araújo, 2009, González-García et al., 2012). However, Tuna and Hulteberg (2014) indicated that bio-ethanol produced by fermentation is far behind the thermo-chemical synthetic fuels due to the higher energy consumption and the lower conversion rate. The theoretical conversion efficiency for ethanol is limited to about 48%, while the current pre-commercial installations are however far from reaching these fuel yields (Pa et al., 2011). Furthermore, the current production cost of ethanol is usually higher than that of gasoline (Demirbas, 2011). That is mainly because of the high the costs for feedstocks and plant investments (Hamelinck et al., 2005, Piccolo and Bezzo, 2009, Demirbas, 2011).

1.3.3 Other Potential Utilisation Pathways

The char produced from woody biomass via pyrolysis can be a potential soil amendment to improve the physiochemical properties of soils and crop yields and reduce GHG emissions (Greenhalf et al., 2013, Irfan et al., 2016). Up to 50% of the carbon in woody biomass can be locked in stable biochar. Biochar is also a highly stable substance with a high capacity to retain nutrients (Greenhalf et al., 2013). Furthermore, recent research showed the evidence of biochar being able to absorb herbicides and metals (Park et al., 2011). Hammond et al. (2011) and Ibarrola et al. (2012) noted that the biochar for soil amendment could even achieve higher GHG emissions reduction than direct combustion for bioenergy substitution. However, Peters et al. (2015) and El Hanandeh (2011) argued that the utilisation of pyrolysis char for energy generation was more favourable than its use as biochar for soil system when other environmental aspects such as acidification, eutrophication, and abiotic depletion impacts are considered. On the other hand, research has shown that there is high uncertainty in the economic profitability of biochar to soil systems (e.g. Galinato et al., 2011, Shackley et al., 2011, Yoder et al., 2011, Homagain et al., 2016).

Composting is another option to manage biodegradable biomass. The biomass composting is based on the aerobic degradation of organic matter, which involved several biological processes in producing the final product (i.e. compost) (Vandecasteele et al., 2016). The compost can be used for different purposes, such as agriculture, horticulture, and landscaping fraction. However, compared to other biomass materials, woody biomass is usually recalcitrant and relatively difficult to be digested (Zhu and Pan, 2010). From the environmental point of view, although biomass composting may offer considerable environmental benefits, applying biomass for energy production may achieve better results, particularly when displacing electricity from fossil fuel (El Hanandeh, 2015). In addition, recent studies also found that the woody biomass composting might be not economically viable without external subsidies due to the high feedstock cost (Galgani et al., 2014).

1.4 Aims and Objectives

Wood product substitution is an important long-term management strategy for mitigating climate change. Increasing use of wood-based material could reduce the net GHG emissions from the building and construction sector because of the relatively low energy requirement during manufacturing stage compared to other materials, such as concrete and steel (Gustavsson et al., 2006). In addition, wood substitution in long-life-span products results in accumulated carbon storage (Sathre and Gustavsson, 2009), which is seen as a potential climate mitigation strategy (Cabeza et al., 2014). Buchanan and Levine (1999) indicated that wood building required much lower process energy and released less GHG emissions than other materials such as brick, steel, and concrete for typical forms of building construction. Goverse et al. (2001) claimed that 50% reduction in CO_2 emissions could be achieved technically by utilisation of timber to substitute concrete in-house structure and other elements design, such as walls and windows frame. Buchanan and Levine (1999) estimated that 17% increase in wood usage in the New Zealand building industry could result in 1.5% reduction in national total GHG emissions. However, the environmental impacts caused by wood product manufacturing, mainly due to chemicals consumption in wood preparation, is a concern (Rivela et al., 2007, Gonzalez-Garcia et al., 2009). Meanwhile, using woody bioenergy instead of natural gas, conventional gasoline, lignite and other fossil fuel or non-renewable energy resource can deliver environmental benefits (Lindholm, 2010, Valente et al., 2011, González-García et al., 2012). From the economic point of view, converting forest biomass into bioenergy products is seen as an opportunity to generate additional revenue streams for the forestry sector, while improving the economic viability of thinning and other forestry management operations (Cambero and Sowlati, 2014). Fuel production from forestry biomass is also consistent with the Queensland Bio-futures plan (The Department of State Development, 2016). Biofuel is cost competitive energy source and provides net benefits on its environmental cost (Hill et al., 2006). However, the actual economic and environmental performance depends largely on the energy conversion technology followed.

Due to the recent global economic downturn and the increased awareness of environmental issues, reducing the life cycle costs and lowering the environmental impacts of products or processes is a key strategy for industries to survive in the future market (Deng et al., 2013). Therefore, the economic and environmental performance of potential technologies is fundamental to determine the best management approaches to the forestry sector. Life cycle assessment (LCA) is widely used as an environmental tool for decision-making, while Life cycle costing (LCC) is an economic analysis tool that can be integrated with LCA (Treasury NSW, 2004) to make sustainable decisions. In order to maximise the utility of the plantation early stage, an optimisation framework is required to be developed.

Therefore, this study aimed to optimise the economic and environmental performance of the hardwood plantation by maximising the second thinning utility. The following research questions were developed:

1. What are the feasible applications from the utilisation of hardwood thinned logs?

2. What are the environmental performances of these identified applications for managing timber from mid-thinning?

3. How does the environmental performance of the identified applications compare to their traditional alternatives?

4. What are the life cycle costs of these applications?

5. Which optimisation approach is suitable for identifying the optimal utility of the resources?

6. Which assemblage of management strategies would result in the optimal utilisation of the resources?

Consequently, the following objectives need to be achieved in order to realise the research questions.

1. To identify value add timber applications from hardwood plantation mid-thinning, which realise both environmental and economic benefits.

2. To assess the environmental performance of different applications (identified in Objective 1) using life cycle assessment method.

3. To assess the economic performance of different applications (identified in Objective 1) using life cycle costing method.

4. To build a mathematical optimisation model with the objectives of minimising life-cycle environmental impact and life-cycle cost of the system.

5. Solve the mathematical model to identify the optimal combination of applications that realise the lowest environmental impact and life-cycle cost.

This study focuses on the South-east Queensland hardwood plantation. Therefore, the hardwood plantation thinning, timber application manufacturing technologies, materials, and transportation considered in this research are typical for the South-east Queensland, Australia.

1.5 Thesis Layout

This Ph.D thesis is presented as a series of papers. The thesis includes an extended abstract followed by eight main chapters and references. Figure 1.1 shows the conceptual framework of this research.

The first chapter provides a brief overview of the current situation of hardwood plantation in Australia, as well as the importance of developing appropriate applications to utilise hardwood thinning in early stages effectively. A background of Australia hardwood plantation statistics, current issues of hardwood plantation in Australia both from environmental and economic perspectives are included. This chapter also identifies potential applications for the utilisation of wood from forestry thinning. Finally, the aims and objectives and thesis layout are presented. A summary of the thesis chapters is also outlined.

Chapter Two discusses the tools and techniques used in this study. Life cycle assessment software and life cycle costing model are introduced. The formula of the optimisation model system is described, while the objective function of the model, constraint, and variables are defined.

Chapter Three and Four focus on one of the viable utilisations of wood thinning that consists of two manuscripts: (Lu and El Hanandeh, 2016a, Lu and El Hanandeh, 2016b). The first manuscript (Chapter Three) presents an LCA of manufacturing veneer-based composite (VBC) hollow wood utility poles from hardwood mid-rotation thinning. The detailed hardwood plantation thinning process and engineered wood product manufacturing processes were discussed and assessed. Three different end-of-life scenarios including landfilling, energy recovery and recycling for particleboard were analysed and compared. The second manuscript (Chapter Four) extended the first study by integrated LCA with LCC analysis. The economic benefits from carbon sequestration were also taken into account. The VBC utility pole was then compared to the conventional materials for the manufacture of utility poles. The results showed that VBC could be a viable and sustainable alternative to conventional options from both environmental and economic perspectives.

Chapter Five consists of the manuscript of the published journal article: (Lu et al., 2017b). The manuscript applies LCA and LCC to evaluate options for Australian residential multi-stories apartment building frame construction. The evaluated options included the LVL manufactured from hardwood plantation thinning (a newly developed material), final harvesting and softwood plantation as well as steel and concrete. The results showed that the LVL had lower impacts than steel and concrete. Particularly, the LVL from hardwood plantation thinning has the best environmental performance and the lowest life-cycle cost. Monte Carlo analysis further reviled that uncertainties in the system did not affect the overall economic and environmental performance of the options. The results indicated that the LVL from hardwood mid-rotation thinning could provide a suitable substitute to conventional building materials from an environmental and economic perspective.

Chapter Six consists of the manuscript of published journal article: (Lu and El Hanandeh, 2017). The manuscript combined LCA and LCC for the evaluation and comparison of 6 bio-energy conversion technologies from hardwood plantation thinning. After considering the uncertainties in the bio-energy production systems, the woodchips gasification (WCG) was identified as the best option. Densification of biomass through pelleting did not improve the environmental performance of the extra energy requirements during manufacturing and transportation outweighed the benefits gained from the improved energy density. On the other hand, the production of liquid fuels fared less favourably due to high processing requirements and low conversion ratios.

Chapter Seven consists of a manuscript submitted and currently under review by the Resource, Conversion and Recycling journal. The manuscript developed a dual-objective multi-period linear programming optimisation framework for the management of low-value forestry by-products regarding the environmental impact during the forestry thinning process and the low financial return in the early stages of timber plantation. Eight utilisation scenarios were considered, including two engineered wood products and six bioenergy applications. The two objectives included minimising the environmental impact and economic cost. The LCA and LCC analysis tools were applied to assess the eight options. The results showed that optimisation of the wood resource should consider strategic mid-range planning periods as the constraints may change over time. The effects of uncertainty in the weightings assigned to each objective were tested by varying the weights over the ranges from 10-90%. The solution was stable for all balanced weight ranges (30-70%).

Chapter Eight presents general conclusions for the work reported in this study. It also presents limitations and recommendations for future research. Additionally, the last chapter summarises all the references that have been involved in the entire study.

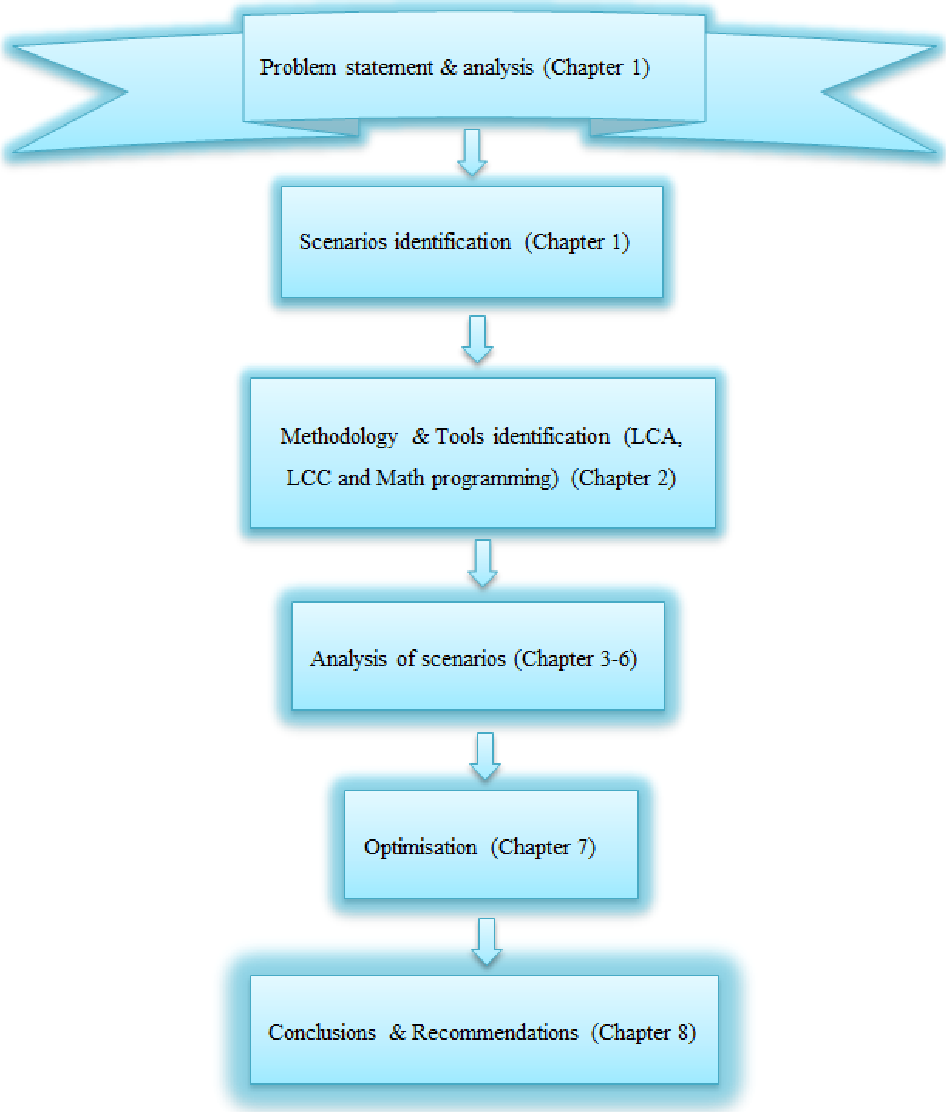

Figure 1.1. **Conceptual framework of this research**

CHAPTER 2 - RESEARCH METHODOLOGY

This chapter provides a general view of the applied research methodologies in the study. This research was conducted as a series of published papers; each chapter in this thesis corresponds to a paper which is a self-contained unit, including the relevant literature review, methods, and discussion. Therefore, this section only provides an overview of the methods common to all chapters including life cycle assessment (LCA); life cycle costing (LCC) and mathematical programming (single and multi-objective optimisation). The software tools and methods used in this study are also described.

2.1 Methodological Framework

This study addressed the optimised assemblage of different management strategies for hardwood plantation thinning applications, within the system boundaries from hardwood seedling, thinning, products manufacturing and service to final disposal. Literature reviews, life cycle assessment and costing analysis, mathematical modelling and statistical simulation as well as scenario analysis were used in this research. Eight possible management options were identified at the beginning, while a series of studies were conducted to evaluate the environmental and economic performance of these scenarios. The scenarios were derived based on two distinct valorisation pathways: the first intends to preserve wood as close to its natural state as possible by upgrading thinned longs into higher quality engineered wood (VBC and LVL); the second is valorisation through energy recovery. Multi-objective optimisation was applied to identify the best combination of options that minimise the life cycle cost and environmental impacts simultaneously.

2.2 Life Cycle Assessment

The International Organisation for Standardisation (ISO) adopted an environmental management standard in the 1990s as part of its 14000 standards series, with the 14040 series focusing on establishing methodologies for LCA (ISO14040, 2006). ISO14040 (2006) defined the LCA as the "quantitation of the materials and energy inputs, outputs and the potential environmental performance of a product or a service throughout its entire lifecycle" (ISO14040, 2006). After more than 20 years of development, LCA has not only been applied to compare different processes or products but also for material flow, energy flow and information flow for a unit product (Klein et al., 2015). LCA considers the sources of raw materials; distribution and transportation; production and maintenance of products; wastes management; and final disposal (Klein et al., 2015, May et al., 2012). LCA has been widely used in the forestry sector to evaluate the impact on both environment and humans from forest plantations and applications management since the early 1990s (Klein et al., 2015, ISO14040, 2006). Nowadays, there is an increasing interest in the development of LCA in the forestry industry (May et al., 2012, Klein et al., 2015).

According to ISO14040 (2006), an LCA study consists of four main phases: (1) Goal and scope definition; (2) Life Cycle Inventory (LCI) analysis; (3) Life Cycle Impact Assessing (LCIA), and (4) Interpretation of results (shown in Figure 2.1).

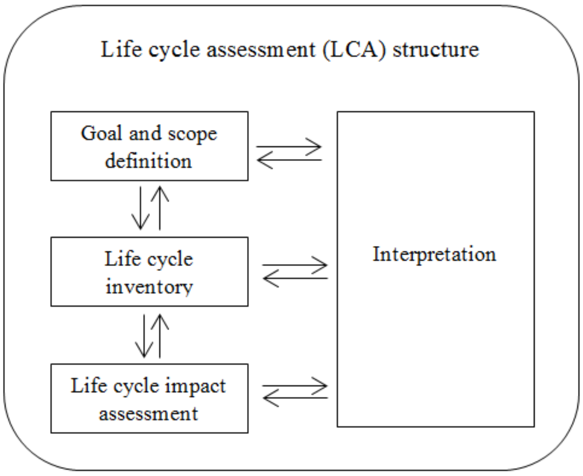

Figure 2.1 LCA framework according to ISO14040 (2006)

2.2.1 Goal and Scope Definition

In the goal and scope definition phase, the functional unit (FU), system boundaries and the procedures of the LCA study are defined (ISO14040, 2006). This stage also includes consideration of how the results are reported and which comparisons are drawn between products and services (ISO14040, 2006). The FU is the central element to define the scope of an LCA study (Cabeza et al., 2014). The selected FU should be comparable to those of other systems serving the same functions. Generally, the FU can be chosen as input based or output based (Iribarren et al., 2012). A system process flow diagram is recommended by the ISO14040 (2006) to clarify the object and systems boundary of the LCA study. The diagram should include all the elementary flows to perform a detailed assessment. The raw materials, energy, and water consumption are seen as the inputs, while the outputs are including the final products, co-products, as well as the emission releases to air, water, and soil (ISO14040, 2006). All the inputs and outputs used in the entire life cycle should be involved in the system boundary (ISO14040, 2006). The emissions saving from energy recovery or product substitutions are also suggested to be considered (Griffin et al., 2016).

2.2.2 Life Cycle Inventory Analysis (LCI)

The LCI analysis has been defined as the stage of recording and tracking the inflow and outflow of the resources and wastes for specific products or processes (ISO14040, 2006, Cabeza et al., 2014). The LCI phase includes collecting data, compiling and modelling the product systems. The outcome of LCI is a quantitative inventory including the total material and energy inputs, products, wastes and emissions (ISO14040, 2006). The quantity of each material is calculated and inputted into an LCA software for the next stage: life cycle impact assessment (LCIA). The data calculation includes validation of collected data, the relating of data to unit process, and the relating of data to the functional unit (ISO14044, 2006).

Eco-invent database contains data sourced from more than 40 countries, including energy, materials, waste management, agricultural products and processes, electronics, metals processing, and transportation (Ecoinvent, 2016). Furthermore, the Australian Life Cycle Assessment Society (ALCAS) established its LCI group in 2002 for establishing the Australian National Life Cycle Inventory Database Initiative (AusLCI) (Tharumarajah and Grant, 2006, AusLCI, 2011). The AusLCI database provides authoritative, comprehensive and transparent environmental information on a wide range of Australian products and services (AusLCI, 2011). Most of the major industry, government and service organisations are involved in the AusLCI database (AusLCI, 2011). There are also some other databases such as the European Life Cycle Database (ELCD) (2013) and U.S. Life Cycle Inventory Database (USDA) (2012). However, the validity of an LCA study for a specific region depends on the quality and specificity of the LCI used (Reap et al., 2008). Therefore, the selection of LCI database is an important decision when undertaking an LCA study. In this study, the LCI data were mainly collected from published literature and proprietary databases such as Ecoinvent (2016) and public databases such as AusLCI (2011). Primary data were used whenever possible for the manufacturing process.

A common issue in LCA studies is the existence of co-products. In such cases, the allocation may become necessary (ISO14040, 2006). Hence, the material and energy flows and the relevant environmental impacts are required to be allocated to the main products and co-products to accurately reflect their contributions to the environmental impact of the system (Cherubini et al., 2011). According to ISO14044 (2006), three steps were outlined to deal with the co-product allocation problem in LCA study: (1) avoiding allocation problem by dividing the complex process into several small sub-processes or by expanding the product system to include the additional functions related to the co-products; (2) if the allocation cannot be avoided, the environmental burdens should be allocated based on the proportions of inputs and outputs resulting from each product (i.e. physical relationships); and (3) if the physical relationship cannot be formed, the allocation should reflect other relationships among inputs and outputs, such as their economic value. In this study, as the thinned logs are a by-product of the forestry industry, the economic allocation was used to distribute emissions from the forestry operations stage between final harvest logs and thinned logs.

2.2.3 Life Cycle Impact Assessment (LCIA)

The LCIA phase aims to evaluate the significance of potential environmental impacts by using the results from the LCI phase (ISO14040, 2006). The LCIA involves a series of elements: selection of environmental impact categories and characterisation models, assignment of LCI results (classification), and calculation of category indicator results (characterisation). In addition, weighting or normalisation are two optional elements of an LCIA (ISO14040, 2006).

Following this environmental mechanism, an impact category indicator result can be selected either at the midpoint or endpoint level (Menoufi, 2011). The midpoints impact category evaluates the primary changes in natural environmental aspects such as climate change, acidification, and human toxicity. Meanwhile, the endpoint category which is also known as the damage-oriented approach generally considers the damaging effect on the environmental aspects (Caserini et al., 2010, Thinkstep, 2015). The endpoint category converts environmental impacts into issues of concern such as human health and natural resources. However, endpoint results usually have a higher level of uncertainty than midpoint results (Caserini et al., 2010, Thinkstep, 2015). The choice of environmental impacts categories to be evaluated usually depends on the goal and scope of the study (ISO14040, 2006). The examples of environmental impact categories include Global Warming, Abiotic Resource Depletion, Land Transformation and Use, Water Depletion, Eutrophication, Acidification, Ozone depletion, Ionising Radiation, Human toxicity, etc. The details of the 14 midpoint impact categories are listed in Table 2.1.

Table 2.1. Commonly used impact indicators and categories (adapted from (ISO14044, 2006, Bengtsson and Howard, 2010b))

Impact Indicator	Impact Category	Damage Categories
The climate change through the release of GHG emissions to the atmosphere. Only fossil carbon emissions are taken into account, while biogenic carbon emissions are not assessed	Global Warming Potential (GWP)	Climate Change
The depletion of minerals and fossil fuel	Abiotic Resource Depletion (minerals and fossil fuel)	Resource Depletion

The damage to ecosystems due to the effects of occupation and transformation of land	Land Transformation and Use	Resource Depletion & Ecological Quality
The total amount of water used by the production system, from cradle (raw material acquisition) to grave (waste management)	Water Depletion	Resource Depletion & Ecological Quality
Releases to air potentially resulting in eutrophication of water bodies	Eutrophication	Resource Depletion & Ecological Quality
Emissions release to air potentially resulting in acid rain	Acidification	Ecological Quality
Fresh-water aquatic eco-toxicity, marine aquatic eco-toxicity and terrestrial eco-toxicity	Eco-toxicity (freshwater, marine and terrestrial)	Ecological Quality
Secondary pollutants formed in the lower atmosphere from NO_x and hydrocarbons in the presence of sunlight.	Photochemical Smog	Ecological Quality & Human Health
The reduction in concentrations of ozone in the stratosphere (ozone layer) when ozone depleting substances (ODS) are released to air	Ozone Depletion	Ecological Quality & Human Health
Impacts arising from releases of radioactive substances as well as direct exposure to radiation	Ionizing Radiation	Ecological Quality & Human Health
Emissions release with potential human toxicity	Human toxicity	Human Health
The primary emissions that cause exposure to these substances are: PM10, PM2.5, trimethylsilyl propionate (TSP), NO_x, NH_3, CO, VOC, and SO_x	Respiratory Effects	Human Health
Noise, or noise nuisance, refers to the environmental impacts of sound.	Noise and Nuisance	Human Health
Health effects from indoor exposure	Indoor Environment Quality	Human Health

The classification process is to assign the LCI data into various environmental impact categories (ISO14040, 2006). For instance, emissions such as CO_2, N_2O, and CH_4 are classified into the GWP category. It is also important to note that emissions may contribute under more than one impact category; for example, NO_x is included under eutrophication, acidification, smog formation and ozone depletion. Characterisation is then applied to characterise each impact quantitatively with a common unit, which allows the effect of the impacts to be evaluated (Bengtsson and Howard, 2010b). For instance, the different GHG emissions (i.e. CO_2, N_2O, and CH_4) are all converted into a CO_2 equivalent (CO_{2-eq}). These values are then summed to represent the effect of GWP (Bengtsson and Howard, 2010b).

Generally speaking, characterisation is the last compulsory stage of an LCA study. Weighting and normalisation are considered as optional LCIA steps. Weighting is the process to allow the different environmental impacts to be weighted relative to each other (ISO14040, 2006). For instance, GWP is ten times worse than acidification potential or human toxicity potential is equally significant to ecosystem quality potential (Bengtsson and Howard, 2010b). In such way, the different environmental impact categories can be added together and convert to a single number to represent the total environmental impact. Normalisation is the step to calculate the magnitude of environmental category indicator results relative to some reference information (ISO14040, 2006). Bengtsson and Howard (2010b) described normalisation as the way to represent the potential impact to ensure all the impacts can be compared on the same scale. Normalisation transforms an environmental indicator result by dividing it by a selected value. Pérez-Fortes et al. (2014) recommended using Impact 2002+ points (pts) method to normalise the overall environmental impact categories in Australian situation. In such way, the overall environmental impact can be determined by the ratio of impact per unit of emission and the total impact generated from emissions per person per year (Jolliet et al., 2003).

The selection of LCIA methods is based on which environmental issues are required for a particular LCA study (ISO14044, 2006). Currently, there are many LCIA methods available, such as ReCiPe, Eco-indicator 99, Impact 2002+, CML, TRACI, EPS2000, EDIP 2003, Lime, EPS 2000, LUCAS, and others (Ren and Su, 2014). The following Table 2.2 presents an overview of the widely used LCIA methods. There are several LCIA methods specific for the Australian region, including Eco-indicator with Australian adjustment; CML 2 baseline 2001-Australian toxicity factor; EDIP with Australian substance; and Impact 2002+ with Australian substance added (Bengtsson and Howard, 2010b).

Table 2.2 An overview of the commonly used LCIA methodologies (adapted from (JRC European Commission, 2010, Menoufi, 2011))

LCIA methods	Origin country	Regional validity	Midpoint/endpoint	Baseline impact categories
CML	Netherlands	Global, except for acidification and photo-oxidant formation (only valid in Europe)	Midpoint	Depletion of abiotic resources, Impact of land use, Climate change, Stratospheric ozone depletion, Human toxicity, Eco-toxicity, Photo-oxidant formation, Acidification, and Eutrophication
Eco-indicator 99	Netherlands	Global impact categories include climate change, ozone depletion, and resources, while other impact categories are limited to the European model. Acidification/eutrophication based on Dutch model, land use based on Swiss model	Endpoint method; however, the method expresses all mid-point indicators per damage category in a common unit	**Midpoint:** Climate change, Ozone layer depletion, Acidification/Eutrophication (combined), Eco-toxicity, Land-use, Mineral resources, Fossil resources, etc. **Endpoint:** Damage to human health, Damage to ecosystem quality and Damage to resources
EPS2000	Sweden	Most of the impact categories can be used globally. Meanwhile, the impacts on biodiversity are limited to Swedish models.	Endpoint	**Endpoint covers:** Human health; Ecosystem production; Biodiversity; and Abiotic stock resource
Impact 2002+	Switzerland	Europe for the basic version. However, a multi-continental version is also available for the assessment of emissions in all the continents.	Midpoint & endpoint	**Midpoint:** Depletion of abiotic resources, Impact of land use, Climate change, Stratospheric ozone depletion, Human toxicity, Eco-toxicity, Photo-oxidant formation, Acidification, and Eutrophication **Endpoint:** Human health, Ecosystem quality, Climate change and Resources
LIME	Japan	Mainly Japan. Environmental impacts of climate change and stratospheric ozone depletion can be used globally.	Combined midpoint and endpoint model	**Midpoint:** Urban air pollution, Global warming, Ozone layer depletion, Human Toxicity, Eco-toxicity, Acidification, Eutrophication, Photochemical oxidant formation, Land Use, Consumption of biotic

				resource, Indoor air pollution, Noise, and Waste **Endpoint:** Human health, Social welfare, Biodiversity, and Primary production
LUCAS	Canada	Global for Climate change and Ozone depletion. Meanwhile, other impact categories are limited to Canada region	Midpoint methodology that will be finally further developed to endpoint	**Midpoint**: Climate change, Ozone depletion, Acidification, Photochemical smog, Respiratory effects, Aquatic eutrophication, Terrestrial eutrophication, and Eco-toxicity (aquatic and terrestrial)
ReCiPe	Netherlands	Europe mainly. Global for Climate change, Ozone layer depletion, and resources	Combining both midpoint and endpoint	Eighteen impact categories are addressed at the **midpoint** level, such as Ozone depletion, Human toxicity, Radiation, Climate change, Marine Eutrophication, Freshwater Eutrophication, etc. **Endpoint**: Human health, Ecosystems, and Resources
TRACI	USA	Mainly for the USA; impact categories such as AP, EP and smog formation are used for North America; ozone depletion and global warming are used globally.	Midpoint	Ozone depletion; Global warming; Smog formation; Acidification; Eutrophication; Human health cancer; Human health non-cancer; Human health criteria pollutants; Eco-toxicity; Fossil fuel depletion.
EDIP2003	Denmark	Europe (up to 44 regions or countries within Europe as well as an average European value). Global for global impact categories	Midpoint, but late in impact pathway (good basis for damage estimation)	**Midpoint:** Global warming, Ozone depletion, Acidification, Terrestrial eutrophication, Aquatic eutrophication, Photochemical ozone formation, Human toxicity, Eco-toxicity, and Noise

According to the "Best practice guide to life cycle impact assessment in Australia" (Grant and Peters, 2016), Global Warming (GWP), Eutrophication (EP), Acidification (AP), Fossil Depletion (FDP) and Human-Toxicity (HTP) were selected for this study. ReCiPe Midpoint (H) life cycle impact assessment (LCIA) method was used for the characterisation of emissions into the relevant impact categories except for the AP and EP categories for which the CML 2001 method was used following the best practice guide (Grant and Peters, 2016). The examples of classifications, characterisations, as well as normalisation factors for the selected environmental impact categories are presented in Table 2.3.

Table 2.3 Commonly used LCI categories in Australia (sourced from (Institute of Environmental Sciences (CML), 2012) and (Goedkoop et al., 2009))

Impact category	Unit	Classifications	Characterisations	Normalisation factor (Australian per capita)
GWP100	kg $CO_{2\ eq}$	Carbon Dioxide (CO_2) Nitrogen Dioxide (NO_2) Methane (CH_4) Perfluorocarbons (PFC_s) Hydrofluorocarbons (HFC_s) Sulphur hexafluoride SF_6	Convert LCI data to CO_2 equivalents For instance: 1kg CO_2 =1kg $CO_{2\ eq}$; 1kg CH_4 = 34 kg $CO_{2\ eq}$; 1kg NO_2 = 298 kg $CO_{2\ eq}$; 1kg SF_6 = 26,087 kg $CO_{2\ eq}$	28,690
AP	kg $SO_{2\ eq}$	Sulfur Oxides (SO_x) Nitrogen Oxides (NO_x) Hydrochloric Acid (HCL) Hydrofluoric Acid (HF) Ammonia (NH_4)	Convert LCI data to SO_2 equivalents For instance: 1kg NO_x = 0.36 kg $SO_{2\ eq}$ 1kg NH_3 = 1.96kg $SO_{2\ eq}$ 1kg SO_2 = 1 kg $SO_{2\ eq}$	123
EP	kg $PO_{4\ eq}$	Phosphate (PO_4) Nitrogen Oxide (NO) Nitrogen Dioxide (NO_2) Nitrates Nitrogen Ammonia (NH_4)	Convert LCI data to phosphate PO_4 equivalents For instance: 1kg PO_4 = 1kg $PO_{4\ eq}$ 1kg NO = 0.13kg $PO_{4\ eq}$ 1kg NO_2 = 0.13kg $PO_{4\ eq}$ 1kg Nitrates = 0.1kg $PO_{4\ eq}$ 1kg Nitrogen = 0.42 $PO_{4\ eq}$ 1kg NH_4 = 0.35kg $PO_{4\ eq}$	19
FDP	kg oil $_{eq}$	The energy content of fossil fuel such as: Crude oil Natural gas Hard coal Brown coal Peat	Convert LCI data to oil $_{eq}$ equivalents. For instance: 1kg Crude oil = 1kg oil $_{eq}$ 1kg Natural gas = 0.84kg oil $_{eq}$ 1kg Hard coal = 0.43kg oil $_{eq}$ 1kg Brown coal = 0.22kg oil $_{eq}$ 1kg Peat = 0.22 kg oil $_{eq}$	6,231

HTP	kg 1,4-DB $_{eq}$	Total emissions release to air, water, and soil	Convert LCI data to 1,4-DB $_{eq}$ equivalents	3,216

2.2.4 Interpretation of Results

In the life cycle interpretation stage, the findings from the life cycle inventory analysis and the life cycle impact assessment are summarised (ISO14040, 2006). The results are reported to identify the significant environmental issues and involve an identification of the significance of the relative contribution of a certain product or a process (ISO14040, 2006). The interpretation phase aims to provide conclusions that are consistent with the defined goal and scope in the first stage, with a discussion of limitations and formulation of recommendations (Cabeza et al., 2014). Data quality analyses are also required in this section included a gravity analysis, uncertainty analysis, and sensitivity analysis (ISO14040, 2006). The findings from the interpretation would be used to reflect the results of the evaluation element (ISO14040, 2006).

2.2.5 LCA Software Review

Several software packages to manage and implement LCA studies are currently available. These can be divided into two major groups: proprietary software packages such as GaBi and SimaPro; and open source software such as OpenLCA and ATHENA. Furthermore, some of the available LCA software packages are industry-specific; for example, ATHENA and BEES while others are generic such as SimaPro and OpenLCA.

ATHENA is a highly rated LCA tool that is usually used in the construction sector (Islam et al., 2015a). The ATHENA has its databases on building materials, energy consumption, construction, maintenance, demolition, transportation and final disposal process (Athena Sustainable Materials Institute, 2017). The databases in Athena are based on data from US and Canadian companies and building practices. It allows modelling of over a thousand structural designs and envelope assembly combinations. It also can be used for comparing different design scenarios (Athena Sustainable Materials Institute, 2017).

BEES is another LCA software that was developed by the National Institute of Standards and Technology. It is a North American environmental impact software program that complies with ISO 14044 (Islam et al., 2015a). It has been widely used to assess the environmental impact of building products (NIST, 2010). BEES analysed a product through the entire lifecycle stages from raw material acquisition via manufacture, transportation, installation, services, to the final disposal. In addition, BEES allows the calculation of economic performance by using the American Society for Testing and Materials (ASTM) standard life-cycle cost method (NIST, 2010, Lippiatt, 1998).

SimaPro (SimaPro, 2017) and GaBi (PE International, 2016) are the two leading software programs used for LCA studies (Herrmann and Moltesen, 2015). Both GaBi and SimaPro are defined as production system modelling program compliant with ISO 14044 standards. GaBi was developed and distributed worldwide by a German company: PE International (2016). Meanwhile, SimaPro is developed by PRe Consultants, which is based in the Netherlands (SimaPro, 2017). Gabi and SimaPro both have up-to-date databases, which also can be edited by the users. The users can also input their own process into the databases. However, compared to the GaBi software, SimaPro involves more environmental impact assessment methods, such as IPCC 2007, EPD, BEES, Ecological Footprint and others (Ren and Su, 2014). Furthermore, the functions of SimaPro are easy to operate compared to GaBi. This powerful and efficient function makes SimaPro more suitable for the user to analyse the products using their own process (Ren and Su, 2014).

OpenLCA is also a professional standard free LCA software. It has the capability of producing reliable LCA models. OpenLCA was first launched in 2006 by GreenDeltaTC Berlin, Germany (GreenDelta, 2014). OpenLCA allows modelling and calculating life-cycle environmental impact based on ISO 14040. It supports the databases such as eco-invent, USDA, ELCD and AusLCI (Ciroth et al., 2008). However, the advantage that OpenLCA has over other commercial packages is being open source and free to use which makes it attractive alternative for academic exercises (GreenDelta, 2014).

2.2.6 LCA Modelling in this Research

In this study, the LCA was conducted using SimaPro v.8.0.4.30 modelling software based following the International Organisation for Standardisation ISO14040 (2006). The functional unit was selected as the treatment of 1 Mg of timber logs from hardwood plantation mid-rotation thinning in South-east Queensland region. The assessment was based on a sixty-year lifetime. The life cycle started from hardwood plantation through thinning, transportation, product manufacturing, service, maintenance, and demolition, to the final waste management. The impact categories considered in this study included GWP, AP, EP, FDP, and HTP. These impact categories were chosen based on the Best Practice Guide to Life Cycle Impact Assessment in Australia (Grant and Peters, 2008). ReCiPe Midpoint (H) life cycle impact assessment (LCIA) method was used for the characterisation of emissions into the relevant impact categories except for the AP category for which the CML 2001 method was used as suggested by Bengtsson and Howard (2010b) and El Hanandeh (2015). Uncertainties and sensitivity analysis were applied to study the effect of uncertainties and assumptions made during the modelling exercise on impacts indicators values.

2.3 Life Cycle Costing Analysis

The Life Cycle Cost of an asset is defined by Treasury NSW (2004) as:

*"The **total cost** throughout its life including planning, design, acquisition and support costs and any other costs directly attributable to owning or using the asset."*

When the Life Cycle Costing is undertaken, all the relevant costs throughout their entire life period will be added together, which include the costs of acquisition, installation, operation, maintenance, refurbishment, discarding, and disposal costs (Rebitzer and Hunkeler, 2003, AS/NZS 4536:1999, R2014). An economic assessment includes analysis and comparison of product options at any stage. The LCC aims to determine the best scenario for a product or a process to meet the identified requirement (Treasury NSW, 2004, AS/NZS 4536:1999, R2014). The life cost planning and life cost analysis are the two main components of the Life Cycle Costing (Treasury NSW, 2004).

- **Life Cost Planning**

The assessment and comparison of options/alternatives are taken consideration during the design/acquisition phase. The similar techniques have been adopted as to discount nominal costs to present values. The application of discounted cost analyses in Life Cost Planning considers all cost components within asset options over the asset's life (Treasury NSW, 2004, AS/NZS 4536:1999, R2014).

- **Life Cost Analysis**

The Life Cost Analysis is a financial management tool to control and manage the ongoing costs of an asset or part thereof enables better prediction and adjustment of the Life Cycle Costing model. The Life cost analysis is based on the LCC Model developed in the Life Cost Planning phase (Treasury NSW, 2004, AS/NZS 4536:1999, R2014).

2.3.1 Steps for Taking an LCC Analysis

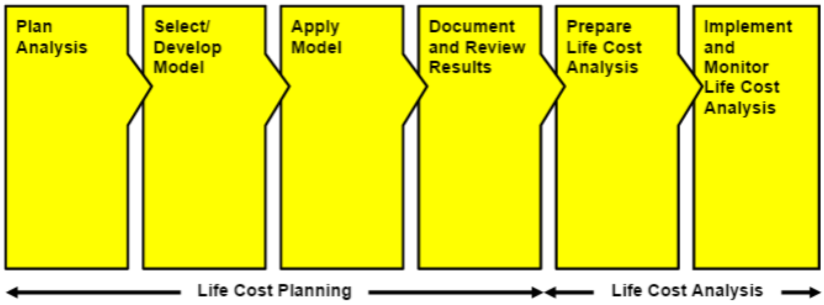

Figure 2.2 The life cycle costing process (Treasury NSW, 2004)

As shown in Figure 2.2, Life Cycle Costing process contains six stages. The first four stages (i.e. (1) Plan Analysis; (2) Select/Develop LCC Model; (3) Apply LCC Model; (4) Document and Review LCC Results) incorporates the Life Cost Planning phase, while the last two stages (i.e. (5) Prepare Life Cost Analysis; (6) Implement and Monitor Life Cost Analysis) comprise the Life Cost Analysis phase (Treasury NSW, 2004, AS/NZS 4536:1999, R2014).

Stage 1 Plan LCC analysis

The LCC analysis starts with the plan development, which aims to address the purpose and scope of the analysis. In the planning stage, the analysis objectives should be defined to support the management decisions. The objectives usually are: (1) determining the LCC for an asset, (2) assessing the impact of alternative courses, and (3) identifying the cost elements (Treasury NSW, 2004, AS/NZS 4536:1999, R2014). In addition, the LCC plan should also define the scope of the analysis and the time period to be considered. Furthermore, the assumptions, limitations, and constraints that may affect the range of acceptable options to be evaluated are also required to be identified (AS/NZS 4536:1999, R2014).

Stage 2 Select/develop LCC model

An appropriate LCC model is either selected or created to satisfy the objectives that have been defined LCC plan. The model typically uses a cost breakdown structure. The cost elements should be identified to ensure no significant impact on the overall LCC of the assets. An economic analysis method is then selected for calculating the cost of every single element in the model. In addition, the uncertainties that may affect the estimation of each cost element should be identified (Treasury NSW, 2004, AS/NZS 4536:1999, R2014).

Stage 3 Applying LCC model/LCC model analysis

A simple spreadsheet or a complex computer simulation can be used to simulate the data into the LCC model. A consistent model must be used for the evaluation of all the alternatives since different models used for each alternative approach might make the comparisons difficult. Most models generate a cash flow scenario based on current costs and then adjust these values for the impact of cost escalation (including inflation impacts) and a discount factor (Treasury NSW, 2004) to account for money-time value factor (Dhillon, 2013). Future costs are discounted to their present value using an appropriate discount rate over the whole lifetime (Deegan, 2008). The discounted cost represents the total amount that has to be reserved today to finance the expenses in future (Deegan, 2008). Meanwhile, the future costs are estimated based on the current price inflated with an estimated future inflation then discounted to present value (Deegan, 2008). Sensitivity analysis is required to assess the effects of the uncertainties arise from assumptions and cost element in the LCC model (AS/NZS 4536:1999, R2014).

Stage 4 Document and review LCC results

In stage 4, results obtained from LCC analysis are documented with clear discussion, conclusion, and recommendations. Both the outcomes and the implications of the analysis associated with the limitations and uncertainties of the results are required to be identified (Treasury NSW, 2004, AS/NZS 4536:1999, R2014).

Stage 5 Prepare life cost analysis

The life cost analysis is prepared by reviewing and developing the LCC model as a real-time cost control mechanism, which requires changes of the costing basis from discounted to nominal costs. It may also require to change the cost breakdown structure and cost elements, to reflect the monitored asset components and the level of the required detail (Treasury NSW, 2004, AS/NZS 4536:1999, R2014).

Stage 6 Implement and monitor life cost analysis

It is necessary for assessing the actual performance of an asset during its operation and maintenance. The continuous monitoring is conducted to identify where the cost savings can be achieved, and it also offers the recommendations for future life cost planning activities (Treasury NSW, 2004, AS/NZS 4536:1999, R2014).

2.3.2 Limitations in LCC

The results from an LCC calculation often bears a significant level of uncertainty due to the insufficient cost data (Sterner, 2000). Insufficient life-cycle cost data would directly result in the limited ability to estimate the future consequences. Hence, many parameters have to be estimated in the LCC calculation that may cause uncertainties, such as the actual lifespan, production cost, operation cost, labour cost, and maintenance costs (Sterner, 2000). Furthermore, the discount rate may also be another element that has a significant influence on the accuracy of the LCC estimation (Copiello et al., 2017). Selection of an appropriate discount rate is the critical part of LCC as if the discount rate is too high may bias decisions in favour of short-term low capital cost options. On the other hand, when the discount rate is too low may result in an undue bias to future cost saving (Sterner, 2000, Copiello et al., 2017). However, the effect of these uncertainties may be mitigated by conducting a sensitivity analysis or a Monte Carlo analysis (Emblemsvag, 2003, Tian, 2013).

A major limitation of LCC is that LCC does not allow a direct comparison of products from different categories (Treasury NSW, 2004). For instance, in this study, the LCC of engineered wood and energy products are incompatible products. The concept of 'substituted values' (SV) was then introduced in this study to overcome this limitation. The 'substituted value' borrows the concept of 'displaced emissions' from the LCA. The SV was calculated as the difference between the LCC of the final product (alternative) and the displaced (substituted) product (i.e. SV equals to the LCC of the alternative product minus the LCC of substituted product). This is in line with the accepted accounting methods of LCA which allows credits for offset emissions. In such way, the benefits from all incompatible products can be equalised and compared on equitable grounds.

2.3.3 LCC in this Research

The LCC was conducted following the Australian/New Zealand Standard: Life Cycle Costing-An Application Guide (AS/NZS 4536:1999, R2014), which took into account all the relevant costs during the entire life stages as mentioned above. Additionally, the environmental aspects and potential impacts associated with the product or within the system are also required to be costed as a part of LCC analysis. For instance, the GWP can be converted to the real cost by using the quantities of $CO_{2\ eq}$ multiply by the carbon price. All the costs were converted to the 2017 Australian dollar Present Value (PV). The LCC of each scenario was calculated by an Excel spreadsheet model.

To calculate the net present value (NPV), future costs were discounted to convert them into the current (2017) Australian dollar values ($AUD). The discount rate represents the rate at which future costs and benefits are discounted compared to the present. The inflation and discount rate were sourced from Reserve Bank of Australia (2015), respectively, which represents the average of the Australian inflation rate and discount rate over the last ten years. The present values of future costs were calculated using current prices, and an estimate of future inflation equation (1), as well as a suitable discount rate equation (2) (Treasury NSW, 2004). When valuing a commercial plantation, forest risk is commonly incorporated in the discount rate.

$FC = PC(1 + f)^n$ …… (1)

Where, *FC* = future cost;

PC = present cost,

f = inflation rate and n= number of years

$PV = \frac{FC}{(1+d)^n}$ …… (2)

Where, *PV* = Present Value;

FC = future cost,

d = discount rate, and

n = number of years

2.4 Optimisation

Optimisation is a process of finding the best solution to a problem that simultaneously meets all imposed constraints (Islam et al., 2015b). The optimisation problem is generally expressed as a mathematical function of decision variables and a set of constraints (Cambero and Sowlati, 2014). The optimisation problem may be formulated as single objective (SOO) or multi-objective (MOO) problem. In SOO, the objective function might be to minimise environmental impact or maximise economic profitability, or any other set objective. However, it is rarely the case that a resource optimisation problem is a single objective problem. More often, optimisation problems involve multiple competing objectives. Islam et al. [10] stated that taking a SOO approach to solve a multi-objective problem is ineffective because it may lead to conflicting results when each objective function is optimised in isolation of the others. Therefore, a MOO approach is recommended because it offers designers a better understanding of the design space (Cocco et al., 2014). In MOO, more than one objective function is maximised or minimised simultaneously.

Huppes and van Oers (2011) and Ren et al. (2010) pointed out that either maximising or minimising the contradictory objective functions simultaneously present particular challenges. The trade-off method is usually suggested to deal with the contradictory objective functions (Wang et al., 2005, Alarcon-Rodriguez et al., 2010).

Generally, in order to solve a MOO, two approaches can be applied (Konak et al., 2006). The first approach is the ε-constraint method that generates Pareto optimal curves showing the potential trade-offs and compromises among them (e.g. Santibañez-Aguilar et al., 2011, You et al., 2012, Kanzian et al., 2013, Pérez-Fortes et al., 2014). A Pareto optimal set is a set of solutions that are not dominated with respect to each other. While moving from one Pareto solution to another, there is always a certain amount of loss in one or more objectives to achieve a certain amount of gain in the other(s) (Konak et al., 2006). Other approaches are to combine all individual objective function into a single composite function or moving all but one objective to the constraint set (Konak et al., 2006). In the first case, a weighted–sum approach is usually applied by assigning a weight to each objective function (e.g. Diakaki et al., 2008, Islam et al., 2015a, Islam et al., 2015b). Currently, there is no particular rule to select the appropriate weighting for each of the objective function (Weng, 2005). A random weighting is usually applied for generating a set of solutions (Kantor et al., 2012, Islam et al., 2015a, Islam et al., 2015b). However, the drawback of this case is that small changes in the weighting may cause significantly different solutions (Konak et al., 2006). In the latter case, the objectives are moved to the constraints set with a constraining value. For instance, both Ganjidoost (2012) and Islam et al. (2015b) converted a MOO problem to a SOO by only including the economic objective function while setting the environmental objective function as the constraint. In both of these two cases, the MOO question would return a single solution rather than a set of solution that required to be trade-offs.

Selection of optimisation software usually depends on the scale of problems, complexity and decision-making preferences (Islam et al., 2015b). MATLAB is a multi-paradigm numerical computing environment, which is intended primarily for numerical computing. It contains an optional toolbox (optimisation toolbox). This toolbox can be used for mathematical programming, which includes functions for linear, quadratic, binary integer programming, nonlinear optimisation, and multi-objective optimisation. This optional toolbox is widely used for standard and large-scale optimisation project (Islam et al., 2015b). On the other hand, *LINGO* and *LINDO* are the commonly used packages in optimisation depend on the problem size and its complexity. *LINDO* is an open source that available to all users. *LINDO* Systems provide fast, easy to use tools that help to minimise or maximise the objective functions subject to a set of constraints using linear, quadratic or integer programming (Islam et al., 2015b). The constraints may be linear equalities or inequalities. *LINDO* software can manage an optimisation problem with up to 150 constraints and 300 variables. *LINGO* is another optimiser from *LINDO* Systems with a set of solvers for linear, integer, and nonlinear models. *LINGO* is more advanced and comprehensive than *LINDO*, which is more appropriate for the more complex problem. It can use up to 1000 constraints and 200 variables (Islam et al., 2015b).

2.4.1 Optimisation Model Formulation

A multi-objective Linear Programming (MOLP) model was used to optimise the utilisation of forestry biomass in this study. Two objective functions (i.e. environmental impact and life cycle costing) were minimised in this study. The variables of the optimisation model were the different forestry biomass application options. The simulation software "*LINDO*" was used to solve the problem. Constraints on design variables essentially delineate the bounds of the feasible range for each variable. Therefore, the constraints of the model were classified into three groups including the total feedstocks availability, final product requirements, and environmental and economic targets.

The decision variables x_i represents the quantities of woody biomass allocated to option i. The two competing objectives in the model were to minimise (1) life-cycle costs and (2) environmental impacts. This research focused on cost minimisation rather than profit maximisation. Nonetheless, the possible incomes (e.g. revenue from material recycling, energy recovery and life cycle cost saving from the substituted products) have to be deducted from the life cycle costs. The life cycle costs (i.e. initial investment, raw material extraction, manufacturing, maintenance, removal, and disposal costs), and environmental related cost (e.g. carbon cost) implied by one functional unit of hardwood thinning application i were represented respectively by means of the data c_i^1 and c_i^2. The economic coefficient c_i^1 was calculated using life cycle costing (LCC) approach and adjusted using the present value method. The environmental coefficient c_i^2 can be determined using life cycle assessment (LCA). Furthermore, a total feedstock supply **s** was determined according to the actual feedstock production. In this constraint, q_i was defined as the amount of feedstocks used by one Functional unit of wood product i. Hence, assuming linear relations, the optimisation of the use of application i to satisfy the feedstock supply s can be formulated as a MOLP as follows:

Min $\sum_{i=1}^{n} c_i^1 x_i$ Economic Objective function…………………….. (1)

Min $\sum_{i=1}^{n} c_i^2 x_i$ Environmental Objective function……………….. (2)

Subject to $\sum_{i=1}^{n} q_i x_i \leq s$ Satisfy supply constraint……………….. (3)

In selecting the optimal combination of the life cycle costs, the MOLP optimisation model may be re-written in equation (4). Regarding the environmental assessment, five different environmental impact categories were considered. Impact 2002+ points method was used for the normalisation of the five different impact categories into one same unit. The normalised factor was determined by the ratio of the impact per unit of emission and the total impact generated from emissions per person per year in Australia (Jolliet et al., 2003).

Normalisation was applied again when combining the environmental and economic objective functions into a single objective function. The economic and environmental normalised values were calculated by using the actual values from each option divided by the optimal value calculated from the economic and environmental objective functions, respectively, following the procedure described by Islam et al. (2015b) and Azapagic and Clift (1999). In this study, the weighted sum approach was applied. A weight was assigned to each normalised objective function so that the problem is converted to a single objective problem with a scalar objective function. Hence, Equations (1 & 2) are then re-written to minimise the multiple variables (cost and multiple environmental impact indicators) as:

$$\mathrm{Min}[w_1 * \sum_{i=1}^{n} (c_i^1\, x_i) + w_2 * \sum_{i=1}^{n} (c_i^2\, x_i)] \ldots\ldots\ (4)$$

Where $(c_i\, x_i)$ is the normalised objective function, while $w_1 + w_2 = 1$.

In this model, various weightings were applied, following the approach developed by Islam et al. (2015b). For the first composite objective function, a weighting of 50% was applied to both economic and environmental functions. For the rest of composite objective functions, various sets of weightings were applied, from 10% to 90% for environmental impacts and from 90% to 10% for life cycle cost by increment or decrement by 10% each time. In each case, the weightings must be sum to 100%. Varying the weighting for one objective can assess whether the best design is sensitive to the weighting (Khan and Ardil, 2009).

2.5 Summary

This chapter describes the methodological framework and the methods used in the study. It discusses the approaches to take environmental and economic analysis and optimisation for hardwood plantation thinning management options. The LCA was conducted using SimaPro v.8.0.4.30 modelling software. Five environmental impact factors were assessed. The LCC was conducted by including all relevant costs throughout the entire life stages. The life cycle GHG emissions costs were also taken into account. All the money values were converted to the 2017

Present Value (PV) in the Australian dollar. The linear programming (LP) was used for optimising environmental and economic performance. Variables were minimised or maximised as a linear function of the outputs of the activities. *LINDO* package was chosen for conducting optimisation studies. The tools and techniques presented here were demonstrated through each chapter.

CHAPTER 3 – RESEARCH ARTICLE 1

STATEMENT OF CONTRIBUTION TO CO-AUTHORED PUBLISHED PAPER

Chapter 3 includes a pre-print of a co-authored paper titled "Life Cycle Assessment of ACQ-treated veneer composite (VBC) Hollow Utility Poles from Hardwood Plantation Mid-thinning" which was published in the Sustainable Production and Consumption Journal. My estimated contribution to the paper is 65%. I contributed to research design, data collection, modelling, analysis, and writing the manuscript.

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(Signed) ________________________________ (Date) ______________

(Countersigned) __________________________ (Date) ______________

Corresponding author of paper:

(Countersigned) __________________________ (Date) ______________

Supervisor:

Life Cycle Assessment of ACQ-Treated Veneer Based Composite (VBC) Hollow Utility Poles from Hardwood Plantation Mid-Thinning

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Abstract

Hardwood plantations are slow to mature with low financial returns in the early stage. Veneer Based Composite (VBC) products developed from mid-thinning may improve the industry's profitability and win new markets. Due to the increasing demand for utility poles and the banning of native forests logging in Australia, VBC poles may become a viable alternative to native hardwood poles. Alkaline copper quaternary (ACQ) preservative treated VBC pole was assessed using life cycle assessment (LCA) methodology. The manufacturing processes considered were based on the current technologies in Queensland. VBC pole life cycle stages assessed include mid-thinning, manufacturing, service-life, and disposal. Three end-of-life scenarios were considered: landfilling, incineration for energy recovery and recycling as particleboard. The functional unit used in this assessment is a 1-metre-length pole with 115-mm internal diameter and 15-mm wall thickness. Global Warming Potential (GWP100), Fossil Depletion Potential (FDP), Acidification Potential (AP), Eutrophication Potential (EP), and Ecological Toxicity Potential (ETP) were quantified. Results indicated that landfilling and incineration outperform the recycling option. Incineration scenario performed slightly better under the GWP100 (0.3659kg-CO_2-Eq), AP (2.12g-SO_2-Eq), FDP (0.360kg-Oil-Eq) and EP (3.81g-PO_4-Eq). Meanwhile, the landfilling scenario had slightly less impact in ETP (12.32-CTUe). Despite generating valuable products, the burdens caused by secondary manufacturing and transportation overweighed the credits earned from recycling. ACQ treatment, Phenol-formaldehyde (PF) resins production and transportation distances were identified as significant parameters affecting the final result. Sensitivity analysis indicated that EP was sensitive to change in ACQ consumption; ETP was affected by PF resin use while changing distances of transporting product affected GWP100, AP and FDP.

Key Words:

Life Cycle Assessment (LCA), hardwood thinning, veneer based composited (VBC), Alkaline copper quaternary (ACQ), pole.

3.1 Introduction

During the last decade, Australian hardwood plantations have increased by approximately 150%, mainly due to changes in government policies. Currently, Australia has approximately 150 million hectares of forest, dominated by more than 700 different species of Eucalyptus. About two million hectares of these forests are managed as timber plantations; with approximately 49% being hardwood plantations (Australian Government Department of Agriculture Fisheries and Forestry 2010). Timber plantations present an opportunity to increase Australian long-term wood supply while contributing significant social, economic and environmental values (Australian Government Department of Agriculture Fisheries and Forestry, 2014).

3.1.1 Background

It is essential to produce high-quality logs at an early age (30 to 35 years). Therefore, pruning and thinning are practised during the early stage of the plantation, in order to increase available light, moisture and nutrients for the rest of trees (McGavin et al., 2006b). Approximately, 50% of the trees are typically cut from the third year during the first thinning, with another 30% removed in the second thinning (10 to 15 years). The removed trees during the second thinning usually have a Diameter at Breast Height Over Bark (DBHOB) of 0.15m to 0.3m. However, these thinned logs have low commercial value because they do not have viable markets (Underhill et al., 2014). Veneer Based Composite (VBC) products have been developed as high-value applications from plantation second thinning, which can be applied for replacing various engineered wood products (McGavin et al., 2006b, Underhill et al., 2014).

Utility pole is defined as a column to support various public utilities, such as cable, fibre optic cable, and related equipment such as streetlights. Utility poles are usually sourced from native forest hardwood species (Gilbert et al., 2014b). Hardwood is seen as the most appropriate solution for manufacturing utility poles. Compared to the alternative materials, such as steel and concrete, hardwood has the lowest environmental and economic burdens (Francis and Norton, 2006). However, due to growing environmental awareness and concerns over the sustainability of forestry practices, agreements have been signed in Australia to phase out logging of native forests. Thus, the supply of hardwood utility poles will decrease sharply, while the demand is expected to dramatically increase (Francis and Norton, 2006). Looking for new materials to satisfy the shortage of poles presents a big challenge (Australian Government Department of Agriculture Fisheries and Forestry, 2014).

3.1.2 Demand for Utility Poles in Australia

More than 5 million timber utility poles are currently in-service in Australia, while most of them were produced from native hardwood forest. However, up to 70% of the total in-service poles are ageing poles and reaching the end of life, which may cause severe consequences in Australian utility system (Francis and Norton, 2006). An investment of approximately 1.75 billion dollars is required to replace these 3.5 million ageing poles. Thus, 175 million dollars is required per annum to achieve this target over the next decade. In addition, new utility poles are also required to satisfy the urban expansion, which will represent an additional cost (Francis and Norton, 2006).

The newly developed veneer based composite (VBC) hardwood hollow utility poles may present an alternative to replace traditional hardwood poles, which also could offer positive environmental and economic performances from the early stage during hardwood plantation in Queensland. The VBC poles are manufactured from Gympie messmate (Eucalyptus *cloeziana*) plantation mid-thinning, a traditional hardwood species used for manufacturing poles in Australia (Gilbert et al., 2014b). Nevertheless, the sustainability of the product from life cycle perspective has not been assessed. For this reason, a full LCA is developed to assess the environmental performance of VBC hardwood hollow utility poles from hardwood plantation mid-thinning.

3.1.3 Timber Veneer Processing

Veneer is a thin slice of timber with a thickness from 0.3mm to 6.0mm, depending on its natural property and applications (Nolan et al., 2005, Skates et al., 2012). Rotary peeling, slicing and half-round slicing are the three common methods for cutting veneers. The logs need to be preconditioned before cutting, which involves debarking and cutting the log to the certain length. The logs are then pre-softened by immersing in hot steam (Ozarska, 2003). This step controls the moisture content in logs, to ensure the veneers are easily peeled (Skates et al., 2012). Due to the difficulty of using traditional peeling technology for peeling thinned logs with a small diameter and low quality, the spindle-less veneer lathe is used to produce unbroken continuous veneers (Gilbert et al., 2014a).

Timber veneer from hardwood plantation is an innovative solution to satisfy the demand of the solid wood product. The suitability of timber from Eucalyptus plantation used for structural laminated veneer lumber (LVL) applications was reported previously by Carrick and Mathieu (2005) and Nolan et al. (2005). More recent research from Queensland Department of Primary Industries (DPI) indicated that they had successfully manufactured and tested engineered wood products such as LVL beams from thinned Eucalyptus trees (McGavin et al., 2006b). Skates et al.

(2012) explored the potential applications of engineered wood from hardwood mid-thinning. In addition, Underhill et al. (2014) mentioned the suitability for manufacturing utility pole.

3.1.4 Aims of this Study

This paper aims to conduct an LCA study to investigate the environmental performances associated with VBC hardwood hollow utility poles manufactured from Gympie messmate plantation mid-thinning. The environmental burdens of three different disposal scenarios are assessed to determine appropriate end-of-life treatment. Results of this study may have help hardwood plantation managers and utility companies to facilitate a better sustainable management of plantations and utility networks.

3.2 Methods

The goal of this study is to provide a scientifically based and comprehensive assessment of the environmental performance of the VBC hardwood hollow utility pole manufactured from hardwood plantation mid-thinning over its life cycle. The scope of this study includes ‘cradle to grave’ life cycle inventory of VBC hollow utility poles from hardwood plantation second thinning. The timbers are obtained from the hardwood plantation located in South East Queensland. Primary data was used whenever possible, secondary data on background operations and energy was collected from various sources in the literature as described later in Section 3-Life Cycle Inventory.

For a collection of life cycle inventory inputs and outputs, the functional unit is defined as a small diameter pole (1 meter in length, with a nominal internal diameter of 115 mm and 15 mm wall thickness). The VBC utility pole is formed in two half-pole jointed together, while each half pole is compressed by 9 pieces of hardwood veneers.

- Functional unit: one VBC utility pole
- Service life: 25 years
- Geographic boundary: South East Queensland, Australia

The system boundary begins with hardwood seedling and ends with the VBC pole being disposed of in a landfill, incinerated or recycled. Planting the seedlings, forest management (i.e. site preparation, first thinning and fertilisation), and second thinning are considered in the system boundary of the plantation. Transporting thinned logs from the plantation site to the mill is accounted for the pole manufacturing. Seedlings and the fertiliser and energy consumptions are considered as inputs in the first stage. The VBC timber pole manufacturing process is divided into several small process units: transportation, debarking and cutting, pre-conditioning, logs peeling, veneer drying, compressing, trimming, VBC utility pole forming, and transporting to the utility lines. The final stage includes pole services in utility system and final disposal (e.g. landfilling, incineration for energy recovery, and recycling). The system boundary is presented in Appendix A. As the thinned log is a by-product of the forestry industry, the allocation is required. In this study, the economic allocation was used following May et al. (2012). In economic allocation, impacts were distributed among co-products according to their market value.

The impact categories investigated in this LCA include global warming potential (GWP100), eutrophication potential (EP), acidification potential (AP), fossil depletion potential (FDP) and eco-toxicity potential (ETP). These impact categories were chosen based on the Best Practice Guide to Life Cycle Impact Assessment in Australia (Grant and Peters, 2008). OpenLCA software was used to conduct the LCA analysis (GreenDelta, 2014). ReCiPe Midpoint (H) life cycle impact assessment (LCIA) method was used for the characterisation of emissions into the relevant impact categories except for the AP category for which the CML 2001 method was used as suggested by Bengtsson and Howard (2010b) and El Hanandeh (2015).

3.3 Life Cycle Inventory (LCI)

The inventory analysis involves data collection and analysis for the cradle-to-grave life cycle of the VBC utility pole. For each stage of the product lifecycle, inputs of energy and raw materials, outputs of products, waste, and emissions releases to the environment are determined. In addition, hardwood plantation system is assumed to be in a steady state both with respect to carbon stocks and management operations. This assumption has been used either explicitly or implicitly in most other forestry LCIs (e.g. Schweinle and Thoroe, 1997, Sonne, 2006, Dias et al., 2007). Life cycle inventory (LCI) data are collected from different sources, including published literature and online databases, such as European Life Cycle Database (ELCD) (2013), U.S. Life Cycle Inventory Database (USDA) (2012) and AusLCI (2011).

3.3.1 Hardwood Plantation Operations

The specific processes during establishment include seedling production, site preparation, slash burning (burning of residues) and planting. Management process includes the chemical application of herbicides or fertiliser, fire prevention and control and construction of forest roads. Thinning process starts from first thinning to second commercial thinning. The electricity consumption for greenhouse operations, fertiliser used for seedling production and growth, and the diesel fuel and lubricants required to power equipment during site preparation, fertilisation, and thinning are considered as the inputs in plantation management. The primary output in this analysis is the green logs from second thinning. The other products, such as non-merchantable slash and brunches are usually treated by mechanical activities or prescribed fire on site, which is not included in the system boundary.

The operations and environmental impacts associated with the plantation operations may fluctuate based on different technologies, management procedures, and plantation scale (Irland et al., 2001). Therefore, this LCA is based on data representing the current state of the art of hardwood plantation operations in Australia. The LCI data of thinned logs, energy and materials consumption and the emissions releasing through operation processes are tracked from different published sources related to Australian hardwood plantation.

3.3.1.1 Equipment in Thinning

Generally, plantation-thinning activities include four components:

- Timber felling;
- Processing (cutting the logs into appropriate lengths and removing non-valuable parts);
- Haulage (transporting logs from felling to a loading point); and
- Loading (moving logs to trucks).

This analysis is based on data for the most common mechanised thinning system in Australia. A cutting device with a crawler woods tractor is usually employed during mechanised felling to cut and bunch trees. Transporting those thinned trees to a landing (skidding), and using another machine that can de-limb and process trees into logs at the landing (May et al., 2012).

3.3.1.2 LCI of Hardwood Plantation Operations

Table 3.1 summarises LCI inputs based on published LCI data for Australian forestry and wood products. The inventory data are adopted from May et al. (2012), who used economic allocation to distribute emissions among products. A details discussion of inputs is provided below.

Table 3.1 Average fuel and materials consumptions of hardwood plantation per m^3 in QLD region (sourced from May et al. (2012))

Resource	Unit	Quantity
Natural resources		
Land	ha year m^3	0.08
Water	ML/m^3	0.23
CO_2 sequestration	kg/m^3	1,030
Energy		
Diesel	L/m^3	6.8
Aviation fuel	L/m^3	0.02
Direct energy	MJ/m^3	263
Indirect energy	MJ/m^3	191
Materials		
Lubricant	L/m^3	0.12
Tyres	kg/m^3	0.09
Steel	kg/m^3	0.08
Gravel	kg/m^3	640
Fertiliser		
N	kg/m^3	0.04
P	kg/m^3	0.11
K	kg/m^3	0.08
Herbicide		
Glyphosate	g/m^3	11.4
Simazine	g/m^3	19.2
Triclopyr	g/m^3	1.4

- Land use

Land use per unit wood production for hardwood plantations, as estimated from the total plantation area and the total annual volume harvested, which is approximately 0.08 ha year m^3 in QLD regions (May et al., 2012).

- Water

The average annual rainfall in Queensland is 1310 mm and the estimated annual evapotranspiration (ET) is 1010 mm. Therefore, in terms of total wood production, forest ET is 0.79 ML m^{-3}. However, relative to ET from grassland, net water use is 0.23 ML m^{-3} in QLD regions (Zhang et al., 1999, Zhang et al., 2001).

- Fuel and energy

Direct energy consumption (mostly diesel) for unit wood production is approximately 6.8 L m^{-3} for hardwood plantations, which is equivalent to 262 MJ m^{-3}. The fuel and energy inputs for hardwood logs are mainly due to thinning and haulage process. Energy and materials used for upstream processes (e.g. energy consumed in processing, manufacture or transport of fuel, fertiliser or other materials) are considered as indirect inputs (May et al., 2012). In this study, the indirect input energy consumption is approximately 191MJ m^{-3}.

- Chemicals and materials consumptions

N, P and K fertiliser are applied during hardwood plantations. Triple superphosphate, ammonium sulphate, urea and potassium sulphates are the most commonly used fertiliser forms. Herbicides considered in this LCA including glyphosate, simazine and triclopyr, which are widely used in plantations. Other material inputs in plantations are mostly related to machinery maintenance.

3.3.2 VBC Utility Poles Manufacturing

The ACQ-treated VBC utility poles manufacturing process are based on current technology in Queensland Government Salisbury Research Facility Centre and the LCI data is collected from different published sources as detailed in the following sections.

3.3.2.1 Transportation

Thinned logs usually transported to the mill by trucks. The Gympie Messmate logs used are sourced from plantations in the Gympie region, South East Queensland (Underhill et al., 2014). These harvested logs are transported to Queensland Government Salisbury Research Facility for processing. The average haul distance from the forestry operations site to Salisbury Research Facility is approximately 170 km. The hardwood green log density ranges from 634 to 810kg/m^3 depending on the growing conditions and species (Bootle, 1983, McGavin et al., 2006b). An average density of 722 kg/m^3 is used in this study.

Transporting the thinned logs and adhesives is using B-Doubles (consisting of a prime mover and two trailers with a load of 45 Mg), with a full load for delivery and a 10% return load are assumed in this study. The LCI of transportation is calculated using tonnage of logs freighted multiplied by distance travelled.

3.3.2.2 Debarking and Cutting

The debarking and cutting process is mechanically removing the bark from the natural logs and cutting them to proper length before being rotary peeled. Co-products generated include bark that used as wood fuel in a boiler, while the other non-valuable waste is sent to the landfills. Inputs during this process include electricity for operating equipment and diesel fuel consumed by log hauliers.

3.3.2.3 Steam Pre-conditioning

Pre-conditioning of logs is heating the wood logs with steam at high temperature around 80 degrees Celsius to improve the quality of peeled veneer. This step makes logs easier to peel, reduces veneer breakage, and results in smoother, higher quality veneer (Puettmann et al., 2013).

3.3.2.4 Log Peeling

The pre-conditioned logs are conveyed to a veneer lathe. Spindle-less veneer lathe is employed for logs peeling in this process, which converts most of a log into green veneers (80 to 90%). A unpeeled core with a diameter of 45 mm is left after peeling (Underhill et al., 2014). The veneers are then moved by conveyor to a clipper to cut into appropriate widths. Wood waste from peeling process represents about 30% by mass of the outputs.

3.3.2.5 Veneer Drying

Drying hardwood veneers consume approximately 75% of the heating energy in the entire manufacturing processes. Bergman and Bowe (2011a) and Bergman and Bowe (2011b) reported that 3,773 MJ of energy is consumed per cubic metre of engineered wood for drying. Currently, wood fuel used represented most of the total energy requirement in Salisbury Research Centre with the remainder sourced from natural gas and electricity. Veneers are dried in continuous direct-fired dryers under a temperature between 150 ° and 185 ºC. This step reduces the moisture content of green veneer from approximately 50% to 5% (oven-dry basis). The density of oven-dry hardwood veneer is estimated as 613 kg/m^3 by Bergman and Bowe (2011a), Bergman and Bowe (2011b).

3.3.2.6 Alkaline Copper Quaternary (ACQ) Treatment

Chromated copper arsenate (CCA)-treated wood was widely used until 2004 for residential and industrial applications. CCA was replaced by alkaline copper quaternary (ACQ) because of health concerns (Janin et al., 2011). ACQ treatment is water-based wood preserving method. ACQ is made up of copper, bactericide and fungicide, which ensures wood resistant to biological attack. A quaternary ammonium compound (quat) acts as a biocide to increase the tolerance of treated timber to copper-resistant bacteria and fungi, in addition, to perform as an insecticide (Lebow, 2004).

ACQ was defined with a retention rate of 2.4 - 6.4 kg active ingredient per cubic meter of lumber, as outlined by the American Wood Protection Association (AWPA) standard UC4A (AWPA, 2010a, AWPA, 2010b). An average retention rate 4.4kg/m^3 of ACQ is used for treating one cubic metre of wood veneer.

Inputs and outputs for ACQ production include the basic components of copper, quaternary, and ethanolamine. One kg of ACQ, the formulation actually includes 0.668 kg of copper oxide, 0.332 kg of quaternary ammonium compound (quat), and 1.81 kg of ethanolamine (Ayres et al., 2013).

3.3.2.7 Phenol-formaldehyde (PF) Adhesive

Phenol-formaldehyde (PF) is water-based glue commonly used for the production of plywood, laminated veneer lumber (LVL), and oriented strand board (OSB) (Kline, 2007, Wilson and Dancer, 2007, Wilson and Sakimoto, 2007). The inputs to produce 1.0 kg of neat PF resin consist of two primary chemicals phenol and methanol dissolved in water. The LCI data for the composition and production of PF adhesive were sourced from Wilson (2009).

3.3.2.8 Compressing

The dried veneers are glued by PF adhesive into half-shapes around a mandrel, while the grains of veneers are orientated in the same direction to form a veneer stack. Teflon sheets are used to cover the veneer stack to reduce the friction when the stacks shape around the mandrel. A series of flexible outer straps are required to cover the stack, and then the veneer stack is compressed between the mandrel and the outer straps by a hydraulic jack driving the mandrel down. The circular hollow sections for casting utility pole can be manufactured by changing the shape of inner mandrel (Underhill et al., 2014). Emissions are generated during the compressing process are mainly caused by PF resin. The density of the VBC product is approximately 610 kg/m^3. This process consumes approximately 15% of heating energy for producing hardwood VBC utility poles in Queensland.

3.3.2.9 Trimming

After compressing, the VBC pole is sent to a finishing area for trimming and sanding. Wood wastes materials include veneers trim and sawdust is recycled in the boiler.

3.3.2.10 VBC Pole Forming

The process involves the forming from veneers of circular hollow structures that consist of half round sections. Each section is formed separately around one of two different diameter mandrills, depending on their position within the structure. This produces an inner and outer pair of half round sections, which are then cut to the appropriate size and bonded together in one 'glue-up' using PF resin to form the hollow structure. According to the current manufacturing process in QSRC, VBC utility pole is formed by two half-pole butts jointed together, while 9 pieces of hardwood veneers are used per half-pole. Each pole functional unit contains 0.006 m^3 of dry and trimmed VBC panel.

3.3.3 Summary of LCI per Functional Unit

The production of 1 function unit of hardwood VBC pole consumes 0.0142 m^3 of green logs. The main inputs considered in the LCI analysis are the hardwood logs from second thinning into the veneer process, leading to veneer input into the veneer based composite process. The logs are green and include wood and bark with a moisture content of approximately 50%. The veneer is mainly taken from the veneering process as a dry veneer that imported into VBC pole manufacturing. The main inputs to manufacture one functional unit of average Australian VBC pole are listed in Table 3.2.

Table 3.2 Inputs and outputs to produce one functional units of VBC pole average Australian veneer based composite including energy from veneer production (adapted from (Ayres et al., 2013, Tucker

et al., 2009, Wilson, 2009, Bergman and Bowe, 2011a, Bergman and Bowe, 2011b, Puettmann et al., 2013))

Material Inputs	Value/function unit	Unit
Hardwood thinned log	0.0142	m^3
Phenol Formaldehyde Adhesive	0.360	kg
ACQ	0.03	kg
Flour	0.0343	kg
Filler	0.0243	kg
Phenolic Overlay sheets	0.0655	kg
Acrylic Putty	0.0036	kg
Phenol Formaldehyde Putty	0.0012	kg
Transportation	1.91	tkm
Total Energy consumption		
Electricity	0.9123	kWh
Natural Gas	0.0312	m^3
LPG	0.0086	L
Diesel Fuel	0.0135	L
Wood fuel	5.688	kWh
Water	2.0083	L
Output Products		
VBC Pole	0.006	m^3
Wood waste	2.7875	kg

The output from this system is VBC pole, which is a high-value product. The outputs from every cubic metre of hardwood-thinned logs include 0.43m^3 of VBC pole (approximately 70 functional units). The wood waste (2.78kg/pole) is sent to the boiler for energy recovery during the manufacturing process. The other waste generated during the pole manufacturing process is transported to landfill for final disposal.

3.3.4 VBC Utility Pole Service Life

The ACQ-treated VBC utility pole service stage includes transportation of pole, installation in the utility use location, maintenance during its service life, and demolition at the end of use. Steel bolts used to attach VBC pole and other hardware are installed by the utility, but they are not considered in the LCI analysis due to the fact that steel bolts used to mount cross-arms generally are the same for all types of poles (Bolin and Smith, 2011).

3.3.4.1 VBC Pole Installation and Transportation

Transportation-related inputs and outputs are included in this life cycle stage. An average distance of 100km is assumed for pole transportation, which translates to 0.37 tkm per functional unit of the pole. Transportation data are sourced from AusLCI (2011).

3.3.4.2 Regular Inspections and Maintenance

The time of VBC poles service in a utility line depends on a number of factors. Utility poles are often removed before the end of their useful service life due to road widening or infrastructure upgrading. In this study, an average service life of 25 years is assumed for VBC poles according to results of the interview with Queensland Government Salisbury Research Centre. Most utility poles have regular inspection programs around 8 - 12 years. Inspections include physical testing for indications of decay (Mankowski et al., 2002). Maintenance treatments are surface treatments (applied at and just below ground-line) or internal treatments. Surface applied materials are copper, boron, and/or sodium fluoride in pastes that both coat and diffuse into the wood and are wrapped with a waterproofing material.

An inspection and maintenance program is included in this LCI. Each pole is assumed to be inspected and maintained every 10 years during the service life. The treatment model assumes 0.069 litter/pole of paste is needed per treatment, consisting 2% of copper, 43% borate (DOT), 10% petroleum (as a surrogate for other possible fossil fuel derived ingredients), water, and mineral filler/thickeners. Based on data published by Bolin and Smith (2011), the treatment per functional unit consists of 0.0014 L copper, 0.0297 L borate and 0.0069 L petroleum.

3.3.5 End of Service Life

At the end of service life, poles may have recycling value as treated wood, such as fence posts or landscaping or as fuel to produce process heat and/or electricity. Some utility companies also simply dispose of the used poles as solid waste in landfills. Disposition, as modelled in this study, is based on the system boundaries including incineration for energy recovery, recycling as particleboard and disposal in landfills.

3.3.5.1 Scenario I – Landfilling

Disposal stage begins with the VBC pole waste transports to landfill and includes processing of the landfilling and emissions over the landfill life. Collection and transportation distance between utility lines and landfill is assumed to be approximately 100km. The landfill results in 40.164 kg of wood carbon released as carbon dioxide (biogenic), while 8.453kg of methane released per 1000kg of wood waste (AusLCI, 2011). At landfills with CH_4 capture infrastructure, typically up to 75 % of the CH_4 generated is assumed to be captured by the collection system (Bracmort et al., 2009). An analysis of the primary metal components in the wood conducted by Dubey et al. (2009) indicated that copper and boron concentrations in the ACQ-treated wood samples were 3750+/-125 mg Cu/kg and 510+/-35 mg B/kg. These concentrations are consistent with the manufacturer rated concentration of 4 kg/m^3.

The landfilling considered in this study includes 100 years of product life after disposal. This time frame is consistent with other solid waste studies (e.g. Diaz and Warith, 2006, El Hanandeh and El-Zein, 2010). The LCI of landfill construction and closure is adopted from Mènard et al. (2003). Landfilling emission outputs data of wood waste is sourced from AusLCI (2011). The LCI data for VBC pole in a landfill is presented in Table 3.3.

Table 3.3 LCI data for wood waste in landfill in QLD region per functional unit (data adapted from (Mènard et al., 2003, Dubey et al., 2009, AusLCI, 2011))

Input Flow	Value/function unit	Units
Wood and wood-waste, at landfill	3.71	kg
Electricity, low voltage, Australian	0.003	kWh
Transportation	0.371	tkm
Diesel, burned in building machine/GLO U	0.143	MJ
Output Flow	**Value**	**Units**
Carbon dioxide, biogenic	0.149	kg
Methane	0.031	kg
Non-methane volatile organic compounds, unspecified origin	0.014	kg
Copper	0.015	kg
Boron	0.002	kg

3.3.5.2 Scenario II – Combustion for Power Generation

VBC Poles recycled for power generation is assumed to be combusted in large cogeneration or utility type boilers that include scrubbers or electrostatic precipitators. The collection and transportation distance is assumed to be 100km approximately from utility to power plant. During the energy recovery process, the wood carbon is released as biogenic carbon dioxide and the combusted preservative carbon will release as fossil carbon dioxide (Bolin and Smith, 2011).

ACQ-treated woods during combustion were investigated by Lin et al. (2007). The measurement of the CO gas from combusting ACQ treated wood was about 179.3 ppm and there was no SO_2 gas being emitted during combustion. NO_x is the main source of pollution from burning ACQ treated wood (Kercher and Nagle, 2001, Humphrey, 2002, Lin et al., 2006, Lin et al., 2007). The maximum NO_x of the ACQ treated wood was about 23.5-26.5 ppm. By clarifying the residual elements (inorganic and organic materials) of discarded ACQ treated wood products; the results indicated the char of the ACQ left an amount of inorganic metal elements, Cu (50.14%) (Kercher and Nagle, 2001, Lee et al., 2005, Lin et al., 2006). The other components of ACQ were volatilized with the increase in temperature during combustion (Lin et al., 2007).

The LCI of wood waste incineration is based on Australian published sources (Ximenes, 2007, Tucker et al., 2009, DCCEE, 2011a, DCCEE, 2011b, NSW Environment Protection Authority, 2012). Electricity from treated wood incineration is assumed to be distributed to the main grid and industrial customers. The utilities used in the waste incineration plant, the bottom ash from incineration and air pollution residues are included in the system. The bottom ash (approximately 220kg/Mg of treated wood product) is disposed of in a landfill. The LCI data of ACQ-treated wood in incineration is presented in Table 3.4.

The energy recovery through wood waste combustion is modelled as energy credit due to the avoidance of production of materials from virgin feedstock (coal and natural gas) and the gains of electricity. The energy content of wood waste is assumed to be 9.5MJ/kg, while the efficiency of power generation from wood waste is 20% based on NSW Environment Protection Authority (2012). Thus, 7.05MJ of electricity is generated per functional unit of a utility pole, while the heat waste is approximately 28.20MJ/functional unit.

Table 3.4 LCI data of wood waste for energy generation in QLD region per functional unit (adapted from (Lin et al., 2007, Ximenes, 2007, Tucker et al., 2009, DCCEE, 2011a, DCCEE, 2011b, NSW Environment Protection Authority, 2012))

Input Flow	**Value/ Function Unit**	**Units/Pole**
Wood and wood-waste	3.71	kg
Transportation	0.453	tkm
Output Flow		
Electricity	7.05	MJ
Heat, waste	28.20	MJ
Carbon dioxide, biogenic	6.056	kg
Nitrogen dioxide	0.004	kg
CO	0.001	kg
NO_x	0.0001	kg
Cu	0.012	kg
Bottom ash	0.816	kg

3.3.5.3 Scenario III –Recycling for Secondary Use - Particleboard

Another potential disposal route for discarded treated VBC pole is to be recycled as valuable products. Preservative treated wood waste recycling into particleboard is one possibility according to Clausen et al. (2001). Daian and Ozarska (2009) noted that despite the high cost involved to clean up the solid or chemical contaminants in engineered wood waste, recycling wood waste as particleboard is still a feasible end of life management scenario. Therefore, ACQ-treated VBC poles are assumed to be recyclable as particleboard in this paper.

The VBC pole inputs are cleaned, chapped and flaked into particles, dried and then sprayed with liquid adhesive. Before drying to a moisture content of between 3-5%, all wood is screened and refined and milled by hammer mills to correct shapes and sizes (Wilson, 2010). All particles are sent through dryers. The energy input for drying the wood particles is highly dependent on the moisture content of the wood inputs. Process waste (wood waste) is used as wood fuel in the boilers. The heat and steam generated are used during particles drying. Natural gas is used as the supplementary fuel due to lack of wood fuel to burn in the boiler. The environmental loads associated with both industrial wood and industrial residue wood are sourced from Ecoinvent

(2016). The consumption of wood per m^3 particleboard is approximately 1.39 m^3 of wood materials. Finally, at the end-of-life, particleboards are disposed to landfill equipped with an emission collection system within an average transportation distance of 100km. The associated flows for production and recovered shredded wood for use in the production of new particleboard are summarised in Table 3.5.

Table 3.5 Input materials associated with production of shredded recovered wood for recycling per functional unit [1] (adapted from (Merrild and Christensen, 2009, Tucker et al., 2009, Wilson, 2010, DCCEE, 2011a, DCCEE, 2011b, NSW Environment Protection Authority, 2012))

Input Material	**Value/function unit**	**Unit/pole**
Waste VBC pole	3.71	kg (606kg/m^3)
Collection and transportation	0.8162	tkm
Diesel for chipping	0.0223	L
Diesel for screening	0.0048	L
Diesel consumed to refine and mill wood	0.0074	L
LPG	0.0311	L
Natural gas for drying	0.3941	MJ
Wood fuel	0.8375	MJ
Electricity consumed to refine and mill wood	0.1626	kWh
PF resin	0.259	kg
Output Material		
Particleboard	2.6594	kg
Solid waste	0.8003	kg

3.4 Results

To assess the processes that result in environmental impact from ACQ-treated VBC utility pole, impact indicator values are added for the entire life cycle stages. The impact indicator values at each of the four life cycle stages and a total for the cradle-to-grave life cycle of ACQ-treated VBC pole are reported in Table 3.6.

[1] The produced particleboard was assumed to substitute the oriented strand board, the avoided emissions were considered.

Table 3.6 LCA result of different disposal scenarios per functional unit

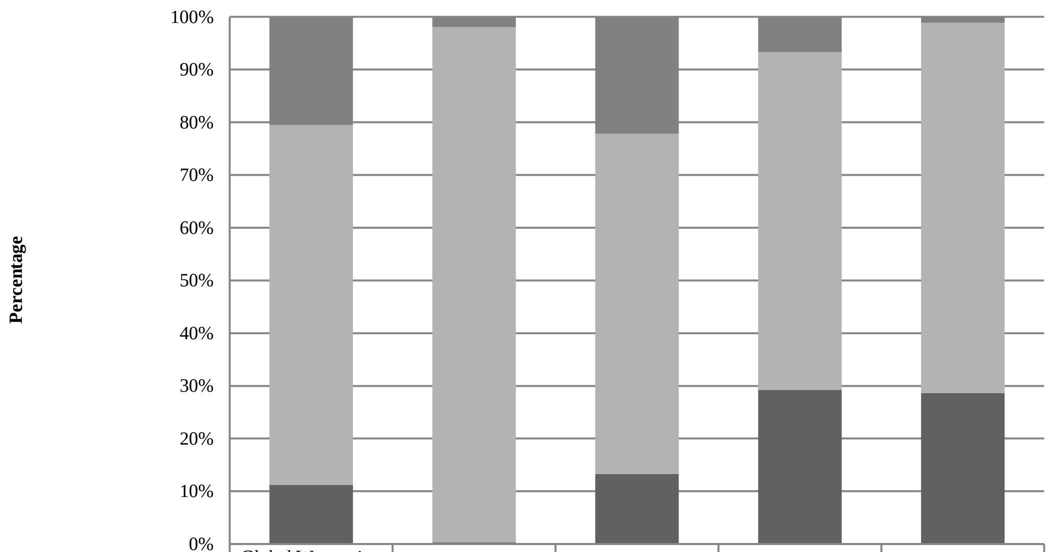
Impact category	Units	Disposal Scenarios		
		Landfilling	**Incineration**	**Particleboard**
GWP	kg $CO_{2\ eq}$	0.64	0.37	0.72
AP	kg $SO_{2\ eq}$	2.21×10^{-3}	2.12×10^{-3}	2.79×10^{-3}
EP	kg $PO_{4\ eq}$	3.90×10^{-3}	3.81×10^{-3}	4.03×10^{-3}
FDP	kg Oil $_{eq}$	0.37	0.36	0.43
ETP	CTUe	12.32	12.34	12.46

According to the results, landfilling and incineration have similar environmental impacts on all assessed categories except GWP. Recycling VBC poles into particleboard are clearly the poorest performer. Compared to landfilling scenario, energy recovery via incineration presents less environmental impact in GWP (0.37 kg $CO_{2\ eq}$) mainly due to avoided methane release. Energy recovery from waste incineration also contributes to slightly lower impact on fossil depletion potential (0.36 kg Oil $_{eq}$) and acidification potential ($2.12\text{x}10^{-3}$ kg $SO_{2\ eq}$) due to the extra energy generation, hence avoiding the emissions releases related to acidification from power generation. Additionally, incineration has less impact on eutrophication potential ($3.81\text{x}10^{-3}$ kg $PO_{4\ eq}$) which may be attributed to the destruction of ACQ preservative during waste combustion. Landfilling scenario has slightly less impact on Eco-toxicity potential (12.32 CTUe) attributed to avoided secondary transportation (transporting ash to the landfill after waste incineration). While, it is clear that recycling is the least environmentally sound option for end of life management of VBC poles, based on the given results, it is very hard to declare either incineration or landfilling as the preferable as both perform very closely.

Bolin and Smith (2011) used LCA to compare the environmental impacts of pentachlorophenol (Penta) treated solid wooden utility pole to steel and concrete poles. Although it is difficult to compare our results to Bolin and Smith (2011) as their study focused on softwood solid poles from mature trees and used US emission factors, our study indicates that VBC hollow poles may have higher environmental burdens than Penta-treated softwood solid pole when considering landfilling as the end of life treatment. On the other hand, our results are in general agreement with Bolin and Smith (2011) findings that wooden pole has lower environmental impacts than concrete and steel utility poles.

Wood waste recycling as particleboard has a higher impact than the other two scenarios on all assessed categories. Despite generating extra credits such as valuable products, the negative impacts caused by secondary manufacturing and transportation overweighed these credits. Nevertheless, recycling scenario contributes economic benefits, but life-cycle cost analysis is not included in this study.

Figure 3.1 shows the percentages of contributions to emissions from first three stages, including thinning logs production, VBC poles manufacturing and service in utility line. VBC poles manufacturing present highest impact on GWP, acidification potential and fossil depletion potential mainly attributed to long-distance transportation and fossil fuel consumption. In addition, most of the eutrophication impacts are caused by ACQ preservative treatment in VBC pole manufacturing stage. Furthermore, PF resin consumption represents approximately 60% of total eco-toxicity potential during pole manufacturing, followed by hardwood logs thinning process, ACQ treatment as well as transportation.



	Global Warmming Potential	Eutrophication	Acidification	Fossil depletion	Eco-toxicity
In Service	0.05972	0.00007	0.00035	0.02219	0.14066
VBC Pole Manufacturing	0.19795	0.00356	0.00102	0.21278	8.55389
Hardwood thinning	0.03265	0.00001	0.00021	0.09698	3.48795

Figure 3.1 Percentage of emissions from first three stages

Figures 3.2 to 3.6 present comparisons of different emissions generated only from the final disposal stages for the three End-of-life scenarios. Landfilling provides the better environmental performance in eco-toxicity potential than the other two scenarios. Energy recovery via incineration option performs better in global warming potential, acidification potential, fossil fuel depletion potential and eutrophication potential. Recycle option represents more impact than the other two scenarios on all assessed categories.

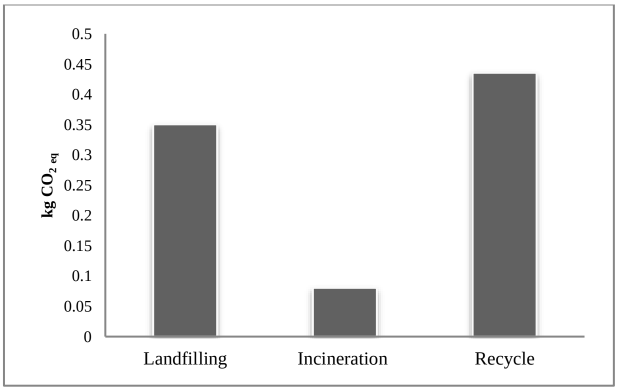


Figure 3.2 Comparison of GWP100 potential

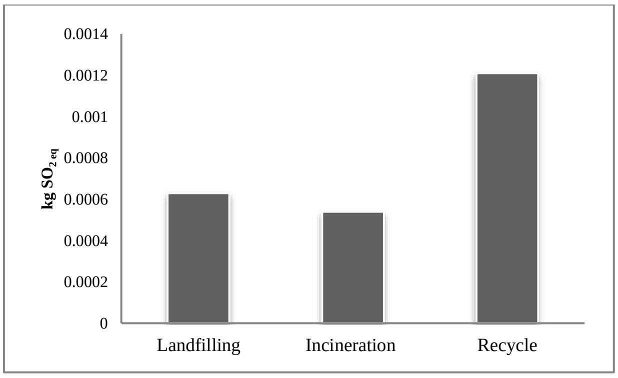


Figure 3.3 Comparison of Acidification potential

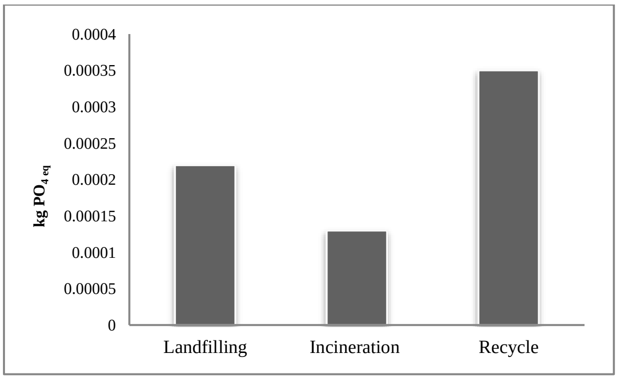


Figure 3.4 Comparison of Eutrophication potential

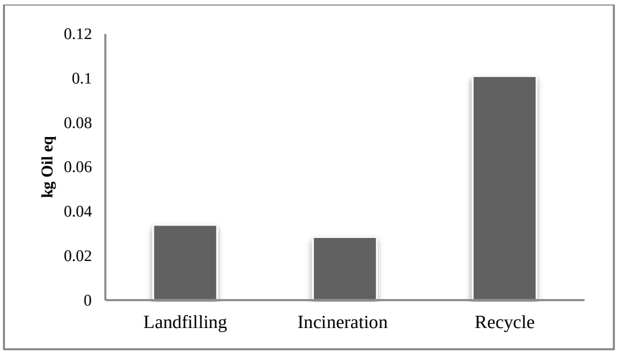


Figure 3.5 Comparison of Fossil Depletion potential

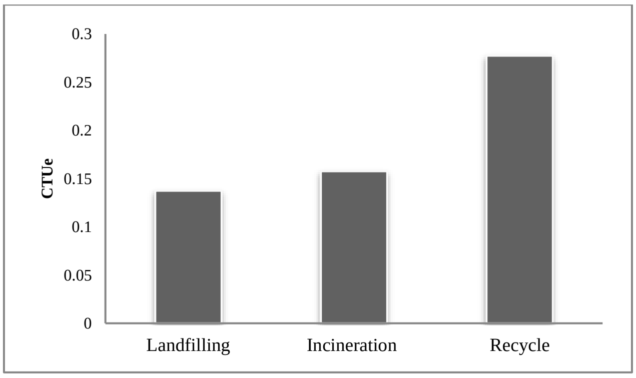


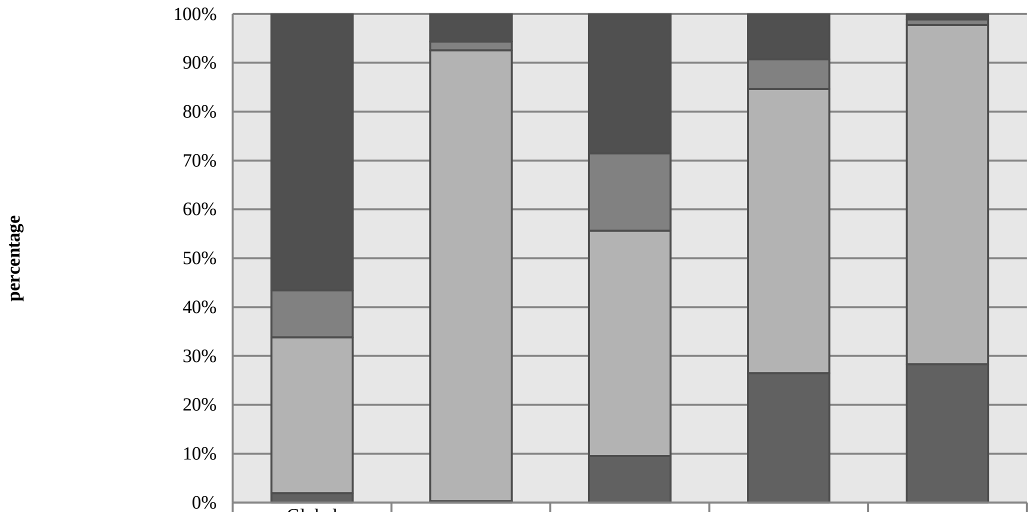
Figure 3.6 Comparison of Eco-toxicity potential

3.4.1 Data Quality Analyses

Base on ISO14040 (2006), a data quality analysis is provided in this LCA. The gravity analysis, uncertainty analysis and sensitivity analysis are detailed and discussed in the following sections.

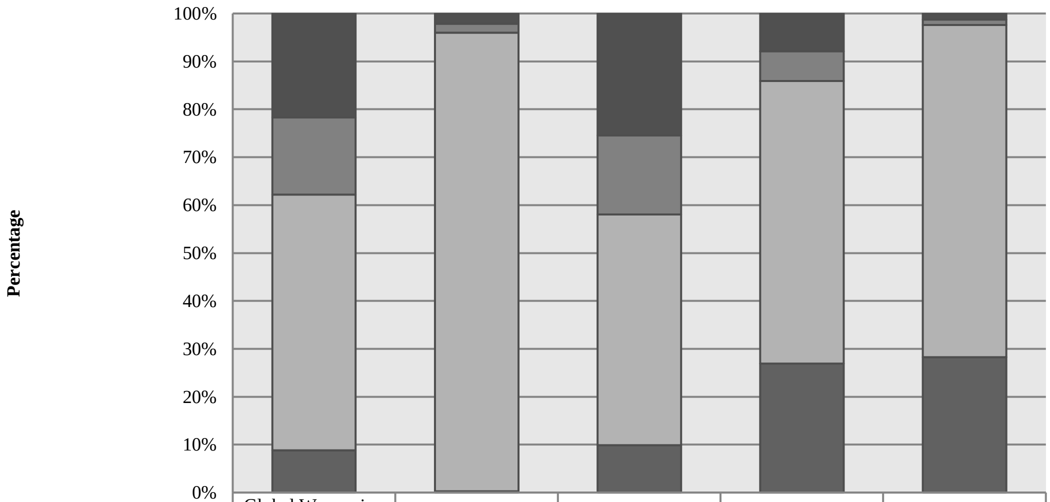
3.4.1.1. Gravity analysis

Figure 3.7 and Figure 3.8 present the different emissions generated during each stage in landfilling and incineration scenarios. VBC pole manufacturing and final disposal stages contribute most of the environmental impacts, particularly due to transportation, fossil fuel consumption, PF resin production and ACQ use. Pole manufacturing is responsible for most environmental impact under eutrophication potential, fossil fuel consumption and eco-toxicity categories because of ACQ preservative usage, energy consumption (diesel fuel, electricity and natural gas) and PF resin production. Eco-toxicity impacts are mostly contributed by hardwood plantation thinning and pole manufacturing stages.



	Global Warmming Potential	Eutrophication	Acidification	Fossil Depletion	Eco-toxicity
Landfilling	0.35064	0.00022	0.00063	0.03395	0.13750
In Service	0.05972	0.00007	0.00035	0.02219	0.14066
VBC Pole Manufacturing	0.19795	0.00360	0.00102	0.21278	8.55389
Hardwood thinning	0.01194	0.00001	0.00021	0.09698	3.48795

Figure 3.7 Comparison of emissions from different life stages for landfilling scenario



	Global Warmming Potential	Eutrophication	Acidification	Fossil depletion	Eco-toxicity
Incineration	0.08042	0.00008	0.00054	0.02852	0.15750
In Service	0.05972	0.00007	0.00035	0.02219	0.14066
VBC Pole Manufacturing	0.19795	0.00356	0.00102	0.21278	8.55389
Hardwood thinning	0.03265	0.00001	0.00021	0.09698	3.48795

Figure 3.8 Comparison of emissions from different life stages for incineration scenario

- In landfilling scenario, final disposal stage contributes the largest impact on GWP in the whole life cycle. More than 40% of GWP is attributed to methane release from the decay of the treated wood in the landfill. However, this result is subject to the assumptions, especially related to GHG emissions generation and capture. The second large contributor to GWP is transportation process, which presents approximately 30% of total GHG emissions. The rest of GWP impacts are caused by fossil fuel energy consumption during VBC pole manufacturing process. Due to the fact that most of the heating energy used in drying is sourced from wood fuel. Therefore, the CO_2 emission generated is biogenic and not counted as GHG emissions. Therefore, the drying stage contribution to the GWP category is smaller than expected. Meanwhile, more than 55% of GWP is attributed to transportation process in incineration scenario. GHG emission release during the incineration of wood waste only presents 5% of total GWP as the CO_2 generated from the combustion of wood is biogenic. Fossil fuel energy used for the manufacture of VBC poles contributes approximately 35% of total GWP100.

- For both landfilling and incineration scenarios, more than 90% of the emissions potentially resulting in eutrophication impact caused by ammonia evaporation during the ACQ preservative consumption; and the rest of the contributions are due to the disposal and transportation process.

- Emissions releases related to acidification occur in all life stages. A significant contributor is transportation process, which caused approximately 35% of total impact in both landfilling and incineration scenarios. In addition, PF resin manufacturing contributes another 35% of impact to acidification potential. Diesel fuel, natural gas consumption and final disposal contribute the remainder.

- Fossil fuel and energy inputs in the entire life cycle mostly occur during the manufacturing stage. Approximately 60% of fossil depletion impact result at the natural gas and diesel fuel usage during the VBC pole manufacturing, particularly due to veneer drying in both landfilling and incineration scenarios. The rest of impacts are mainly caused by transportation process. However, due to energy recovery through wood waste incineration, the energy credit due to the avoidance of using fossil fuel contributes to the abatement of fossil depletion potential. Hence, incineration scenario performs slight better result than landfilling scenario.

- PF resin manufacturing contributes 60% of the emissions potentially resulting in ecological toxicity in both scenarios. Approximately 25% of emissions occur during the hardwood thinning processing while ACQ is responsible for 10% of the ecological toxicity impact. However, due to the need for secondary transportation to deliver the bottom ash to the landfill after wood waste combustion, incineration scenario has a slightly higher impact than landfilling under eco-toxicity potential impact.

3.4.1.2. Uncertainty analysis

Uncertainty in the LCA model is introduced due to various factors such as assumptions made based only on professional judgment and incomplete process data (e.g. Copper released during ACQ treatment, PF resin manufacturing, transportation distances, and disposal). These uncertainties may impact the final results because of variations in production facility containment structure integrity, production facility, manufacturing process, regional location of the treating facility, and disposal site characteristics.

The LCI of landfilling and incineration are based on AusLCI (2011), however, different assumptions result in variability of impact indicator values, especially for GWP100. For instance, some landfills in Australia install methane collection systems to recover methane, thus methane emissions from landfills are decreased. Because of these uncertainties, further analysis was conducted as part of the sensitivity analysis. The uncertainties most likely to impact the results of this LCA study for both landfilling and incineration options are discussed below.

1. ACQ Production

The preservative manufacturers did not provide detailed LCI input and output data for ACQ production, this study made assumptions and used analogous processes to estimate the inputs and outputs for ACQ manufacturing process based on the data given in the published relevant literature. Assumptions used in this LCA may largely affect the total eutrophication and eco-toxicity potential.

2. PF Resin Consumption

The LCI data of PF resin consumption is collected from different sources in the relevant literature. The sources provided a wide range of the PF consumption (10 to 60 kg/m^3) during the compressing process. This study made assumptions and used the weighted average method to estimate the PF resin consumption for VBC panel manufacturing process. The uncertainty of these differences of PF resin consumption is large and may impact the eco-toxicity potential.

3. Transportation

Transportation required during pole manufacturing, service, and final disposal are estimated based on the assumptions made. The uncertainty of these assumptions is large because of variations in the regional location of the plantation site, manufacturing facility and disposal area. The transportation process has been considered as one of the largest contributors in the whole life stages, which may significantly affect the global warming potential, acidification potential and fossil fuel depletion.

4. Disposal and Releases

For assessing the disposal stage, some assumptions created have a significant impact on indicator values, especially for GHGs releasing during the decay of wood in landfills. For instance, an assumption was made that the ACQ-treated wood is degraded to the same degree as untreated wood in this LCA. This assumption used may result in uncertainty when calculating GHGs, as VBC product includes composition resins, preservative, and inorganic compounds would affect the emissions generated during decay in landfills (Ximenes et al., 2015). Further analysis should undertake to investigate the uncertainties as part of the sensitivity analysis.

3.4.1.3 Sensitivity analysis

Sensitivity analysis is completed to determine the effects of assumptions change on LCA results. Items or categories, which show high sensitivity effect on impact indicator, are discussed in details below. Additional information and model results are included.

1. ACQ Retention

As a sensitivity analysis, ACQ retention in VBC products is adjusted, as undertreating and over treating of VBC pole may occur. The baseline used in this assessment is 4.4 kg/m^3. Two cases are modelled for sensitivity, including: 1) under-treatment at 2.4kg/m^3 and 2) over-treatment at 6.4kg/m^3. As expected, changing the amount of preservative affects LCA results. Since production of ACQ is ecological toxicity and eutrophication intensive, increasing the ACQ content from 2.4 to 6.4kg/m^3 results in increasing total ecological toxicity by 8% and eutrophication by 76% in landfilling scenario, while increasing total ecological toxicity by 8% and eutrophication by 78% in the incineration scenario (Shown in Figure 3.9). Eutrophication is most dramatically impacted because ACQ contains a significant amount of NH_4^+.

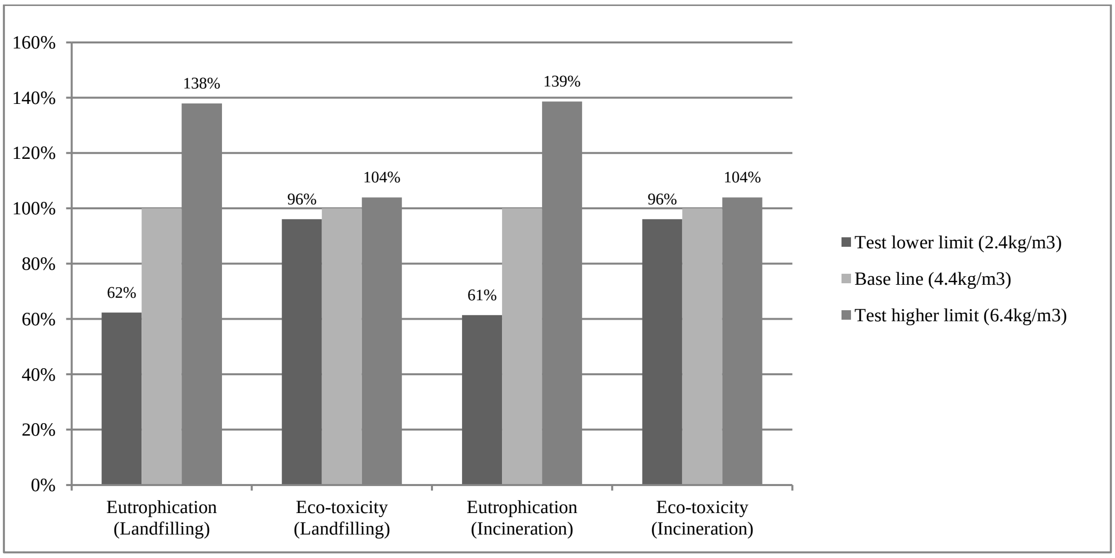


Figure 3.9 Sensitivity analysis for landfilling and incineration scenarios: ACQ retention

2. PF Resin

PF resin consumption in VBC products is adjusted due to the different manufacturing technologies available. The baseline used in this assessment is 53.6kg/m^3. Two cases are modelled for sensitivity analysis, including: 1) at 10kg/m^3 and 2) at 60kg/m^3. For landfilling option, increasing the PF resin consumption, from 10 to 60kg/m^3, increases total global 22% on global warming potential, 34% on acidification, 37% on fossil depletion and 59% on ecological toxicity. Eutrophication potential is not significantly affected by changing the quantity of PF production, which only increases 4%. Meanwhile, it increases total GWP100 by 38%, acidification by 33%, fossil depletion by 38% and ecological toxicity by 59% in incineration scenario. The eutrophication potential is not significantly affected by changing the quantity of PF production, which only increases 4% in total (shown in Figure 3.10).

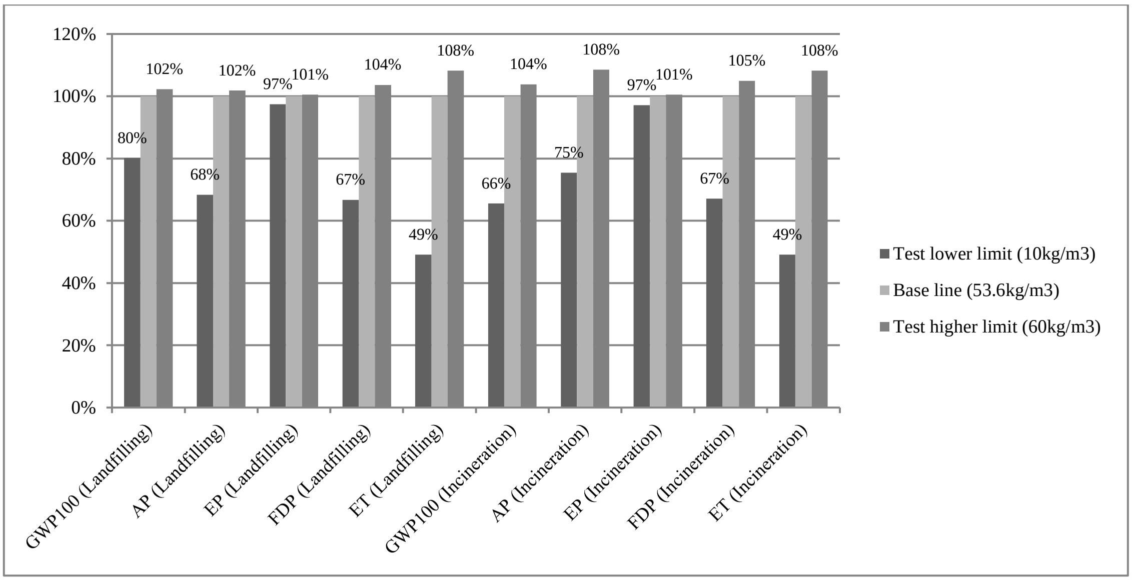


Figure 3.10 Sensitivity analysis for landfilling and incineration scenarios: PF resin production and consumption

3. Transportation

In this study, transportation distance of product from manufacturing facility to the utility system is assumed as 100 km per functional unit with 10% return load. However, distances in the range of 50 -150 km are reported. As expected, changing the transportation distance from 50km to 150km increase total global warming potential by 9%, acidification impact by 16% and fossil depletion by 6% in landfilling option. In the case of incineration, it resulted in increasing 16% on global warming potential, 17% on acidification impact and 6% on fossil depletion potential. Figure 3.11 shows the effect of transportation on different impact categories.

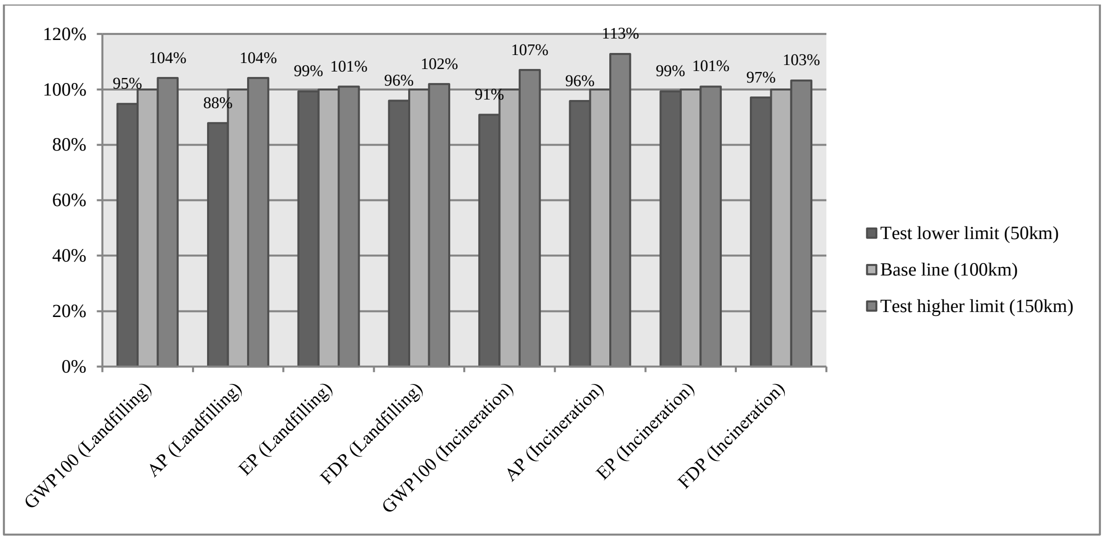


Figure 3.11 Sensitivity analysis for landfilling and incineration scenarios: transportation distances

4. Landfilling Disposal

This LCA assumes 77% of the carbon absorbed by wood is sequestered after decomposition in a landfill as it was reported in Barlaz (1998). However, preservative and other inorganic compounds in the disposed of wood are expected to retard carbon releasing when compared to untreated wood (Bolin and Smith, 2011). Two cases are modelled for sensitivity; 50% and 90%. The results confirm that higher sequestration rate reduces the GWP impact indicator by approximately 14% while reducing sequestration to 50% increase the GWP impact by 30% when compared to the baseline model (shown in Figure 3.12). Nevertheless, a field investigation of wood decay in Australian landfills revealed that wood deposited in landfills have remained virtually intact (Ximenes et al., 2015). Therefore, it is most likely that the GWP of the landfilling option is much smaller than what is predicted in this LCA study.

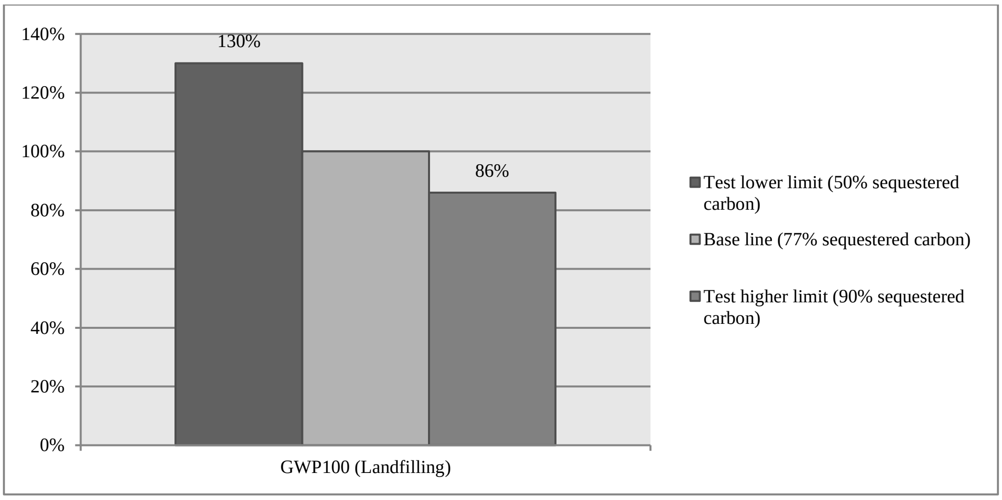


Figure 3.12 Sensitivity analysis for landfilling scenario: landfilling disposal

3.4.2 Limitations

The scope of the study is limited to boundaries established in the goal and scope documented in this study. Limitations included reliance on the published and publically available information in many instances. Such information is assumed to be accurate. The life cycle inventory completed for VBC pole manufacturing is designed to represent the typical or average product on the market. Nevertheless, VBC pole-manufacturing impacts may vary according to the technology and scale used. Inventory data for the wood waste disposal are sourced from the Australian LCI database. This study only assessed one secondary use of end of life VBC pole (particleboard); other secondary use scenarios are not considered. Although recycling may have economic benefits, the life cycle cost was not evaluated in this study. Further studies using life cycle costing is needed to gain a comprehensive understanding of the system. This LCA focused on ACQ-treated VBC pole. While portions of this LCA may apply to pole treated with other preservatives, the overall conclusions only apply to ACQ-treated VBC products.

3.5 Conclusions

This LCA study assessed the environmental impact of VBC hollow pole using life cycle methodology. Three different disposal scenarios for the end of life treatment were considered, namely: landfilling; incineration; and recycle as particleboard. Landfilling and incineration options showed better environmental performance than recycling scenario. Although recycling and reuse of VBC poles generate extra credits due to avoid the use of virgin material, these scenarios resulted in higher negative impacts due to secondary manufacturing and transportation emissions. The environmental impacts were mostly caused by the VBC pole manufacturing process. Long distance transportation and fossil fuel consumption due to pre-conditioning, veneer drying and compressing are the main contributors to the global warming potential, acidification and fossil depletion potential. Production of PF resin is a major contributor to ecological toxicity. In addition, ACQ preservative used for VBC pole treatment has a significant effect on eutrophication impact due to ammonia release.

This study focuses solely on the environmental performance of the VBC pole. It is recommended that the economic and technical feasibility of the alternatives should be considered. Furthermore, other energy utilisation alternatives should be evaluated and perhaps an update to the industry best practice may be promoted. Further research is needed to evaluate the environmental, economic and social impacts to achieve a comprehensive understanding of the viability of VBC hollow utility pole as an alternative for Australian utility networks.

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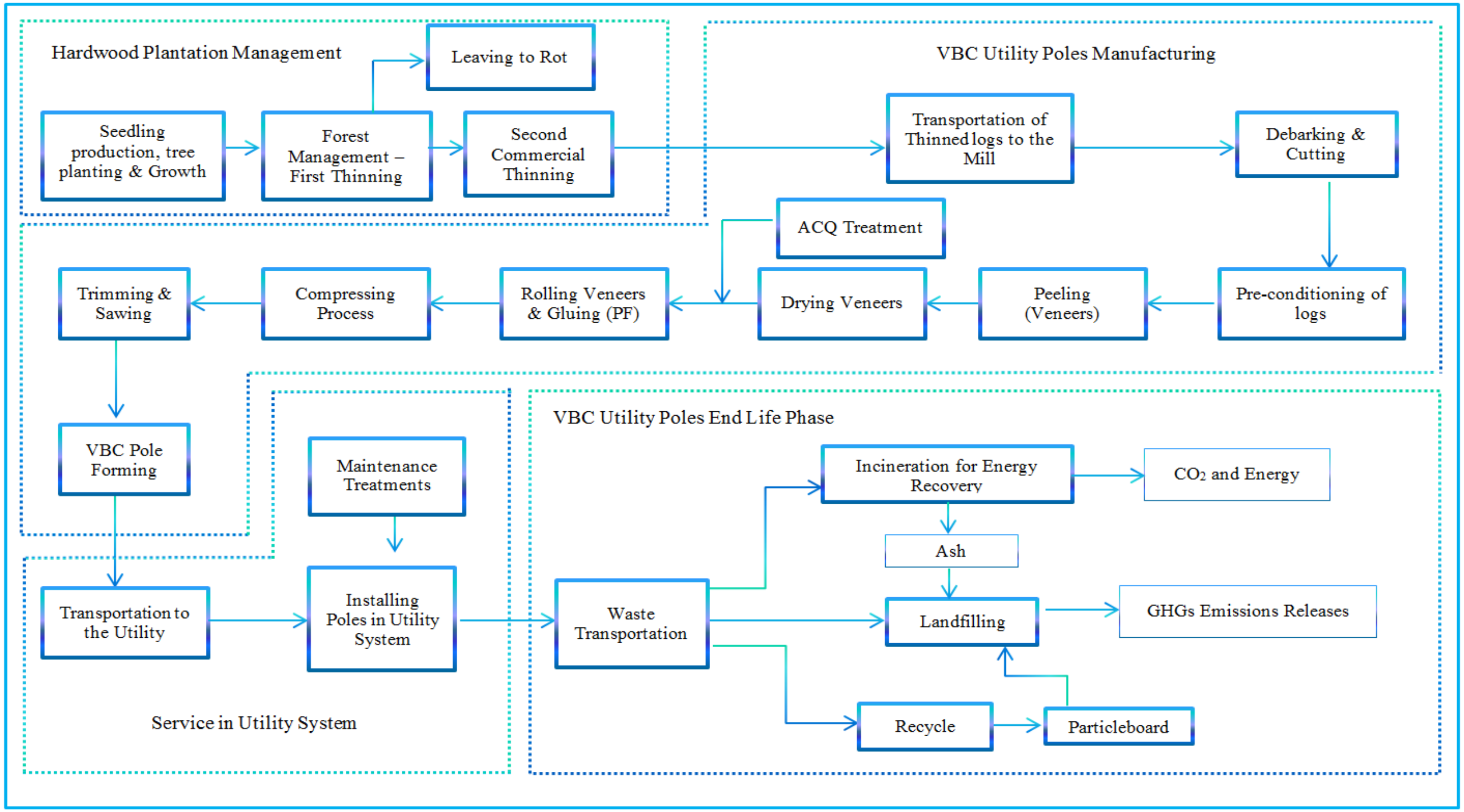
Appendix

Appendix A. System boundary of VBC utility poles



CHAPTER 4 - RESEARCH ARTICLE 2

STATEMENT OF CONTRIBUTION TO CO-AUTHORED PUBLISHED PAPER

Chapter 4 includes a pre-print of a co-authored paper titled “Environmental and economic assessment of utility poles using life cycle approach” which was published in the Clean Technologies and Environmental Policy Journal. My estimated contribution to the paper is 65%. I contributed to research design, data collection, modelling, analysis, and writing the manuscript.

Paper citation: Lu, H. R., & El Hanandeh, A. (2016). Environmental and economic assessment of utility poles using life cycle approach. Clean Technologies and Environmental Policy, 4(19), 1047-1066. doi: 10.1007/s10098-016-1299-4.

(Signed) ________________________________ (Date) ______________

(Countersigned) __________________________ (Date) ______________

Corresponding author of paper:

(Countersigned) __________________________ (Date) ______________

Supervisor:

Environmental and Economic Assessment of Utility Poles Using Life Cycle Approach

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Abstract

Due to increasing demand for utility poles and the banning of native forests logging in Australia, it is necessary to find sustainable alternatives to round wood utility poles. Currently, steel and concrete are the most common alternatives. Veneer Based Composite (VBC) is a newly developed product made from hardwood plantation mid-thinning. To assess the viability of VBC, comparative Life cycle assessment (LCA) and life cycle costing (LCC) analysis were conducted. Two end-of-life scenarios for VBC pole were assessed: incineration with energy recovery and landfilling. Five impact categories were considered: global warming (GWP); acidification (AP); eutrophication (EP); fossil depletion (FDP) and human toxicity (HTP). VBC pole with incineration showed the best environmental performance, particularly on GWP (63.22kg-CO_2-Eq), AP (0.29kg-SO_2-Eq), FDP (30.78kg-Oil-Eq) and HTP (2.27kg-1,4-DB-Eq), which are less than half of concrete and steel poles. However, VBC had higher EP than concrete and steel due to use of adhesives and preservatives. VBC pole also had the lowest LCC ($1529); due to use of low-value materials and lower manufacturing cost. The LCC showed that both VBC scenarios performed equally on economic grounds. Sensitivity analysis showed that service life was the most sensitive parameter affecting both environmental and economic results, especially the VBC. Transportation distances and fossil fuel consumption also had significant effects on LCA result. Monte Carlo analyses further revealed that despite the high levels of uncertainties in the input parameters, the overall ranking of the options remained the same with VBC being the best performer and concrete the least.

Key Words:

VBC; concrete; Steel; pole; LCA; LCC

4.1 Introduction

Utility poles are a significant component in the Australian infrastructure sector. Timber is typically used for manufacturing utility power poles due to its low initial cost, natural insulation properties, and ease of transport (Shafieezadeh et al., 2014). There are more than 5 million timber utility poles in service in Australia, with approximately 75,000 new poles being installed annually (Kent, 2006). However, due to growing environmental awareness and concerns about the sustainability of Australian natural forests, agreements were signed to phase out logging of native forests; as a result limiting the supply of hardwood utility poles significantly (Francis and Norton, 2006). Finding new materials to satisfy the shortage of poles presents a significant challenge to the industry (Australian Government Department of Agriculture Fisheries and Forestry, 2014). Steel and concrete poles are viewed as the most viable options for replacing natural hardwood poles. Concrete poles can be supplied in specific sizes that offer strength properties that match or exceed those of native hardwood pole. Furthermore, concrete poles have high structural reliability and low maintenance costs (Stone and Wood, 2010). Steel poles have a relatively low maintenance cost, lightweight, consistent performance, and long lifespan (Deegan, 2008). Veneer based composite (VBC) manufactured from low-value hardwood thinning logs is seen as an opportunity to manufacture hollow utility poles (Underhill et al., 2014), which may offer an opportunity to replace the traditional hardwood poles (Gilbert et al., 2014a). Deegan (2008) emphasised that the argument for a material to be used in the utility network should be made based on both environmental and economic perspective. However, existing studies focus only on either the environmental impacts or economic analysis but rarely combining the two criteria. Furthermore, the existing studies focused on well-established alternatives such as steel, round wood and concrete and ignored emerging materials such as the VBC. Therefore the objective of this study is to conduct a comparative Life Cycle Assessment (LCA) and Life Cycle Costing (LCC) analysis of three alternatives to hardwood utility poles: VBC, steel and concrete in South-east Queensland region. Results of this study might have implications for Australian forestry practices and utility companies to facilitate a better sustainable management strategy for hardwood plantations and utility networks.

4.1.1. Environmental Aspects

Wood product substitution is an important long-term management strategy for mitigating climate change. Existing LCA studies indicated that significant energy saving and greenhouse gas (GHG) emissions reduction can be realised by substituting wood to the alternative materials. For example, LCA studies have shown that the use of wood products as a substitute material for steel and concrete in the construction sector may realise more than 80% reduction in GHG emissions (Gustavsson et al., 2006, Eriksson et al., 2007, Werner and Richter, 2007).

Sedjo (2002) compared the energy consumptions and GHG emissions during pole manufacturing using wood and other alternative materials. According to Sedjo (2002), less GHG emissions and energy were required during manufacturing of wood utility poles compared to steel and concrete. However, Sedjo (2002) excluded the impact from wood preservative used in this study. Although, most of the LCA studies indicated that there is significant energy saving and GHG emissions reduction by manufacturing wood poles instead of alternative materials, the environmental impacts caused by chemicals consumption in the wood preparation phase are still a concern (Nebel et al., 2006, Rivela et al., 2007, González-García et al., 2009). Different types of preservatives are used for treating wood utility poles to protect it from microbial and fungal decay (Chirenje et al., 2003). Bolin and Smith (2011) conducted a full LCA to evaluate the environmental impacts associated with pentachlorophenol (Penta)-treated wooden utility poles and compared it to concrete and steel poles. Their results indicated that Penta-treated poles had lower environmental impacts than concrete and steel poles. Lu and El Hanandeh (2016b) found that the use of Alkaline Copper Quaternary (ACQ) preservative is a major contributor to eutrophication and human toxicity potential of the ACQ-treated VBC utility pole.

The environmental benefits of using a different material for manufacturing utility poles have been argued in various studies due to their different manufacturing processes, lifespans, treatment and maintenance requirements and disposal options (e.g. Sedjo, 2002, Kuenniger and Richter, 1995, Deegan, 2008, Bolin and Smith, 2011). In order to aid utility industries in their decision making about mitigating environmental burdens, and identify appropriate alternatives to overcome the limited supply of wooden utility poles in Australia, there is a need to evaluate any newly developed material and compare its environmental performance to the available options.

4.1.2 Costs of Wood Products vs Competing Materials

In a comparison of total cost of wooden, concrete and steel applications, wood has been commonly seen as a cost-effective option (Petersen and Solberg, 2005). There are a few studies that compared the total cost of wood-based products to other materials such as steel, aluminium or concrete in the building and construction sectors. For instance, Petersen and Solberg (2005) concluded that wood as a building material is competitive on price compared to concrete or steel structure. Sathre (2007) also indicated that the total cost of the wood building was lower than other common non-wood building materials, mainly due to the lower energy cost for material manufacture.

A number of studies comparing life cycle cost of utility poles manufactured by different materials were also found in the literature. For example, Deegan (2008) noted that steel poles are more expensive than wooden poles due to higher transport costs. However, steel poles typically have a longer lifespan than other materials. Steel poles can also be mostly recycled at the end of their life, which could lead to considerable financial benefits. Concrete poles are relatively more expensive than other materials because of their higher transportation cost owing to their heavy weight (Protection, 2014). Francis and Norton (2006) suggested that although timber composite poles might cost more than natural round-wood poles; they are still expected to be less expensive than most non-timber poles because some of the manufacturing costs associated with producing reliably strong and durable composite poles are often offset by more economical raw resources. Deegan (2008) concluded that the life-cycle costs of engineered wood pole can vary considerably and is affected by raw material costs, manufacturing cost (i.e. labour, machine depreciation and general factory overheads), energy required to manufacture the pole, waste generated during the manufacturing process, costs of transportation, costs of installing the pole, maintenance costs associated with using the pole (maintenance costs will be higher in hard-to-access areas); environmental impacts associated with the maintenance (e.g. the consequences of chemical treatments), and the disposal costs. However, there are no existing studies that focused on comparing the economic impact of preservative treated engineered wood poles as LCA studies do not typically include economic aspects. Hence, the existing challenge is to combine traditional LCA with economic analysis in order to gain a comprehensive view of the real cost of the different utility pole alternatives. Therefore, a comprehensive life cycle study is required to fill the knowledge gaps on both environmental and economic impacts of utility pole alternatives.

4.2 Methods

4.2.1 Life Cycle Assessment and Life Cycle Costing Analysis

A scientifically based and comprehensive comparison of the environmental and economic performance of the VBC hardwood hollow utility pole and its alternatives (steel and concrete poles) was conducted. This study used ‘cradle to grave’ LCA and LCC analysis based on the International Organisation for Standardisation (ISO14040, 2006) and Life Cycle Costing Guideline (Treasury NSW, 2004). Primary data was used whenever possible, secondary data on background operations and energy was collected from various sources in the literature as described in the following sections.

4.2.1.1 Life Cycle Assessment

This study follows the ISO14040 (2006) standards. OpenLCA software was used to conduct the LCA analysis (GreenDelta, 2014). The impact categories considered in this study included global warming potential (GWP100), eutrophication potential (EP), acidification potential (AP), fossil depletion potential (FDP) and eco-toxicity potential (ETP). These impact categories were chosen based on the Best Practice Guide to Life Cycle Impact Assessment in Australia (Grant and Peters, 2008). ReCiPe Midpoint (H) life cycle impact assessment (LCIA) method was used for the characterisation of emissions into the relevant impact categories except for the AP category for which the CML 2001 method was used as suggested by Bengtsson and Howard (2010b) and El Hanandeh (2015).

4.2.1.2 Life Cycle Costing Analysis

According to Life Cycle Costing Guideline (Treasury NSW, 2004), LCC estimates all relevant costs throughout the life period. It includes manufacturing costs, maintenance and replacement costs, energy costs, and residual values. Costs are estimated at their Present Value (PV). Based on Deegan (2008), the life-cycle costs of a power pole should include: the raw material costs, the manufacturing cost (labour and machinery), energy cost, pollution treatment cost, transportation cost, installation, maintenance, and removal cost, environmental impacts cost, as well as the disposal costs (and revenue). For analysing the economic viability of different utility poles, an Excel spreadsheet model was used to calculate the net present value (NPV), derived from estimated cash flows over the simulation period of 60 years. The lifespan of the steel and concrete pole was estimated to be 60 years based on the average lifetime reported in the literature (Bolin and Smith, 2011). The service life of VBC pole was assumed to be 25 years according to Lu and El Hanandeh (2016b). Therefore, 2.4 of VBC poles were considered in 60 years' simulation period. The economic assumptions and parameters for this analysis are listed in Table 4.1.

Table 4.1 Economic parameters for Life-Cycle Costing Analysis*

Parameter Value	Parameter Value	Sources
General inflation rate	3%	Reserve Bank of Australia (2015)
Discount rate	4.9%	Reserve Bank of Australia (2015)
Diesel fuel cost	$1.37/L	Australian Institute of Petroleum (AIP) (2016)
Gasoline	$1.43/L	Australian Institute of Petroleum (AIP) (2016)
Natural gas	$10/GJ ($0.387/m^3)	Australian Institute of Petroleum (AIP) (2016)
LPG	$0.65/L	Australian Institute of Petroleum (AIP) (2016)
Electricity cost	22.24 ¢/kWh	Queensland Government Department of Energy and Water Supply (2015)
Carbon Price	$29/tonne CO2-eq	The Treasury Australian Government (2016)
Transport	0.098$/tkm	Higgins et al. (2015)

* All the prices are expressed in constant dollar terms and discounted to the current financial year.

4.2.2 Scope and Definition

4.2.2.1 System Boundaries

This study evaluated the environmental impact and life-cycle cost of three utility poles made of VBC, steel or concrete from raw materials extraction, pole manufacturing, installation, servicing, dismounting, to final disposal. Transportation processes between the different stages were included. The final treatment scenarios included were: landfilling and incineration with energy recovery for the ACQ-treated VBC poles; material-recycling for the steel poles and landfilling for concrete poles. The utility poles comprise a base region to mount the pole in the ground and a top section to provide a columnar pole assembly. The cross arms to support power cables were also included in this study. However, steel bolts used to attach cross-arms are generally the same for all poles; hence the LCI of steel bolts were excluded. In addition, the LCC also includes the cost of GHG emissions for each option; negative costs are assigned to offset GHG emissions. The LCC data of material, labour and disposal were derived from relevant literature. The operational energy costs data and manufacturing data were collected from the published sources listed in Table 4.1 above. Operational energy costs data included the local energy prices as well as its service charges. The year of 2016 was considered as the base year for life cycle costing analysis. The prices used in this study were based on average market prices in Australia. The assessments were based on 60 years' simulation period.

4.2.2.2 Functional Unit (FU)

A 12.5-meter long (VBC, steel and concrete) utility pole for power transmission was considered as one functional unit in this study because it is the most common length of a utility pole in Australia (Francis and Norton, 2006).

4.2.3 Life Cycle Inventory (LCI)

Data sources and analysis for the cradle-to-grave life cycle of the VBC, concrete and steel utility pole are presented the following subsections. For each stage of the product lifecycle, inputs of energy and raw materials, outputs of products, waste, and emission releases to the environment are collated. The life cycle inventory (LCI) data for upstream processes were collected from different sources, including published literature and online database such as, European Life Cycle Database (ELCD) (2013), U.S. Life Cycle Inventory Database (USDA) (2012) and the Australian National Life Cycle Inventory AusLCI (2011). Electricity grid mix used in this modelling exercise is based on the South East Queensland grid which is comprised of 68% coal-power, 20% natural gas, 5% Hydropower, 3% biomass and the remaining is sourced from other sources, such as solar, wind and waste (AusLCI, 2011).

4.2.3.1 LCI of Concrete Pole

The dimensions of the concrete pole were designed based on Australian/New Zealand Standard for Concrete utility services poles (AS/NZS 4065, 2000). The concrete pole assessed in this study was 12.5 m in length with a 254 mm internal diameter and 312 mm external diameter hollowed structure utility pole. The steel reinforcement includes four strands of 15.2 mm longitudinal reinforcement, a spiral of 3.2 mm-diameter wire with 76.2 m pitch. Concrete components include water, cement, and coarse and fine aggregate. The LCI data on cement, concrete, aggregate, and steel were sourced from the U.S. Life Cycle Inventory Database (USDA) (2012). The concrete pole manufacturing process required electricity and natural gas to heat the concrete for accelerated curing, as well as diesel, and water inputs. The LCI of concrete pole manufacturing was sourced from AusLCI (2011). In this study, all the transportation processes were modelled as by road using the B-double truck. This process started from transporting concrete pole manufacturing component to the casting plant, transportation of concrete poles to the utility system and ultimately to the use site, as well as transporting end-of-life poles to landfill for final disposal. The total transportation distance was estimated as 270 km.

In the final disposal stage, although concrete recycling has been receiving more attention recently, the current practice is still limited in Australia (Senaratne et al., 2016). Furthermore, Bolin and

Smith (2011) also indicated that concrete poles manufactured from high strength concrete are unlikely to be recycled due to the low value of recovered products and the difficulty of the recovery process. This LCA study assumed that all the used concrete was sent to landfill for final disposal, meanwhile, 85% of the reinforcing steel was recycled according to Buchanan et al. (2012). The LCI for concrete pole disposal in landfills involves the inputs and outputs for construction and closure of landfill facilities proportional to the mass of disposed waste poles. The LCI of disposing of concrete in the landfill was sourced from AusLCI (2011). In addition, concrete poles were modelled with inspections occurring once every 25 years. The inputs and outputs for regular inspection and maintenance were included.

4.2.3.2 LCI of Steel Pole

According to the Australian/New Zealand Standard Structural design AS/NZS 4065 (2000) requirements for utility services poles, steel utility poles are typically manufactured from a minimum of 3.048-mm thick sheet steel. Utility poles are generally tapered with the same dimensions as wood poles of similar class and length. The steel used for pole manufacturing was assumed to be 100% new steel. The steel poles are a hollow structure with top caps and a bottom plate. A concrete butt was assumed to be used for supporting the pole. Additionally, the poles were assumed to be hot-dip galvanized with zinc in order to reduce the speed of corrosion.

The LCI data of steel pole manufacturing was sourced from published LCI data on steel production and galvanizing (Ecoinvent, 2016, AusLCI, 2011). Electricity consumption during manufacturing was estimated to be 807.45 kWh per FU of steel utility pole according to Ecoinvent (2016). Approximately 159.28 litres of water were used during hot-dip galvanizing of the steel pole. On average, 2.98 kg of zinc was required for hot dip galvanizing one functional unit of a steel pole. The process of galvanizing steel required pre-heating treatment to between 700 and 800ºC, while keeping the galvanizing liquid around 450 ºC during the galvanizing process. Zinc was assumed to be continually released during the service life due to corrosion and weathering (Hedberg et al., 2013), and it was modelled in the LCI as a release to the ground. The transportation processes considered in this study including transporting raw material to manufacturing facilities, transporting poles to the utility system as well as transporting used poles to recycling sites. Total transportation distances were assumed to be 270 km by truck. In final disposal stage, steel poles were assumed to be 100% recycled. The LCI data for scrap steel recycling was sourced from AusLCI (2011) and Yellishetty et al. (2011). The average service life of 60 years was assumed for steel poles with inspections occurring once every 25 years. It was further assumed that concrete and steel utility poles, designed to provide the same strength as VBC poles, were installed at equivalent spacing. The details LCI of concrete and steel pole are presented in Table 4.2.

Table 4.2 LCI of concrete and steel utility poles

Inputs material and energy	Unit/FU	Concrete Pole	Steel Pole
Electricity, high voltage, Queensland	kWh	703.56	807.45
Natural gas, combusted in industrial boiler and equipment	m^3	43.4	51.53
Diesel fuel, combusted in industrial boiler and equipment	L	5.28	1.90
Gasoline, combusted in industrial equipment	L	2.35	4.49
Coal combusted in industrial boiler	kg	0.17	0.1
Transportation	tkm	578.13	162.36
Zinc	kg	-	2.98
Steel	kg	85.57	260.88
Water	L	267.94	159.28
Cement	kg	370.8	18.63
Aggregate	kg	1075.61	-

4.2.3.3 LCI of VBC Utility Pole

A hollow structure VBC pole was assessed in this study. The external diameter of the VBC pole was 325 mm with 30 mm wall-thickness that is formed by two half-pole jointed together (Gilbert et al., 2014b). The LCI inputs of this stage were based on published data for Australian forestry and wood products. The inventory data of VBC manufacturing, shown in Table 4.3, were based on Lu and El Hanandeh (2016b). Traditionally, Chromated Copper Arsenate (CCA) treatment was used for utility poles. However, due to inefficient results and potential health concerns, the safer ACQ preservatives are now becoming the industry common practice (Janin et al., 2011) and therefore were used in this study. Furthermore, hollow structure poles are typically connected to steel or concrete butt rather than being buried in the ground directly (Francis and Norton, 2006). According to AS/NZS 4065 (2000), a concrete base section is formed and embedded in the ground to support the VBC utility pole. For a pole around 12.5 m high with a working load of approximately 25 kN, the butt diameter has to be more than 499 mm with a nominal depth of 1.85m in the ground. Concreting should extend from the butt to a point 350 mm below the final ground line, hence approximately 39.73 kg of concrete was estimated to be used for each functional unit of VBC pole (AS/NZS 4065, 2000). A cross-arm attached to the upper portion of a utility pole carries power transmission lines. Cross-arms are sawn and surfaced to 89 mm wide by 114 mm thick by 2.44 m long with the same material (Piao and Monlezun, 2010). In addition, the regular inspection and maintenance of utility poles were also considered in this study. Each pole was assumed to be inspected and maintained every 10 years during the service life. The treatment model assumed that 3.91 litter/pole of paste was needed per treatment, consisting of 2% copper, 43% borate (DOT), 10% petroleum, water, and mineral filler/thickeners (AWPA, 2010a, AWPA, 2010b).

Table 4.3 LCI of VBC utility poles (adapted from Lu and El Hanandeh (2016b))

Material Inputs	**Value/FU**	**Unit**
Hardwood thinned log	0.806	m^3
Phenol Formaldehyde Adhesive	20.385	kg
ACQ	1.699	kg
Concrete	39.73	kg
Steel	0.063	kg
Transportation	108.154	tkm
Energy Consumption		
Electricity	51.660	kWh
Natural Gas	1.767	m^3
LPG	0.489	L
Diesel Fuel	0.095	L
Wood fuel	0.072	MJ
Water	0.063	L

Lu and El Hanandeh (2016b) compared different end of life management options for VBC pole and concluded that landfilling and incineration were the two most appropriate scenarios for final disposal. Therefore, these two options were considered in this study.

Disposal Options 1 - Landfilling

Disposal stage begins with the VBC pole waste transports to landfill and includes processing of the landfilling and emissions over the landfill life. The landfilling considered in this study included 100 years of product life after disposal. This time frame was consistent with other solid waste studies (e.g. Diaz and Warith, 2006, El Hanandeh and El-Zein, 2010). Up to 75% of the

CH_4 generated was assumed to be captured by the collection system based on Bracmort et al. (2009). The LCI data for VBC pole in a landfill is presented in Table 4.4.

Table 4.4 LCI of wood waste in landfill per FU (Adapted from Lu and El Hanandeh (2016b))

Input Flow	**Value/FU**	**Units**
Wood waste, at landfill	210.08	kg
Transportation	21.010	tkm
Output Flow		
Carbon dioxide, biogenic	8.439	kg
Methane	1.758	kg
Non-methane volatile organic compounds	0.793	kg
Copper	0.852	kg
Boron	0.113	kg

Disposal Option 2- Incineration for Energy Recovery

VBC Poles were combusted in large cogeneration or utility type boilers that included scrubbers or electrostatic precipitators. Electricity from treated wood incineration was assumed to be distributed to the main grid in South East Queensland. The bottom ash (approximately 220kg/Mg of treated wood product) was assumed to be disposed of in a landfill. The LCI data of ACQ-treated wood in incineration is shown in Table 4.5 based on Lu and El Hanandeh (2016b). The energy recovery through wood waste combustion was modelled as energy credit due to the avoidance of production and combustion of materials from virgin feedstock (i.e. Over 68% of electricity is generated by coal-power, 20% gas, 5% from Hydropower, 3% biomass and the rests are from other sources, such as solar, wind and waste) according to the current Queensland electricity grid mix. The energy content of wood waste was assumed to be 19.5MJ/kg, while the efficiency of power generation from wood waste is 20% based on NSW Environment Protection Authority (2012). It was estimated that 227.5 kWh of electricity would be generated per functional unit of a utility pole.

Table 4.5 LCI of wood waste for energy generation per functional unit (Adapted from Lu and El Hanandeh (2016b))

Input Flow	Value/FU	Units
Wood and wood-waste	210.08	kg
Transportation	25.653	tkm
Output Flow		
Electricity	227.5	kWh
Carbon dioxide, biogenic	342.921	kg
NO_2	0.227	kg
CO	0.059	kg
NO_x	0.005	kg
Copper	0.680	kg
Bottom Ash	46.206	kg

4.2.3.4 LCC of Concrete Utility Pole

The lifecycle cost during concrete utility poles manufacturing stage was estimated at $776.95 in this study. The cost breakdown for manufacturing one FU of a concrete utility pole is presented in Appendix 1, including the cost of energy, raw materials, transportation, as well as labour cost. Due to the lack of data on the installation cost of concrete poles, this study estimated the cost for concrete pole installation inspection and maintenance as $1,071 per pole (2016 price) based on assumptions following Salman and Li (2016). The cost for final removal and treatment of each ageing concrete pole after 60 years' lifespan was estimated at $357.68 and $38.57 respectively in present value based on Salman and Li (2016). Hence the total cost for pole removal and disposal was $396.25. According to the CO_2 equivalent generated through whole life cycle shown in LCA result, the total carbon cost in 60 years was calculated as approximately $6.60 per FU of the concrete pole. Furthermore, this study assumed 85% of reinforced steel was recycled; hence $7.04 would be paid back for recycling each functional unit of the concrete pole. In addition, Yellishetty et al. (2011) estimated that the CO_2 emission offset was approximately as 900 kg per tonne of steel. The CO_2 offset then contributes to approximately $0.63 cost saving. The total LCC of one functional unit of a concrete utility pole was then estimated at $2,243.32 in present value.

4.2.3.5 LCC of Steel Utility Pole

The life cycle costs for manufacturing one FU of steel utility pole were listed in Appendix 2. The costs of energy consumption, materials as well as labour and manufacturing facilities were all included. The life cycle cost during steel pole manufacturing stage was calculated as approximately $736.93. The installation, inspection and maintenance costs of steel poles were estimated by Salman and Li (2016), which was equivalent to $940.74 Australian dollar in 2016. The total carbon cost in 60 years was estimated at $4.30 based on our LCA result. Approximately $5538 was estimated to be spent for removing and disposing of one functional unit of steel utility pole at the end of its service life, which equals to $313.93 in present value. Hyder Consulting Encycle Consulting & Sustainable Resource Solutions (2011) reported the current ballpark price of recycled scrap steel was around $250/tonne. This study assumes 100% of steel was recycled; hence $25.25 would be paid back for recycling each functional unit of a steel pole. The CO_2 offset then contributes to approximately $2.27 cost saving. Therefore, the total life-cycle cost of one FU of steel utility power pole was estimated at $1,968.38. The detailed life-cycle costs of both concrete pole and steel poles are presented in following Table 4.6.

Table 4.6 Detailed LCC of concrete and steel pole in 60-year-lifespan

Items	Concrete Pole	Steel Pole
Pole material and manufacturing cost	$776.95	$736.93
Installation cost	$1,071.20	$940.74
Removal cost & Disposal cost	$396.25	$313.93
Carbon cost	$6.59	$4.30
Revenue	$7.67	$27.52
Total LCA	$2,243.32	$1,968.38

4.2.3.6 LCC of VBC Utility Pole

For assessing the economic performance of VBC utility poles, the total life-cycle cost was broken into several stages presented in following sections. The costs of plantation thinning, energy and materials, manufacturing, disposal, as well as carbon emission cost were included throughout the whole life cycle. The cost data for hardwood plantation thinning management were sourced from an Australian forestry study conducted by Paul et al. (2013a) and Paul et al. (2013b). The VBC utility poles manufacturing costs were obtained from published sources as detailed in following sections.

Stage One - LCC of Hardwood Plantation Thinning

The costs for hardwood establishment and management (the first stage) including, annual administration, forestry road construction, site preparation, seedling, fertiliser and herbicide applications, pruning, and thinning process are listed in Appendix 3. An average productivity of hardwood plantation (Gympie Messmate) was estimated as 174.85m^3/ha in the first 15 years during the mid-thinning following Meadows et al. (2014). The land cost for hardwood plantation in Australia was estimated at $1250/ha following Paul et al. (2013a) and Paul et al. (2013b). For evaluating the labour cost, the numbers of the full-time equivalent job (FTEs) 1000 ha^{-1} was used following Paul et al. (2013a) and Paul et al. (2013b). One FTE equates to1855 h/year was assumed (Australian Bureau of Statistics, 2014). For the various forestry activities, Paul et al. (2013a) and Paul et al. (2013b) estimated that 50.17 FTEs were required for generating 1 ha of hardwood thinned logs, including estate management, hardwood planting, seedlings production, weed control, site preparation, planting, fertiliser application and thinning. The total employment for 1 FU of thinned logs was then calculated as 0.23FTEs which equals to 0.43 hours. The average wage of $27.64/hour for labour in Australian forestry sector was obtained from Rawlinsons Construction Cost Guide (Rawlinsons, 2016). The total labour cost for generating 1 FU of thinning logs was then calculated as $14.19 in present value. The life cycle cost during hardwood plantation thinning was then estimated as $121.52 per VBC pole in 2016 present value. Therefore, the total cost in the 60-year life cycle for raw material extraction from hardwood thinning was estimated at $217.96.

Stage Two - LCC of VBC Pole Manufacturing

The costs for VBC pole manufacturing were collected from published sources. The manufacturing facilities and labour cost were assumed as 30% of total manufacturing cost based on the average cost for the manufacturing sector in Australia (Australian Bureau of Statistics, 2012). The details costs were listed in Appendix 4.

The cost during VBC pole manufacturing stage was estimated at $230.40 in 2016. As this study assumed 60 years surveyable time and while the VBC pole assumed service life is 25 year; another two VBC poles are required in the 25th year and 50th year. However, LCC only considered within 60-year-life cycle and therefore, the LCC of the third pole was calculated on a pro-rata basis (0.4 FU). The life cycle cost was then calculated as $413.25 for pole manufacturing stage in 60 years reflecting the present value in 2016.

Stage Three: Cost for Installation, Replacement and Maintenance

The cost for hollow structure VBC pole installation was assumed less than that of timber poles due to reduced weight which results in less transportation cost (Salman and Li, 2016). The cost of installation of VBC poles was assumed as half of that of timber poles, which was approximate $310/pole. The replacement cost for VBC pole included the cost of removal and transport to the final disposal site. The removal cost was estimated at $82.70/pole by Salman and Li (2016). Within the 60 years-life-cycle, there are two replacements for VBC utility poles. Furthermore, this study assumed $40 will be spent for pole maintenance every 10 years according to Ausgrid (2015). The total cost for installation, replacement and maintenance was estimated at $871 totally.

Stage Four - Cost for Final Disposal

Option 1 - Landfilling

Construction and Demolition Waste Guide (2012) reported that the costs for landfills ranged between $42 and $102 per tonne of waste in Australia, depending on the level of management controls. This cost includes the cost of land, approvals, leachate collection, gas recovery, amenity management, site operations, capping & remediation and post-closure maintenance (Construction and Demolition Waste Guide, 2012). The average cost was assumed as $77 for treating 1 tonne of solid waste in this study. The cost for treating VBC poles in 60 years was estimated at $18.37. The GHG emissions realised through 60 years' life cycle were 74.995 kg/CO_2-eq per pole in landfilling option, hence the carbon cost was estimated at $3.48 in 60 years. Therefore, the total cost of disposing of a pole in landfilling scenario is $21.85.

Option 2 - Incineration for Energy Recovery

Waste incineration for energy recovery is a financially costly waste disposal option (State Government of New South Wales, 2000). The New South Wales Alternative Waste Management Technologies and Practices Inquiry estimated that the net financial cost of such facilities in Australia in 2000 was between $180 and $260 per tonne of waste (State Government of New South Wales, 2000). The Bottom ash is generally disposed of in municipal waste landfills, which costs $77 per tonne to dispose of to landfill.

In this study, the total cost in 60 years for incineration treatment was calculated as $84.51 with further $3.73 spent for disposal of bottom ash in a landfill. The GHG emissions realised was approximately 63.22 kg/$CO_{2\text{-}eq}$ per pole in incineration scenarios, which contributed around $3.14 towards carbon cost over the 60 years period. However, the energy recovery from incineration of each utility pole was estimated at 227.5 kWh, which contributed $57.46 financial return in total. Furthermore, according to Australian Government Department of the Environment (2014), 0.81kg of $CO_{2\text{-}eq}$ is generated for producing 1kWh of electricity in Queensland. The total GHG offset for displacing electricity from fossil fuel was estimated to be $6.07. Therefore, the total cost for this end of life scenario was estimated at $27.85.

Total Life Cycle Costs of VBC Utility Poles in Two Scenarios

The total life-cycle cost of VBC pole summarised all the cost and took off the financial return from 60 year-life-span as presented in following Table 4.7. The total life-cycle costs were estimated at $1,524.06 and $ 1,529.88, respectively in landfilling and incineration scenarios in 60-year-lifecycle.

Table 4.7 Total LCC breakdown of VBC pole in landfilling and incineration scenarios

Cost & Revenue	Landfilling Option	Incineration Option
Hardwood Plantation Thinning	$217.96	$217.96
VBC Pole Manufacture	$413.25	$413.25
Installation, Maintenance and Removal	$871.00	$871.00
Final Disposal	$18.37	$88.24
Carbon cost	$3.48	$3.14
Revenue (energy recovery)	-	$57.64
Revenue (GHG offset)	-	$6.07
Total LCC	$1,524.06	$1,529.88

4.2.4 Results and Discussions

4.2.4.1 LCA Results

To assess the overall environmental impacts of the different utility poles, impact indicator values were added throughout the four stages: material extraction; poles manufacturing; service in the utility system and final disposal. The impact indicator values for VBC, steel and concrete poles are presented in Table 4.8. Due to the different service lifetimes, the impact indicator values of VBC poles are converted to 60 years, in order to compare with the alternatives.

Table 4.8 LCA results of utility poles in four scenarios

Impact category	Units/FU	VBC Pole		Concrete Pole	Steel Pole
		Landfilling	Incineration		
GWP	kg $CO_{2\ eq}$	74.995	63.22	220.644	143.186
AP	kg $SO_{2\ eq}$	0.425	0.29	1.228	1.059
EP	kg $PO_{4\ eq}$	0.530	0.492	0.227	0.159
FDP	kg Oil_{eq}	40.715	30.78	124.713	112.087
HTP	kg 1,4-DB_{eq}	3.736	2.27	12.108	7.712

According to the results (Table 4.8), the concrete pole had the largest environmental burden on GWP, AP, FDP and HTP mainly due to the significant transportation, fossil fuel consumption, as well as use of energy-intensive materials during the poles manufacturing process. A Large quantity of fossil fuel consumption during manufacture of concrete pole and transportation process contributes higher impact on all indicators compared to steel poles, particular on GWP, EP and HTP. The AP (1.059 kg-$SO_{2\text{-}eq}$) and FDP (112.087 kg-$Oil_{\text{-}eq}$) from steel pole were slightly better than that of the concrete pole. The most signification impact of steel pole came from steel manufacturing and energy consumption in the whole life cycle. However, this study assumed the 100% of steel was recycled at the end of the steel pole service life, which contributed significant GHG emissions offset. Among all alternatives, ACQ-treated VBC poles presented the best results on most impact categories compared to concrete and steel utility poles, except EP due to use of PF resin and ACQ preservative. The energy required for casting concrete and steel poles are mostly sourced from fossil fuel attributed to electricity and natural gas consumption. However, the manufacture of ACQ-treated VBC utility poles used both fossil fuel and biomass (wood waste). According to Intergovernmental Panel on Climate Change (IPCC) (2007), CO_2 emissions from burning biomass were not counted towards the GWP.

In the case of VBC pole, energy recovery via incineration presented less environmental impact on GWP100 (63.22 kg-CO_2-$_{eq}$) than landfilling scenario, mainly due to avoided methane release and GHG emissions offset via energy recovery in the waste treatment stage. Compared to the landfilling scenario, energy recovery from waste incineration contributed to the significant reduction on all impact factors, particular on AP (i.e. reduced by 32%), FDP (i.e. reduced by 25%), and HTP (i.e. reduced by 40%). The extra energy generation via waste incineration helped to avoid the environmental burdens from power generation using fossil fuel. Additionally, incineration scenario had relatively less impact on eutrophication potential (0.492kg-PO_4-$_{eq}$) which was attributed to the destruction of ACQ preservative during waste combustion.

4.2.4.2 LCC Results

The LCC results indicated that the total life-cycle costs of VBC poles in both ends of life scenarios (landfilling and incineration) were much lower than concrete and steel poles. Over the 60 years life cycle, the total LCC of VBC poles were estimated at $1,524.06 and $1,529.88 for landfilling and incineration scenarios, respectively. The LCC of steel pole was estimated at $1,968.38, which is approximately 30% higher than VBC poles. Meanwhile, concrete pole presented the highest life-cycle cost of $2,243.32, which is about 45% higher than that of VBC pole and 15% higher compared to a steel pole. One of the reasons for cost savings of using VBC poles was due to its lower manufacturing, installation and removal cost. VBC pole is lighter than concrete pole and required less material and energy during manufacturing, which also contributed to the lower transportation cost. In addition, VBC pole was sourced from low-value hardwood plantation thinned logs that resulted in lower cost of raw material. Furthermore, due to the low GHG emission released throughout its life cycle, VBC pole has a relatively lower carbon cost. When the VBC pole reached its end-of-life, it could contribute to financial payback from electricity generation via waste incineration. However, the carbon sequestered and stored in VBC poles were not considered in the total life-cycle cost due to its short lifespan (25 years). Nevertheless, the GHG emissions quantified in LCA was using the IPCC method in terms of the 100-year global warming potential.

The total life-cycle cost of VBC pole in incineration scenario was slightly higher than landfilling scenario due to higher treatment cost. Despite the high financial return from extra energy generation and GHG offset, the costs of waste incineration operation still slightly overweighed credits earned from energy recovery. However, the life-cycle costs between these two disposals scenarios in VBC poles were insignificant, which can be ignored. Steel pole was seen as a better option compared to the concrete pole. The steel poles lightweight contributed to lower life-cycle cost on material and energy, transportation, installation, removal and disposal. Additionally, the high recycling rate of scrap steel and GHG emission offset associated with material recycling contributed to the net savings. The high life cycle cost of the concrete pole was mainly attributed to installation and removal cost. During the pole manufacturing stage, concrete poles also required more transportation process than other options.

4.3 Uncertainty & Sensitivity Analysis

The input data involved uncertainty due to variability and assumptions made about input and output parameters in the LCC and LCA models. This section aims to identify the most uncertain factors that significantly affect both environmental and economic performance throughout the whole life cycle of VBC, concrete and steel poles. Sensitivity analysis was conducted to indicate the changing of impact indicator values affecting by assumptions and uncertainties on the final results. To better understand the response of the system to the effect of combined uncertainties, Monte Carlo simulations were conducted.

4.3.1 Uncertainty and Sensitivity Analysis in LCA

Throughout the modelling exercise, average values were used and assumptions were made about service life; transportation distance, energy resource consumption, as well as the carbon sequestration rate of wood in a landfill, thus introducing a level of uncertainty in the outcome of the LCA exercise.

4.3.1.1 Service Life

The service life of VBC, concrete and steel pole in utility system depend on a number of factors. Poles may be removed from service before their useful service life; for example, road widening. In this study, an average service life of 60 years was assumed for both concrete and steel poles, while the lifespan of VBC pole was assumed to be 25 years based on professional judgement and preliminary experimental result from Queensland Government Salisbury Research Facility Centre (Lu and El Hanandeh, 2016b). Spencer and Elder (2009) reported that the service life of concrete and steel poles vary between 50 and 70 years, a variation of ±17% from the assumed average service life. On the other hand, the VBC pole is a new product and its durability under actual service conditions has not yet been confirmed. Therefore, we considered higher variability in our analysis (±20%) for all materials.

Sensitivity analysis indicated that changes in service life affect all impact indicators. Increasing the service life of VBC pole by 20% will reduce all impact indicators by 17%, while reducing the lifespan by 20% will cause 25% more impact on all factors in both landfilling and incineration scenarios. Changes in the lifespan of concrete and steel poles led all impact indicators to change proportionally, for example a +20% change in lifespan caused impact indicators to increase by 20%. The results highlight that the environmental impacts are more sensitive to shorter service life in the case of the VBC pole scenarios. The higher sensitivity to the shorter service life is attributed to the need to install an extra pole during the service period considered (60 years) and the associated processes such as manufacturing, transportation and disposal. Nevertheless, the results suggest that even though the VBC pole was sensitive to service life, the overall environmental performance did not change.

4.3.1.2 Transportation Distance

Transportation distance during whole life cycle were estimated and obtained by an Internet highway directions interface (Google maps directions, 2016). The transportation distance for one FU of pole during the whole life was 270 km. However, the estimated route kilometres may not yield a real representation of actual transportation distance. The uncertainty of the assumptions is large because of variations in regional location of the plantation sites, manufacturing facilities and disposal sites. A coefficient of variation (CoV) of 20% is usually assumed for the estimation of transportation distance (Sonnemann et al., 2003). In the sensitivity analysis, two cases were modelled for comparison: **1)** shorter distance of 220 km and **2)** longer distance of 320 km. As expected, changes in transportation distances affected all impact categories as shown in Figure 4 (a, b & c). VBC poles, both scenarios, were the most sensitive to the change in transportation distances while steel pole was the least sensitive. However, in absolute values, concrete continued to exhibit the highest impacts. This is due to the heavy weight of concrete pole leading to higher transportation emissions due to consumption of diesel fuel. Transportation contributed large proportion of the impacts in the all life stages of VBC pole due to fossil fuel consumption. In the case of VBC, incineration scenario was more sensitive to change in transportation distance than the landfilling scenario, particular AP impact category. This can be attributed to the relative contribution of transportation emissions to the overall impacts of each scenario. For example, the GHG emissions from transportation are similar in the case of incineration and landfilling; however, the proportion of the overall GHG emissions is larger in the case of incineration. As discussed in the results earlier, VBC incineration has smaller GHG emissions than VBC landfilling scenario because of the avoided energy and landfill gases emissions. In turn, the effect of GHG emissions from transportation on the GWP is more prominent. Furthermore, our analysis also revealed that when longer transportation distances were considered, the gap in environmental impacts of steel pole and VBC shrinks. As a result, while it is clear that concrete is the least favourable option from LCA impacts; the performance of the VBC pole will largely depend on the transportation intensity.

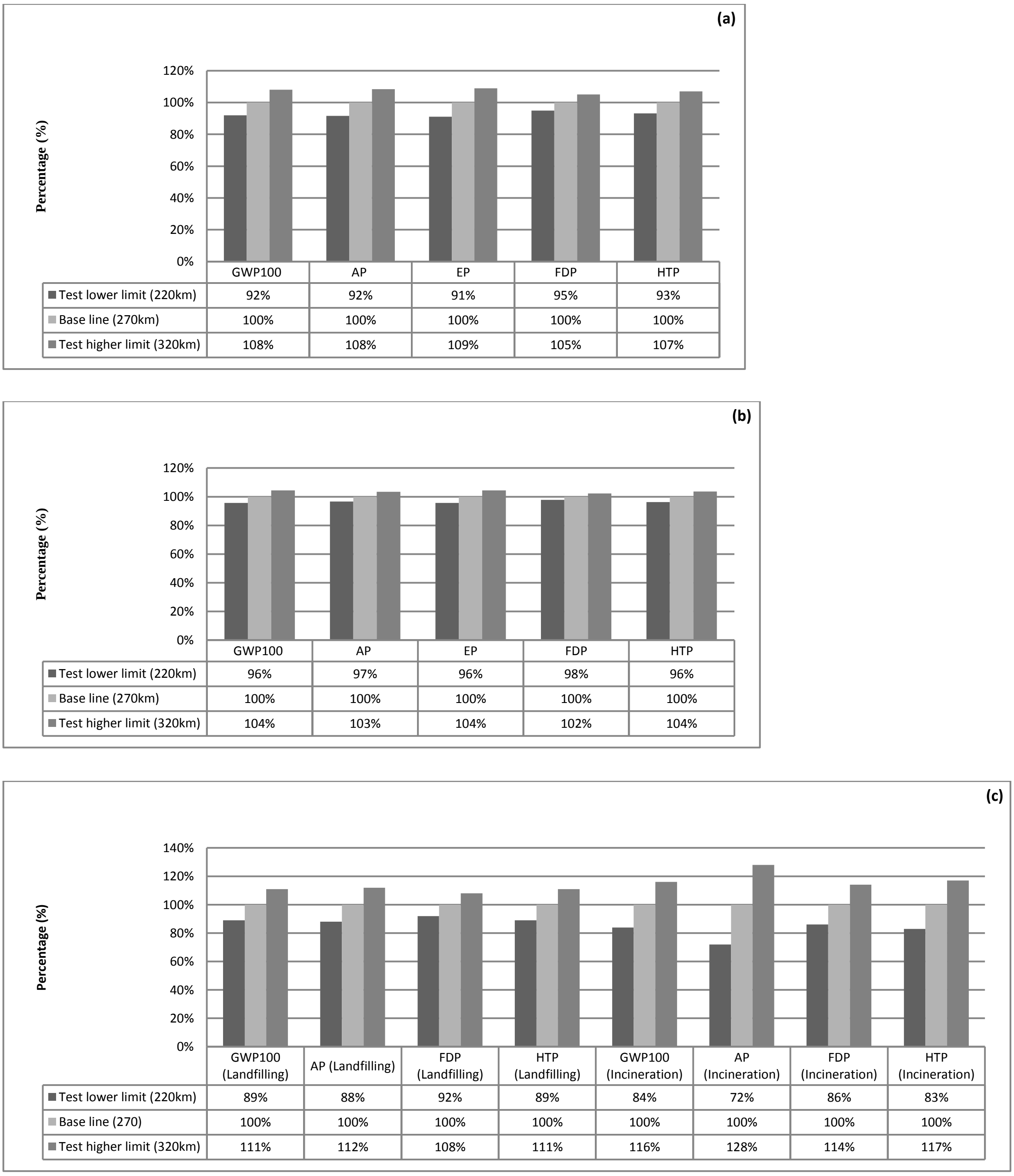

Figure 4.1 Sensitivity analysis of results to transportation distances: (a) concrete pole, (b) steel pole and (c) VBC pole landfilling and incineration scenarios

4.3.1.3 Energy Consumption

Significant differences may occur in energy consumption mainly due to the changes in technology and changes in the structure of the economy (Voigt et al., 2014). Therefore, further analysis was conducted to determine the sensitivity of the LCA results to energy consumption. Voigt et al. (2014) reported that the energy consumption in manufacturing and service sector fell approximately 20% during the last decade due to the greater improvement in technology and economy structure. In this sensitivity analysis, two cases were modelled for comparison, including (1) energy consumption saving of 20%, (2) increased energy consumption by 20%. Changes in energy consumption had greater effect on concrete and steel pole results. This is due to the higher energy consumption during concrete and steel manufacturing leading to significant effects on all impact indicators. A change of ±20% in energy consumption had between -13 and 12% changes in all impact categories for the concrete pole. The EP, FDP and HTP were the most sensitive in the case of steel pole. On the other hand, FDP was the most sensitive in the case of steel pole (-16% to 15%) followed by HTP and EP. Both scenarios of VBC showed little sensitivity to changes in energy consumption (<6%) for all impact categories. This can be attributed to the high proportion of biomass energy usage in the manufacturing process of the VBC pole.

4.3.1.4 VBC Pole Decay in Landfill

This LCA assumed 77% of the carbon in the wood was sequestered in landfill according to Barlaz (1998). However, Bolin and Smith (2011) indicated that preservative and other inorganic compounds in the disposed wood are expected to retard carbon release when compared to untreated wood. Field investigation of wood decay in Australian landfills revealed that wood deposited in landfills have remained virtually intact (Ximenes et al., 2015). Therefore, the sensitivity of the LCA results to carbon sequestration rate was tested using 50% and 90% sequestration rates. The results confirmed that changing sequestration rate to 50% from the baseline (77%) can increase the GWP impact indicator by approximately 30%, while increased sequestration rate (90%) will reduce GWP by 14%. The effect of the change in sequestration rate is proportional to the change in the GWP impact category. This is mainly due to landfill gas emissions, particularly fugitive methane, which is released during the anaerobic decomposition of wood in the landfill.

4.3.2 Effect of Combined Uncertainties on Overall Performance of Alternatives

Monte Carlo analysis was conducted to test the effect of the overall uncertainties in the systems on the LCA results. The effect of uncertainties by running 1000 Monte Carlo Simulations allowing multiple uncertain parameters to vary simultaneously, and drawing random samples from pre-defined distributions as described in Table 4.9. The Monte Carlo analysis was run in the OpenLCA 1.4.1 software. These parameters were derived based on uncertainty and sensitive analysis described in section 3.1. In addition, the percentage of renewable energy for Queensland electricity production was also considered. The data of electricity generation used in this study was based on ‘electricity, high voltage, Queensland’ from AusLCI (2011), which indicated that 12% of the total power generation came from renewable sources. Australian Energy Statistics

(2016) reported that the share of renewable energy might rise to 15% of total electricity generation in Australia. Therefore, the renewable energy content in electricity generation was also included in this analysis. For the VBC poles, factors such as ACQ preservative retention and PF resin consumption were also reported to affect the LCA results (Lu and El Hanandeh, 2016b). ACQ retention in VBC products in this assessment was 4.4 kg/m^3. However, ACQ was defined with a retention rate of 2.4 - 6.4 kg active ingredient per cubic meter of lumber, as outlined by the American Wood Protection Association (AWPA) standard UC4A (AWPA, 2010a, AWPA, 2010b). Lu and El Hanandeh (2016b) also indicated that the PF resin consumption per cubic meter usually varies between 10 to 60 kg during the pole manufacturing process. Therefore, these two factors were also included in the Monte Carlo Analysis. Table 4.9 shows a summary of the uncertain parameters included in the Monte Carlo simulation.

Table 4.9 Parameters used in the Monte Carlo analysis

Parameter	Mode	Unit	Distribution type
Lifespan	VBC pole: 25	years	Normal: μ=25, σ=5
	Concrete & steel poles: 60	years	Normal: μ=60, σ=12
Transportation distance	270	km	Normal: μ=270, σ=50
Energy consumption	100	%	Normal: μ=100, σ=20
Renewable energy content in electricity generation	12	%	Normal: μ=12, σ=3
ACQ retention in VBC pole	4.4	kg/m^3	Triangle: min=2.4; max=6.4
PF consumption in VBC pole	53.6	kg/m^3	Triangle: min=10; max=60
Carbon sequestration rate (VBC pole landfilling scenario only)	77	%	Triangle: min=50; max=90

Table 4.10 presents a summary of the Monte Carlo simulation results including the mean, standard deviation (SD), 95th percentile and coefficient of variation (CoV). The results suggest that the compared to steel and concrete, VBC (both scenarios) are more vulnerable to the effects of overall uncertainties in the system in all impact indicators as shown by the higher CoV values. The Monte Carlo analysis showed that the VBC incineration scenario is more sensitive to the effects of the combined uncertainties than the landfilling scenario, particular the AP and HTP impact categories. Despite the high level of uncertainty in AP and HTP of the VBC pole incineration scenario, the results suggested that there was 99.9% likelihood that the AP and HTP will be less than that of

the VBC landfilling scenario. In concrete and steel poles cases, the impact of the combined uncertainties was more pronounced on GWP and HTP impact indicators, while the other impact categories were less sensitive to the changes. Table 4.10 also suggests that the overall ranking of the four scenarios assessed was not affected by the combined uncertainty; with concrete being the least favourable and VBC incineration scenario being the most environmentally favourable option.

Table 4.10 Summary of Monte Carlo simulation results in LCA

Options	Impact	Units/FU	Mean ± E*	SD*	95^{th} percentile	CoV*
VBC Pole in Landfilling Scenario	GWP	kg $CO_{2\ eq}$	75.66 ± 1.54	24.76	119.00	33%
	AP	kg $SO_{2\ eq}$	0.43 ± 0.01	0.14	0.68	33%
	EP	kg $PO_{4\ eq}$	0.55 ± 0.01	0.15	0.80	28%
	FDP	kg $Oil_{\ eq}$	39.86 ± 0.72	11.62	59.64	29%
	HTP	kg 1,4-DB $_{eq}$	3.81 ± 0.07	1.2	5.85	32%
VBC Pole in Incineration Scenario	GWP	kg $CO_{2\ eq}$	63.43 ± 1.53	24.67	105.70	39%
	AP	kg $SO_{2\ eq}$	0.29 ± 0.01	0.13	0.50	45%
	EP	kg $PO_{4\ eq}$	0.52 ± 0.01	0.15	0.77	29%
	FDP	kg $Oil_{\ eq}$	29.51 ± 0.62	10.07	46.93	34%
	HTP	kg 1,4-DB $_{eq}$	2.27 ± 0.06	0.98	3.89	43%
Concrete Pole	GWP	kg CO2 $_{eq}$	228.70 ± 1.99	32.1	290.90	14%
	AP	kg SO2 $_{eq}$	1.21 ± 0.0074	0.12	1.38	10%
	EP	kg PO4 $_{eq}$	0.22 ± 0.0012	0.020	0.25	9%
	FDP	kg $Oil_{\ eq}$	123.60 ± 0.56	10.80	135.70	9%
	HTP	kg 1,4-DB $_{eq}$	12.03 ± 0.13	2.10	15.97	17%
Steel Pole	GWP	kg $CO_{2\ eq}$	153.2 ± 1.56	25.19	168.30	16%
	AP	kg $SO_{2\ eq}$	1.16 ± 0.08	0.13	1.26	11%
	EP	kg $PO_{4\ eq}$	0.17 ± 0.001	0.018	0.18	10%
	FDP	kg $Oil_{\ eq}$	118.50 ± 0.59	9.48	128.20	8%
	HTP	kg 1,4-DB $_{eq}$	8.87 ± 0.084	1.35	10.85	15%

*E: confidence interval (95%), SD: standard deviation, CoV: coefficient of variation

4.3.3 Uncertainty and Sensitivity Analysis in LCC

The uncertainty and sensitivity analysis were also performed to evaluate the effect of various uncertainties and assumptions in this economic modelling on the LCC results. Parameters affecting the life-cycle cost of the utility poles included the service lifetime (t), the inflation rate (i), discount rate (r), labour cost, as well as the financial return from carbon offset.

4.3.3.1 Service Life

To examine the influence of the service life on the life-cycle cost, a sensitivity analysis was performed using the same parameters used in the LCA analysis (±20%). Results indicated that an increase in the service life of VBC utility pole from 25 years to 30 years resulted in a decrease in the total life-cycle cost by more than 24% for both VBC scenarios. On the other hand, the reduction of the service life to 20 years led to an increase of 13%. Meanwhile, the changes in service life had relatively less effect on life-cycle costs of concrete and steel poles. Increasing the service life from 60 to 72 years caused the life cycle cost of concrete and steel poles to decrease by approximately 19%. In contrast, the reduction of service life from 60 years to 48 years caused around 15% increase in the life cycle cost of the concrete and steel poles. The LCC was more sensitive to increased service life than service life reduction. This is counter-intuitive to what is expected from the financial PV formula which includes the time (service life in our case) as an exponent in the denominator. However, the increased service life of VBC pole from 25 to 30 years would require only 2 poles to be used to cover the 60 years simulation period rather than the 2.4 poles (in the base case scenario) and 3 poles in the case of the reduced service life of 20 years. Similarly, a reduction in the service life of steel and concrete poles would necessitate an extra installation during the simulation period. As a result, the overall savings due to the avoided LCC of the third pole can explain the higher effect of increased service life.

4.3.3.2 Inflation Rate

The inflation rate in Australia fluctuated between 1 and 5% during the last decade (Reserve Bank of Australia, 2015). Therefore, to examine the sensitivity of the results to changes in the inflation rate, sensitivity analyses have been carried out using the extreme limits (1 and 5%). As expected, an increase in the inflation rate would lead to the increment of the life-cycle cost. The total life-cycle costs of VBC pole increased by more than 55% in both landfilling and incineration scenarios. Meanwhile, the total life-cycle cost of concrete and steel pole increased by approximately 40% over the 60-year life cycle. On the other hand, the life cycle costs of VBC poles would be reduced by approximately 30% for both scenarios when the inflation rate changed to 1%. At the same time, the life cycle costs of concrete and steel poles have decreased by approximately 20%.

Results indicated that the changes in inflation rate have more impact on VBC pole scenarios than concrete and steel poles. That is because, in VBC pole scenarios, two extra poles are required to be installed in 25th and 50th year during the 60 year period. Despite the higher sensitivity of VBC poles scenarios to the inflation rate, it continued to deliver lower life cycle cost than the concrete and steel options.

4.3.3.3 Labour Cost in Manufacturing Sector

Another parameter that has an influence on life-cycle costs is the labour cost. Australian Bureau of Statistics (2012) reported that the labour cost in Australian manufacturing industry accounted from 23 to 41% of the total manufacturing cost during last ten years. In order to examine the effect of labour cost on the life-cycle cost two cases were assessed: 20 and 40% of total manufacturing cost. As expected, a decrease in the labour cost would lead to a reduction in the LCC for all options. The total life-cycle cost of VBC pole would increase by 4% in both VBC pole scenarios if the labour cost increases from 30 to 40%. Meanwhile, the total life-cycle costs of concrete and steel pole increased by 6%. On the other hand, labour cost reduces from 30 to 20%, would lead to 3% reduction in the total life-cycle costs of VBC poles in both incineration and landfilling scenarios. Meanwhile, the LCC of concrete and steel poles would reduce by 4% and 5%, respectively. The changes caused by uncertainties in labour cost were not obvious in all options. However, the life-cycle cost of concrete and steel poles were more sensitive to changes in labour cost than that of VBC pole options due to the higher manufacturing cost.

4.3.3.4 Discount Rate

During the last ten years, the Australian discount rate fluctuated between 3 to 7% (Reserve Bank of Australia, 2015). To examine the influence of the discount rate on the total life-cycle cost, sensitivity analyses were carried out with the discount rates 3 and 7%. A decrease in the discount rate from the baseline (4.9%) to 3% would lead to an increase in the life-cycle cost by 48% for the VBC options and 40% for the concrete and steel poles. On the other hand, the life cycle costs of VBC poles would be reduced 25 % for the VBC options if the discount rate changed to 7%. At the same time, the life cycle costs of concrete and steel poles would be reduced by 12% and 10%, respectively. Results indicated that this uncertainty have higher effect on VBC pole scenarios than concrete and steel poles because of the capital injections required at the 25th and 50th year for the installation of replacement poles.

4.3.3.5 Carbon Price

The carbon prices in 2016 fluctuated between 23 and \$31/tonne $CO_{2\ eq}$ reported by The Treasury Australian Government (2016). In order to examine the influence of the carbon price on the total life-cycle cost, sensitivity analyses were carried out with the carbon price at \$23/tonne $CO_{2\ eq}$ and \$31/tonne $CO_{2\ eq}$. Results indicated that the uncertainties in carbon price only contributed minor effects on the total life-cycle cost of all scenarios. The life cycle cost of VBC pole in both two scenarios only changed less than \$1. Compared to VBC pole options, concrete pole and steel poles options had relative larger effects on its life cycle cost due to the largest quantity of GHG emissions released. However, the uncertainties were only about \$2 to \$3 different.

4.3.3.6 VBC pole decay in Landfill

As previously mentioned, the carbon sequestration rate in landfill was tested using 50% and 90% sequestration rates in LCA. The results confirmed that changing sequestration rate to 50% from the baseline (77%) could increase the life cycle cost by \$1.04, while increased sequestration rate (90%) will reduce life cycle cost by \$0.49. The changes attributed to the carbon sequestration rate were negligible.

4.3.4 Monte Carlo Simulation in LCC

The overall uncertainties and variations in the LCC were modelled and analysed simultaneously by a Monte Carlo Simulation procedure. One thousand (1000) simulations were run in Excel spreadsheet model. Parameters considered in the Monte Carlo Simulations are presented in Table 4.11.

Table 4.11Parameters used in the Monte Carlo analysis

Parameter	**Mode**	**Unit**	**Distribution type**
Lifespan	VBC pole: 25	years	Normal: μ=25, σ=5
	Concrete & steel poles: 60	years	Normal: μ=60, σ=12
Inflation rate	3	%	Normal: μ=3, σ=2
Discount rate	4.9	%	Normal: μ=4.9, σ=2
Labour cost	30	%	Normal: μ=30, σ=10
Energy consumption	100	%	Normal: μ=100, σ=20

Carbon Price	29		\$/tonne $CO_{2\ eq}$	Triangle: Min =23; Max=31
GWP	VBC pole landfilling option	75.66	kg $CO_{2\ eq}$	Normal: μ=75.66, σ=24.76
	VBC pole Incineration option	63.42	kg $CO_{2\ eq}$	Normal: μ=63.43, σ=24.66
	Concrete pole option	218.40	kg $CO_{2\ eq}$	Normal: μ=218.40, σ=25.69
	Steel pole option	148.70	kg $CO_{2\ eq}$	Normal: μ=148.70, σ=12.70

The simulated results including, 95% confidence interval of the mean, standard deviation (SD), 95^{th} percentile, as well as the coefficient of variation (CoV) are presented Table 4.12. The results suggest that the overall uncertainties have a similar effect on all scenarios. Figure 4.2 presents the cumulative frequency of life cycle cost of the four options. VBC poles in both landfilling and incineration scenarios are still the best options from LCC perceptive. Although the CoV from both VBC scenarios were large, there is still 95% likelihood that the life cycle cost of VBC pole scenarios will be less than \$3201.08. On the other hand, the life cycle cost of the concrete pole is still the most expensive option among all, followed by a steel pole. Results indicated that there was a high probability that the VBC pole incineration option has lower life-cycle cost than that in landfilling scenario. Additionally, the life cycle cost of concrete pole only has a remote possibility to be less than that of a steel pole. Therefore, the Monte Carlo analysis suggests that the overall ranking of the three materials assessed was not significantly affected by the combined uncertainty. However, the VBC pole incineration scenario may have slightly less lifecycle cost than that in landfilling scenario.

Table 4.12 Monte Carlo simulation results in LCC

Options	LCC/(2016)	95% confidence interval (Mean±E)*	SD*	95^{th} percentile*	(CoV)*
VBC Pole Landfilling scenario	AUD ($)	1725.91 ± 48.93	789.43	3181.56	46%
VBC Pole Incineration scenario	AUD ($)	1734.58 ± 49.50	798.54	3201.08	46%
Concrete pole	AUD ($)	2663.85 ± 87.60	1413.36	4677.37	53%
Steel Pole	AUD ($)	2298.33 ± 69.40	1119.60	4002.37	49%

*E: standard Error, SD: standard deviation, CoV: coefficient of variation

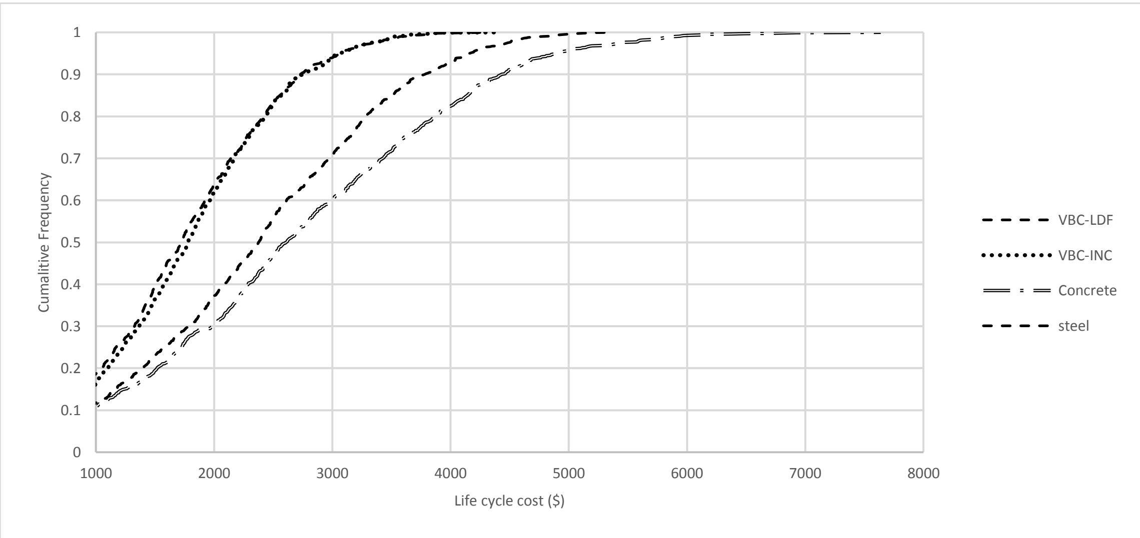

Figure 4.2 Monte Carlo simulation results in LCC

By conducting the uncertainty and sensitivity analysis, service life was found to be the most significant factor affecting both environmental and economic results. Transportation distance had more effect on VBC poles scenarios followed by concrete and steel poles due to its large contribution to all impacts in all life stages. Changes to fossil fuel consumption had more effect on concrete and steel poles options due to the higher energy requirement in the manufacturing stage. The VBC poles scenarios showed little sensitivity to the changes in energy consumption due to the high proportion of biomass energy substitution. Furthermore, the carbon sequestration rate in the landfill had a significant impact on the environmental performance of the VBC pole landfilling scenario mainly due to landfill gas emissions released during the anaerobic decomposition of wood in the ground. On the other hand, sequestration rate had little impact on the economic performance. Monte Carlo Analysis in LCA indicated that the overall ranking of the four scenarios assessed was not affected by the combined uncertainties; with concrete remaining the least favourable and VBC incineration scenario being the most environmentally favourable option. Inflation and discount rates, although they had no impact on the environmental performance of any of the assessed options, they presented significant impact on the overall life-cycle costs of all assessed scenarios. The effects of the uncertainties in carbon price and carbon sequestration rate in the landfill were not obvious on the total life-cycle cost. Monte Carlo Analysis showed that the overall ranking of the four options assessed was not affected by the effect of the combined uncertainties. The VBC options were almost indistinguishable in their LCC performance. Nevertheless, from the environmental perspective, the VBC with incineration option had a clear advantage over the landfilling.

4.3.5 Limitations

The scope of this life cycle study was limited to boundaries established in the goal and scope documented in this study. Limitations included reliance on the published and publically available information in many instances. Such information was assumed to be accurate. The life cycle inventory completed for different utility poles manufacturing was designed to represent the typical or average product on the market. Nevertheless, utility poles manufacturing impacts might vary according to the technology and scale used. Inventory data for different waste disposal were sourced from the Australian LCI database. This study only included two different end-of-life scenarios of VBC pole. In addition, the used concrete was only assessed as solid waste disposed of in a landfill. This comparative life cycle study solely focused on ACQ-treated VBC, concrete and steel pole.

4.4 Conclusions

This study compared four alternative options for utility pole using life cycle approach. A newly developed material, VBC pole made from hardwood forestry thinning, was compared to the more mature options of steel and concrete. The result of this study indicated that the VBC utility pole had outstanding performance on both environmental and economic perspectives. Low energy intensity and utilization of low-value material in the manufacturing of the VBC were the most contributing factors to the savings in both the life cycle cost and environmental impacts. Additionally, using biomass fuel to substitute fossil fuel as an energy source during VBC pole manufacturing led to significant reduction in environmental impact. The concrete pole was the least favourite option due to high environmental impact and life-cycle cost. The largest impacts of the concrete pole were mainly attributed to using energy-intensive materials and its heavy weight, which required more energy during manufacturing and transportation. Steel pole performed slightly better than concrete pole mainly due to its lighter weight, which led to a reduction in the energy and material consumption, as well as transportation. The financial returns from steel recycling and GHG emission offset also contributed to net saving. Two ends of life scenarios were considered for the VBC pole: incineration and landfilling; both were equally efficient from a life-cycle cost perspective. However, incineration was a better performer from life cycle assessment perspective as it had lower environmental impacts. Inflation and discount rates, labour cost and the service life were the main factors affecting the economic performance of all options. Energy recovery contributed to both environmental impact and economic saving during waste pole incineration. Therefore, VBC utility pole with incineration and energy recovery scenario may present a viable option to be considered by utility network operators.

4.5 Recommendations

Utility network operators should consider the use of innovative materials to improve the sustainability of the power distribution network.

The manufacture of the three types of utility poles should strive to reduce energy and raw material inputs through conservation and innovation. For instance, the raw materials should be sourced from locations close to point of manufacturing facilities, which could help to reduce the emission generation and cost due to long distance transportation. Additionally, using bio-energy products to replace fossil fuel or electricity as the main energy source could significantly reduce environmental impacts.

Treated-wood pole service life varies greatly and often is a function of proper inspection and maintenance. Improved inspection and maintenance programs should be used to maximize the durability, thereby reducing environmental impacts and life cycle cost. Furthermore, utility companies should look at ways to minimize end of life impacts through energy recovery or other innovative recovery option.

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Appendix

Appendix 1. Life cycle cost of manufacturing 1 Functional Unit of concrete utility pole

Items	Price (In referenced year)	Cost/FU converted to 2016 Present Value (PV)
Electricity	22.24 ¢/kWh	$156.47
Natural gas	$0.387/m3	$16.80
Diesel fuel	$1.37/L	$7.23
Gasoline	$1.43/L	$3.36
Transportation	$0.098/tkm	$56.66
Water	$0.00363/L	$0.973
Cement	$0.425/kg	$157.60
Steel	$0.9/kg	$77.01
Aggregate	$63/tonne	$67.76
Labour cost	$233.08/pole	$233.08
Total manufacturing cost		$776.95

Appendix 2. Life cycle cost of manufacturing 1 Functional unit of steel utility pole

Items	Price (In referenced year)	Cost/FU converted to 2016 PV
Electricity, at grid	$0.2224/kWh	$179.58
Natural gas, combusted in industrial boiler	$0.387/m3	$19.94
Diesel fuel, combusted in industrial boiler	$1.37/L	$3.01
Gasoline, combusted in industrial equipment	$1.43/L	$7.44
Transportation	$0.098/tkm	$15.91
Zinc	$2.69/kg	$8.02
Steel	$0.9/kg	272.19
Water	$0.00363/L	$0.58
Cement	$0.425/kg	$9.18
Labour cost	$221.08/pole	$221.08
Total manufacturing cost		736.93

Appendix 3. Life cycle cost of hardwood plantation thinned logs per FU of pole

Items	**Price (In referenced year)**	**Cost/FU converted to 2016 Present Value (PV)**
Estate Management	$15/ha/annual	$1.04
Forest planning and approval	$75/ha/annual	$5.19
Seedlings	$535/ha/annual	$37.01
Planting	$25/ha/annual	$1.73
Site preparation	$187.5/ha/annual	$12.97
Fertiliser application	$140/ha/annual	$9.69
Planting	$185/ha/annual	$12.80
Thinning Process	$306/ha/annual	$21.14
Land cost	$1250/ha	$5.76
Labour cost	$27.64/hour	$14.19
Total		$121.52

Appendix 4. Life cycle cost of VBC pole manufacturing per FU of pole

Items	**Price (In referenced year)**	**Cost/FU (2016 PV)**
Labour cost	$69.12/pole	$69.12
Electricity cost	22.24 ¢/kWh	$12.55
Materials cost		
Cost of ACQ	$15.75/ kg	$29.24
Phenol Formaldehyde Adhesive	$3.86/kg	$85.98
Cement	$0.4/kg	$17.37
Acrylic Putty	$0.75/kg	$0.17
Transportation	0.098$/tkm	$11.58
Natural Gas	$10/GJ or ($0.387/m^3)	$0.75
LPG	$0.65/L	$3.50
Diesel Fuel	$1.37/L	$0.14
Total		**$230.40**

CHAPTER 5 – RESEARCH ARTICLE 3

STATEMENT OF CONTRIBUTION TO CO-AUTHORED PUBLISHED PAPER

Chapter 5 includes a pre-print of a co-authored paper titled "Assessment of bioenergy production from mid-rotation thinning of hardwood plantation: life cycle assessment and cost analysis" which was published in the Clean Technologies and Environmental Policy Journal. My estimated contribution to the paper is 65%. I contributed to research design, data collection, modelling, analysis, and writing the manuscript.

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Assessment of Bio-Energy Production from Mid-Rotation Thinning of Hardwood Plantation: Life Cycle Assessment and Cost Analysis

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Abstract

Forestry thinning logs, a low-value by-product of the forestry industry, presents an opportunity for bioenergy production. It can be converted into solid, liquid, and gaseous fuels via different conversion techniques. Comparative life cycle assessment (LCA) and life cycle costing (LCC) analysis were conducted to evaluate six options: woodchip gasification for power generation; wood pellets gasification in combined heat and power plant; wood pellet combustion for domestic water and space heating; pyrolysis for power generation; pyrolysis with bio-oil upgrading to transportation fuels and ethanol production for transportation fuel mix. The functional unit used in this study was the treatment of 1 Mg of biomass. Global warming; acidification; eutrophication; fossil depletion, human toxicity and land use impact categories were considered. The LCC also included greenhouse gas (GHG) emissions costs. The effects of uncertainties in the system on the overall performance of the scenarios were also evaluated. The results showed that all options except for ethanol production are GHG emission negative. Woodchips gasification performed best in all environmental impact categories and had the lowest LCC ($177.6/Mg). Biomass drying consumed more than 50% of the energy required for all options except for the production of liquid transportation fuels via upgrading of pyrolytic oil, in which case the fuel upgrading process was the most energy intensive. In terms of energy return, all options, except electricity production through pyrolysis, offered a positive return. The results highlight the importance of using biomass with least possible processing in order to maximise environmental and energy return and minimise LCC.

Key Words:

Life Cycle Assessment; Life Cycle Costing; Hardwood thinning; Bioenergy; Biomass

5.1 Introduction

Timber plantations contribute to the Australian long-term wood supply and deliver significant economic and environmental benefits. During the last decade, the Australian plantations sector grew by approximately 150% to reach 2 million hectares of which 1 million hectares are hardwood; mainly due to changes in government policies (Meadows et al., 2014). The hardwood plantation estate in South-east Queensland alone was estimated to cover 53,500 ha in 2012 (Meadows et al., 2014). Currently, hardwood plantations focus on the production of high-quality logs; in order to achieve this goal, thinning operations are carried out to remove trees with obvious defects during the plantation life. Approximately 50% of the trees are removed in the third year in what is known as the first thinning; another 30% is removed during the second thinning when the trees are around 10 to 15 years old. The rest of trees are left to mature and usually harvested at 30 to 35 years old. The annual productivity from hardwood plantation thinning was estimated to be approximately 11.5 m^3/ha (Underhill et al., 2014); corresponding to approximately 615,250 m^3 annually. The removed trees usually have low commercial value and as a result contribute very little towards the economic sustainability of the plantation operation (Underhill et al., 2014). To ensure the sustainable expansion of the Australian hardwood plantation industry, value-add products need to be developed to reduce wastage and satisfy the projected growth in the biomass-to-energy market (Australian Energy Statistics, 2016).

Heat and/or power generation from short rotation forestry to replace fossil fuel can realise significant environmental benefits. Wood can be continually replenished, thus leading to a sustainable and dependable supply of energy while helping to reduce greenhouse gases (GHG) emissions (Wahlund et al., 2004, Rosen et al., 2005, Solomon and Luzadis, 2008). Using low-value forestry thinning as a source of energy can reduce resource wastage and increase the financial return through timber plantation management (Munsell and Fox, 2010). Currently, there are many bio-energy technologies available for biomass conversion. For example, woodchips can be used either directly by combustion and gasification, or by upgrading to other forms, such as wood pellets, bio-oil, or liquid biofuels to substitute fossil fuel (Searcy et al., 2007). Wood pellets have been portrayed as a suitable method to substitute fossil fuel for energy generation because of their high energy density, low moisture content, and convenient transportation and storage (Goh et al., 2013, Tarasov et al., 2013). Bio-oil produced from pyrolysis is a promising alternative to fossil crude oil (Xiu and Shahbazi, 2012). Fast pyrolysis is commercially available and is promoted as an effective technology for converting biomass into liquid fuels (Zhong et al., 2010). The crude bio-oil can be directly used through co-combustion in conventional fossil fuel power plants or in existing industrial boilers for electricity and heat production (Fan et al., 2011). However, Chang et al. (2013) noted that the chemical composition of the pyrolytic bio-oil is different from that of petroleum-derived oils due to the high moisture content and oxygenated organic compounds content. Nevertheless, hydro-treatment can be used to upgrade low quality crude bio-oil to petroleum and petroleum-like products which can be used as a substitute for gasoline or diesel in boilers, furnaces, engines, and turbines (Xiu and Shahbazi, 2012). However, the commercial viability of pyrolysis oils from hydro-processing has not yet been achieved and there is limited demand for pyrolysis oils in the market (Iribarren et al., 2012). Bio-ethanol produced through fermentation is another option and bio-fuel is widely used in the transportation sector (Demirbas, 2011). Given the many options available for biomass utilisation and their potential economic and environmental impacts, choosing the optimal energy conversion method requires a balance between the economic and environmental benefits. Therefore, it is important to evaluate the environmental and economic efficiency of each option using objective approaches such as life cycle assessment (LCA) and life cycle costing (LCC) analysis.

A survey of the literature of biofuel life cycle assessment studies is summarised in Appendix 1. Most of the LCA studies conducted on biofuel systems are attributional LCA (aLCA). The system boundaries commonly included raw material extraction to the final use of bioenergy products. The selection of a functional unit (FU) that enables equitable comparison between alternatives is essential. Currently, there is no common functional unit used across all studies. However, Cherubini and Strømman (2011) indicated that the functional units used can be classified into four categories: (1) input unit related, (2) output unit related, (3) unit of agricultural land area, and (4) yearly basis related functional unit. However, some studies reported final outcomes using two or more functional units (Stichnothe and Azapagic, 2009).

The land area related FU is an important parameter since the production area of different biomass feedstocks may be the biggest bottleneck for the bioenergy production (Peters et al., 2015, Kim and Dale, 2005). However, this type of functional unit is more relevant to the case where more than one biomass is used. A few LCA results were reported on a yearly basis (e.g. Stichnothe and Azapagic, 2009, Cherubini and Ulgiati, 2010). This type of functional unit is used in studies characterised by multiple final products since it allows avoiding an allocation step (Cherubini and Strømman, 2011). The input and output related functional units are chosen by the majority of the studies (e.g. Dornburg and Faaij, 2001, Wahlund et al., 2004, Hill et al., 2006, Lardon et al., 2009, Stichnothe and Azapagic, 2009, Collet et al., 2011, Guest et al., 2011, Pa et al., 2011, Iribarren et al., 2012, Kalinci et al., 2012, Esteban et al., 2014). Among the output related LCAs, the energy output per (MJ) was commonly used as the functional unit to characterise the final bioenergy products (e.g. Iribarren et al., 2012, Dornburg and Faaij, 2001, Kalinci et al., 2012). Meanwhile, regarding the studies on transportation biofuels only, the use of different units is evenly distributed. However, Cherubini et al. (2009) indicated that the utilization of functional unit such as per 1 MJ of fuel is not appropriate for biofuel production, although it is adopted by several case studies. The mechanical efficiency can vary among different types of fuels; hence 1 MJ may result in different driving distances. Singh et al. (2010) and Campbell et al. (2011) suggested that the LCA results for transportation biofuels should be expressed on a per vehicle-km basis because this ensures that all the life cycle stages (distribution and biofuel combustion) are included. Therefore, the biofuel mechanical efficiency is considered, while the results are comparable with conventional fossil systems (Singh et al., 2010, Campbell et al., 2011).

In addition, the output related functional unit is usually used in studies characterised by similar final products. For instance, Iribarren et al. (2012) set up the FU related to the production of gasoline and diesel (biofuel production), meanwhile, Dornburg and Faaij (2001) used the fossil primary energy savings from produced electricity and heat (both can be measured by MJ) as the functional unit. However, a proper functional unit should be considered when dealing with allocation issues, especially for systems with multiple products. For instance, existing LCA studies on bio-refineries managed to avoid allocation by selecting a proper functional unit by reporting final results per unit of input biomass (e.g. Uihlein and Schebek, 2009, Pettersson and Harvey, 2010) or on a yearly basis (e.g. Cherubini and Ulgiati, 2010).

With regards to the performance of the bioenergy products, a few life cycle studies concluded that bio-energy lead to significant GHG emissions reduction mainly due to carbon cycling and fossil fuel substitution (e.g. Laird et al., 2009, Fan et al., 2011, Thakur et al., 2014, Shen et al., 2015). Feedstock transportation process is usually signalled as the main contributor to energy consumption and GHG emissions in the entire life cycle of bio-energy products (Thakur et al., 2014). The economic performance of bio-energy production is affected by biomass procurement costs, transportation costs, technical maturity, conversion rate and capital cost (Muñoz et al., 2015). Solid woody biomass is generally considered a cost-competitive energy source (Hill et al., 2006). Yuzugullu (2013) reported that the cost of producing 1 kWh of electricity from woody biomass was about $0.06, which is comparable to other electricity production systems. However, this is not the case for liquid biofuels from woody biomass. Bio-ethanol production from woody biomass has been impeded by high capital costs, technological inefficiency, and lack of governmental incentives and subsidies (Schmidt et al., 2010, Demirbas, 2011, Sunde et al., 2011).

In Australia, a limited number of studies were conducted to analyse the environmental and economic impacts of different bio-energy products from biomass. For instance, Rodriguez et al. (2011a), Rodriguez et al. (2011b) analysed the viability of small-scale bioelectricity and ethanol production from lignocellulosic biomass in the Green Triangle region of South Australia. More recently, Murphy et al. (2015) and Hayward et al. (2015) studied the sustainability of the production of aviation fuels from lignocellulosic biomass in Queensland region. The studies indicated that under the pertaining economic conditions which then included government incentives such as waiver of the excise tax on ethanol and renewable energy credits, bioenergy production is economically viable and will result in significant GHG emission reduction. However, the studies focused on single bio-energy product and did not take into account impact categories other than GHG emissions. Therefore, the existing literature lacks a comparative analysis that addresses both environmental and economic criteria of the different woody biomass energy production methods. This study fills this gap by conducting a comparative life cycle study to investigate the environmental and economic efficiency of the available options for converting woody biomass into bioenergy products. This study focused on the utilisation of low-grade woody biomass generated from the hardwood forestry thinning operations in South East Queensland (SEQ) region. Nevertheless, the study is relevant nationally and internationally as it presents a novel method to integrate economic and environmental assessment using life cycle approach. The results of the study are also relevant to other waste and low-grade products from the softwood and pulp-wood plantations.

5.2 Methods and Materials

5.2.1 Environmental and Economic Assessment

The environmental and economic impacts of six different bioenergy applications were conducted following 'cradle to grave' LCA and LCC analysis based on the International Organisation for Standardisation ISO14040 (2006) and the Australian/New Zealand Standards Lifecycle costing AS/NZS 4536:1999 (R2014). This paper also demonstrates how to internalise the environmental cost by incorporating the life cycle GHG emissions into the LCC method.

5.2.1.1 Life Cycle Assessment

The LCA was conducted using SimaPro v.8.0.4.30 modelling software. Global warming potential (GWP), eutrophication potential (EP), acidification potential (AP), fossil depletion potential (FDP) human toxicity potential (HTP) were assessed based on the Best Practice Guide to Life Cycle Impact Assessment in Australia (Grant and Peters, 2008). Furthermore, land use potential (LUP) was also assessed due to its high relevance to the study. ReCiPe Midpoint (H) life cycle impact assessment (LCIA) method was used for the characterisation of emissions into the relevant impact categories except for the AP category for which the CML 2001 method was used as recommended by Bengtsson and Howard (2010b).

5.2.1.2 Life Cycle Costing Analysis

The LCC accounts all relevant costs, such as acquisition, operation, maintenance and disposal throughout the entire life stages (AS/NZS 4536:1999, R2014). Lu and El Hanandeh (2016a) emphasised that the life cycle cost related to environmental impacts should also be included in order to internalise the environmental cost. Therefore, the life cycle GHG emissions costs were also included. All the costs were converted to the Australian dollar in 2017 Present Value (PV). An Excel spreadsheet model was used to calculate the net present value (NPV). Table 5.1 presents the main economic assumptions and parameters used in this analysis.

Table 5.1 Main economic assumptions and parameters for Life Cycle Cost Analysis*

Parameters	Value	Sources
Inflation rate	3%	(Reserve Bank of Australia, 2015)
Discount rate	4.9%	(Reserve Bank of Australia, 2015)
Diesel fuel cost	$1.23/L	(Australian Institute of Petroleum (AIP), 2016)
Gasoline	$1.25/L	(Australian Institute of Petroleum (AIP), 2016)
Natural gas	$10/GJ ($0.387/m^3)	(Australian Institute of Petroleum (AIP), 2016)
LPG	$0.65/L	(Australian Institute of Petroleum (AIP), 2016)
Electricity cost	$0.2224/kWh	(Queensland Government Department of Energy and Water Supply, 2015)
Carbon Price	$29/t CO_2-eq	(The Treasury Australian Government, 2016)
Transport cost	$0.098/tkm	(Higgins et al., 2015)

* All the prices are expressed in constant dollar terms and discounted to the current financial year.

5.2.1.3 Functional Unit and System Boundaries

In this study, multiple final products are included (heat, electricity and liquid fuels); thus, the selected functional unit must be able to reflect the diverse output. As per the earlier discussion, input related functional units are the most commonly used in these cases. Therefore, in this study, the functional unit is selected as the treatment of 1 Mg of green hardwood logs produced from the thinning operation (density: 944kg/m^3) in South East Queensland region, Australia. The system boundaries diagram is shown in Figure 5.1. An average lifespan of 30 years was assumed for bio-energy conversion plants. As the thinned logs are a by-product of the forestry industry, the economic allocation was used to distribute emissions from the forestry operations stage (May et al., 2012).

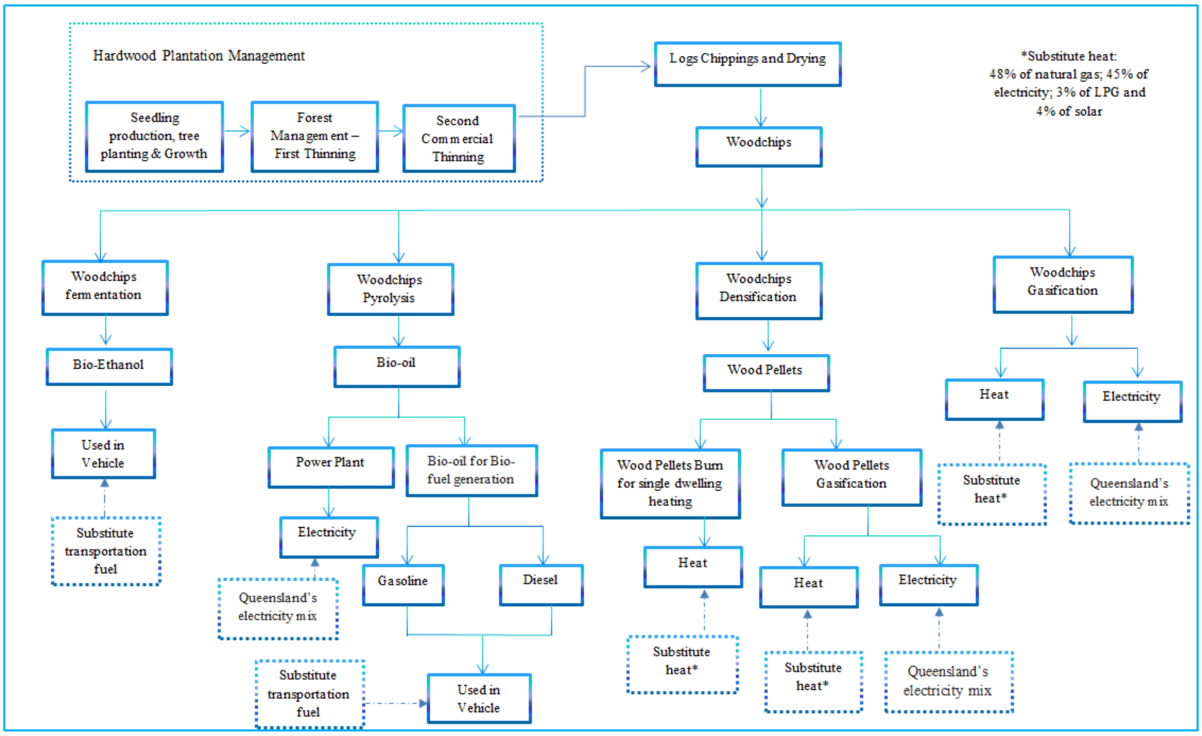


Figure 5.1 **System boundary of woody bioenergy production options**

5.2.1.4 Energy Conversion Technologies

The first step in the conversion process of woody biomass involves chipping the logs into small size woodchips for further processing. Four technologies were considered in this study: gasification; combustion; pyrolysis and fermentation.

Gasification - two options were considered for this technology: (1) woodchips gasification for power and heat generation and (2) wood pellets gasification. In both cases, a biomass integrated combined cycle (BIGCC) production technology was assumed because it is the most common technology in Australia (Hobson et al., 2003). Such plants are more efficient than traditional steam engines for power generation (Dornburg and Faaij, 2001, Marbe et al., 2004).

Combustion - domestic space and water heating was assessed as an option for wood pellets use because of the high efficiency usually achieved by modern domestic biomass boilers (Ammann et al., 2009).

Pyrolysis- under this technology, two options were assessed: (1) pyrolysis oil (bio-oil) was assumed to be converted to electricity via co-combustion in conventional liquid fossil fuels power generation plants, gas turbine combined cycle (GTCC) and diesel generators and (2) bio-oil generated from pyrolysis upgraded via hydrogenation processes to produce liquid fuels.

Fermentation- woody biomass can be converted to transportation liquid fuel products, such as ethanol (Demirbas, 2009). Piccolo and Bezzo (2009)) identified hydrolysis and subsequent fermentation to be the best available ethanol process technology.

5.2.2 Life Cycle Inventory (LCI)

For each stage of the product life cycle of each option, inputs of energy and raw materials, outputs of products, waste, and emissions releases to the environment were determined. In keeping with the convention of other forestry life cycle studies, the hardwood plantation system was assumed to be in a steady state (Dias et al., 2007, Lu and El Hanandeh, 2016a). Life cycle inventory (LCI) about the specific conversion processes were collected from the literature as listed in the relevant sections. Inventory data about the background and downstream processes were sourced from Ecoinvent database (Ecoinvent, 2016) and the Australian National Life Cycle Inventory (AusLCI, 2011).

5.2.2.1 LCI of Hardwood Plantation Thinning Operations

This LCA is based on data representing the current state of the art of hardwood plantation operations in Australia. The specific processes during establishment include seedling production, site preparation, slash burning and planting. Management processes include the chemical application of herbicides and fertiliser, fire prevention, construction of forest roads and control. The electricity consumption for greenhouse operations, fertiliser used for seedling production and growth, and the diesel fuel and lubricants required to power the equipment during site preparation, fertilisation, and thinning are considered as inputs to the plantation management. The primary output, for this study, is green logs from second thinning. Other products, such as non-merchantable slash and brunches are usually treated by mechanical activities or prescribed fire on site, which is not included in the system boundary. The LCI data of thinned logs, energy and materials consumption and the emissions released through the operation processes are shown in Appendix 2.

The life cycle cost for producing 1m^3 of green hardwood (Gympie Messmate) thinned logs was estimated by Lu and El Hanandeh (2016a), based on an average cost and productivity of hardwood plantation thinning in South-east Queensland. The cost of hardwood plantation establishment, management, operation, transportation, labour cost as well as land were all taken into account.

The average life cycle cost during hardwood plantation thinning was estimated at $36.97 per Mg of green logs (on site) over the 30-year simulation period.

5.2.2.2 LCI of Wood Chipping and Drying

In order to improve the efficiency of the woody biomass combustion and transportation, thinned logs extracted from the forest must be chipped (Esteban et al., 2014). In all scenarios, the thinned logs are assumed to be transported from the hardwood plantations site in the Gympie region, South East Queensland to the nearby wood chipping industry. The average haul distance from the forestry operations site to the wood chipping mill was estimated to be 20 km. It is assumed that logs were transported using both truck and train, with a full load for delivery and a 10% return load (Lu and El Hanandeh, 2016b). An industrial high-productivity wood chipper was assumed to be used for the wood-chipping operation that consumed 13.6 kWh electric energy and 1.23 L diesel fuel for chipping 1 Mg of wood (Higo and Dowaki, 2010). During chipping up to 2% of the wood is lost as dust and fines waste (Whittaker et al., 2011).

Green hardwood chips typically contain about 50% moisture. Bergman and Bowe (2011a) reported that 3,773 MJ of energy is consumed per cubic metre of hardwood for drying. This study assumed green logs were dried in continuous direct-fired dryers under a temperature between 150 º and 185 ºC with 80% of the energy supplied by woody biomass and the rest sourced from electricity and natural gas (Tucker et al., 2009). Drying reduces the moisture content to 10-12% (Oven-dry basis) with a density of ~700 kg/m^3. The transportation process for woodchips includes a mixture of rail and truck to either power plant for electricity generation or sent to a mill to generate wood pellets. The average haul distance was assumed to be 150 km. Table 5.2 shows a summary of the LCI of wood chips production.

Table 5.2 LCI of wood-chipping and drying per FU of thinned logs (Data adapted from (Higo and Dowaki, 2010, Whittaker et al., 2011, Esteban et al., 2014))

Parameter	Unit	Amount
Input material or energy		
Thinned logs (green)	Mg	1

Diesel fuel for on-site transportation	L	0.62
Diesel fuel for chipping	L	1.23
Electricity	kWh	177.08
Natural gas	MJ	149.53
Wood fuel	MJ	3133.51
Lubricating oil	L	0.003
Transport to chipping mill	tkm	22
Output product and waste		
Wood chips (dry)	kg	700
Wood dust (waste)	kg	20

The life cycle cost during this stage was contributed by wood chipping, transportation, as well as drying. The average cost of chipping and transportation was estimated at $10.32/Mg of wood logs. The average LCC of woodchips drying was estimated as $22.25/Mg of thinned logs. Therefore, the average LCC during this stage was estimated as $32.56/Mg of green thinned logs.

5.2.3 Bioenergy Applications

5.2.3.1 Woodchips gasification for electricity generation (WCG)

Gasification is a thermo-chemical process which occurs at moderately low temperatures in a limited oxygen environment. Through gasification, woody biomass can be converted into syngas which can be used for electricity production (Huber et al., 2006). The dried woodchips were assumed to be transported to a biomass integrated gasification combined cycle (BIGCC) plant. Gasification of biomass in a combined heat and power (CHP) plants allows reaching maximum electrical efficiencies up to 34%, and thermal efficiency up to 53% (Guest et al., 2011).

A medium size (5 MW) fixed-bed gasification plant with annual electricity and heat generation of 43,800 MWh and 68,276 MWh, respectively was assessed in this study. The gross calorific energy content of hardwood chips (moisture content 10%) was assumed to be 16.2 MJ/kg based on NSW Environment Protection Authority (2012). The ash and solid waste generated via wood gasification were assumed to be 9% of the total weight of woodchips (Kim and Song, 2014) and was assumed to be landfilled with average transportation distance of 50 km. Table 5.3 presents the LCI of woodchips gasification for the treatment of 1 Mg of woody biomass for energy production. The electricity and heat produced were assumed to displace electricity from the main electricity grid mix and industrial heat requirements mainly from electricity (42%) and natural gas (58%) (Kenway, 2013).

Table 5.3 LCI of wood gasification per FU (Data adopted (Kalinci et al., 2012, François et al., 2013, Yang and Chen, 2014))

Parameter	Unit/FU	Amount
Use of resources		
Wood chips (dry)	kg	700
Cement	kg	0.86
Steel	kg	0.27
Aluminium	kg	0.0023
Iron	kg	0.0034
Transport to bioenergy facilities	tkm	115.5
Energy		
Electricity	kwh	95.77
Natural gas	kwh	33.52
Product		
Electricity produced	MJ	3,856
Heat produced	MJ	6,010
Waste		
Ash and solid waste (Landfill)	kg	60.34
Wastewater to treatment plant	L	60.34
Sludge to incinerator	kg	2.90

In order to calculate the life cycle cost, the levelised electricity cost (LEC), was estimated using the International Energy Agency (IEA) costing model (IEA, 2010). The levelised energy cost is equal to the present value of the sum of discounted costs divided by the total production adjusted for its economic time value. To calculate the levelised equivalent cost of electricity, heat credit is subtracted from the total unit cost. However, since heat and power are genuine co-products, this approach is discouraged (IEA, 2010). In order to generate comparative results to compare with other energy products such as heat, transportation fuel or bio-oil, the quantity of heat from CHP was also taken into account; the LEC is calculated using the following equation:

$$LEC = \frac{\sum_{t=1}^{n} \frac{I_t + O\&M_t + T_t + F_t + C_t + D_t + W_t}{(1+r)^t}}{\sum_{t=1}^{n} \frac{E_t}{(1+r)^t}}$$

E_t: The amount of energy produced in year "t";

$(1+r)^t$: The discount factor for year "t";

I_t: Investment costs in year "t";

$O\&M_t$: Operations and maintenance costs in year "t";

F_t: Fuel costs in year "t";

C_t: Carbon costs in year "t";

D_t: Decommissioning cost in year "t".

T_t: Transportation costs in year "t".

W_t: Waste management cost in year "t".

The annual biomass feed rate in a 5 MW CHP plant is approximately 28.62 thousand dry tonnes. The capital cost including gasifier, boiler, steam turbine, auxiliary equipment, grid connection, civils and infrastructure was estimated at $27 million (Stucley et al., 2012). The initial investment I_t was also calculated to be approximately $0.9 million per year in 2012, which is equal to $1.013 million per year in present value. M_t represents the maintenance costs, which was estimated at $1.80 million/year in 2017 PV. The operating costs (O) for a bioenergy plant including labour and consumable items were estimated at $1.5 million/year based on current Australian bioenergy production cost model (Rodriguez et al., 2011b, Stucley et al., 2012). The feedstock expenditure F_t, was estimated based on the life-cycle cost of the woodchips multiplied by the amount of biomass required to run the plant. Decommissioning costs D_t was estimated at 5% of construction costs (IEA, 2010). The carbon price was assumed to be $29/t CO_2-eq (The Treasury Australian Government, 2016). Thus, the carbon cost in this study was calculated by using carbon price times the quantities of CO_2 eq reported in our LCA results. Furthermore, the cost for landfills was assumed as $77 per tonne of waste in Australia (Construction and Demolition Waste Guide, 2012). The E_t is the amount of energy generated every year. The average life cycle cost for producing 1 MJ of energy was calculated as $0.018 (i.e. 0.39 MJ of electricity, 0.61 MJ of heat based on energy production ratio). Therefore, the average LCC of woodchips gasification for energy production was calculated as $177.60/FU of hardwood thinning logs.

Currently, there is no nationally accepted protocol for apportioning either the energy inputs or GHG emissions arising from the generation of thermal and electrical energy supplied by CHP systems. The Energy Efficiency Council (2013) suggests the proportion method (also called as exergy method) for calculating the emission allocations using the following equation.

$$\text{Emissions}_{\text{Heat}} = \text{Emissions}_{\text{Total}} \times \frac{\frac{Heat\ Output}{Efficiency\ heat}}{\frac{Heat\ output}{Efficiency\ heat} + \frac{Electricity\ output}{Efficiency\ electricity}}$$

And Emissions $_{\text{Total}}$ = Emissions$_{\text{Heat}}$ + Emissions$_{\text{Electricity}}$

Where:

Emissions$_{\text{Total}}$ = total emissions from CHP plant

Emissions$_{\text{Heat}}$ = emissions share attributable to heat production

Emissions$_{\text{Electricity}}$ = emissions share attributable to electricity production

Emissions$_{\text{Heat}}$ = assumed efficiency of typical heat production

Emissions$_{\text{Electricity}}$ = assumed efficiency of typical power production

The calculated result indicated that Emissions$_{\text{Heat}}$ = Emissions$_{\text{Electricity}}$ = 0.5. Therefore, the average life cycle costs for producing 1MJ electricity and heat were estimated at $0.023 and $0.015, respectively.

5.2.3.2 Wood pellet production

To produce wood pellets, dried chips arriving at the pellet mill are ground using a hammer mill to reduce the size to particle size between 6.4 or 3.2 mm, which are then compressed to form pellets (Magelli et al., 2009). The produced wood pellets are cooled down to ensure the quality and durability of the product before being delivered using the pneumatic conveying system to a storage facility (Magelli et al., 2009, Pa et al., 2011). The heating value of the wood pellets was estimated as 16.65 MJ/kg (Ghafghazi et al., 2011). The LCI of pellet manufacturing is listed in Table 5.4.

Table 5.4 LCI of pellet production process associated with 1 Mg of hardwood thinned logs (data sourced from Magelli et al. (2009))

Parameter	Unit	Amount
Use of resources		
Wood chips (dry)	kg	700
Aluminium	kg	0.013
Steel	kg	0.06
Cast iron	kg	0.025
Chromium steel	kg	0.001
Reinforced steel	kg	0.015
Concrete	kg	0.043
Transport to wood pelleting mill	tkm	124.73
Energy Use		
Electricity	kWh	133.65
Natural gas	MJ	563.32
Diesel	L	6.09
Product		
Wood Pellet	kg	679.7
Waste		
Solid waste	kg	0.25

The general cost for wood pellets production from dry woodchips was obtained from Stucley et al. (2012) and Mobini et al. (2013), who estimated that the feedstock and transportation cost constituted approximately 40% of the total production cost. The rest of the costs are related to processing, capital investment, spare parts and consumables, labour costs, as well as distribution cost (Mobini et al., 2013). Based on this assumption, the total wood pellet production cost from 1Mg of thinned logs (green) was then estimated at $173.8.

Option 1: Pellet Combustion in Domestic Boiler (WPC)

The most common use of wood pellets is direct burning for heating. Recent research shows that the energy associated with residential water heating represents 23% of Australian household energy usage (Kenway, 2013). In this scenario, wood pellets were assumed to be used in individual households for space and water heating (El Hanandeh, 2015). Wood pellets are transported via truck for an average distance of 50 km. Burning wood pellets in the household boiler are assumed to comply with the Australian/New Zealand Standard AS/NZS 4013 (2014). For biomass with 10% moisture content, the thermal efficiency of 85% was assumed (El Hanandeh, 2015). The total heat production was then calculated as 9,619 MJ. For woody feedstock using households, the average annual wood pellet intake rate was estimated at 1.9 tonnes per household based on the average Australian household heat demand (Australian Energy Statistics, 2016). The emission released from burning 1 Mg of wood pellets includes 62.5 kg of CO, 21.75 kg of NO_x, 0.225 kg of SO_2, 4 kg of PM_{10}, 5 kg of VOC, as well as 40 kg of ash (AS/NZS 4013, 2014). This study assumed that the ash is deposited in a nearby landfill within 50 km radius. It is further assumed that the heat output from wood pellet displaced energy from the average domestic water heating composition: 48% of natural gas; 45% of electricity; 3% of LPG and 4% of solar (El Hanandeh, 2015). The average life-cycle cost of wood pellet combustion in the domestic boiler (WPC) includes the costs of pellet production, and distribution, domestic wood pellet boiler, carbon cost, as well as waste treatment. The cost of a home water heating boiler was estimated to be $10,000 and last for 30 years (Eco-Home-Essentials, 2016). Thus, the average life-cycle cost of the domestic boiler for burning wood pellet from 1FU of thinned logs was estimated at $91.6. Therefore, the total life-cycle cost of WPC option was then estimated as $270/Mg of thinned logs, while the average energy production cost was calculated as $0.028/MJ of heat.

Option 2: Wood Pellet Gasification (WPG)

In this scenario, wood pellets were assumed to be transported 50 km by truck to a gasification plant using CHP technology. Ash and solid waste generated were assumed to be 8% and 0.87% of total weight of wood pellets, respectively. The ash and solid waste were assumed to be sent to a landfill for final disposal with an average distance of 50 km. Gasification emissions were based on wood gasification in a commercial fixed bed gasifier (Koroneos et al., 2008). During the wood pellet gasification, 3,850 MJ of electricity and 6,000 MJ of heat are produced. The energy produced was assumed to displace electricity from the main electricity grid mix and industrial processing heat. The average cost for energy production was estimated at $0.031/MJ (allocated as $0.040/MJ of electricity and $0.025/MJ of heat) based on the LEC formula that mentioned in section 2.3.1. Consequently, the average LCC of wood pellet gasification for energy generation was then calculated as $305/FU of woody biomass, including the cost of pellet production, gasification, transportation, carbon cost and waste management.

5.2.3.3 Woodchips Pyrolysis

Pyrolysis is the thermo-chemical decomposition of biomass in absence of oxygen to produce char, gas and liquid fractions (Bridgwater, 2012). The proportions of the pyrolysis products vary according to process conditions. Fast pyrolysis can achieve maximum yield of liquid at around 500 °C (Demirbas and Balat, 2007). The liquid produced after condensation and filtering is called bio-oil or bio-crude which is a potential substitute for petroleum. Upgrading of the bio-oil to petroleum compatible products is difficult due to the complex composition of the bio-oil (Czernik and Bridgwater, 2004). This study assessed two applications of bio-oil: the first is direct use as a substitute for heavy fuel oil for electricity generation and the second is upgrading to liquid transportation fuel through hydro-processing. As the commercial viability of hydro-processing has not yet been achieved (Iribarren et al., 2012); this paper used data from laboratory scale experiments.

Option 1 - Pyrolysis for Electricity Generation (PyEl)

Inputs for this step obtained from reports by Demirbas and Balat (2007), Fan et al. (2011) and Peters et al. (2015) are summarised in Table 5.5. The obtained yields were assumed to be 68.8% bio-oil, 14.3% gas and 16.9% char. The produced gas and char were assumed to be burned in the combustor for woodchips pre-treatment and to maintain the process (Peters et al., 2015). The energy densities of char and bio-oil were estimated as 24.5 MJ kg^{-1} and 19 MJ kg^{-1}, respectively (Stucley et al., 2012, Peters et al., 2015).

Table 5.5 LCI of converting 1FU of thinned wood logs to bio-oil through pyrolysis for electricity generation

Materials	Quantity	Unit
Woodchips (dry)	700.00	kg
Water	18.25	L
Energy		
Electricity	143.08	kWh
Natural gas	0.32	kg
Transport to bioenergy facilities	124.73	tkm
Products		
Pyrolysis Bio-oil	502.27	kg
Char	48.12	kg
Waste to treatment		
Solid waste	6.36	kg

This study assumed that an integrated biomass to electricity facility was used as described by Iribarren et al. (2012). The electricity generation efficiency from bio-oil and char was assumed to be 34% (Fan et al., 2011). Therefore, 1 FU of woodchips could only convert to 3,638 MJ of electricity theoretically. The generated electricity was assumed to displace electricity from the main grid mix.

The production costs were calculated using a standard levelised cost formula (IEA, 2010), which has been discussed previously. The capital cost for fast pyrolysis unit was calculated based on the equation suggested by Bridgwater (2012) and Hayward et al. (2015):

$$K_{AUD\ million\ 2012} = 2.302 * f^{0.67}$$

Where K is the capital cost and f is the biomass feed rate in dry ton yr^{-1}. In order to be comparable to other scenarios, the biomass feed rate was assumed to be 28.62 thousand dry tonnes per year. The capital cost was then calculated as $24.51 million in 2017 value, while the annual operation and maintenance (O&M) costs were estimated at 2.93 million in present value (Hayward et al., 2015). The cost of feedstock was calculated based on the life-cycle cost of woodchips times the annual biomass feed rate. The average life cycle cost for converting 1 FU of woody biomass for electricity generation was calculated as $204. In addition, the average cost for producing 1 MJ of electricity was estimated at $0.056.

Option 2 – Pyrolysis for Liquid Transportation Fuel (PyLT)

The crude bio-oil obtained from fast pyrolysis was assumed to be upgraded to synthetic fuels in a hydrothermal liquefaction plant. The raw bio-oil is converted into gasoline and diesel blend-stocks via hydro-upgrading. The hydrothermal liquefaction plant consists of three principal processing steps: hydro-treatment; product separation (including hydrocracking); and steam reforming. The bio-oil from pyrolysis plant is pressurised with hydrogen up to 170 bar and fed to the reactor. The bio-oil in the reactor is stabilised under 270 °C for further processing. In the next step, the stabilised oil is deoxygenated under more severe conditions (150 bar, 350 °C) through a combination of hydro-deoxygenation and decarboxylation processes. Then the deoxygenated oil is distilled in order to reduce the oxygen content to below 2%. It was estimated that 0.1kg catalyst consisting of 11% Mo, 4% Co and 7% S on an Al_2O_3 is required for each metric ton of feedstock (Marafi et al., 2010, Peters et al., 2015). In the hydrocracking process, large hydrocarbons are broken up under severe conditions into smaller molecules. The cracked oil then enters the fractionation process to separate the gasoline and diesel fractions. The required hydrogen is produced from internal gases and external natural gas in the steam reformer (François et al., 2013).

The final products are dispatched gasoline and diesel blend-stocks with corresponding heating densities of 44.40 MJ/kg and 42.79 MJ/kg, respectively (Iribarren et al., 2012). This study assumed that the produced gasoline and diesel are perfect substituted to conventional petroleum products and therefore displace equivalent quantities of the corresponding products. The fuels are transported to regional storage with an average distance of 50 km for use as vehicle fuel. Emissions generation during diesel and gasoline combustion were based on Ecoinvent database (Ecoinvent, 2016). The summary of LCI is presented in Table 5.6.

Table 5.6 LCI of 1FU of woody biomass for biofuel production from bio-oil (Data sourced from (Iribarren et al., 2012, Peters et al., 2015, Demirbas and Balat, 2007)

Materials	**Quantity**	**Unit**
Bio-oil	502.27	kg
Process Water	184.17	L
Catalyst	0.05	kg
Energy		
Electricity	92.08	kWh
Natural gas	2511.35	MJ
Transportation	35.16	tkm
Products		
Dispatched gasoline blend-stock	4102	MJ
Dispatched diesel blend-stock	4269	MJ
Waste to treatment		
Wastewater	242.76	kg

The hydrothermal liquefaction plant is a novel technology. Therefore, the costs in this study were estimated using the method suggested by Hayward et al. (2015) as follows:

$$K_{AUD\ million\ 2012} = 2.302 * f^{0.54}$$

where 0.54 is the scaling factor for capital cost estimation. Therefore, the capital cost of the hydrothermal liquefaction plant was calculated as $15.85 million in 2017 present value. In addition, the O&M cost was estimated at $2.93 million/year. The average life cycle cost for converting 1 FU of woody biomass to biofuel production was then calculated as $276. The average cost for producing 1 MJ of energy was then estimated at $0.033.

5.2.3.4 LCI of Ethanol Production via Fermentation (EthP)

The woody biomass enzymatic hydrolysis with fermentation to bio-ethanol was assessed in this study. The resulting ethanol was assumed to be used as fuel mix to produce E85 fuel. The ethanol conversion process starts with woodchips pre-treatment and conditioning, followed by enzymatic hydrolysis and fermentation process. Finally, the produced ethanol is distilled and dehydrated in order to achieve 99.5% purity. The solid waste from the production process was assumed to be landfilled and the bottom liquid waste, generated in the distillation and evaporation processes, treated in wastewater treatment plant. The LCI data, including enzymes production and ethanol production, was sourced from González-García et al. (2012) and summarised in Table 5.7. The produced bio-ethanol can be blended with gasoline to produce E85 and delivered from the refinery to local petrol stations by trucks. In this study, the produced ethanol was assumed to displace gasoline based on energy content value.

Table 5.7 LCI for 1FU of lignocellulosic for ethanol production (sourced from González-García et al. (2012))

Materials	Unit	Amount
Inputs from technosphere		
Woodchips	kg	700
Vinyl acetate	kg	0.33
Sulphuric acid	kg	18.85
Lime	kg	13.46
Diammonium phosphate	kg	0.23
Enzyme	kg	75.38
Nutrient feed	kg	0.26
Energy		
Electricity	kWh	242.31
Steam	GJ	1.35
Transport to bioenergy facilities	tkm	118.46
Transport to gasoline station	tkm	11.04
Outputs to technosphere		
Ethanol (99.5%) (26.8MJ/kg)	MJ	5385
Wastes to treatment		
Gypsum (to Landfill)	kg	26.92
Ash (to Landfill)	kg	15.62

In order to estimate the life cycle cost, an advanced lignocellulosic bioethanol plant with annual feedstock rate of 28.62 thousand dry tonnes was assessed. The capital cost was estimated at $23.77 million AUD in 2017 with the annual O&M cost of $2.26 million (Arif et al., 2014).

Therefore, the average life-cycle cost for treating 1 FU of woody biomass for bio-ethanol production was estimated at $183. Hence, the average life-cycle cost for producing 1 MJ of ethanol was estimated at $0.034.

5.3 Results

Table 5.8 presents a summary of the results. Woodchips gasification (WCG) is the best performing option because it has the lowest environmental impacts and life cycle cost. However, choosing the runner-up is less straightforward because of the conflicting performance across the environmental and life-cycle cost. The WPG alternative ranks second on the environmental criteria but has the highest life-cycle cost. On the other hand, EthP ranks the second lowest life-cycle cost but is the worst performer on the environmental criteria. The results also show that solid fuel options (WCG, WPG and WPC) have lower environmental impacts compared to liquid fuel options (PyLT and EthP).

Both gasification options (WCG and WPG) performed almost equally in the GWP impact category, however, WCG has a clear advantage when considering the other environmental impact categories. Although pelletising increased the energy density, the emissions and costs associated with the additional processing of the biomass outweighed the benefits gained from the densification of the fuel. Comparing WPG to WPC reveals that the WPC option has a lower LCC but delivers lower environmental value. This can be explained by 1) in the case of WPC the displaced product (heat) contains significant proportion generated from biomass and gas as opposed to the electricity, in the case of WPG, which is mainly generated from coal and 2) the energy conversion efficiency of the technology compared to the displaced system (i.e, CHP has higher efficiency than the typical coal-fired power plant while domestic biomass boiler has comparable efficiency to gas or electric water heater). Furthermore, the WPC option has higher human toxicity impact due to increased VOC, PAH and PM emissions from biomass combustion (Caserini et al., 2010).

With regard to liquid fuel options, ethanol production has lower life-cycle cost than pyrolysis with upgrading to dispatch gasoline and diesel fuel. Nevertheless, EthP has higher environmental impacts than PyLT due to lower yield combined with the lower heating value of ethanol compared to diesel and gasoline. A particular concern is that EthP is the only option that results in added burdens in the GWP which makes it unfavourable option despite its lower LCC. Reduction of GHG emissions, and in turn reducing the GWP, is the common argument used for promoting biofuels. However, as ethanol production from woody biomass is an emerging technology, advancements are expected to improve the efficiency, thus enhancing its environmental performance and life-cycle cost in the future (González-García et al., 2012).

The production of electricity through pyrolysis (PyEl) had lower GWP impact and life cycle cost that producing liquid fuel (PyLT). These benefits can be directly correlated to the higher conversion efficiency of the CHP plant as well as the displaced fossil fuel (mainly coal in the case of electricity). In general, upgrading biomass to liquid fuels seems to result in lower environmental benefits than the more direct utilisation options.

Table 5.8 LCA & LCC result from different bio-energy conversion scenarios per functional unit

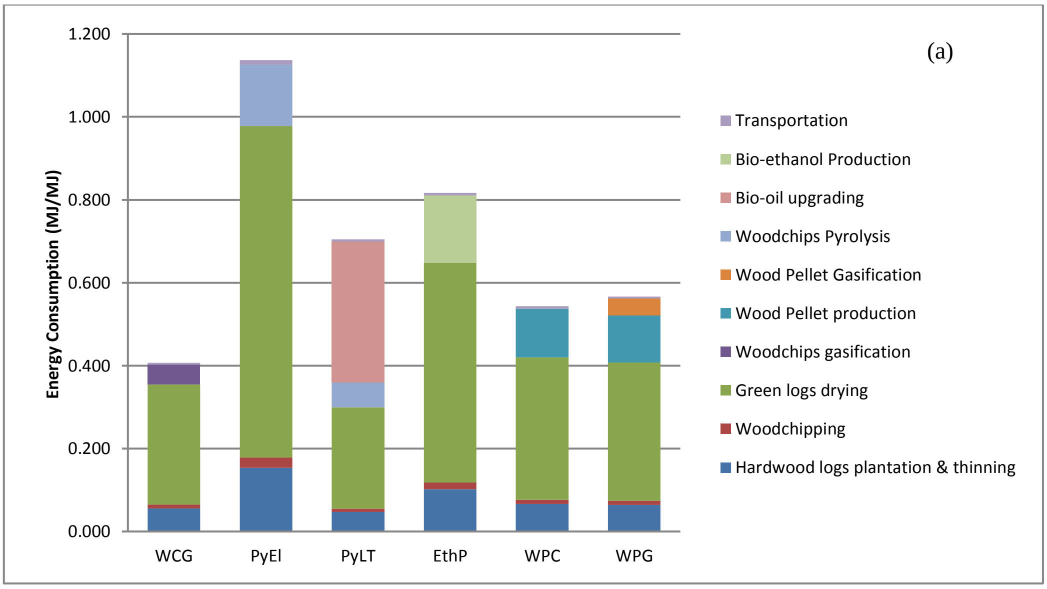
Impacts	Units/FU	WCG	PyEl	PyLT	EthP	WPC	WPG
GWP	kg $CO_{2\ eq}$	-1.08E3	-294	-269	14.9	-767	-1.07E3
AP	kg $SO_{2\ eq}$	-0.942	0.107	-0.77	0.097	-0.342	-0.673
EP	kg $PO_{4\ eq}$	-0.574	-0.071	0.164	0.33	-0.233	-0.466
FDP	kg Oil $_{eq}$	-374	-5.75	-51.6	-10.3	-253	-333
HTP	kg 1,4-DB $_{eq}$	-41.6	40.9	35.8	88.5	37.6	-0.329
LUP	m^2*yr	24.8	46.8	68.1	94.1	40.5	28.1
LCC	$/FU	177.60	204	276	183	270	305

5.3.1 Further discussion based on energy return

Figure 5.2 (a & b) shows the breakdown of the energy and cost required for the production of 1 MJ energy over the entire life cycle of each option. As shown in Figure 5.2 (a), the WCG is the most energy-efficient option while PyEl is the least energy efficient option. All options, except for PyEl, had net positive energy return. In fact, PyEl delivers net energy loss of 0.126 MJ which makes it an undesirable option from energy return perspective. The energy inefficiency of PyEl means more feedstock is required for producing the same quantity of energy compared to the other options, which would also cause the high energy input during the bio-oil production phase. The energy required during green logs drying is the most significant contributor to the total energy consumption in all options, especially in the PyEl case where the drying process (both wood fuel and fossil fuel consumption) contributes to approximately 70% of its total energy requirement for generating 1 MJ of electricity. Nevertheless, the PyEl option still shows negative GHG emissions in our final result. This paradoxical result is explained by the accounting method used for GHG emissions from biomass. According to the Intergovernmental Panel on Climate Change (IPCC) (2007), carbon dioxide emissions originating from the combustion of biomass do not count towards the GWP impact because they have been recently absorbed from the atmosphere and will be taken up by the next generation biomass. In the PyEl option, up to 80% of energy consumption during the drying process is sourced from wood waste residues (biomass); accordingly, the associated CO_2 emissions are classified as biogenic and do not count towards the GWP impact (Intergovernmental Panel on Climate Change (IPCC), 2007). However, if the CO_2 emissions from wood combustion were not taken up by the next generation of plantation within 100 years, then they will be treated as abiotic emissions adding up to 314.5kg of $CO_{2\text{-eq}}$ to the GWP impact. As a result, the PyEl option may become a net contributor to GHG emissions.

Figure 5.2 (a) also revealed that biomass drying was the most energy consuming process, except in the case of PyLT where bio-oil upgrading to transportation fuel accounted for 34% of the total energy requirement. Referring to Figure 5.2 (b) it can be derived that the O&M cost and the feedstock & transportation costs are the two most significant contributors to the total life-cycle cost in all options. On the other hand, the decommissioning cost and carbon cost are negligible in all options.

Considering energy output rather than biomass input reveals that WPC and WPG have lower LCC per MJ produced than PyEl and PyLT. In the other words, although WPC and WPG had higher LCCs for treating 1 Mg of thinned logs, the higher energy output compensates for the added cost. As such, it can be inferred that solid fuels are more efficient utilisation of biomass than conversion to liquid fuels.



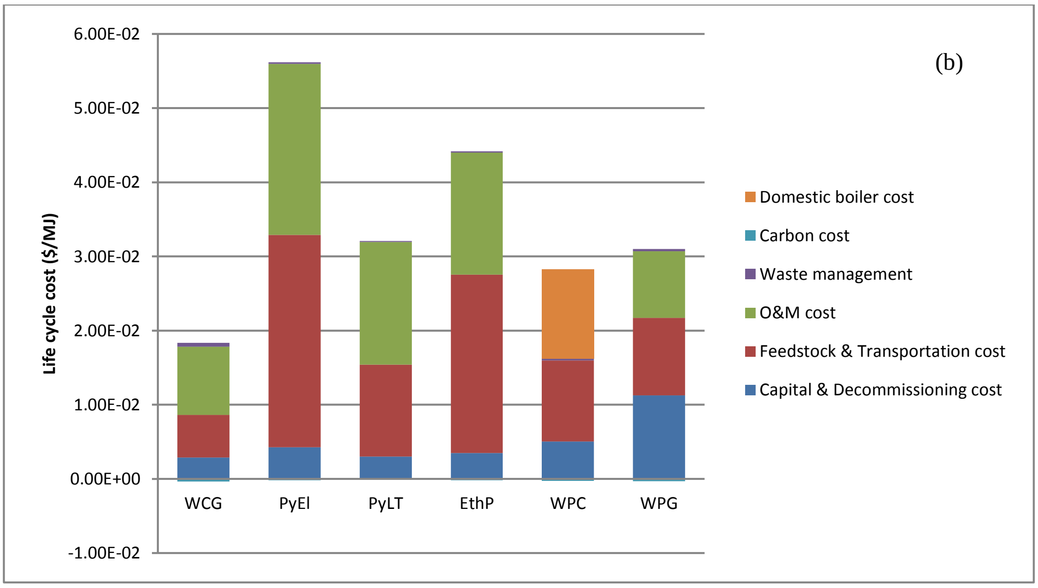


Figure 5.2 (a). **Breakdown of energy requirement for converting 1MJ of energy in each option** (b). **Breakdown of LCC for converting 1MJ of energy in each option**

5.4 Uncertainty and Sensitivity Analysis

Variability in the input data and assumptions made in the environmental and economic models may affect the final results. The uncertain factors that may significantly affect the LCA and LCC results were identified. Sensitivity analysis was carried out to investigate the effect of individual factors. Monte Carlo simulations were also conducted to better understand the effect of the combined uncertainties on the results.

5.4.1 Uncertainty and Sensitivity Analysis in LCA

Transportation distances, energy consumption and energy conversion efficiencies can affect the results of the LCA and LCC models. Therefore, the sensitivity of the results to each of these factors is discussed in the following sections.

5.4.1.1 Transportation distance

Transportation distances were estimated using Google maps (Google maps directions, 2016), however, variations are expected. Sonnemann et al. (2003) suggested that a 20% coefficient of variation may be used to estimate the actual transportation distance. Variation in transportation distances did not have a significant impact on the GWP and LUP impact categories in all options. However, changes in transportation distances affected the AP (± 33% to ± 43%), EP (±22% to ±49%) and HTP (±33% to ±58%) impact categories of the PyEl, WPG and WPC options.

Furthermore, sensitivity analysis revealed that shorter transportation distances caused the gap in environmental impacts of all scenarios to shrink. Therefore, transportation is a critical factor in the environmental performance of all assessed options.

5.4.1.2 Energy consumption

Over the last decade, technological improvements contributed 20% energy saving in the Australian industrial sector (Voigt et al., 2014). Therefore, we tested the effect of ±20% change in energy consumption on the performance of the assessed options. Sensitivity analysis showed that changes in energy consumption had more impact on the PyEl option, followed by WPG and WPC options. In particular, AP (±45% to ±50%), EP (±15% to ±48%) and HTP (±23% to ±54%) impact categories were the most sensitive to variation in energy consumption. The relatively higher energy usage during the conversion process is responsible for their higher sensitivity to variation in energy consumption. The GWP and LUP were relatively less sensitive (<20%) for all options.

5.4.1.3 Bio-energy conversion rate

The bio-energy conversion rate may vary because of the different technologies and future technological advances. Hayward et al. (2015) noted that the actual bio-energy conversion rate may vary by 10%. Accordingly, a sensitivity analysis was conducted by varying the feedstock input by ±10%. The result showed that changes in conversion rate would affect all impact categories proportionately by ±10%.

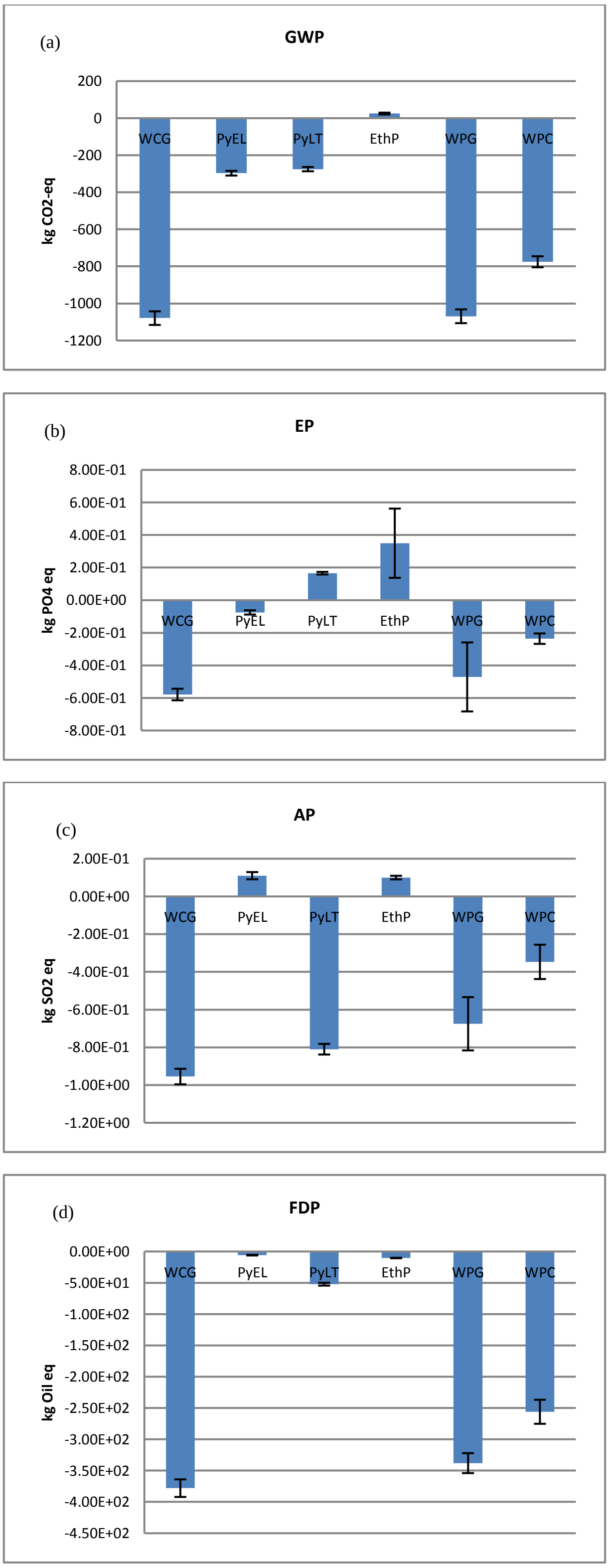
5.4.1.4 Effect of Combined Uncertainties on Overall Performance of Alternatives

One thousand (1000) Monte Carlo Simulations were conducted using SimaPro v.8.0.4.30 modelling software based on the ISO14040 (2006) by allowing all uncertain parameters to change simultaneously (GreenDelta, 2014). Table 5.9 presents the parameters input in the Monte Carlo Analysis, as identified from the sensitivity analysis. Additionally, the anticipated change in renewable energy share in the Queensland's electricity mix was also included. A higher renewable energy content rate of 15% of total electricity generation was considered in Monte Carlo Simulation as suggested by Australian Energy Statistics (2016). The LCI data for renewable energy was sourced from AusLCI (2011).

Table 5.9 Parameters used in the Monte Carlo analysis

Parameter	Mode	Unit	Distribution type
Transportation distance	100	%	Normal: μ=100, σ=20
Energy consumption	100	%	Normal: μ=100, σ=20
Bioenergy conversion rate	100	%	Normal: μ=100, σ=10
Renewable energy content in electricity generation	12	%	Normal: μ=12, σ=3

The Monte Carlo analysis results are presented in Figure 5.3. The results suggest that WPC, WPG and EthP options are more sensitive to the overall uncertainties in all impact indicators. Monte Carlo Analysis indicated that the uncertainties will particularly have more effect on the EP and HTP impact categories of the EthP option. The GWP, EP, HTP, and LUP impacts of the EthP are likely to exceed the impacts of the PyLT option. In both wood pellets options (WPC and WPG), the effect of the overall uncertainties was more pronounced on the AP, EP, FDP and HTP impacts, while the GWP and LUP were less sensitive to the changes. The simulation results also suggested that the overall ranking of all scenarios were remaining the same.



(a)
GWP
kg CO2-eq
200
0
-200
-400
-600
-800
-1000
-1200
WCG
PyEL
PyLT
EthP
WPG
WPC
(b)
EP
kg PO4 eq
8.00E-01
6.00E-01
4.00E-01
2.00E-01
0.00E+00
-2.00E-01
-4.00E-01
-6.00E-01
-8.00E-01
WCG
PyEL
PyLT
EthP
WPG
WPC
(c)
AP
kg SO2 eq
2.00E-01
0.00E+00
-2.00E-01
-4.00E-01
-6.00E-01
-8.00E-01
-1.00E+00
-1.20E+00
WCG
PyEL
PyLT
EthP
WPG
WPC
(d)
FDP
kg Oil eq
0.00E+00
-5.00E+01
-1.00E+02
-1.50E+02
-2.00E+02
-2.50E+02
-3.00E+02
-3.50E+02
-4.00E+02
-4.50E+02
WCG
PyEL
PyLT
EthP
WPG
WPC

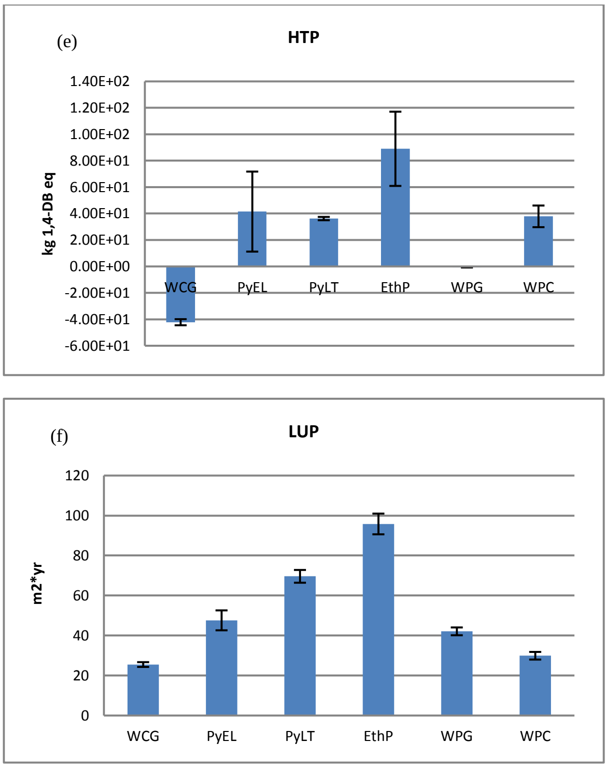


Figure 5.3 (a, b, c, d, e & f) Environmental performance of each option under uncertainty conditions

5.4.2 Uncertainty and Sensitivity Analysis in LCC

Uncertainty and sensitivity analysis were performed to evaluate the effect of the assumptions made in LCC modelling. The inflation rate, discount rate, capital cost, O&M cost, carbon price and bio-energy conversion efficiency were tested.

5.4.2.1 Inflation rate

The Australian inflation rate fluctuated from a minimum of 1% to a maximum of 5% over the past decade (Reserve Bank of Australia, 2015). Sensitivity analyses showed that an increase in the inflation rate by 2% could result in 33% increase in the LCC of all options. On the other hand, a reduction of 1% would lead to 22% decrease in LCC of all scenarios.

5.4.2.2 Discount rate

The reported Australian discount rates fluctuated between 3% and 7% over the last decade (Reserve Bank of Australia, 2015). Sensitivity analyses showed that a 2% reduction could result in 33% increase in the LCCs for all options. Meanwhile, the LCCs of all options would reduce by 22% if the discount rate reaches 7%.

5.4.2.3 Capital cost and O&M cost

The capital and O&M cost were sourced from published data. Huang et al. (2015) suggested that actual costs may differ from the estimated cost by 30%. Sensitivity analysis results showed that changing capital and O&M cost by ±30% had more influence on WPC and WPG options which changed by ±14% and ±15%, respectively; meanwhile, the effect on PyLT option was ±11%.

5.4.2.4 Bio-energy Conversion Rate

Technical efficiency indirectly affects the LCC by changing the quantity of feedstock for the final product. Uncertainties in bio-energy conversion rate from woody biomass are usually between 10 and 20% (Brinsmead et al., 2015, Hayward et al., 2015). Sensitivity analyses showed that a ±15% variation in the technical efficiency will result in proportional changes in the unit cost.

5.4.2.5 Carbon Price

The Australian carbon prices were reported to vary from AUD 23 to 31 per Mg of CO_2 equivalent in 2016 (The Treasury Australian Government, 2016). Sensitivity analysis suggested that changes in carbon price will have negligible effects on the LCC of all scenarios.

5.4.2.6 Monte Carlo Simulation in LCC

Monte Carlo Simulations was also performed to test the effect of combined uncertainties on the LCC results. One thousand (1000) simulations were run in Excel spreadsheet model. The parameters tested are summarised in Table 5.10.

Table 5.10 Parameters used in the Monte Carlo analysis

Parameter	**Mode**	**Unit**	**Distribution type**
Inflation rate	3.0	%	Normal: μ=3.0, σ=2.0
Discount rate	4.9	%	Normal: μ=4.9, σ=2.0
Capital and O&M cost	100	%	Normal: μ=100, σ=30
Bioenergy conversion rate	100	%	Normal: μ=100, σ=10
Carbon Price	29	\$/tonne $CO_{2\ eq}$	Triangle: Min=23; Max=31

Transportation distance	100		%	Triangle: Min=80; Max=120
GWP	WCG	-1.08E3	kg $CO_{2\ eq}$	Normal: μ=-1.08E3; σ=36.8
	PyEl	-297	kg $CO_{2\ eq}$	Normal: μ=-297; σ=12.6
	PyLT	-275	kg $CO_{2\ eq}$	Normal: μ=-275; σ=11.5
	EthP	25	kg $CO_{2\ eq}$	Normal: μ=25; σ=4.94
	WPG	-1.07E3	kg $CO_{2\ eq}$	Normal: μ=-1.07E3; σ=37.3
	WPC	-775	kg $CO_{2\ eq}$	Normal: μ=-775; σ=29.6

Figure 5.4 shows the cumulative frequency of the LCCs for all options. As evident in Figure 5.4, the WCG and EthP options continue to be the leading options on the economic criteria, with 95% likelihood that the life cycle cost will be less than $399/FU and $427/FU, respectively. The worst performing option is the WPG with 95% likelihood that the life cycle cost will be greater than the other options. The differences between the remaining alternatives were not as pronounced. Nevertheless, there is 15% chance that WPC and PyLT switched positions, while PyEl maintained the third position.

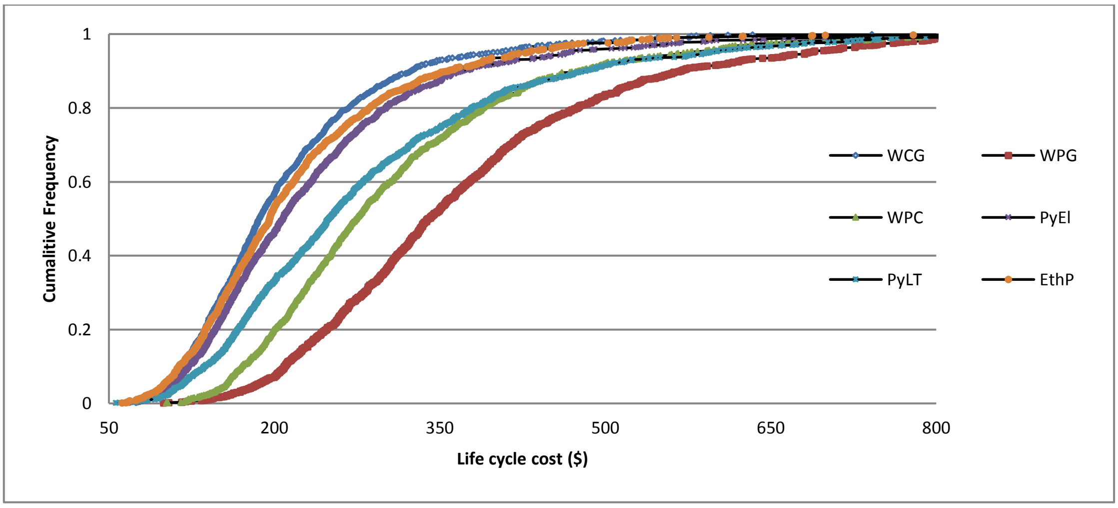


Figure 5.4 Monte Carlo simulation results in LCC

5.4.3 Limitations

This study was limited to the goal, scope and system boundaries used in the LCA and LCC. The simulation model was conducted based on the South-east Queensland region. Availability and accuracy of collected data, published data and publically available databases may also impose limitations. In addition, the life cycle inventories completed in this study for different bioenergy products were assumed to represent the typical or average products on the market. However, the actual bioenergy production impacts from both environmental and economic perspectives may change depending on the technology, scale and location. Furthermore, this study assumed the system is in steady state and all carbon emissions released from the burning of the harvested biomass will be taken up by the next generation of the plantation. However, if carbon uptake rate of the next generation was slower than the rate at which being released, then this may alter the results because some of these emissions will now count towards the global warming potential.

5.4.4 The Implication in Global Energy Sector

Biomass is an important renewable energy source; it plays an essential role in the strategy to mitigate climate change impacts arising from the energy sector. In 2010, woody biomass supplied 9% of annual global primary energy, and it is expected to satisfy 18% of the world's primary energy requirements in 2050 (Lauri et al., 2014), hence significantly reducing global reliance on fossil fuels. This study demonstrated a novel method which can be extended to other regions to evaluate the environmental and economic impacts of different bio-energy conversion scenarios from locally available low-value sources.

Globally, bioenergy production from renewable biomass resource has been accelerated by the enabling environment and biofuel policies (de Wit et al., 2013). In the European Union (EU), the total primary energy production from biomass has increased by 53% over the period between 2005 and 2010, and it is expected to double by 2020 (de Wit et al., 2013). However, due to limited regional resources, the woody biomass feedstocks are mainly imported from the neighbouring countries, Russia and North America (Moiseyev et al., 2014). Hence, the production of bio-energy will become relatively more expensive due to the higher transportation cost and will entail higher transportation emissions which may counter its intended benefits as climate mitigation strategy. Additionally, sensitivity analysis has shown that increased transportation distances will have a particular effect on the AP, EP and HTP impacts. Furthermore, the growing demand for feedstocks imports to the EU region may lead to direct and indirect land use change, which could offset the positive impacts from substituting fossil fuel energy. This is especially so, if the feedstock was purposely planted for energy displacing food crops. This study showed that carbon price did not have a significant effect on the life-cycle cost of the assessed bio-energy options which make the results globally relevant in the absence of universal climate change policy.

5.5 Conclusions

This study compared six bioenergy conversion scenarios using life cycle approach. The result of this study showed that, after considering the uncertainties in the bio-energy production systems, woodchips gasification (WCG) had the best environmental, life cycle cost performance and energy return. Its high energy conversion rate, low energy consumption, and the relatively low capital and O&M costs contributed significant savings to both the economic and environmental performance. The extra energy requirements during manufacturing and transportation processes outweighed the environmental benefits gained from the improved energy density of wood pellets (WPC and WPG) and led to higher life-cycle cost, lower energy return and lower environmental benefits. Pyrolysis for liquid transportation fuels (PyLT) ranked fourth in the environmental criteria and fifth in the economic criteria. At the same time, pyrolysis for electricity generation (PyEl), although avoided the secondary upgrading process of the bio-oil, led to higher environmental burdens due to its low energy conversion rate. Ethanol production for transportation fuel blends (EthP) has shown high environmental impact. Monte Carlo analysis indicated that when the combined effects of uncertainties in the system were considered, the overall rankings of options were not significantly affected. The results highlighted the difficulty in ranking the options due to the conflicting performance of the alternatives under the different impact categories. To achieve an objective ranking, we recommend that proper valuation of different impact categories should be conducted in order to internalise the LCA cost into the LCC model; alternatively, proper weighting for each impact category should be developed in order to achieve complete ranking.

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Appendix

Appendix 1. Reviewed related literatures

Authors & Year	**Assessed Bio-energy applications**	**Functional Unit**	**System Boundaries**	**Types of LCA**
Peters et al. (2015)	Bio-char from biomass	1 ha of agricultural area	Includes all processes from agricultural production until the final product substitution.	aLCA and only applied cLCA in biochar application
Iribarren et al. (2012)	Gasoline and diesel	1 t of biofuel products	From the feedstock plantation to the blending facility	aLCA
Dornburg and Faaij (2001)	Heat and/or power	Fossil primary energy savings	Raw material extraction to bioenergy applications	cLCA
Kalinci et al. (2012)	Hydrogen production from biomass gasification	1 MJ/s of hydrogen produced	From feedstock extraction to hydrogen production	aLCA
Pa et al. (2011)	Heat	Per unit of energy produced on a yearly basis (MJ/year)	From raw materials extraction to heat production	aLCA
Lardon et al. (2009)	Biodiesel production from microalgae	The LCA is the combustion of 1 MJ of fuel in a diesel engine	From the extraction and production of raw materials, to the use of product in the engine.	aLCA
Singh et al. (2010)	Ethanol production	Distance travelled (km)	From the raw materials extraction to final use of energy product	aLCA with a discussion of future trend
Kim and Dale (2005)	Biofuels	1 ha of arable land producing biomass for biofuels	From the raw materials extraction to final use of bio-fuel product	aLCA

Stichnothe and Azapagic (2009)	Bioethanol production	Two FUs were used: i) treatment of 190,000 t of MSW/year ii) MJ of fuel equivalent	From feedstock extraction to final production	aLCA
Campbell et al. (2011)	Biodiesel production from microalgae	Enough fuel in an articulated truck to transport one tonne of freight one kilometre (tkm)	From the raw materials extraction to final use of bio-fuel product	aLCA
Collet et al. (2011)	Biogas production (methane from algae)	1 MJ produced by combustion in an internal combustion engine	From feedstock production to final use of bio-energy product	aLCA
Cherubini and Ulgiati (2010)	Bioethanol, bioenergy and biochemicals	The amount of agricultural residues treated per year by each biorefinery system	From the raw materials extraction to final product application	cLCA
Hill et al. (2006)	Biodiesel and ethanol biofuels	Per unit of energy gained by producing the biofuel	From the raw materials extraction to final product produced	aLCA
Esteban et al. (2014)	Heating	kW hour of thermal energy generated by the boiler	From the forest chips production to the use for residential heating boilers.	aLCA
Guest et al. (2011)	Heat and electricity	1 MJ of electricity and 1 MJ of district heating delivered to the end user (two functional units)	From forest waste extraction to final bioenergy application (electricity and heat)	aLCA
González-García et al. (2012)	Ethanol production	Two FUs were used: i) 1 kg of ethanol ii) 1 km distance driven	From the raw materials extraction to final product application	aLCA

Appendix 2. Average fuel and materials consumptions of hardwood plantation per m^3 in QLD region

Inputs	**Unit per m^3**	**Quantity**
Energy		
Diesel	L	6.8
Aviation fuel	L	0.02
Direct energy	MJ	263
Indirect energy	MJ	191
Materials		
Lubricant	L	0.12
Steel	kg	0.08
Gravel	kg	640
Fertiliser		
N	kg	0.04
P	kg	0.11
K	kg	0.08
Herbicide		
Glyphosate	g	11.4
Simazine	g	19.2
Triclopyr	g	1.4

CHAPTER 6 – RESEARCH ARTICLE 4

STATEMENT OF CONTRIBUTION TO CO-AUTHORED PUBLISHED PAPER

Chapter 6 includes a pre-print of a co-authored paper titled "A comparative life cycle study of alternative materials for Australian multi-storey apartment building frame constructions: Environmental and economic perspective" which was published in the Journal of Cleaner Production. My estimated contribution to the paper is 55%. I contributed to research design, data collection, modelling, analysis, and writing the manuscript.

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Supervisor:

A Comparative Life Cycle Study of Alternative Materials for Australian Multi-Storey Apartment Building Frame Constructions: Environmental and Economic Perspective

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Abstract

The building construction sector contributes to a quarter of the total Australian Greenhouse gas emissions. These emissions are mainly attributed to the use of energy-intensive materials. To achieve better environmental benefits and cost saving, the utilisation of wood-based construction materials is currently attracting attention. However, the manufacturing of engineered wood products consumes large quantities of chemicals and energy, which may have adverse environmental impacts. Therefore, a life cycle study was conducted to compare various materials for constructing the structural frame of a 4-storey apartment building compliant with the Australian building codes. Five alternatives were assessed: Laminated Veneer Lumber (LVL) manufactured from early to mid-rotation hardwood plantation logs (LVL_m), LVL manufactured from mature hardwood plantations (LVL_h), LVL manufactured from mature softwood plantations (LVL_s), concrete and steel. The functional unit was defined as the whole building structural frame. Global Warming Potential (GWP), Acidification, Eutrophication, Fossil Depletion, Human-toxicity Potential (HTP) and Life Cycle Cost (LCC) were evaluated. The LVL generally performed better than concrete and steel structural products. Particularly, LVL_m had the lowest GWP (2.84E4±233kg-$CO_{2\text{-eq}}$) and LCC ($128,855±2,797), which were less than a quarter of the concrete option. However, the usage of chemical preservatives and phenol-formaldehyde adhesive during the LVL production and treatment caused the HTP impact to be higher than the steel option. Monte Carlo Analysis showed that while the LVL options presented a higher sensitivity to the combined uncertainties, the overall ranking of the five options remained the same. Therefore, the inclusion of wood-based material in structural elements may significantly contribute to reducing the environmental impacts and the LCC of the construction sector.

Key Words:

Laminated Veneer Lumber; Concrete; Steel; Life Cycle Assessment; Life Cycle Costing; Sustainable Constructions

6.1 Introduction

The Australian housing market, in particular, the mid-rise apartment buildings, is witnessing a boom due to a sustained population growth and overseas investments combined with historically low-interest rates (Australian Construction Industry Forum (ACIF), 2014). A recent research pointed out that the annual growth rate in the market of multi-level apartment buildings reached 14.5% in 2015 and further increased by 4.1% in 2016 (Australian Construction Industry Forum (ACIF), 2014). This dramatic growth is associated with a proportional increase in the environmental impacts of the sector, with the majority of emissions being attributed to the use of energy-intensive materials such as concrete and steel (Cabeza et al., 2014). On the other hand, wood and wood-based structural products are promoted as sustainable and renewable building construction materials (Bribián et al., 2011) and their use has been increasing steadily over the past few decades (Wang et al., 2014). This trend is now further supported by the new Australian building code (National Construction Code Series 2016) which allows timber buildings of up to 25 m high to be designed under the 'Deemed to satisfy' provisions (Australian Building Codes Board (ABCB), 2016). With this drastic change, the use of wood or wood-based materials in mid-rise constructions is expected to significantly increase in the near future. However, an increase in wood uses can only be justified if there is a corresponding increase in the availability of long-term sustainably managed forests (Buchanan and Levine, 1999).

Excessive logging and unsustainable forestry practices may lead to deforestation, which is a major contributor to climate change (Chakravarty et al., 2012). More than 40% of the Australian native forests were lost since the last century due to excessive timber harvesting (Bradshaw, 2012). In response to growing environmental concerns about the sustainability of forestry practices, timber plantations are currently being promoted as a long-term wood supply strategy in Australia (Bradshaw, 2012). Generally, only high-quality timber from plantations can be used to produce sawn timber for structural purposes. The low-quality logs are usually excluded due to their high proportion of natural defects (e.g. knots, grain deviation, gum veins, etc.) and reduced mechanical properties (McGavin et al., 2006a). Additionally, the use of sawn timber usually has limitations, such as the unavailability of large timber sections that meet the spans required for modern building components (McGavin et al., 2006a). Nevertheless, researchers such as McGavin et al. (2013), Gilbert et al. (2014b) and Gilbert et al. (2017) have shown that the aforementioned low-quality logs can be converted to high performing engineered wood structural sections. Engineered wood products are manufactured by bonding together wood boards, veneers, strands or flakes using adhesives to form panels or other shaped structural products (Lam, 2001, Barbu et al., 2017). By randomising wood defects, engineered wood structural products have less variability in their mechanical properties and usually higher strength than sawn timber (Barbu et al., 2017). Today, engineered wood products such as Laminated Veneer Lumber (LVL), Oriented Strand Lumber (OSL) and Cross-Laminated Timber (CLT) have gained popularity in the construction sector and are increasingly accepted as cost-competitive building materials (Mallo and Espinoza, 2015).

Wood or engineered-wood products are generally identified as the most sustainable structural materials and strongly recommended for substituting high energy intensive products in building constructions (Knowles et al., 2011, Wang et al., 2014). Increasing use of wood-based materials could reduce the net Greenhouse gas (GHG) emissions from the building and construction sector because of their relatively low energy requirement during the manufacturing stage as opposed to other building materials, such as concrete and steel (Gustavsson et al., 2006). In addition, wood substitution in long-life-span products results in accumulated carbon storage (Sathre and Gustavsson, 2009), which is seen as a potential climate mitigation strategy (Cabeza et al., 2014). An early study conducted by Buchanan and Levine (1999) indicated that for typical forms of building construction, wood buildings require significantly lower process energy and released less GHG emissions than buildings constructed from other materials such as brick, steel and concrete. They estimated that a 17% increase in wood usage in the New Zealand building industry could result in 1.5% reduction in the national total GHG emissions. Goverse et al. (2001) claimed that 50% reduction in CO_2 emissions could be achieved technically by utilising timber as a substitute to concrete in the main and secondary structural elements in houses.

Life Cycle Assessment (LCA) is commonly used to gain a better understanding of the environmental impacts of wood-based materials, when compared to other materials, in the construction industry. Börjesson and Gustavsson (2000) conducted a comparative LCA of both wood and concrete frames in a built multi-storey building and found that the primary energy used during the production of the concrete building was 60–80% higher than the wooden one. The better performance of the wood building was attributed to the use of forestry residues in the production, as well as using the wood waste for energy recovery at the end of life. Furthermore, Börjesson and Gustavsson (2000) claimed that the environmental impact of the wood frame building strongly depended on wood handling options during the final disposal stage. Perez et al. (2009) calculated the embodied energy and Global Warming Potential (GWP) of four alternative theoretical office building designs and found that engineered wood design had the best performance compared to concrete and steel buildings. Nevertheless, only looking at a single environmental indicator, such as GWP, is not sufficient (Robertson et al., 2012). Particularly, some researchers are concerned that chemical consumption (e.g. preservative and resin) in the manufacturing phase of engineered wood products may lead to various environmental impacts related to human health or ecological concerns (Nebel et al., 2006, Rivela et al., 2007, González-García et al., 2009). The use of multiple indicators is then required to better inform the environmental decision-making process. Hence, Robertson et al. (2012) conducted an LCA study of a typical mid-rise office building in North America by including 11 environmental impact indicators such as GWP, Ozone depletion, human health, ecological toxicity, and acidification. Results showed that the laminated timber building had the lowest environmental impact in 10 out of the 11 impact categories. However, the embodied energy was almost identical to that of the concrete option, mainly due to a heavy timber-frame design and associated use of adhesive resins during the manufacture frame elements. However, Perez-Garcia et al. (2007) argued that the substitution of sawn wood joists by engineered I-joists in residential homes has very little effect on the environmental performance indices as the use of resins and energy in the latter product was offset by its greater material efficiency. Their results also showed that wooden houses have outstanding performances when compared to concrete and steel houses on embodied energy, GWP, air emission index and water emission index. More recently, Bolin and Smith (2011) also indicated that although borate-treated lumber structural frame consumed chemical preservatives, the cradle-to-grave life cycle impacts of borate-treated lumber frames were still less than that of galvanised steel frames. Their results also indicated that the lumber production and preservative treatment were not the main impactors. Similar to the findings in Börjesson and Gustavsson (2000), Bolin and Smith (2011) found that the disposal stage significantly contributed to the environmental impact in the borate-treated lumber structural frame option, particularly on GHG emission, fossil fuel use, acid rain potential and ecological impact. These impacts are attributed to the landfill construction, carbon release during wood decomposition and associated transportation process.

The economic performance of building constructions from different materials also received considerable attention (Dakwale et al., 2011). Life Cycle Costing (LCC) analysis is usually integrated with an LCA to evaluate the overall performance of the building sector. For instance, Islam et al. (2014) combined LCA and LCC analyses to evaluate the influence of alternative wall assemblages in a typical Australian double-storey townhouse. The system boundaries included the building construction, maintenance and replacement operations, and final disposal stages. However, the forestry phase was excluded in the study. In the waste disposal stage, the aged timber materials were assumed to be landfilled, without considering alternative options such as energy recovery and recycling. Nässén et al. (2012) compared the net present cost of concrete and wood buildings on the total material, energy and carbon dioxide costs. Their results showed that the difference in material costs between the two options was small; hence it was unclear whether wood buildings would be a more cost-effective option. However, their LCC study did not include the cost of labour, construction, maintenance and demolition. Furthermore, this study was conducted in 2012 and may not represent the current economic performance. Kim et al. (2013) conducted an LCA and LCC study to quantitatively compare the environmental burden and construction cost of two conventional types of structural frame by focusing on the construction stage. The cost of the CO_2 emissions was accounted. Their results indicated that the LCC of concrete buildings was about 10% lower than the one of steel buildings. Timber and engineered-wood structural products were not considered. More recently, Hossaini et al. (2015) integrated both LCA and LCC to systematically assess the overall performance of concrete and wood mid-rise building constructions in Canada. The LCC of the buildings included the design, construction, operation, maintenance, demolition and final waste management. Their case study results showed that the timber building had slightly lower LCC when compared to the alternative. Hossaini et al. (2015) further claimed that the environmental and economic performances were highly dependent on the service life operational energy, rather than on the structural materials.

In summary, most of the previous studies focused on either environmental or economic analysis, while only limited studies attempted to integrate both LCA with LCC. Wood-based structural products are commonly identified in the literature as an environmentally competitive building material, particularly due to their lower GHG emissions and fossil fuel energy saving. However, chemical consumption (i.e. preservative and resin) during the manufacturing of engineered-wood products as well as their end-of-life treatment were usually pointed out as the major contributors to the environmental impact. In addition, the use of different building structural solutions may also affect the environmental performance of the building. For instance, researchers who used heavy timber structural solutions claimed the fossil depletion potential was higher than conventional buildings (Robertson et al., 2012). On the other hand, the economic performance of using wood-based products in building constructions was reported to be only marginally better than the conventional materials. However, these results were based on incomplete LCC assessment and some studies ignored important factors such as labour cost, demolition, disposal and potential revenue from recycling and energy recovery. Furthermore, none of the previous studies included the potential revenue from the GHG emissions offset due to recycling or energy recovery. Additionally, the previous comparative life cycle studies usually considered wood materials from a single type (e.g. hardwood or softwood) and many did not include the plantation stage. Moreover, none of the previous studies included a comparison of engineered wood products manufactured from low commercial value thinned logs and mature logs.

This paper aims to bridge some of the aforementioned gaps in the literature by quantitatively assessing the environmental and economic impacts of using engineered wood structural products from low-grade timber as alternative structural products in Australian residential apartment building constructions. The result will be compared to the most commonly used materials for this type of constructions: concrete and steel. Three types of LVL manufactured from (i) hardwood plantation thinned trees, (ii) hardwood plantation final harvesting and (iii) softwood plantations were considered. Life cycle assessment and costing methods are applied to evaluate the impacts of material selection for the structural frame of a case-study multi-storey residential building compliant with the Australian building codes.

The outcomes of this research have implications for both the forestry and building industries. Results of this study can be potentially used to support residential building designers and builders in making informed choices on the environmental and economic impacts of the selected construction materials. Additionally, this can enhance the economic performance of the forestry sector by developing a new market for the low-value plantation thinned timber logs from plantation early stages.

6.2 Case Study

Australia has ratified international accords to reduce the national GHG emissions. In this regard, the Australian Government has set a target to reduce GHG emissions by 26-28 percent from 2005 levels by 2030 (Australian Government Department of the Environment and Energy, 2015a). Australian building construction sector is seen as the major GHG emissions contributor, responsible for approximately a quarter of the national carbon emissions (Basaglia et al., 2015). The majority of these emissions are attributed to the residential building constructions (Basaglia et al., 2015). Hence, this sector offers significant potential for reducing GHG emissions.

In 2016, the number of annual dwelling approvals in Queensland was reported as approximately 3,500/annual (Australian Bureau of Statistics, 2017). The apartment building is popular and covers approximately 40% of the new dwellings in Queensland (Citi/ABS, 2016). Therefore, a four-storey residential apartment building was used in this paper as a case study. The building is assumed to be built in Brisbane, the Queensland state capital city, Australia. The building was designed by a Chartered Engineer following the relevant Australian standards: AS 1720.1 2010 [2] in reference to AS/NZS 4357.0:2005 (R2016) [3]; AS3600 (2009) [4] and AS4100 (1998) [5]. In this study the following scenarios are assessed: a concrete frame; steel frame and LVL frame. The LVL scenario includes three sub-scenarios: (i) sections manufactured from mid-rotation hardwood plantation thinned trees, henceforth referred to as LVL_m, (i) sections manufactured

[2] AS 1720.1 2010 - Timber Structures

[3] AS/NZS 4357.0:2005 (R2016) - Structural Laminated Veneer Lumber - Specifications

[4] AS3600 (2009) - Concrete Structures

[5] AS4100 (1998) - Steel Structures

from the final harvest of hardwood plantations (mature trees), henceforth referred to as LVL_h, and (iii) sections manufactured from mature softwood plantations, henceforth referred to as LVL_s.

Sketches of the building plan and structural frame are shown in Figure 6.1. The building components considered in this study were limited to the structural columns, beams and connections such as base plates, connecting plates and bolts for the steel and LVL frames. All other building elements (wall, floor, roof, etc.) were assumed to be the same in all scenarios. The weight of the structural frame in all designs differs considerably, especially for the concrete solution. However, the weight of the frame of each solution, when compared to the overall weight (by adding all building components, such as common slabs, walls and roof) and design live load, is relatively small (approximately 10%). Therefore, we assumed that the foundations are the same for all designs. As such, foundations were excluded from the system boundaries.

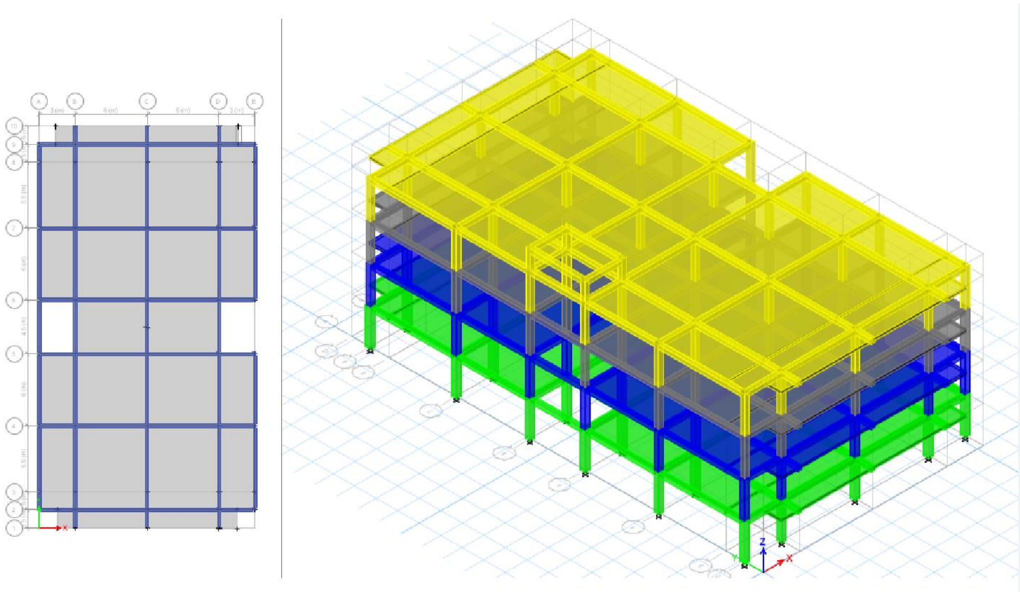


Figure 6.1 Four stories residential apartment building structure and frame design

The specific dimensions of columns and beams for each solution are presented in Appendix A - Summary report of design. Appendix B details the building frame, including both column and beam designs. In reference to Figure 6.1, the land area was calculated as 603 m^2 (i.e. 33.5 m × 18 m) and the building is 13.5 m in height. The material consumptions were calculated according to the structural dimensions. For the hardwood LVL options (both LVL_m and LVL_h), the total amount of green logs required was estimated as 70 tonnes (t). Meanwhile, LVL_s option only requires 54 t of green logs because of the lower density of softwood trees. The steel required for the connections was estimated as 3.25 t in all three LVL options. In the case of the concrete frame option, the total concrete required was calculated as 267.4 t, with 20.8 t of steel for reinforcement.

On the other hand, the total steel consumption was calculated as only 38.72 t in the steel frame building including all connections.

6.3 Methods

To conduct the life cycle exercise, the frame of the building was designed according to the relevant Australian code. Quantity surveying was then used to calculate the total amount of each material used in each scenario. The materials and energy consumptions were then inputted into the software to assess the life cycle emissions. The LCC of each option was then calculated and compared as discussed in the following sections.

6.3.1 Environmental and Economic Assessment

6.3.1.1 LCA

The LCA was conducted using SimaPro v.8.0.4.30 modelling software, which is based on the ISO14040 (2006). Life Cycle Inventories (LCI) of the manufacturing of the three LVL products were based on primary data, while other data were collected from the relevant literature and LCI databases. GWP, Eutrophication Potential (EP), Acidification Potential (AP), Fossil Depletion Potential (FDP) and Human-Toxicity Potential (HTP) were evaluated in this study (Grant and Peters, 2008). ReCiPe Midpoint (H) life cycle impact assessment (LCIA) method was used for the characterisation of emissions into the relevant impact categories except for the AP category for which the CML 2001 method was used (Bengtsson and Howard, 2010b, El Hanandeh, 2015)

6.3.1.2 LCC

The LCC was carried following the AS/NZS 4536:1999 (R2014), which takes into account all the relevant costs during the entire life stages. Furthermore, the LCC arising from environmental perspectives, such as carbon emission costs, was also included as suggested by Lu and El Hanandeh (2016a). All the costs have been converted to the 2016 Australian dollar Present Value (PV). The LCCs of each scenario was calculated by an Excel spreadsheet model. The main economic assumptions and parameters used in the LCC are summarised in Table 6.1.

Table 6.1 Economic parameters main economic assumptions and parameters for LCC analysis *

Parameters	Value (2016 PV)	Sources

General inflation rate	3%	Reserve Bank of Australia (2015)
Discount rate	4.9%	Reserve Bank of Australia (2015)
Electricity cost	22.24 ¢/kWh	Queensland Government Department of Energy and Water Supply (2015)
Carbon Price	\$29/t CO_{2}-$_{eq}$	The Treasury Australian Government (2016)
Transport cost	0.098\$/tkm	Higgins et al. (2015)

* All the prices are expressed in constant dollar terms and discounted to the 2016 financial year.

6.3.2. Functional Unit and System Boundaries

The functional unit of this study was referenced as the entire structural frame of the case-study residential apartment building. The study provides a ‘cradle-to-grave’ LCA of the LVL, concrete and steel structural members only based on resources from South-east Queensland. The system boundaries consider the following life-cycle stages: raw material extraction, product manufacturing, construction, maintenance, demolition, as well as end-of-life treatment. However, operation phase was excluded, since this study solely focused on the structural frame, while all other elements remaining the same. Thus, we assumed that different frame materials would not affect the thermal or operational performance of the building.

In case of the LVL scenarios, the system boundaries begin from timber plantation and end with harvesting, second thinning or final harvesting depending on the products. The logs were then transported to the mill to produce LVL sections for building constructions. The structural products were transported to the building construction site using road truck. During the end-of-life stage, the most environmentally feasible option was selected. For LVL options, incineration with energy recovery was chosen (Lu and El Hanandeh, 2016b). In case of the steel frame, steel was assumed to be 100% recycled. Concrete was assumed to be sent to landfill with 85% of the reinforcement steel bars recovered and recycled (Buchanan et al., 2012). The service life of the frame was set to sixty (60) years based on Australian average building service life (Aye et al., 2012).

6.3.3 LCI Analysis

This section analyses the LCI data of each stage of the products. The energy usage and raw material consumption, final products, waste, and emissions released were all included. The life cycle cost related to each scenario was also considered.

6.3.3.1 LCI of LVL Options

The LCI for the LVL options starts from the plantation stage, including seedling production, site preparation, tree planting, application of herbicides and fertiliser usage. The plantation stages end up to second thinning for the LVL_m and final harvesting for the LVL_h and LVL_s. Electricity used during greenhouse operations, diesel fuel and lubricants used to power equipment for a series of forestry machinery are also accounted. The outputs are the green thinned logs from mid-rotation thinned and mature hardwood and mature softwood logs from final harvesting.

Environmental Inventory

The LCI of plantation operations in South-east Queensland, Australia, was collected from published LCI from Australian forests and wood products (May et al., 2012). The hardwood logs from plantation thinning were treated as co-products of the forestry operations. Consequently, the environmental impacts were split between thinned and mature logs in the ratio of 1:30, based on the estimated market value of the products. The LCI of producing 1 m^3 of mature hardwood and softwood logs was collected from May et al. (2012). The quantity of fertiliser and herbicide consumption were also considered. Softwood plantations require relatively less energy and material input compared to hardwood plantations. The production of 1 m^3 of mature hardwood logs requires 45% more energy than of softwood (May et al., 2012)

The LVL product manufacturing process was sourced from Lu and El Hanandeh (2016a) and summarised in the system diagram shown in Figure 6.2. LCI data of LVL manufacturing from hardwood log was collected based on our primary data combined with recently published relevant literature (Bergman and Bowe, 2011a, Puettmann et al., 2013, Tucker et al., 2009), which represent common manufacturing practices from Australian wood industries. Meanwhile, the LCI of the production of the structural LVL_S (i.e. from mature softwood) was sourced from the Ecoinvent Database (Swiss Centre For Life Cycle Inventories, 2016). The drying process is an energy-intensive process that allows reducing the moisture content of the timber logs to 12% (Bergman and Bowe, 2011a). While wood fuel constitutes up to 80% of the total energy requirement, the remainder is sourced from natural gas and electricity. The dried veneers are then glued using Phenol-formaldehyde (PF) adhesive and pressed to form LVL products (Kline, 2007). Finally, the LVL products are sent to a finishing area for trimming and sanding.

The air-dried density of the hardwood and softwood LVL products is considered herein to be approximately 787 kg/m^3 and 606 kg/m^3, respectively. Wood wastes (i.e. veneer trims and sawdust) generated from the manufacturing stage were treated as feedstocks for energy substitution. The transportation process required to produce 1 m^3 of LVL products and deliver it to the construction site was also taken into account. The total transportation distance was estimated to be 250 km, from logs harvesting to the final disposal.

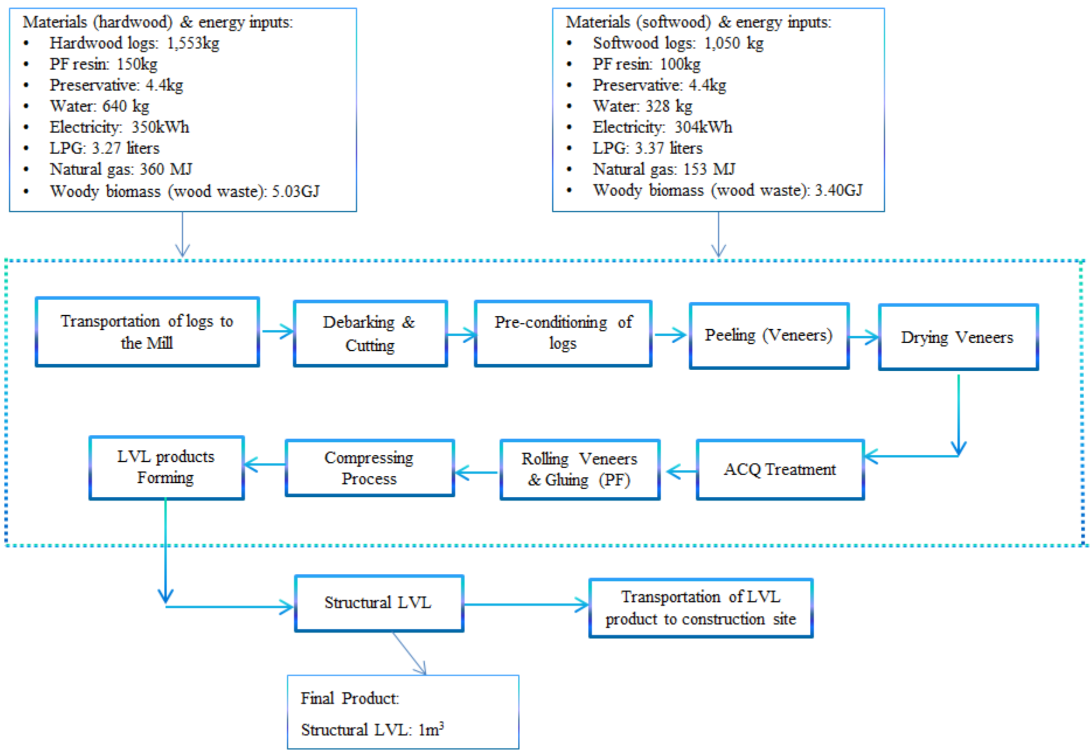


Figure 6.2 LCI of 1m^3 of hardwood or softwood LVL product during manufacturing stage

In addition, 3.25 tonnes of steel were used in the LVL frame options for connections. The life cycle energy inputs for manufacturing 1 tonne of the steel connections is 3,223 kWh of electricity, 10,284 MJ of natural gas, 7.4 L of diesel and 18.2 L of gasoline per tonne (Lu and El Hanandeh, 2016a). The steel products were assumed to be sourced and manufactured locally in Australia. Transport of steel from the extraction point to the building construction site includes a mixture of rail and truck. Transport of steel from the point of use to recycle sites was assumed by truck. The total transportation distance was estimated as 2,000km.

This study assumed that the LVL products will be combusted in a large cogeneration power plant at the end of the product life, and the produced electricity is distributed to the local main grid. The bottom ash produced from the wood waste incineration is assumed to be deposited in the local landfill. The LCI data of incinerated engineered wood products is adopted from Lu and El Hanandeh (2016b). Typically, 20% energy recovery efficiency is realised for electricity generation from wood waste (Lu and El Hanandeh, 2016a). The recovered energy is treated as a credit due to the avoidance of combusting virgin feedstock according to the current Queensland electricity grid mix. The CO_2 offset from steel recycling was also included as credit. The LCI data for scrap steel recycling was sourced from published literature (AusLCI, 2011, Yellishetty et al., 2011).

Economic Inventory

The life cycle cost of the wood frame apartment buildings included the cost of raw materials extraction (logs and steel), products manufacturing, construction, maintenance, demolition, transportation, and final disposal. The carbon offset from wood waste incineration and steel recycling were also included in this LCC. The LCC of 1 m^3 of mature hardwood logs from plantation was estimated at $151, including the forestry processing cost from seeding to final harvesting, as well as labour and land cost (Lu and El Hanandeh, 2016a). Meanwhile, the LCC of softwood plantation logs was calculated as $108/$m^3$ in present value based on the LCI data from May et al. (2012). The production cost of 1 m^3 of mature hardwood and softwood LVL were then calculated as $752 and $606, respectively, including labour cost, materials and energy usage cost (Lu and El Hanandeh, 2016a). The production of 1 m^3 of LVL from thinned logs was estimated to be $508/$m^3$. Furthermore, the labour costs during frame construction and demolition stages were estimated following the Rawlinsons Construction Cost Guide (Rawlinsons, 2016). The maintenance program for all scenarios was estimated at 7.5% of total LCC, the average maintenance cost in the building construction sector in Queensland (Islam et al., 2015a).

During the final disposal stage, the cost included both demolition and waste treatment. Wood waste incineration in Australia was estimated at $350 per tonne in present value (Lu and El Hanandeh, 2016a). The generated Bottom ash is disposed of in municipal waste landfills, which costs $90 per tonne in present value according to Construction and Demolition Waste Guide (2012). The energy recovery from the wood waste incineration and GHG offset for displacing electricity from fossil fuel was estimated to be $60 per tonne (Lu and El Hanandeh, 2016a). The carbon credit was calculated based on the LCA results by multiplying the quantity of $CO_{2\ eq}$ offset by the current carbon price (i.e. $29/tonne) (The Treasury Australian Government, 2016). Furthermore, the price of recycled scrap steel was approximate $290/tonne in 2016 present value (Hyder Consulting Encycle Consulting & Sustainable Resource Solutions, 2011). Hence, at the end of the 60-year service life, $97 would be paid back from recycling 1 tonne of steel and $8.6 from CO_2 offset (Yellishetty et al., 2011). The summary of cost for each of the LVL options is presented in Table 6.2. The construction cost includes material, energy, and labour cost during this stage.

Table 6.2 LCC details in three LVL building frame options

Cost & Revenue	LVL_m	LVL_h	LVL_s
Construction cost	$79,039	$100,755	$87,761
Maintaining cost	$8,995	$10,799	$9,726
Demolition & waste management	$36,539	$36,539	$34,668
Carbon cost	$864	$1,389	$1,018
Revenue (energy recovery)	$4,664	$4,664	$2,884
Revenue (GHG offset from electricity generation)	$492	$492	$267
Revenue (steel recycling)	$315	$315	$315
Revenue (GHG offset from steel recycling)	$28	$28	$28
Total LCC	$119,939	$143,983	$129,679

6.3.3.2 LCI of concrete option

The LCI data of concrete product manufacturing was sourced from AusLCI (2011). The energy input to manufacture 1 m^3 of concrete structural products requires 1077.28 kWh of electricity, 3320 MJ of natural gas, 8.16 L of diesel fuel and 3.52 L of gasoline (AusLCI, 2011). The materials required for 1 m^3 of concrete with steel reinforcement are 186.68 kg of steel, 410.24 L of water, 567.68 kg of cement as well as 1,646.88 kg of aggregate. The transportation distance was estimated to be 250 km in total. This includes the material extraction and manufacturing, transporting the products to the building construction site, and final disposal. Although concrete waste can be reused in road construction, this is not a common practice in Australia due to the low value of the recovered material and the difficulty in recycling concrete. As a result, most concrete waste is disposed of in landfills (Senaratne et al., 2016). Nevertheless, in our study, we assumed that 85% of the high-value reinforcement bars are recovered and recycled (Buchanan et al., 2012). The environmental impacts from landfill facilities construction and closure were assumed to be proportional to the mass of the disposed waste.

The production cost of 1 m^3 of reinforced concrete was estimated at $1,065, including material, energy consumption, equipment and labour cost (Lu and El Hanandeh, 2016a). In this study, the construction cost included both concrete production and on-site labour cost. The total carbon cost for the entire frame over 60 years life cycle was calculated as $5,336. Disposal of the waste concrete in landfill costs $8,131 in present value. Approximately, $1,712 in revenue is expected from recycling steel at the end-of-service-life. Additional $461 is expected as credit due to the

CO_2 emission offset from steel recycling. The total LCC of one functional unit of the concrete frame was then estimated at $477,102 in present value. The summarised LCCs of the concrete frame building option is presented in Table 6.3.

6.3.3.3 LCI of steel structural

The LCI data of steel structural products manufacturing was sourced from AusLCI and Eco-invent database (AusLCI, 2011, Ecoinvent, 2016). The energy required for 1 m^3 of steel products during manufacturing stage included 25,876 kWh of electricity, 82,273 MJ of natural gas, 59 L of diesel and 145.5 L of gasoline. At the end of the building life, the steel frame was assumed to be 100% recycled (Bolin and Smith, 2011).

The life cycle cost for producing structural steel products was estimated at $2,877.60/tonne, which included the cost of energy consumption, raw materials as well as labour and manufacturing facilities (Lu and El Hanandeh, 2016a). The total steel frame production cost in this scenario was calculated as $111,421 based on the material consumption. The on-site construction labour cost was also estimated and added to the total construction cost according to Rawlinsons (2016). The carbon cost was estimated at $4,147 based on LCA results. Furthermore, the demolition and waste management cost was estimated at $48,395. The current price of recycled scrap steel was approximate $290/tonne in 2016 present value (Hyder Consulting Encycle Consulting & Sustainable Resource Solutions, 2011). Hence $3,750 would be paid back from steel recycling, while the CO_2 offset contributing approximately $417 cost saving. Therefore, the total LCC of one FU of steel structure building frame was estimated at $219,216. The detailed life cycle costs of the steel frame option are presented in Table 6.3.

Table 6.3 Detailed LCCs of concrete and steel building frames in 60-year-lifespan

Items	Concrete Frame	Steel Frame
Construction cost	$370,581	$154,321
Maintenance cost	$36,116	$16,441
Demolition & waste management	$66,935	$48,395
Carbon cost	$5,336	$4,147
Revenue from recycling	$1,712	$3,750
Revenue from carbon offset	$154	$338
Total LCA	$477,102	$219,216

6.4 Results and Discussions

The LCA and LCC results are presented in Figure 6.3 and Figure 6.4 shows the details of contributions of the emissions to each impact factor of the five assessed scenarios. Concrete had the largest environmental burden in all impact categories. The steel frame presented the second highest environmental burdens, except for HTP. Overall, the LVL options were the best performers in all environmental impact categories, except for the HTP. Among the LVL options, LVL_m was the option with the lowest environmental burdens. Interestingly, the LCC values are in agreement with the environmental performance of the options, as shown in Figure 6.3 (f).

The concrete's poor performance was attributed to the intensive energy demand during concrete ingredients manufacturing, emissions during cement manufacturing process, as well as extensive materials consumption, thus requiring significant transportation. Figure 6.4 (d) indicates that the energy consumption during the manufacturing of the structural products contributes to approximately 50% of the total impact in all categories. The second major emission generator is the extraction of the raw materials, followed by transportation. In the case of the steel frame, steel extraction and production stages were the main contributors to the environmental impacts, while the emissions released during transportation were negligible, as shown in Figure 6.4 (e). However, the steel used for the building frame is recyclable, which brought significant environmental saving, such as carbon offset due to a reduced energy requirement of recycled steel compared to virgin steel. Our results showed that steel recycling contributed to approximately a 20% reduction in both GWP and EP, 30% and 10% offset to the AP and FDP impact, respectively.

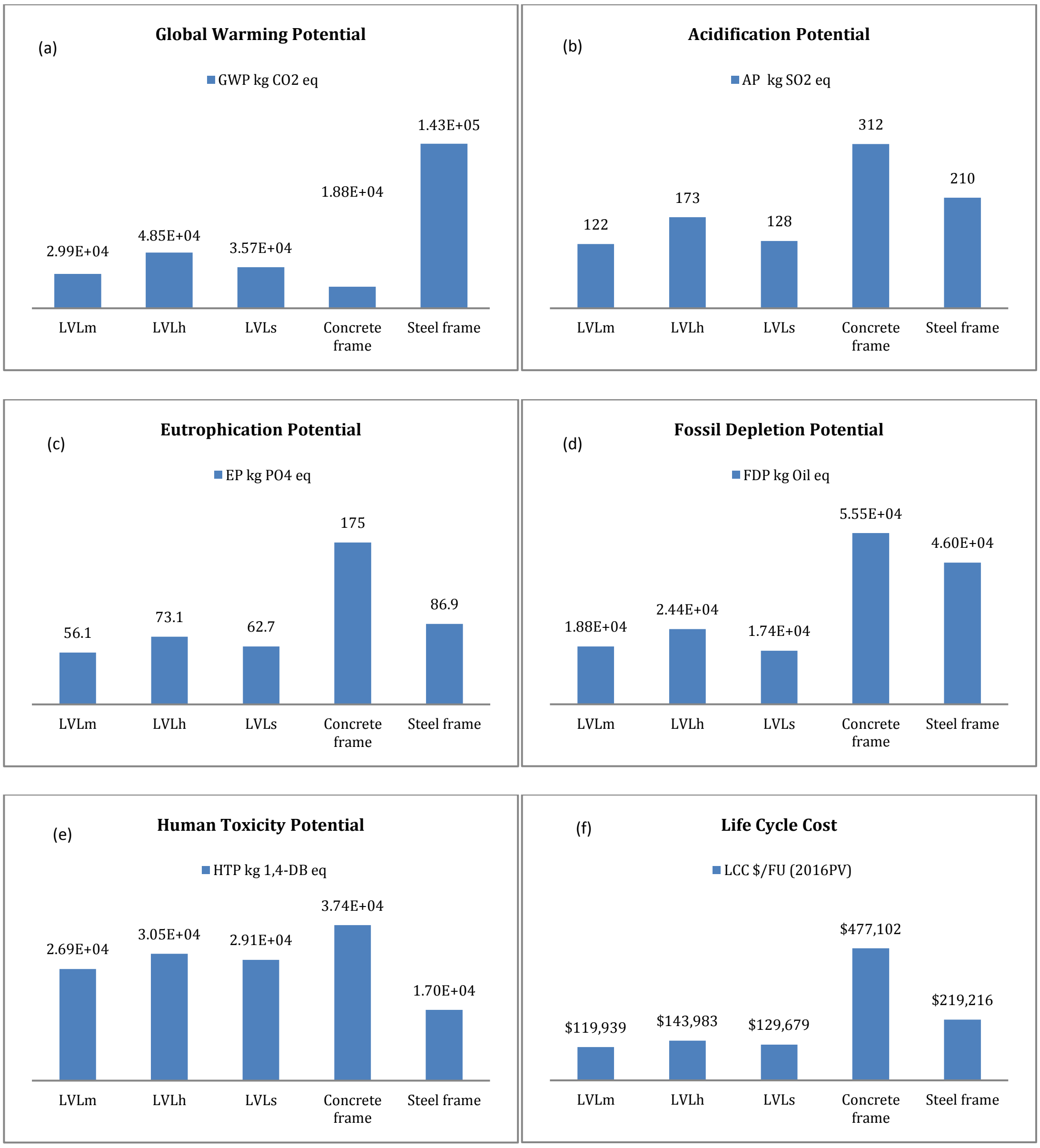


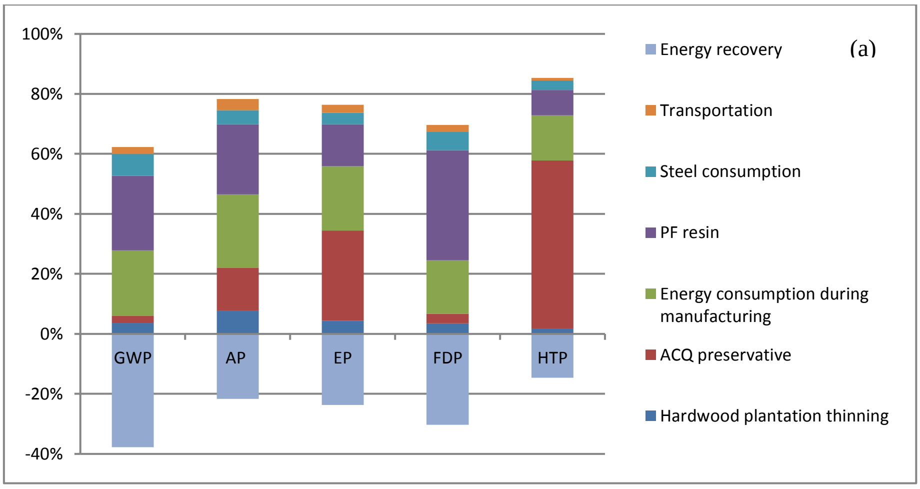
Figure 6.3 Life cycle results of building frame design in five scenarios (a) GWP, (b) AP, (c) EP, (d) FDP, (e) HTP and (f) LCC

The relatively high HTP impact of that LVL options was mainly attributed to the use of preservative during the manufacturing stage, which contributed to more than 50% of the total HTP in all LVL options. Additionally, the preservative consumption also contributed to about 15% and 25% to the AP and EP impacts, respectively. PF resin was another major environmental impact source. According to Figure 6.4 (a, b & c), the use of PF resin was responsible for 20% of the GWP and AP impacts, and more than 35% of the FDP impact. Although there was a significant amount of energy required for wood drying, the use of wood waste for energy production during the LVL manufacturing significantly reduced the environmental impacts associated with the energy demand. Thus, the total energy consumption in the LVL options only contributed to about

20% of the environmental impacts. The energy credit gained from wood waste incineration further helped to reduce the environmental burdens due to the avoidance of using fossil fuel for power generation. More than 30% reduction of GWP and 20% reductions of AP, EP, and FDP may potentially be realised during the energy recovery stage at the end of the LVL frame life.

Comparing the three LVL options, raw material sourced from low value thinned logs (i.e. LVL_m) presented relatively less environmental impact than that from mature hardwood logs due to lower emissions during the plantation stage (i.e. shorter harvesting cycle and allocation of a higher proportion of the emissions to the final harvest which has a higher value). Thus, the LVL_m option had significantly lower impacts compared to LVL_h option, particularly on GWP (reduced by 38%), AP (reduced by 30%) and EP (reduced by 23%). LVL_s option had better environmental performance than LVL_h, mainly due to emission saving during the plantation stage (shorter rotation) and lower PF resin consumption in the manufacturing stage. Comparing LVL_s to LVL_m, the lower emissions in the manufacturing phase of LVL_s (as a result of reduced use of resin and lower energy requirement) were overweighted by the emissions from the softwood plantation stage, thus leading to relatively higher overall environmental impacts, except for the FDP impact category.

The LCC results indicated that the concrete building frame ($477,102) was the most expensive option among all, followed by steel frame option ($219,216). The former price is due to the concrete option consuming large quantities of materials. Furthermore, the concrete frame required more processing and handling of heavier materials and elements, which contributed to higher cost of labour, maintenance, demolition, disposal, as well as carbon emissions. Although the material and manufacturing cost of producing 1 tonne of structural material is 2.5 times higher for steel than for concrete, the steel frame option required less material, which resulted in a lower material cost when compared to the concrete option. Additionally, the waste steel recycling contributed to higher revenue at the end of life stage. Among all options, the LVL frames were the most cost-effective options, particularly LVL_m. Due to the low cost of the hardwood thinned logs, the LVL_m option was about 20% and 10% cheaper than that of the LVL from mature hardwood and softwood, respectively.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80%
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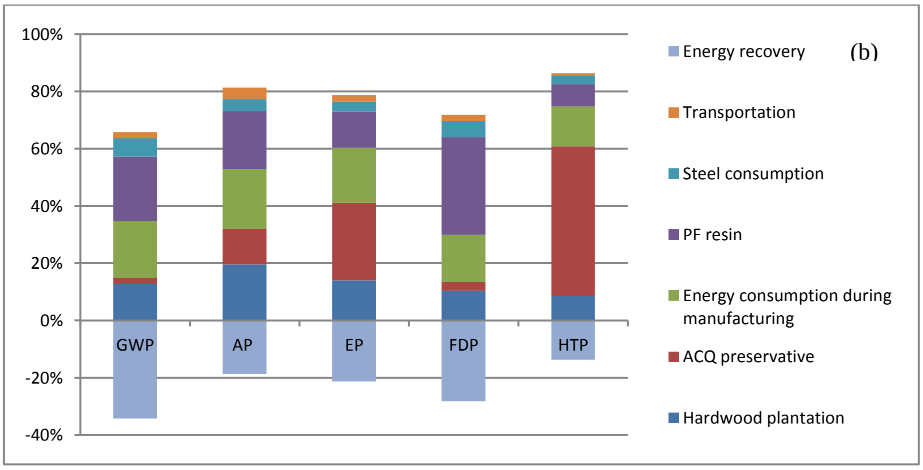
20%
0%
-20%
-40%
GWP
AP
EP
FDP
HTP
Energy recovery
Transportation
Steel consumption
PF resin
Energy consumption during manufacturing
ACQ preservative
Hardwood plantation thinning
(a)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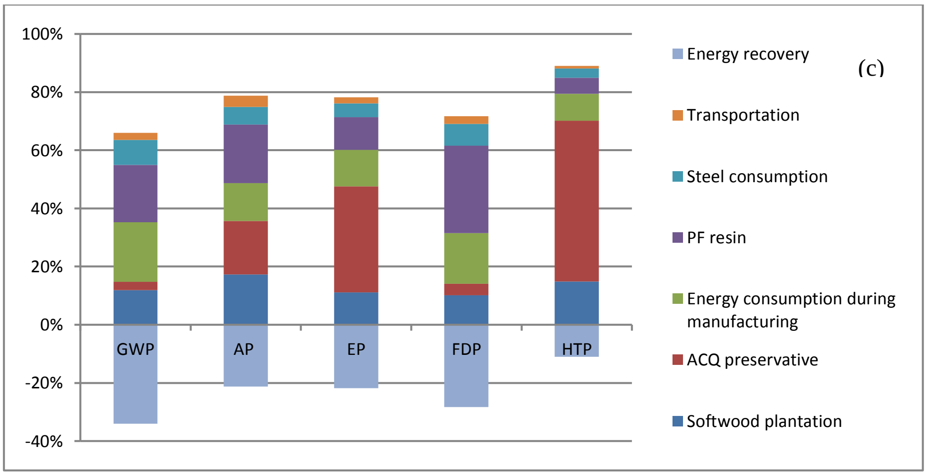
Energy consumption during manufacturing
ACQ preservative
Hardwood plantation
(b)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ACQ preservative
Softwood plantation
(c)

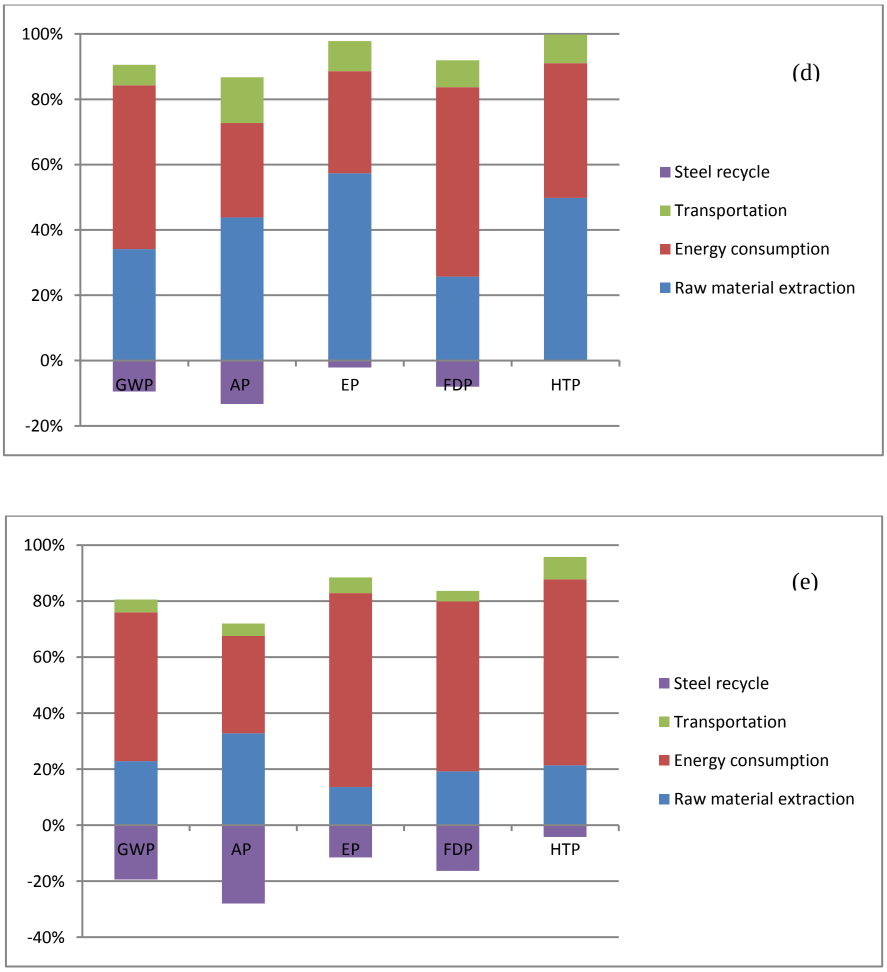

Figure 6.4 (a, b, c, d, e) **Life cycle environmental emissions sources in LVL_m, LVL_h, LVL_s, concrete and steel options**

It is worth to mention that as the results of this study highlighted the environmental and economic benefits of using low-grade wood (hardwood thinned logs) to manufacture value-added structural products, opportunities to use other low-grade wooden materials, such as pulpwood plantation logs, which are currently facing economic challenge due to market glut, may arise.

The result of this study is in general agreement with published literature with regards to use of resin and preservatives being major contributors to the overall impacts. However, this study highlighted that the greater material efficiency, use of low impact raw materials, as well as using wood fuel in the manufacturing process, contributed to the better environmental performance of the LVL options when compared to the steel and concrete one. Bolin and Smith (2011) reported that end of life stage (i.e. landfilling) was the most significant impactor in the case of wood. Our study found that the end of life stage impacts can be significantly reduced if the energy value of the wood was recovered. As a result, extra environmental credit can be gained by substituting the fossil fuel for electricity generation, while avoiding the significant amount of carbon emissions released during the wood decay in the landfill. Furthermore, this study highlighted the importance of including the plantation stage in the assessment as it can contribute up to 15% of the total environmental impacts in the case of products from matured plantations.

From an economic perspective, the study conducted by Hossaini et al. (2015) indicated that timber frames have slightly less LCC compared to the alternatives. Additionally, Nässén et al. (2012) mentioned that the differences in net present cost of concrete and wooden buildings on the total material, energy and carbon dioxide costs were small. However, our study showed that there were significant differences among engineered wood, concrete and steel building frame. These differences are attributed to different system boundaries and economic data. Hossaini et al. (2015) analysed different construction materials for a mid-rise building and indicated that the operation phase accounted for more than 90% to the total LCC of the building. This is likely due to the Canadian climate which requires higher heating demand to maintain thermal comfort. However, in their assessment, Hossaini et al. (2015) included the facade and other non-structural elements that are likely to affect the overall thermal and operational performance of the building. This operation phase was excluded in the system boundaries due to our study only focused on building frame. Therefore, the gaps among engineered wood, concrete and steel building frame options in other stages become more obvious. As such, the construction stage was identified as the highest LCC phase in all options. The LCCs of the LVL options during this stage were far less than that of concrete and steel options. Nässén et al. (2012) used the actual purchase price, which was much higher than concrete. On the other hand, our study found that the LCC of LVL was slightly less than concrete when compared on a mass basis. But more importantly, the LVL options had greater material efficiency compared to concrete. As such lesser amount of LVL is needed than concrete, which resulted in significant LCC saving.

6.4.1 Uncertainty and Sensitivity Analysis

The factors which may have introduced uncertainty in the outcomes, such as transportation distances, energy consumption, as well as the use of PF resin, were estimated using professional judgement. Sensitivity analyses were carried out to identify the effects of changing the previous uncertainty factors. A normal distribution with a coefficient of variation (CoV) of 20% was used to test the sensitivity of the results to transportation distances and energy consumption (Lu and El Hanandeh, 2016a). The results indicated that the uncertainties in transportation distances had an insignificant impact on all impact indicators and for all options (≤5%). Changes in energy consumption had the most impact on the steel frame option which was affected by ±18 to ±20% in all impact factors. All LVL options showed a similar response to the uncertainties in energy consumption, with the highest sensitivity being observed in the GWP (±18%) and FDP (±10%)

impact categories. On the other hand, the GWP and FDP of the concrete frame option only changed by ±14% and ±15%, respectively.

Furthermore, PF resin consumption is also a factor that significantly contributed to the environmental impact. The existing relevant literature show a wide variation on the quantities of resin usage, varying between 180g/m^2 to 360g/m^2 per bond-line, depending on the species of timber, its quality and the final product (Scott, 2005). However, the actual usage of PF resin for hardwood products is 450g/m^2 per bond-line based on our data. The effect of a 20% variation in PF resin usage was tested. As expected, the changes in PF usage could cause approximately 20% variation in both GWP and FDP in mature hardwood logs option, followed by AP (±9%) and EP (±7%). Meanwhile, the LVL frame from thinned logs option was relatively more sensitive to this variation, with the changes in GWP and FDP affected by ±29% and ±23%, respectively, followed by AP (±13%) and EP (±9%). The LVL$_s$ option was the least sensitive to this uncertainty, with the variations of ±10% in both GWP and FDP, while ±7 and ±4% in AP and EP, respectively.

6.4.2 Monte Carlo Simulations in LCA

The effect of combined uncertainties on the environmental performance was assessed by running 1,000 Monte Carlo Simulations in SimaPro v.8.0.4.30 software, which allows multiple uncertainty parameters to change simultaneously from pre-defined distributions, as described in Table 6.4. The parameters included were based on the uncertainty and sensitivity analyses in the previous section. Additionally, the percentage of renewable energy for Queensland electricity was also considered following Lu and El Hanandeh (2016a). The renewable energy contribution for electricity generation with a 25% variation was considered (Australian Energy Statistics, 2016).

Table 6.4 Parameters used in the Monte Carlo analysis

Parameter	Mode	Unit	Distribution type
Transportation distance	100	%	Normal: μ=100, σ=20
Energy consumption	100	%	Normal: μ=100, σ=20
Renewable energy content in electricity generation	12	%	Normal: μ=12, σ=3
ACQ retention in LVL product	4.4	kg/m^3	Triangle: min=2.4; max=6.4
PF consumption in LVL product	100	%	Triangle: min=80; max=120

Table 6.5 presents a summary of the Monte Carlo simulation results. The LVL building frame options are still to be the best environmental performers. However, judging by the CoV values,

their impact indicators had a high sensitivity to the combined uncertainties, particularly GWP, EP and HTP. The analyses also suggested that the LVL frame from thinned logs was more vulnerable to changes in input parameters than the LVL frame from mature logs, especially in the case of AP and HTP impact categories. However, there was still high likelihood that the AP and HTP would be less than the ones of the LVL produced from mature logs. In the concrete frame option, the effects from the combined uncertainties were generally insignificant; however, GWP and HTP were more affected by the overall uncertainties. Although the steel option was comparatively more affected by the uncertainties than the concrete frame option, its overall environmental performance was still better than concrete. The Monte Carlo simulations also indicated that the overall ranking of the five scenarios was not affected by the effect of combined uncertainties; with concrete being the least and LVL from thinned logs option being the most environmentally favourable choices.

Table 6.5 Summary of Monte Carlo simulation results in LCA

Options	Impact	Units/FU	Mean ± E*	SD*	97.5th percentile	CoV*
LVL_m	GWP	kg $CO_{2\ eq}$	2.84E4±233	3.76E3	3.53E4	13.4%
	AP	kg $SO_{2\ eq}$	118±0.53	8.59	136	7.3%
	EP	kg $PO_{4\ eq}$	54.7±0.42	6.75	71	12.3%
	FDP	kg Oil_{eq}	1.82E4±113	1.82E3	2.17E4	10%
	HTP	kg 1,4-DB_{eq}	2.68E4±172	2.77E3	3.29E4	10.3%
LVL_h	GWP	kg $CO_{2\ eq}$	4.57E4±245	3.95E3	5.37E4	8.64%
	AP	kg $SO_{2\ eq}$	168±0.55	8.95	186	5.32%
	EP	kg $PO_{4\ eq}$	71.2±0.39	6.26	85.7	8.79%
	FDP	kg Oil_{eq}	2.37E4±114	1.84E3	2.74E4	7.77%
	HTP	kg 1,4-DB_{eq}	3.01E4±177	2.86E3	3.62E4	9.51%
LVL_s	GWP	kg $CO_{2\ eq}$	3.72E4±206	3.33E3	4.37E4	8.95%
	AP	kg $SO_{2\ eq}$	128±0.47	7.52	143	5.86%
	EP	kg $PO_{4\ eq}$	64±0.37	5.95	76.8	9.3%
	FDP	kg Oil_{eq}	1.81E4±86	1.39E3	2.1E4	7.69%
	HTP	kg 1,4-DB_{eq}	2.94E4±188	3.04E3	3.47E4	10.4%
Concrete frame option	GWP	kg $CO_{2\ eq}$	1.57E5±410	6.61E3	1.7E5	4.21%
	AP	kg $SO_{2\ eq}$	291±0.51	8.29	307	2.85%
	EP	kg $PO_{4\ eq}$	159±0.27	4.29	167	2.7%
	FDP	kg Oil_{eq}	4.58E4±133	2.14E3	5E4	4.68%
	HTP	kg 1,4-DB_{eq}	3.33E4±51	820	3.5E4	2.46%
Steel frame option	GWP	kg $CO_{2\ eq}$	1.14E5±449	7.25E3	1.28E5	6.38%
	AP	kg $SO_{2\ eq}$	156±0.53	8.62	173	5.51%
	EP	kg $PO_{4\ eq}$	64.1±0.29	4.67	73.1	7.28%
	FDP	kg Oil_{eq}	3.65E4±136	2.19E3	4.06E4	6.01%
	HTP	kg 1,4-DB_{eq}	1.27E4±52	837	1.43E4	6.58%

*E: confidence interval (97.5%), SD: standard deviation, CoV: coefficient of variation

6.4.3 Uncertainty and Sensitivity Analysis in LCC

This section examined the effect of the uncertainties on the LCC model. The inflation rate, discount rate and carbon price were considered as the most uncertain factors (Lu and El Hanandeh, 2016a). The Australian inflation rate and discount rate fluctuated over the past decade (Reserve Bank of Australia, 2015). Sensitivity analyses were conducted to investigate the consequence of changing the inflation rate from 1% to 5% and discount rates from 3% to 7% (Lu and El Hanandeh, 2016a). Results indicated that an increase in the inflation rate by 2% could result in a 3-4% reduction in the LCC while a decrease by 2% could lead to 8-13% increase. The LVL_m was more sensitive to the overall uncertainties when compared to LVL_s and LVL_h options. The concrete option was the least sensitive option to the changes in the inflation rate. However, for the steel option, an increase or reduction in the inflation rate would cause the LCC to follow a different trend to the timber and concrete options due to the significant amount of revenue coming from the steel recycling which contributes in a high proportion of the savings in the LCC. Specifically, a reduction of the inflation rate from 3% to 2% would generate a 2% reduction in LCC. Meanwhile, if the inflation rate increases, the LCC of the steel option may be reduced by 8%.

Regarding the uncertainties on the discount rate, the LVL frame options, especially the LVL manufactured from thinned logs, were the most sensitive options to changes. The analysis showed that a change in the discount rate by -2% would increase of the LCCs by 8-11%, while an increase in discount rate by 2% would result in a 3-4% reduction in the LCCs of the LVL options. The concrete option was the least sensitive option to changes in the discount rate, which only varied from -4% to +6%. Similar to the results on the inflation rate, the steel option showed a different trend to the one observed for the timber and concrete options. The reduction in discount rate from the base scenario to 3% caused a 7% reduction in the total LCC while increasing the discount rate to 7% would increase the LCC by 3%.

The change in Australian GHG emissions policy will have a significant impact on carbon price (The Treasury Australian Government, 2016). Therefore, a variation of 20% in carbon price was selected to assess the sensitivity of the results to the carbon price. Changes in carbon price would cause negligible effects on the LCC in all scenarios (<2%).

6.4.4 Monte Carlo Simulations in LCC

One thousand (1,000) simulations were run to test the impact of the combined uncertainties on the overall LCC of the analysed options. The parameters tested are listed in Table 6.6. Virgin and scrap steel prices were determined based on the fluctuations in the global steel market price (SteelBenchmarker, 2016).

Table 6.6 Parameters used in the Monte Carlo analysis

Parameter	Mode		Unit	Distribution type
Inflation rate	3.0		%	Normal: μ=3.0, σ=2.0
Discount rate	4.9		%	Normal: μ=4.9, σ=2.0
On-site labour cost	100		%	Normal: μ=100, σ=20%
Maintenance cost	100		%	Normal: μ=100, σ=20%
Demolition cost	100		%	Normal: μ=100, σ=20%
Carbon Price	29		\$/tonne $CO_{2\ eq}$	Triangle: Min =23; Max=31
Virgin Steel Price	774		\$/tonne	Normal: μ=774, σ=230
Steel Scrap Price	290		\$/tonne	Normal: μ=290, σ= 90
GWP	LVL_m	2.84E4	kg $CO_{2\ eq}$	Normal: μ=2.84E4; σ=3.76E3
	LVL_h	4.57E4	kg $CO_{2\ eq}$	Normal: μ=4.57E4; σ=3.95E3
	LVL_s	3.72E4	kg $CO_{2\ eq}$	Normal: μ=3.72E4; σ=3.33E3
	Concrete frame option	1.57E5	kg $CO_{2\ eq}$	Normal: μ=1.57E5; σ=6.61E3
	Steel frame option	1.14E5	kg $CO_{2\ eq}$	Normal: μ=1.14E5; σ=7.25E3

The Monte Carlo simulations results are listed in table 6.7, while the cumulative frequency of LCCs for the five options are presented in Figure 6.5. Results indicated that the overall ranking of all options remained the same. Although LVL_m had the highest variation (i.e. CoV=35%), there is a 97.5% likelihood that the LCC would be less than $172,065, which is still significantly lower than all other options.

Table 6.7 Monte Carlo simulation results in LCC

Options	**LCC/(2016)**	**95% confidence interval (Mean±E)***	**SD***	**97.5th percentile***	**(CoV)***
LVL_m	AUD ($)	$128,855±2,797	$45,130	$172,065	35%
LVL_h	AUD ($)	$151,237±1,578	$25,463	$185,480	17%
LVL_s	AUD ($)	$140,531±2,346	$37,845	$182,732	27%
Concrete	AUD ($)	$498,698±4,660	$75,182	$573,164	15%
Steel	AUD ($)	$210,678±1,635	$26,373	$239,091	13%

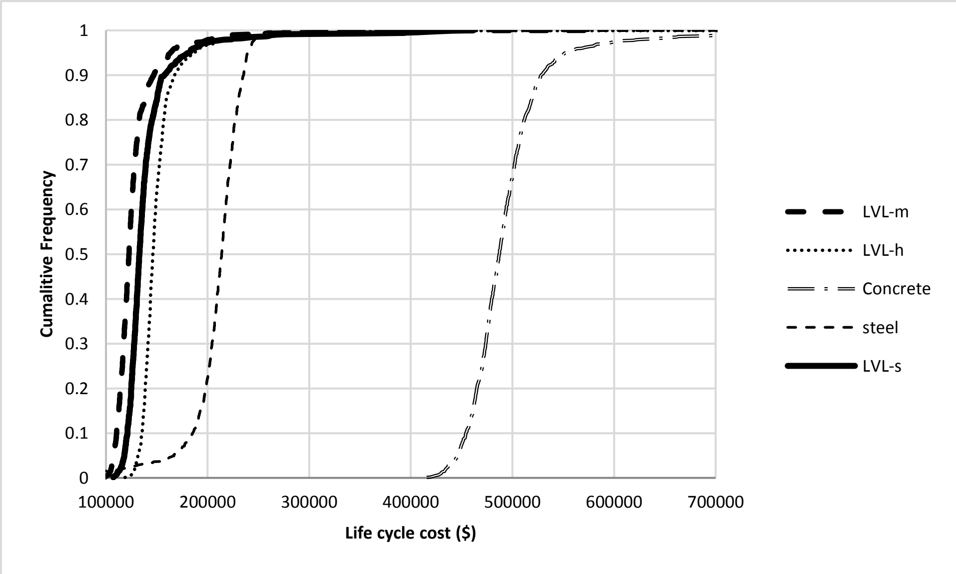

Figure 6.5 Monte Carlo simulation results in LCC

6.5 Conclusions

A representative four-storey frame residential apartment building, compliant with Australian codes, was assessed using LCA and LCC to evaluate the environmental and economic performance of using different building materials in the construction process. The different options for the main structure include concrete, steel and LVL products. LVL manufactured from hardwood plantation early to mid-rotation thinned logs, hardwood and softwood plantations final harvested logs were assessed. The LVL building frame options presented tangible benefits from both environmental and economic perspectives. Although chemical consumptions resulted in environmental burdens in all impact categories, the use of relatively low impact raw materials and wood fuel during the manufacturing process, as well as the environmental credits generated during energy recovery stage, significantly contributed to offset the negative impacts. From the economic perspective, the low construction cost of the LVL options, particularly labour and production costs, led to significant saving compared to the steel and concrete options. Among the three assessed LVL options, the LVL manufactured from plantation thinned logs had the lowest environmental impact and LCC. This resulted from the economic allocation of the emissions which attributes most of the emissions from the plantation stage to the final harvest. On the other hand, the concrete option was the least favourable option due to the extensive material and energy demands. The steel frame option presented much better environmental and economic performance than the concrete one due to lower material and energy consumptions, as well as the greater material efficiency that led to negligible transportation emissions. Furthermore, steel recycling helped to achieve a significant reduction in the overall environmental impact and life cycle cost. Monte Carlo simulations indicated that the overall ranking of the five options was not affected by the uncertainties in the parameters in the system. Therefore, LVL manufactured from hardwood plantation thinned logs is a value add-product which can be used to substitute conventional building materials. In summary, the use of low impact materials can mitigate the environmental impacts of the building construction sector and in turn, contribute to the Australian GHG emissions reduction target.

The scope of the study was limited to the boundaries established in the goal and scope of the life cycle study. The study focused on Australian mid-rise buildings, without considering other types of building such as residential houses and commercial buildings. The lifecycle analysis only focused on the building frame, while other elements (e.g. slabs, roofs and walls) were assumed to be the same for all options. As such, it was assumed that the frame material will not cause a significant difference in the operation of the building. The LCI relied on published or publically available information, which was assumed to be accurate. The manufacturing technologies, energy supply, life cycle cost of labour, construction, maintenance and demolition were based on the Australian case. The final disposal of engineered wood products was solely focused on energy recovery; other recycling scenarios were not evaluated.

Additional research is required to consider other building elements, the operation phase and different types of buildings including non-residential. In addition, an expansion of the current analysis by combining social impact is recommended to address the full scope of sustainable design and construction practices. Further research also requires to include other relevant environmental impact categories, such as solid waste, biodiversity loss, and land use impacts; hence to generate a more comprehensive environmental assessment.

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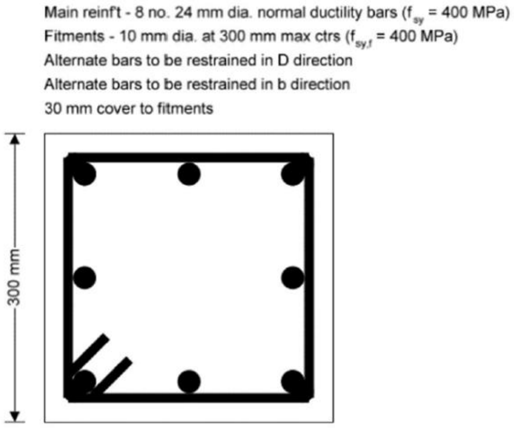
Appendix

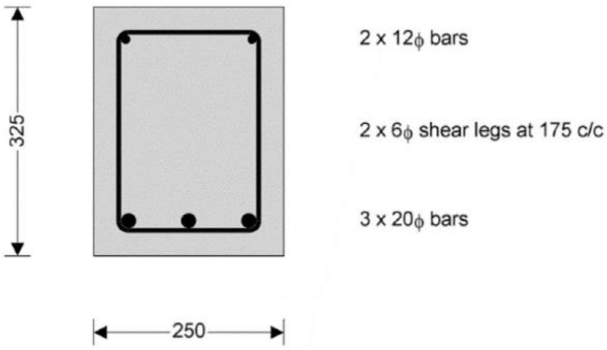
Appendix A - Summary Report of Design

Concrete Structure (dimension in mm)

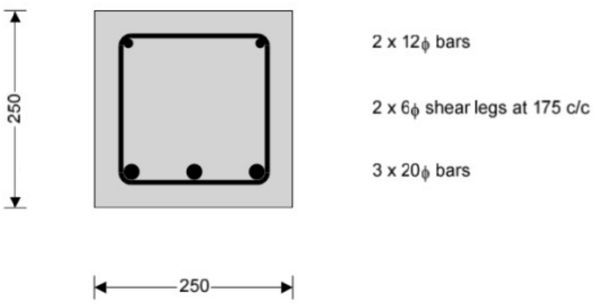
Columns



Beams – Long span (mm)



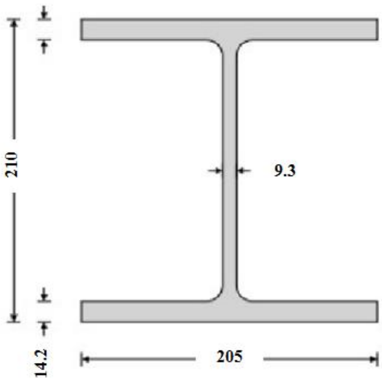
Beams – short span (mm):



Steel Structures:

Column:

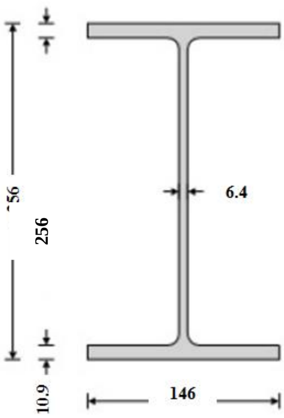
Section details	
Section type	**200x59.5 UC (AISC 1994)**
Steel grade	**350**
From table 2.1: Strengths of steels	
Thickness of material	$t = \max(t_f, t_w) = \mathbf{14.2}$ mm
Yield stress	$f_y = \mathbf{340}$ N/mm^2
Tensile strength	$f_u = \mathbf{480}$ N/mm^2
Modulus of elasticity	$E = \mathbf{200000}$ N/mm^2



Beam Long Span:

In accordance with AS4100-1998 incorporating Amendment No.1 2012

Section details	
Section type	**250x37.3 UB (AISC 1994)**
Steel grade	**350**
From table 2.1: Strengths of steels	
Thickness of material	$t = \max(t_f, t_w) = \mathbf{10.9}$ mm
Yield stress	$f_y = \mathbf{360}$ N/mm^2
Tensile strength	$f_u = \mathbf{480}$ N/mm^2
Modulus of elasticity	$E = \mathbf{200000}$ N/mm^2



Beam-Small Span:

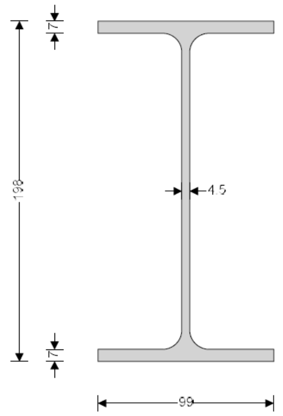
In accordance with AS4100-1998 incorporating Amendment No.1 2012

Section details

Section type	**200x18.2 UB (AISC 1994)**
Steel grade	**300**

From table 2.1: Strengths of steels

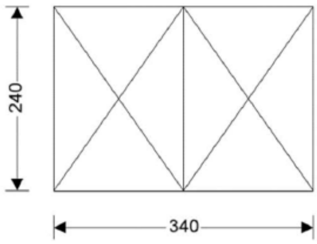
Thickness of material	$t = \max(t_f, t_w) = \mathbf{7.0}$ mm
Yield stress	$f_y = \mathbf{320}$ N/mm^2
Tensile strength	$f_u = \mathbf{440}$ N/mm^2
Modulus of elasticity	$E = \mathbf{200000}$ N/mm^2



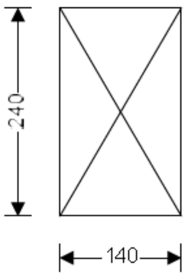
Wood structure: Timber section details

Column (post):

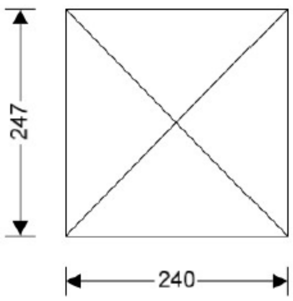
Timber section details	
Breadth of timber sections	b = **170** mm
Depth of timber sections	d = **240** mm
Number of timber sections in member	N = **2**
Overall breadth of timber member	$b_b = N \times b =$ **340** mm
Timber species	**Mixed softwood species (excl.Pinus species)**
Moisture condition	**Seasoned**
Timber strength grade - Table H2.1	**F17**



Beams –short span (mm)

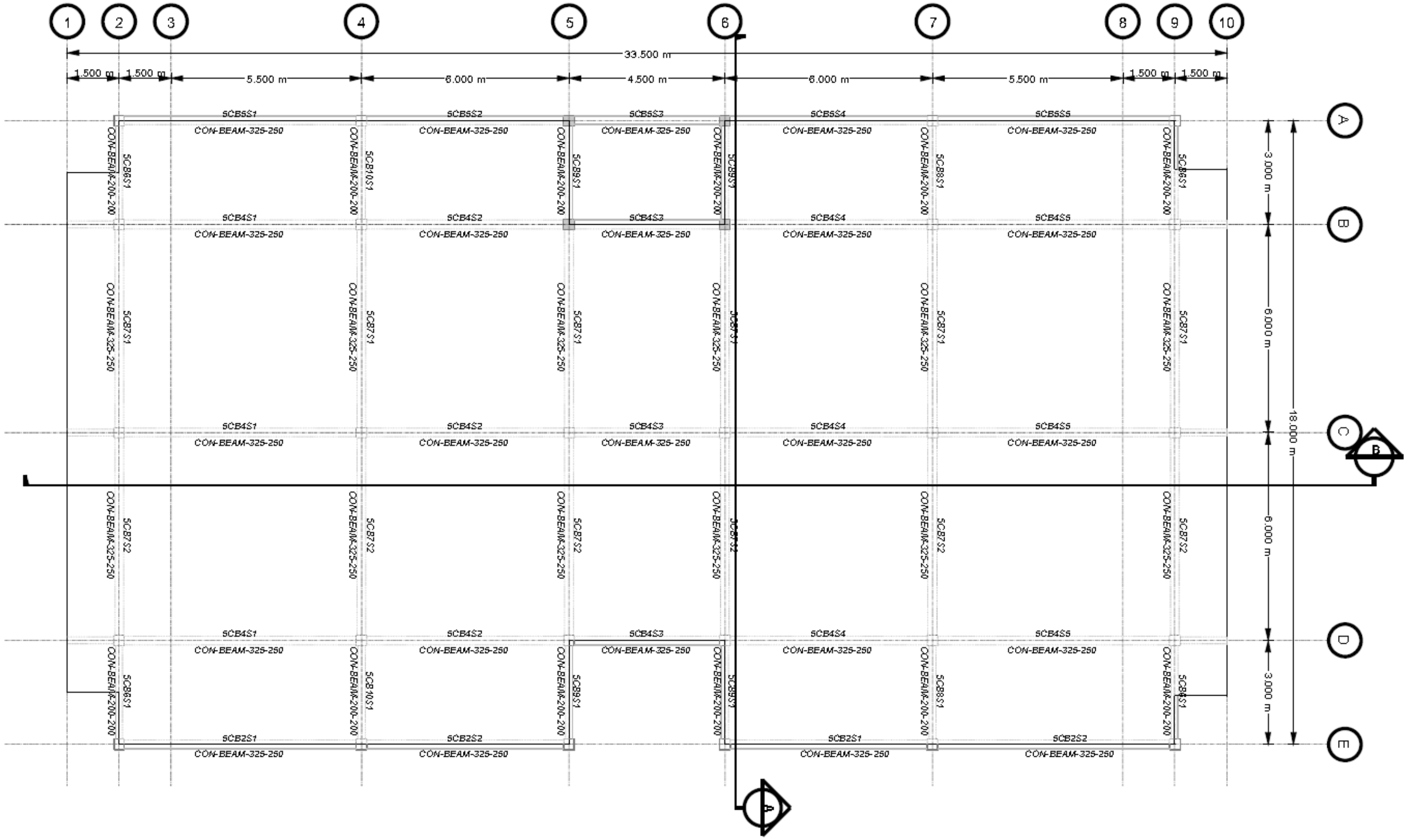


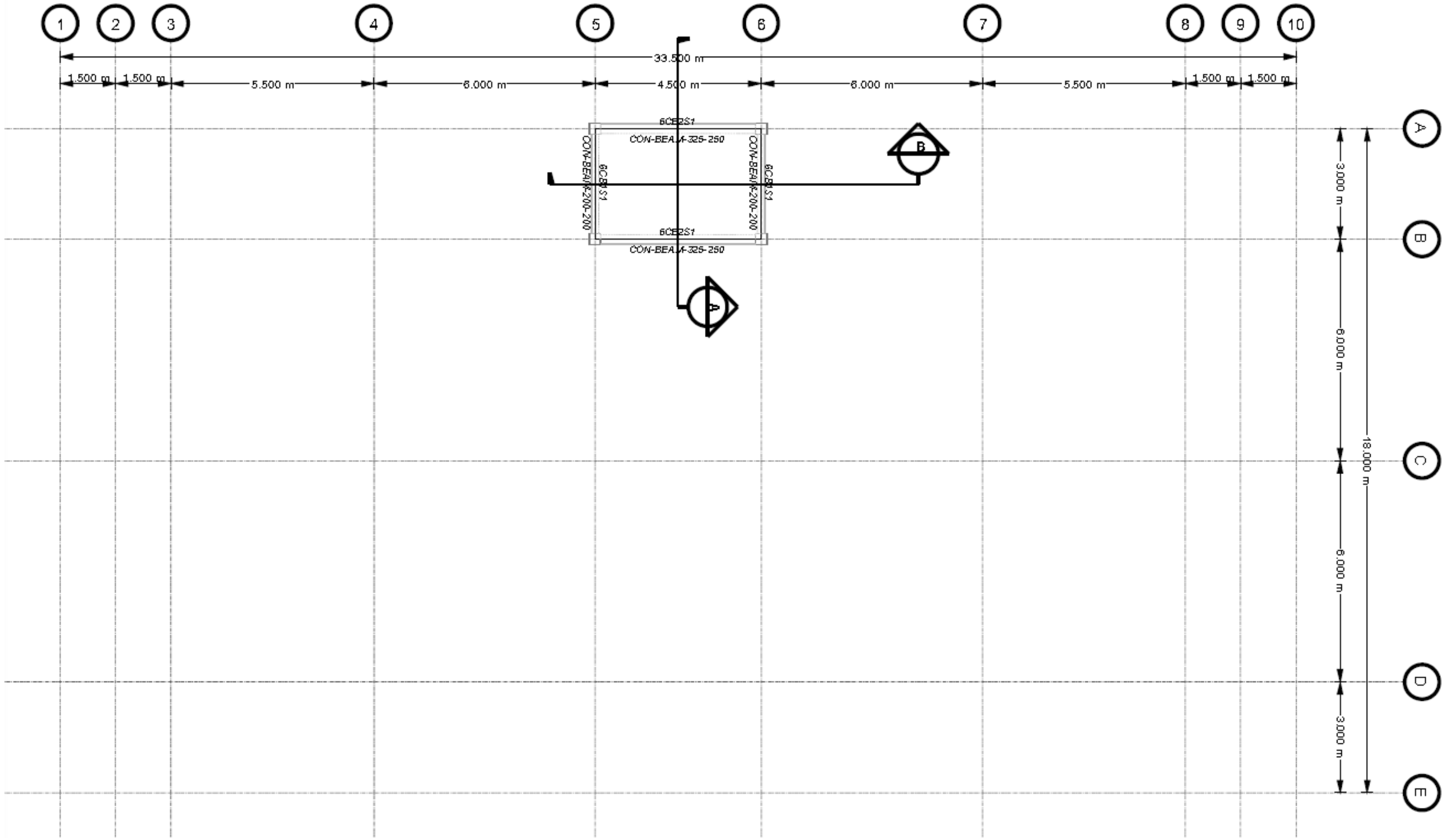
Beams – long span (mm):



Appendix B - Building Frame Design

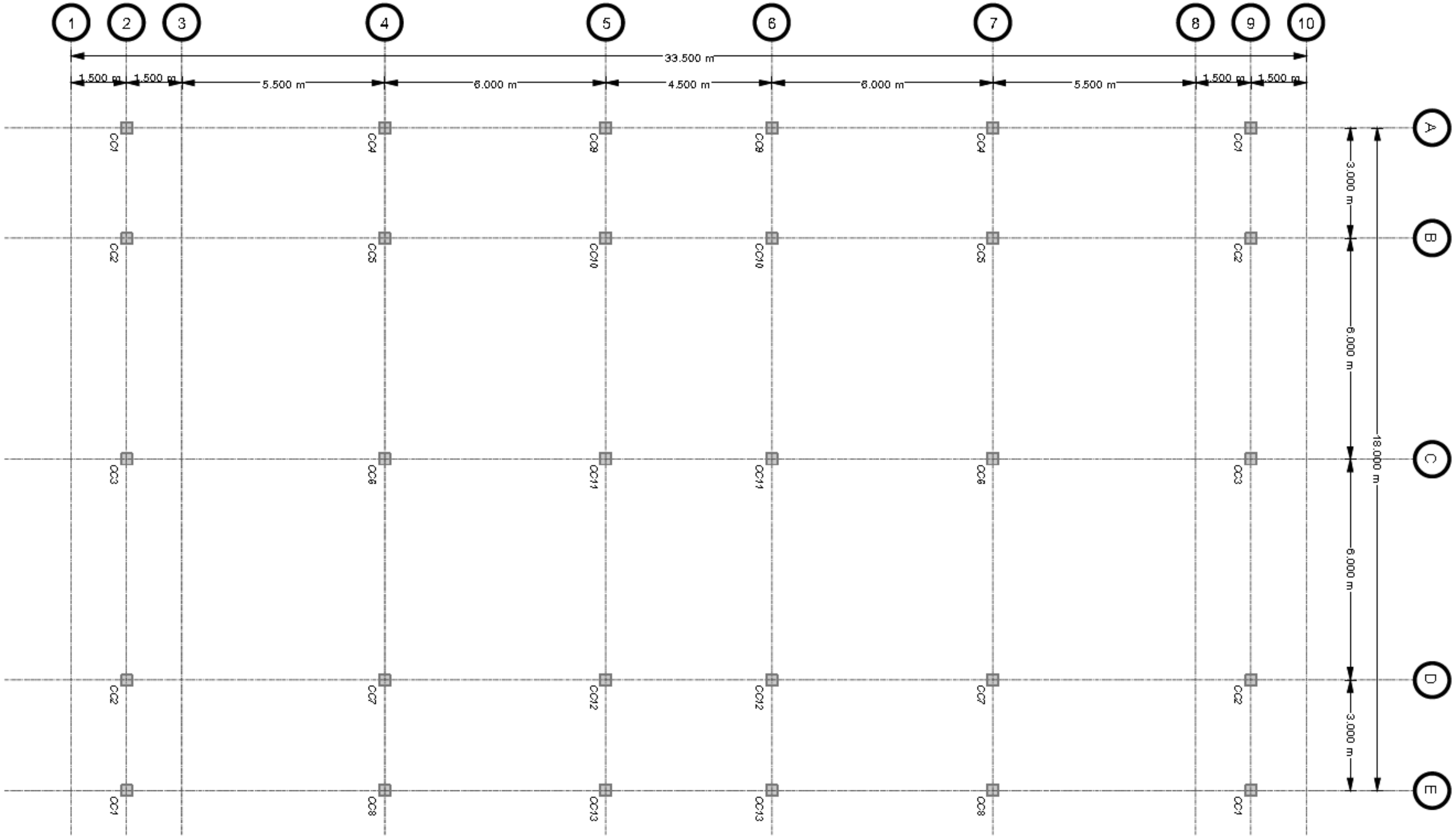
Beams Design





1
2
3
4
5
6
7
8
9
10
33.500 m
1.500 m
1.500 m
5.500 m
6.000 m
4.500 m
6.000 m
5.500 m
1.500 m
1.500 m
6CB2S1
CON-BEAM-325-250
CON-BEAM-200-200
6CB1S1
CON-BEAM-200-200
6CB1S1
6CB2S1
CON-BEAM-325-250
A
B
A
B
C
D
E
3.000 m
6.000 m
6.000 m
3.000 m
18.000 m

Columns Design



CHAPTER 7 – RESEARCH ARTICLE 5

STATEMENT OF CONTRIBUTION TO CO-AUTHORED PUBLISHED PAPER

Chapter 7 includes a pre-print of a co-authored paper titled "Toward Sustainability: Environmental and Economic Optimisation Framework for Australian Hardwood Plantation Mid-Thinning" which is currently under review in the Journal: Resources, Conservation and Recycling. My estimated contribution to the paper is 65%. I contributed to research design, data collection, modelling, analysis, and writing the manuscript.

Paper citation: Lu, HR. & El Hanandeh, A. (under review). Toward Sustainability: Environmental and Economic Optimisation Framework for Australian Hardwood Plantation Mid-Thinning. (Manuscript is under reviewed by Resources, Conservation and Recycling Journal on 14/09/2017).

(Signed) ________________________________ (Date) ______________

(Countersigned) __________________________ (Date) ______________

Corresponding author of paper:

(Countersigned) __________________________ (Date) ______________

Supervisor:

Toward Sustainability: Environmental and Economic Optimisation Framework for Australian Hardwood Plantation Mid-Thinning

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Abstract

A dual-objective multi-period linear programming optimisation framework was developed for the management of low-value forestry by-products. The objectives of the model were to improve the economic performance and reduce the environmental impacts from the early stages of hardwood plantations using life cycle approach. Eight utilisation scenarios were considered, including two engineered wood products and six bioenergy applications. The functional unit for the Life Cycle Costing (LCC) analysis and Life Cycle Assessment (LCA) study was selected as the treatment of 1 Mg of green thinned logs on the forest floor. The costs of the displaced products by the alternatives were also taken into consideration in line with the LCA analysis, to provide a more equitable comparison. The model was solved in six time periods (ten years each) and tested by assigning different weightings for the environmental and economic objectives. The result showed the Laminated Veneer Lumber (LVL) and Woodchips Gasification (WCG) options were the dominating solutions depended on different time periods. The solution favoured energy production with 85% allocation to WCG in the first period. Nevertheless, the quantities of feedstocks allocated to energy production declined progressively over the subsequent periods. The LVL surpassed the WCG and became more dominant option starting from the fourth period. In the final period, 100% of thinned logs were allocated to the LVL option due to the predicted increased demand. The developed framework offers forestry managements the foresight to enhance their long term environmental and economic performance through strategic multi-period optimisation approach.

Key words: Life Cycle Cost (LCC); Life Cycle Assessment (LCA), Multi-objective Optimisation (MOO); Linear Programming (LP); Multi-period

7.1 Introduction

7.1.1 Background

There is an increasing interest in the utilisation of wood and woody biomass to replace energy-intensive infrastructure materials and to produce bioenergy to substitute fossil fuels (Cambero and Sowlati, 2014). Various studies have shown that using engineered wood to substitute energy-intensive materials in the building sector could contribute significant Greenhouse Gases (GHG) emissions saving (e.g. Gustavsson et al., 2006, Upton et al., 2008, Bribián et al., 2011). Additionally, utilising low-value forestry waste or by-products as a source for bioenergy generation can reduce resource wastage and increase the financial return of the forestry industry (Munsell and Fox, 2010). Due to the recent global economic downturn and the increased awareness of environmental issues, reducing the life cycle costs and lowering the environmental impacts of products or processes is seen as a key strategy for industries to survive in the future market (Deng et al., 2013). Therefore, the economic and environmental performance of these available options is fundamental to determining the best management approaches to the forestry sector. Life cycle assessment (LCA) is a widely used method for environmental decision-making (ISO14040, 2006), and it is usually integrated with an economic analysis tool: Life cycle costing (LCC) analysis (Treasury NSW, 2004) to make sustainable decisions. In order to select the best solution from both environmental and economic perspectives, an optimisation framework is usually required (Lu and El Hanandeh, 2017).

Mathematical programming is normally used for selecting the optimal solution to maximise or minimise a quantitative objective considering scarce resources (Cambero and Sowlati, 2014). The optimisation problem is generally expressed as a mathematical function of decision variables and a set of constraints (Cambero and Sowlati, 2014). The problem may be formulated as a single objective (SOO) or a multi-objective (MOO) problem. In a SOO, the objective function might be either to minimise the environmental impact or maximise economic profitability. However, it is rarely the case that a resource optimisation problem is a single objective problem. More often, resource optimisation problems have more than one objective. Islam et al. [10] stated that taking a SOO approach might be ineffective when the multiple objective functions must be optimised simultaneously because it sometimes would lead to conflicting results when each objective function is optimised in isolation of the others. Therefore, a MOO approach is recommended to provide designers with a better understanding of the design space (Cocco et al., 2014).

In a MOO problem, two or more objective functions are optimised (maximised/minimised) simultaneously. Typically, a MOO problem will involve two or more competing objectives; for example, minimising environmental impact while minimising life-cycle cost. A common approach is to simplify the MOO problem into a SOO problem by incorporating the environmental impacts into the economic values (Radics et al., 2016). However, the conversion of environmental attributes to economic costs is complex and can be based on either market price or some wider measure of the social cost (Radics et al., 2016). An example of market costs for environmental attributes is the carbon cost. For instance, Ganjidoost (2012) converted the Global warming potential (GWP) to an economic value; however, due to lack of reliable data on conversion values of other environmental indicators, they were unable to include other environmental impacts in the objective function, rather than treated them as constraints. Islam et al. (2015b) commented that using inappropriate constraints may lead to infeasible outcomes. Therefore, normalising the different environmental impact categories into one unit presents challenges. De Bruyn et al. (2010) developed weighting factors for evaluating different environmental impact categories based on abatement cost and policy targets, which is also known as 'shadow prices'. However, these weighting factors should be ideally based on high-quality primary valuation studies, which are not always available for a given region or country (De Bruyn et al., 2010). More recently, Pérez-Fortes et al. (2014) recommended using Impact 2002+ points (pts) method to calculate the overall environmental impact categories. In such way, the overall environmental impact can be determined by the ratio of impact per unit of emission and the total impact generated from emissions per person per year (Jolliet et al., 2003).

With reference to forestry management, previous studies have applied optimisation on either maximising the production quantity or economic profits of timber plantations by determining the number or species of trees to be planted and the corresponding rotation ages for one full plantation cycle (e.g. Mabvurira and Pukkala, 2002, Palahí and Pukkala, 2003, Vanlisuta and Prombanpong, 2012). The utilisation of MOO to support decisions in forest biomass supply chains is a relatively new research topic (Cambero and Sowlati, 2014). Recently, the integration of economic and environmental objectives in the optimisation of forest biomass supply chains for the production of bioenergy and bioproducts have gained a special attention (e.g. Santibañez-Aguilar et al., 2011, You et al., 2012, Mafakheri and Nasiri, 2014, Yue et al., 2014). A summary of the research efforts made to optimise economic performance and environmental benefits are shown in Appendix 1. Linear programming (LP) and mixed integer linear programming (MILP) are the two most commonly used optimisation techniques for woody biomass supply chain design and management. Kanzian et al. (2013) formulated a MOO framework to maximise the profit and minimise the CO_2 emissions for biomass supply networks from the entire life cycle perspective. MILP was used by applying a binary decision variable to ensure the total amount of feedstock that has to be chipped to avoid transport of both solid and chipped fuel from a single site. Pérez-Fortes et al. (2014) developed a MOO model to assess the optimal woody waste pre-treatment plans in co-combustion plants. The optimisation problem was formulated as a MILP by applying binary variables in the determination of installation of different combustion technologies. Meanwhile, Frombo et al. (2009) used LP optimisation to find the optimum biomass supply for energy production for each individual energy conversion technology (pyrolysis, gasification or combustion). Santibañez-Aguilar et al. (2011) presented an LP to optimise economic profitability and minimise the environmental burden of the bio-refinery plant. The economic objective function considered the availability of biomass, processing limits, and demands for the products in a specific region as well as the costs of feedstocks, products, and processing routes, while the environmental objective function considered the overall environmental impact which was calculated through the Eco-indicator 99 life cycle impact method.

Although a number of optimisation studies focused on supporting decisions in the design, planning and management of forest biomass supply chains, the majority of them were limited to modelling a single application (e.g. Frombo et al., 2009, Kanzian et al., 2013, Pérez-Fortes et al., 2014) or a set of final products in the same category (e.g. Santibañez-Aguilar et al., 2011) without considering other valuable bioproducts such as engineered wood products. Additionally, most of the studies were not conducted from entire life cycle perspective, rather focused only on a few selected stages (e.g. Kanzian et al., 2009, Keirstead et al., 2012, Van Dyken et al., 2010). The final application stage such as the economic and environmental savings arising from final products substitutions was usually excluded. However, we believe these benefits should be also taken into account in order to have an equitable evaluation of the competing products. Therefore, this study aims to develop an optimisation framework using life cycle approach which allows for the comparison of different utilisation pathways including incompatible products (e.g., structural products and bioenergy applications). The framework was then applied to develop a model to optimise the utilisation of hardwood plantation thinning in South-East Queensland region of Australia.

7.2 Problem Statement

Forestry biomass resource has multiple applications such as in the public infrastructure, building construction and bioenergy conversion sectors. Each application may have its potential economic and environmental benefits. Therefore, it is important to find an optimal utilisation of the available resource in order to minimise the life cycle cost and the environmental burdens of the products.

7.2.1 Wood as Construction Material

Arguably, wood is the most sustainable structural material and strongly recommended as a substitute for high energy-intensive products (Knowles et al., 2011, Wang et al., 2014). Increasing use of wood or wood-based materials could reduce the net GHG emissions because of the relatively low energy requirement during wood products manufacturing stage compared to other materials, such as concrete and steel (Gustavsson et al., 2006). In addition, wood substitution in long-life-span products results in accumulated carbon storage (Sathre and Gustavsson, 2009), which is seen as a potential climate mitigation strategy (Cabeza et al., 2014). Traditionally, high-quality timber logs obtained from the forestry are sent to the sawmill for manufacturing sawn timber (Martínez-Alonso and Berdasco, 2015). The sawn timber is usually seen as an important raw material for the construction, furniture, and furniture components sectors. However, the use of timber in its solid form has obvious limitations. For example, the availability of natural timber sections in lengths and cross sections that meet the spans required for modern building components is not always guaranteed. Modern technology has managed to overcome many of these limitations by converting timber into the engineered wood. Engineered wood products enhance the quality by converting logs with defects to more uniform structural products compared to sawn timber, resulting in higher strength and less variability in mechanical properties (Barbu et al., 2013). These kinds of wood-based products are generally manufactured by bonding wood boards, veneers, strands or flakes using adhesives to form panels or other shaped structural products (Lam, 2001, Barbu et al., 2013). Examples of engineered wood applications include laminated veneer lumber (LVL), plywood, oriented strand board (OSB), and veneer based composite (VBC) (Underhill et al., 2014, Kasal et al., 2015).

7.2.2 Wood as Energy Feedstock

Converting forestry woody biomass to bio-energy products for replacing fossil fuel can be another environmentally beneficial option. Woody biomass provides a sustainable and dependable supply of energy feedstock since wood can be continually replenished while contributing significant GHG emission reduction (Wahlund et al., 2004, Rosen et al., 2005, Solomon and Luzadis, 2008). In addition, utilising low-value forestry waste as a source for energy generation can reduce resource wastage and increase the financial return through proper timber plantation management (Munsell and Fox, 2010). Currently, there are many bio-energy technologies available for biomass conversion. For example, wood chips can be used either directly by combustion and gasification, or by upgrading to other forms, such as wood pellets, bio-oil, or liquid biofuels to substitute fossil fuel (Searcy et al., 2007). Wood pellets have been portrayed as a suitable method to substitute fossil fuel for energy generation because of their high energy density, low moisture content, and convenient transportation and storage (Goh et al., 2013, Tarasov et al., 2013). Bio-oil produced from pyrolysis is a promising alternative to fossil crude oil (Xiu and Shahbazi, 2012). Fast pyrolysis is commercially available and it has been promoted as an effective technology for converting biomass into liquid fuels (Zhong et al., 2010). The crude bio-oil can be directly used through co-combustion in conventional fossil fuel power plants or in existing industrial boilers for electricity and heat production (Fan et al., 2011). Furthermore, hydrotreatment allows upgrading low-quality crude bio-oil to petroleum-like products which can be used as a substitute for gasoline or diesel in boilers, furnaces, engines, and turbines (Xiu and Shahbazi, 2012). Likewise, liquid fuel such as bioethanol can be produced through fermentation of biomass (Demirbas, 2011). Given the many options available for biomass utilisation and their competing potential economic and environmental impacts, choosing the optimal supply for each product requires a balance between the economic and environmental impacts.

7.2.4 Environmental and Economic Assessment

In order to calculate the environmental impacts and life cycle cost of each scenario, the LCA and LCC analysis were conducted based on the International Organisation for Standardisation ISO14040 (2006) and Australian/New Zealand Standards for Lifecycle costing AS/NZS 4536:1999 (R2014), respectively. The LCA evaluates the environmental aspects and potential impacts related to a product from raw material extraction to final disposal and treatment. Additionally, the transportation, use/operation and maintenance are all encompassed in the entire lifecycle. SimaPro v.8.0.4.30 modelling software was used in the LCA modelling exercise (GreenDelta, 2014). Global warming potential (GWP), eutrophication potential (EP), acidification potential (AP), fossil depletion potential (FDP) and human toxicity potential (HTP) were assessed based on the Best Practice Guide to Life Cycle Impact Assessment in Australia (Grant and Peters, 2016). ReCiPe Midpoint (H) life cycle impact assessment (LCIA) method was used for the characterisation of emissions into the relevant impact categories except for the AP category for which the CML 2001 method was used as suggested by Bengtsson and Howard (2010b). The LCC accounts all relevant costs, including acquisition, operation, maintenance and disposal throughout the entire life stages. According to Life Cycle Costing Guideline (Treasury NSW, 2004), the LCC does not directly consider the revenue which is assumed to be equal among all options being compared. However, in this study, the final products differ in their values. The concept of 'substituted values' (SV) was then introduced in this study to overcome this limitation. The 'substituted value' borrows the concept of 'displaced emissions' from the LCA. The SV is calculated as the difference between the LCC of the final product (alternative) and the displaced (substituted) product, (i.e. SV equals to the LCC of the alternative product minus the LCC of substituted product). All the costs were converted to the 2017 Present Value (PV). Future values were calculated based on current prices and an estimate of future inflation. The PV of a future cost was calculated based on an estimated future inflation and a suitable discount rate, which has been identified by Lu and El Hanandeh (2017). An Excel spreadsheet model was used to calculate the net present value (NPV).

7.3 General Model Formulations

This study presents a multi-objective linear programming (MOLP) model to optimise the utilisation of forestry biomass in multi-period. Two objective functions (i.e. environmental impact and economic cost) are minimised in this study. The variables of the optimisation model are the different forestry biomass application options. The simulation software "*LINDO*" was used to solve the problem. Constraints on design variables essentially delineate the bounds of the feasible range for each variable. Therefore, the constraints of the model were classified into four groups including the total feedstocks availability, final product requirements, environmental impact reduction and economic targets.

The decision variables x_i represents the quantities of woody biomass consumption in the total amount of available woody biomass for application *i*. The two competing objectives in the model are to minimise (1) economic costs and (2) environmental impacts. Nonetheless, the possible incomes (e.g. revenue from material recycling, energy recovery and life cycle cost saving from the substituted products) have to be deducted from the life cycle costs. The life cycle costs (i.e. initial investment, raw material extraction, manufacturing, maintenance, removal, and disposal costs), and environmental related cost (e.g. carbon cost) implied by one functional unit of hardwood thinning application *i* are represented respectively by means of the data c_i^1 and c_i^2. The economic coefficients c_i^1 were calculated using LCC analysis approach. The net present value approach was used to calculate costs. The environmental coefficients c_i^2 were determined using LCA. The sum of all amounts allocated to the different products may not exceed the total available supply (*s*). Let q_i be the amount of product *i* produced by one Functional Unit of wood logs. Hence, assuming linear relations, the optimisation of the use of application *i* to satisfy the feedstock supply *s* can be formulated as follows:

Min $\sum_{i=1}^{n} c_i^1 x_i$, Economic objective function … (1)

Min $\sum_{i=1}^{n} c_i^2 x_i$, Environmental objective function … (2)

Subject to

Subject to $\sum_{i=1}^{n} q_i x_i \leq s$, Satisfy supply constraint … (3)

where i = 1, 2, 3… n …

However, to solve the above problem using supply and constraint values averaged over the long planning period may produce an inaccurate solution. Especially so, when demand for products may vary over time, the set of decision variables may change over time, and/or the environmental and economic coefficients of products may change over time. Therefore, the problem should be modified to account for strategic planning through breaking the long-term plan into multiple strategic periods. This will avoid the over-allocation of resources and ensure optimal solutions for each period. The above equations (1-3) can be re-written to account for the multi-period optimisation as follows:

$$\text{Min} \sum_{t=1}^{j} \left[\sum_{i=1}^{n} c_{i,t}^{1}\, x_{i,t}\right] \qquad (4)$$

$$\text{Min} \sum_{t=1}^{j} \left[\sum_{i=1}^{n} c_{i,t}^{2}\, x_{i,t}\right] \qquad (5)$$

where i = 1, 2, 3…n …; j = 1, 2, 3…n …

Subject to:

$$\sum_{t=1}^{j} \sum_{i=1}^{n} q_{i,t}\, x_{i,t} \leq s_{i,t}, \qquad (6)$$

In regard to the environmental assessment, five impact factors were considered; therefore, all the impact factors' values were normalised based on the Impact 2002+ points method. The normalised factor was determined by the ratio of the impact per unit of emission and the total impact generated from emissions per person per year in Australia according to Jolliet et al. (2003) and Huppes and van Oers (2011).

In order to convert the MOO to a SOO problem, normalisation was applied to both economic and environmental objective functions, so that categories with different units could be compared on the same scale. The normalised data were calculated using the actual value divided by the optimal value among the alternatives as mentioned by Azapagic and Clift (1999) and Islam et al. (2015b). In this study, the weighted sum approach was used. A weight was assigned to each normalised objective function so that the problem is converted to a single objective problem with a scalar objective function. Hence, Equations (4 & 5) are then rewritten to minimise the multiple variables (cost and multiple environmental impact indicators) as:

$$\text{Min} \sum_{t=1}^{j} \left[w_1 * \sum_{i=1}^{n} (c_{i,t}^{1}\, x_{i,t}) + w_2 * \sum_{i=1}^{n} (c_{i,t}^{2}\, x_{i,t})\right] \dots\dots \; (7)$$

Where, $(c_{i,t}\; x_{i,t})$ is the normalised objective function, while $w_1 + w_2 = 1$

7.4 Case Study

The MOLP optimisation model developed in the previous section was applied to the hardwood plantation mid-rotation thinning in South-east Queensland region. The plantations are mostly spread throughout the state's southern high-rainfall (i.e. more than 800 mm/annum) areas within 150 km of the coast. Approximately 53,500 ha of the hardwood plantation area is managed by high-quality plantations for sawlog production (Meadows et al., 2014). In order to produce high-quality logs at the end of harvesting, thinning operations are usually carried out to remove trees with obvious defects during the plantation life. The annual generation rate of hardwood plantation thinned logs is approximately 11.5 m^3/ha (Meadows et al., 2014). In addition, the State of Queensland (2012) reported that the plantation estate has the potential to increase by 400 hectares per year. The average density of the hardwood thinned logs (i.e. Gympie Messmate) was reported as 944kg/m^3 with 50% moisture content (Lu and El Hanandeh, 2016a).

A number of options were investigated to add values to this resource stream, including multiple pathways for the recovery of their energy value and use to manufacture structural components (e.g. Lu and El Hanandeh, 2016a, Lu and El Hanandeh, 2017, Lu et al., 2017b). The following sections present brief summary of each of the utilisation options considered in this model.

7.4.1 VBC Utility Poles

The utility pole is a significant component in the Australian infrastructure sector. Timber is typically used for manufacturing utility power poles due to its low initial cost, natural insulation properties, and ease of transport (Shafieezadeh et al., 2014). The VBC products made from low-value hardwood thinning logs can be used to manufacture hollow utility poles to replace the traditional hardwood poles from a technical point of view (Gilbert et al., 2014a, Underhill et al., 2014). Lu and El Hanandeh (2016a) reported that from both environmental and economic perspectives, the VBC utility pole is a more competitive product than conventional steel and concrete utility poles.

The LCI of VBC utility pole manufacturing was adopted from Lu and El Hanandeh (2016a). An average lifespan of 25 years was estimated for VBC pole (Lu and El Hanandeh, 2016a). Each pole was assumed to be inspected and maintained every 10 years during the service life (Lu and El Hanandeh, 2016a). The manufactured VBC utility poles were assumed to displace steel and concrete poles. Therefore, the substituted value of the VBC pole was calculated as the difference between the LCC of VBC pole and the weighted average LCC of the concrete and steel poles. The life cycle cost of an equivalent new concrete or a steel utility pole were estimated at $2,783 and 2,442 respectively (Lu and El Hanandeh, 2016a). The quantities of existing employed concrete and steel poles in Queensland are in a ratio of 1.3:1 (Francis and Norton, 2006). Hence, the weighted average LCC of concrete and steel poles was calculated as \$2,635. In order to substitute an equivalent concrete or steel pole, 2.4 VBC poles are required (Lu and El Hanandeh, 2016a). Therefore, an average value of \$738 will be saved by using the VBC utility poles to substitute concrete and steel poles. In addition, the environmental benefits from replacing the conventional utility poles by VBC poles were also taken into account in line with the LCC.

Constraints:

Approximately 65,570 utility poles were estimated to be required every year for replacing the ageing poles in Queensland based on the report by Francis and Norton (2006). Meanwhile, an average increase rate of 4.5% per annual was assumed due to the extension of the new lines (Francis and Norton, 2006). Almost 95% of the required utility poles are sourced from timber, while the remaining (5%) are made of concrete and steel (Francis and Norton, 2006). This study assumes that the VBC poles will compete with the concrete and steel utility poles. Therefore, the VBC poles were assumed to substitute no more than 100% of the average demand for concrete and steel poles during each optimisation period.

7.4.2 LVL for Multi-Storey Building Frame Construction

The building construction sector contributes a significant amount of GHG emissions in Australia (Basaglia et al., 2015). The majority of emissions are attributed to the use of energy-intensive materials such as concrete and steel (Cabeza et al., 2014). Wood and wood-based products are usually promoted as sustainable and renewable building construction materials (Wang et al., 2014). Recently, engineered wood products have gained great successes in the construction sector and increasingly being accepted as a cost-competitive building material (Bribián et al., 2011, Kasal et al., 2015). The new Australian Building Codes Board (ABCB) (2016) allows high timber buildings (up to 25 m high) to be designed under the 'Deemed to satisfy' provisions. Hence, the use of wood in mid-rise building constructions is expected to increase. Therefore, in this study, the LVL products for the structure frame (beams and columns) construction was selected as another option.

The environmental and economic life cycle inventories of LVL product were adapted from Lu et al. (2017a), and Lu et al. (2017b). The LCC of LVL wood frame per FU was calculated by including the cost of raw material extraction, products manufacturing, transportation, and final disposal. The carbon offset and energy recovery from wood waste incineration and steel recycling were also included in the LCC. The LVL building frame was assumed to substitute the conventional concrete, steel and softwood frames with an average lifespan of 60 years.

Nevertheless, no statistical data were available about the percentage of material type usage in building frame to calculate the weighted average for substitution value. Hence, the environmental and economic benefits during the LVL substitution were calculated assuming equal proportions of concrete, steel and timber. The LCA and LCC of an equivalent concrete, steel, softwood and LVL frame were adapted from Lu et al. (2017b).

Constraints:

The number of apartment building approvals in Queensland during the 2016-17 period was approximately 6,000 units (The State of Queensland, 2017). The existing population growth rate, overseas investment, and the historically low-interest rates are expected to push residential construction further, in particular for the multi-level apartments (Lu et al., 2017b). The annual increase rate of 1.8% was assumed based on the average apartment building growth rate during last 15 years in Queensland (The State of Queensland, 2017). Currently, engineered wood-framed buildings represent approximately 10% of the new buildings, with an annual increase rate of 1.2% (Nolan, 2011). The engineered wood frame was estimated to be 70 tonnes of LVL per apartment building (Lu et al., 2017b). The engineered wood was assumed to be used for replacing no more than 100% of total engineered wood demand for apartment building frames construction.

7.4.3 Bioenergy Production

The use of forestry residues as a feedstock for bioenergy production is an emerging technology that could present additional markets for wood residues and provide an additional renewable energy source in Australia (Lu and El Hanandeh, 2017). For the bioenergy conversion scenarios, six options were included following Lu and El Hanandeh (2017). The options are including woodchip gasification in biomass integrated combined cycle (WCG); electricity production via pyrolysis in gas turbine combined cycle (PyEL); liquid fuel production through hydrogenation of pyrolytic oil (PyLT); ethanol production for transport fuel mix through hydrolysis and subsequent fermentation (EthP); power generation from pellets (WPG) and use of wood pellets in domestic water heating (WPC). A medium size (5 MW) power generation plant was assumed for all power generation options in this study. The average lifespan of each bioenergy conversion plant was assumed to be 30 years according to Lu and El Hanandeh (2017). The life cycle costs and environmental impact from the substituted products were assumed to be avoided by all scenarios.

The levelised cost of electricity (LCOE) production in Australia from various sources was reported by Bureau of Resources and Energy Economics (BREE) (2012), including coal-fired electricity (AUD$78–91/MWh), combined-cycle gas turbines (A$97/MWh), wind (A$150–214/MWh), medium-sized (5 MW) solar PV systems (A$400–473/MWh). The existing electricity production in South East Queensland grid is comprised of 68% coal-power, 20% natural gas, 5% Hydropower, 3% biomass and the rests are from other sources, such as solar, wind and waste (AusLCI, 2011). Therefore, the LCOE in QLD was estimated as $0.10/kWh.

In regards to the heat generated from CHP, steam can be extracted from the turbine at the pressure required by the industrial heating process or the heat can be converted to provide cooling using absorption chillers in larger configurations (Energeia, 2011). In this study, the CHP has been assessed against alternative industrial water heating technologies, while the costs savings were based on a comparison of an equivalent amount of hot water and mains electricity. The existing industry water heating relies on 42% of electricity and 58% of natural gas (Kenway, 2013). On the other hand, the heat generated from the biomass domestic water heater was assumed to displace domestic heat: 48% of natural gas; 45% of electricity; 3% of LPG and 4% of solar (El Hanandeh, 2015). The natural gas production cost was reported as $0.033/kWh in Queensland (Commonwealth of Australia 2016). Therefore, the cost-saving was calculated as 0.061/kWh and $0.067/kWh for industrial heating and domestic water heating, respectively.

Constraints:

There are inherent challenges in forecasting future bioenergy markets and demand. The potential for these emerging products to be developed in Australia needs to be evaluated to ensure the produced bio-energy is able to meet future energy demand (Commonwealth of Australia, 2015).

- Electricity:

Australian Energy Statistics (2016) reported that the renewable energy content in Queensland electricity generation was approximately 12%, while biomass contributed 26.9% of the total renewable energy consumption, which is about 42.56 PJ. However, the contribution of the woody biomass from thinning, even when assuming that all are converted to energy (5.69 PJ), it is still far below the total energy required from biomass in Queensland. Therefore, there is no maximum limit of electricity production in this study.

- Heat:

Recent research shows that the energy associated with residential and industrial water heating represents 0.9% and 1.1% of Australian total primary energy consumption, which accounted for 52.48 PJ and 64.14 PJ, respectively (Kenway, 2013). The heat energy consumptions in south-east Queensland were then estimated at 8.41 PJ and 10.27 PJ for domestic and industrial water heating, respectively based on the local population. However, these values are still more than the total energy required from the renewable energy source. Therefore, there is no maximum limit of heat energy production.

- Biofuels (ethanol and biodiesel):

The Australian government has set a target to increase the renewable energy generation from its current level (13.7%) to 23.5% by 2020 (Australian Government Department of the Environment and Energy, 2015b, Australian Energy Statistics, 2016). The annual growth rate was then calculated as 1.8%. The biodiesel and bio-ethanol represented 2.5% and 3.5% of the total renewable energy consumption in South East Queensland, respectively (Australian Energy Statistics, 2016). Therefore, a constraint is set on the total biofuels production (Energy: MJ) not to exceed the annual biofuel demand.

7.4.4 Environmental and Economic Coefficient of the Available Options

A 60-year planning time frame was applied in this study. To ensure that all environmental impacts are comparable on the same scale, the normalised value of mid-point impact was calculated using the impact per unit of emissions divided by the total impact generated from Australian emissions per person per year following to the method described by Jolliet et al. (2003). The GWP, AP, EP and HTP generated from Australian per capita were sourced from Bengtsson and Howard (2010a). The FDP from Australian per capita were calculated based on the data from Australian Energy Statistics (Australian Energy Statistics, 2016). The environmental impacts and substituted value of each option from following Table 7.1 were calculated based on the recently published life cycle studies (Lu and El Hanandeh, 2016a, Lu and El Hanandeh, 2016b, Lu and El Hanandeh, 2017, Lu et al., 2017b).

Table 7.1 LCA and LCC performance

Impact category	Units/FU	Australian per capita	VBC Pole	LVL frame	WCG	PyEl	PyLT	EthP	WPC	WPG
GWP	kg $CO_{2\ eq}$	28,690	-182.67	-376.06	-2,160	-588	-538	29.8	-1,534	-2,140
AP	kg $SO_{2\ eq}$	123	-0.41	-0.52	-1.88	0.21	-1.54	0.19	-0.68	-1.35
EP	kg $PO_{4\ eq}$	19	-0.14	-0.30	-1.15	-0.14	0.33	0.66	-0.47	-0.93
FDP	kg Oil $_{eq}$	6,231	-82.77	-107.15	-748	-11.5	-103.2	-20.6	-506	-666
HTP	kg 1,4-DB $_{eq}$	3,216	84.48	-13.01	-83.2	81.8	71.6	177	75.2	-0.66
Overall LCA	unit/FU		-0.018	-0.054	-0.297	-0.003	-0.008	0.089	-0.142	-0.242
SV	$/FU		-737	-875	8	97	89	30	102	105

7.4.5 Model Formulation for the Case Study

To identify the optimum allocation of the forestry thinning biomass to each of the utilisation options discussed earlier, a constrained multi-periods linear programming multi-objective model based on equation 8 & 9 was formulated. The linear program was solved over 6 intervals of 10 years each. The economic and the environmental objective functions were first solved solely subject to four sets of constraints: total feedstock availability, final products requirements, environmental impact reduction and economic performance target.

The feedstock and product constraints are discussed in the previous sections. The economic target was derived from the market value of the thinned logs which are currently treated as low-value by-products. The present value of low-quality sawlogs was calculated as $57/Mg (green logs) in Australian market based on the report from May et al. (2012). Lu and El Hanandeh (2017) calculated the LCC of production of hardwood thinned logs to be $37/Mg (green) by including the cost of plantation establishment, management, thinning, labour, land and transportation. Thus, the current economic return is around $20/Mg. Therefore, an economic target was set to be at least better than the economic performance of the existing management option. Furthermore, the overall environmental performance needs to meet the GHG emissions reduction target set by the Australian Government. This target requires the national GHG emissions to be reduced by 26-28 percent from the 2005 levels (612 Mt CO_2-e) by 2030 (441 Mt CO_2-e) [61]. To achieve this target an average reduction rate of 0.93% per annual was estimated. We further assumed that this rate would extend beyond 2030.

Objective: Minimise overall economic impact:

$$\text{Min} \quad \sum_{t=1}^{6} [c_{VBC,t}^{SV} \times x_{VBC,t} + c_{LVL,t}^{SV} \times x_{LVL,t} + c_{WCG,t}^{SV} \times x_{WCG,t} + c_{PyEl,t}^{SV} \times x_{PyEl,t} + c_{PyLT,t}^{SV} \times x_{PyLT,t} + c_{EthP,t}^{SV} \times x_{EthP,t} + c_{WPC,t}^{SV} \times x_{WPC,t} + c_{WPG,t}^{SV} \times x_{WPG,t}] \ldots (8)$$

Objective: Minimise overall environmental impact:

$$\text{Min} \quad \sum_{t=1}^{6} [c_{VBC,t}^{LCA} \times x_{VBC,t} + c_{LVL,t}^{LCA} \times x_{LVL,t} + c_{WCG,t}^{LCA} \times x_{WCG,t} + c_{PyEl,t}^{LCA} \times x_{PyEl,t} + c_{PyLT,t}^{LCA} \times x_{PyLT,t} + c_{EthP,t}^{LCA} \times x_{EthP,t} + c_{WPC,t}^{CA} \times x_{WPC,t} + c_{WPG,t}^{LCA} \times x_{WPG,t}] \ldots (9)$$

Subject to the following constraints

1. Forestry raw material availability

$$x_{VBC,t} + x_{LVL,t} + x_{WCG,t} + x_{PyEl,t} + x_{PyLT,t} + x_{EthP,t} + x_{WPC,t} + x_{WPG,t} \leq S_t \ldots (10)$$

Final product required - VBC poles

$q_{VBC,t} \times x_{VBC,t} \leq D_t$ … (11)

Final product required – LVL building frame

$q_{LVL,t} : x_{LVL,t} \leq L_t$ … (12)

Final product required - Biofuels (ethanol and biodiesel)

$q_{PyLT,t}\, x_{PyLT,t} \leq P_t$ … (13)

$q_{EthP,t}\, x_{EthP,t} \leq E_t$ … (14)

Economic target

$$\sum_{t=1}^{n} [c_{VBC,t}^{SV} \times x_{VBC,t} + c_{LVL,t}^{SV} \times x_{LVL,t} + c_{WCG,t}^{SV} \times x_{WCG,t} + c_{PyEl,t}^{SV} \times x_{PyEl,t} + c_{PyLT,t}^{SV} \times x_{PyLT,t} + c_{EthP,t}^{SV} \times x_{EthP,t} + c_{WPC,t}^{SV} \times x_{WPC,t} + c_{WPG,t}^{SV} \times x_{WPG,t}] \leq M_t \ \ldots$$
(15)

Environmental target

$$\sum_{t=1}^{n} [c_{VBC,t}^{LCA} \times x_{VBC,t} + c_{LVL,t}^{LCA} \times x_{LVL,t} + c_{WCG,t}^{LCA} \times x_{WCG,t} + c_{PyEl,t}^{LCA} \times x_{PyEl,t} + c_{PyLT,t}^{LCA} \times x_{PyLT,t} + c_{EthP,t}^{LCA} \times x_{EthP,t} + c_{WPC,t}^{LCA} \times x_{WPC,t} + c_{WPG,t}^{LCA} \times x_{WPG,t}] \leq N_t \ \ldots (16)$$

In order to convert the MOO into a SOO problem, the economic and environmental coefficients were normalised following the method described in the general model formulation sections. The normalised economic and environmental functions were then assigned a random weighting for each, while the sum of the environmental and economic weighting is always "100%". Equation (17) presents the optimisation model for minimising both environmental impact and life-cycle cost.

Objective: Minimise overall impact

$$\text{Min} \quad \sum_{t=1}^{n} [W_a \times (N_{VBC,t}^{SV} \times x_{VBC,t} + N_{LVL,t}^{SV} \times x_{LVL,t} + N_{WCG,t}^{SV} \times x_{WCG,t} + N_{PyEl,t}^{SV} \times x_{PyEl,t} + N_{PyLT,t}^{SV} \times x_{PyLT,t} + N_{EthP,t}^{SV} \times x_{EthP,t} + N_{WPC,t}^{SV} \times x_{WPC,t} + N_{WPG,t}^{SV} \times N_{WPG,t}) + W_b \times (N_{VBC,t}^{LCA} \times x_{VBC,t} + N_{LVL,t}^{LCA}\, x_{LVL,t} + N_{WCG,t}^{LCA} \times x_{WCG,t} + N_{PyEl,t}^{LCA} \times$$

$x_{PyEl,t} + N_{PyLT,t}^{LCA} \times x_{PyLT,t} + N_{EthP,t}^{LCA} \times x_{EthP,t} + N_{WPC,t}^{LCA} \times x_{WPC,t} + N_{WPG,t}^{LCA} \times x_{WPG,t})]$ …… (17)

Where W_a+ W_b= 100%

Table 7.2 Parameters of the optimisation model

Symbol	Description
t	Time intervals, t=1,2,…6
$c_{VBC,t}^{SV}$	Economic coefficient of VBC pole option in stage t
$c_{LVL,t}^{SV}$	Economic coefficient of LVL option in stage t
$c_{WCG,t}^{SV}$	Economic coefficient of WCG option in stage t
$c_{PyEl,t}^{SV}$	Economic coefficient of PyEl option in stage t
$c_{PyLT,t}^{SV}$	Economic coefficient of PyLT option in stage t
$c_{EthP,t}^{SV}$	Economic coefficient of EthP option in stage t
$c_{WPC,t}^{SV}$	Economic coefficient of WPC option in stage t
$c_{WPG,t}^{SV}$	Economic coefficient of WPG option in stage t
c_{VBC}^{LCA}	Environmental coefficient of VBC pole option
$c_{LVL,t}^{LCA}$	Environmental coefficient of LVL option in stage t
$c_{WCG,t}^{LCA}$	Environmental coefficient of WCG option in stage t
$c_{PyEl,t}^{LCA}$	Environmental coefficient of PyEl option in stage t
$c_{PyLT,t}^{LCA}$	Environmental coefficient of PyLT option in stage t
$c_{EthP,t}^{LCA}$	Environmental coefficient of EthP option in stage t
$c_{WPC,t}^{LCA}$	Environmental coefficient of WPC option in stage t
$c_{WPG,t}^{LCA}$	Environmental coefficient of WPG option in stage t
$x_{VBC,t}$	Quantities of thinned logs allocated to VBC pole option in stage t

$x_{LVL,t}$	Quantities of thinned logs allocated to LVL option in stage t
$x_{WCG,t}$	Quantities of thinned logs allocated to WCG option in stage t
$x_{PyEl,t}$	Quantities of thinned logs allocated to PyEl option in stage t
$x_{PyLT,t}$	Quantities of thinned logs allocated to PyLT option in stage t
$x_{EthP,t}$	Quantities of thinned logs allocated to EthP option in stage t
$x_{WPC,t}$	Quantities of thinned logs allocated to WPC option in stage t
$x_{WPG,t}$	Quantities of thinned logs allocated to WPG option in stage t
N_{VBC}^{SV}	Normalised economic coefficient of VBC pole option in stage t
$N_{LVL,t}^{SV}$	Normalised economic coefficient of LVL option in stage t
$N_{WCG,t}^{SV}$	Normalised economic coefficient of WCG option in stage t
$N_{PyEl,t}^{SV}$	Normalised economic coefficient of PyEl option in stage t
$N_{PyLT,t}^{SV}$	Normalised economic coefficient of PyLT option in stage t
$N_{EthP,t}^{SV}$	Normalised economic coefficient of EthP option in stage t
$N_{WPC,t}^{SV}$	Normalised economic coefficient of WPC option in stage t
$N_{WPG,t}^{SV}$	Normalised economic coefficient of WPG option in stage t
$N_{VBC,t}^{LCA}$	Normalised environmental coefficient of VBC pole option in stage t
$N_{LVL,t}^{LCA}$	Normalised environmental coefficient of LVL option in stage t
$N_{WCG,t}^{LCA}$	Normalised environmental coefficient of WCG option in stage t
$N_{PyEl,t}^{LCA}$	Normalised environmental coefficient of PyEl option in stage t
$N_{PyLT,t}^{LCA}$	Normalised environmental coefficient of PyLT option in stage t
$N_{EthP,t}^{LCA}$	Normalised environmental coefficient of EthP option in stage t
$N_{WPC,t}^{LCA}$	Normalised environmental coefficient of WPC option in stage t
$N_{WPG,t}^{LCA}$	Normalised environmental coefficient of WPG option in stage t
S_t	Average annual available feedstock in stage *t*
D_t	Average annual utility pole demand in stage *t*
$q_{VBC,t}$	Quantities of VBC poles produced from 1 FU of wood logs in stage t
$q_{LVL,t}$	Quantities of LVL product manufactured from 1 FU of wood logs in stage t

$q_{PyLT,t}$	Quantities of bio-diesel generated from 1 FU of wood logs via PyLT option in stage t
$q_{EthP,t}$	Quantities of bio-ethanol generated from FU of wood logs via EthP option in stage t
L_t	Average annual LVL demand in stage *t*
P_t	Annual biodiesel demand (PyLT) in stage *t*
E_t	Annual bioethanol demand (EthP) in stage *t*
M_t	Economic target in stage *t*
N_t	Environmental target in stage *t*
W_a	Environmental weighting, W_a = 1 - W_b
W_b	Economic weighting, W_b = 1- W_a

7.5 Results and Discussions

Table 7.3 shows the optimal results by considering either environmental or economic objective function independently. The "best" design depends on which single objective function was selected to be optimised. Bioenergy conversion (WCG) option was the most favourable scenario in all of the 6-time intervals when solely optimising the environmental objective (independently of the economic objective). Almost all the available thinned logs were allocated to the WCG option, while approximately 2% of logs were allocated to engineered wood (LVL) option due to the economic constraint. On the other hand, when the economic performance was set as the single objective function, the minimum economic cost was achieved by allocating the logs to the VBC and LVL options. However, because of the limited market demand, only a part of the thinned logs can be utilised for the VBC and LVL production during the first five periods. The rest of thinned logs would have to be left unallocated as all the energy options would violate the economic target constraint. Furthermore, LVL became the only feasible option in the last period. The solutions from environmental SOO and economic SOO showed conflicting results. There was no single best design that minimises both objective functions at the same time.

Table 7.3 SOO results: optimal environmental and economic performance per FU

Impact	Units	Period 1	Period 2	Period 3	Period 4	Period 5	Period 6
Environmental	unit/FU	-0.1355	-0.1355	-0.130	-0.127	-0.123	0.118
Economic	$/FU	-845	-846	-852	-840	-840	875

To optimise both two objectives simultaneously, both economic and environmental objective functions were normalised by dividing the optimal values from the SOOs (in Table 7.3) as described in the model formulation section. The model was then solved in six time periods and the robustness of the solution was tested by assigning different weightings to the two objective functions. According to the result shown in Table 7.4, the WCG and LVL were identified as the only two options in the solution when the economic and environmental objectives were equally weighted. Furthermore, the results showed a downtrend in the percentage of logs allocated to WCG option as the time went on. LVL surpassed WGC and became the more favourable option after the third period and eventually becoming the only option in the solution set during the sixth period. This solution was stable for the economic function weight range 40-70%. In addition, for this weighting range, the allocation of logs to the LVL option was controlled by the market demand constraint. However, when the environmental weighting approached 70%, the LVL option lost favour to the WCG but retained a small allocation due to the economic target. On the other hand, when the economic weighting factor approached 80%, the amount of thinned logs allocated to WCG were slightly reduced and moved to VBC option as the time went on. However, due to the market demand of utility poles, the share of VBC remained insignificant up to the fifth period (<5.5%). Nevertheless, the LVL option became the sole solution in the last period. This is because the LVL has lower SV than the VBC option.

The results showed that optimisation of the resource should consider strategic mid-range planning periods as the constraints may change over time. The weighting assigned to each objective function also affects the final allocation of resources. Nevertheless, the effect of the environmental weighting was less prominent over the realistic weighting range (30-60%). Only for extreme environmental weights (≤20% or ≥70%) would the final solution be significantly affected or the solution set would change to include or exclude options form the balanced solution.

Table 7.4 MOO results for hardwood plantation thinning waste management design

Environmental: Economic	**20%:80%**	**30%:70%**	**40%:60%**	**50%:50%**	**60%:40%**	**70%:30%**	**80%:20%**
Stage 1 Year (1-10)	LVL: 15% WCG: 84% VBC: 1%	LVL: 15% WCG: 85%	LVL: 15% WCG: 85%	LVL: 15% WCG: 85%	LVL: 15% WCG: 85%	LVL: 2% WCG: 98%	LVL: 2% WCG: 98%
Stage 2 (Year 11-20)	LVL: 23% WCG: 76% VBC: 1%	LVL: 23% WCG: 77%	LVL: 23% WCG: 77%	LVL: 23% WCG: 77%	LVL: 23% WCG: 77%	LVL: 2% WCG: 98%	LVL: 2% WCG: 98%
Stage 3 (Year 21-30)	LVL: 34% WCG: 64.5%	LVL: 34% WCG: 66%	LVL: 34% WCG: 66%	LVL: 34% WCG: 66%	LVL: 34% WCG: 66%	LVL: 2% WCG: 98%	LVL: 2% WCG: 98%

	VBC: 1.5%						
Stage 4 (Year 31-40)	LVL: 51% WCG: 45.5% VBC: 3.5%	LVL: 51% WCG: 49%	LVL: 51% WCG: 49%	LVL: 51% WCG: 49%	LVL: 51% WCG: 49%	LVL: 2% WCG: 98%	LVL: 2% WCG: 98%
Stage 5 (Year 41-50)	LVL: 77% WCG: 17.5% VBC: 5.5%	LVL: 77% WCG: 23%	LVL: 77% WCG: 23%	LVL: 77% WCG: 23%	LVL: 77% WCG: 23%	LVL: 2% WCG: 98%	LVL: 2% WCG: 98%
Stage 6 (Year 51-60)	LVL: 100%	LVL: 100%	LVL: 100%	LVL: 100%	LVL: 100%	LVL: 2% WCG: 98%	WCG: 98%

The previous forestry studies solely focused on optimisation of wood supply chains for energy productions, with a wide range of objective functions, such as minimising cost, GHG emissions, or maximising the numbers of accrued local jobs (e.g. Santibañez-Aguilar et al., 2011, You et al., 2012, Kanzian et al., 2013). The variables were related to operations, such as selection of feedstock, processing technology, location, transportation mode, …etc. (Santibañez-Aguilar et al., 2011, e.g. You et al., 2012). The optimisation framework introduced in this study complement the existing literature by including both bioenergy production and engineered wood products in the system, which provides a more comprehensive decision-supporting tool to rationally manage the low-quality forestry resource. Furthermore, the optimisation framework proposed in this study allows for the long-term planning using strategic multi-period approach; thus complementing the short-term operational and tactical approach common in the literature. A major limitation to being able to compare products from different categories is being able to select a representative functional unit. To overcome this hurdle, the potential economic credits from alternative substitutions were incorporated into the LCC, in line with the accepted accounting methods of LCA which allows credits for offset emissions. In such way, the potential benefits from all the products can be calculated and compared on equitable grounds. This method can be applied to conduct a meta-analysis of the existing comparative LCA and LCC studies, thus allowing easier comparisons between different final products given that system boundary issues are dealt with.

This research has implications for the Australian forestry sector, particularly hardwood plantation management. The results of the study are also relevant, locally and globally, to other timber residues and low-grade products such as pulp-wood plantations which are currently facing a market glut. This study confirms the feasibility of using forestry residue as feedstock to substitute fossil fuel energy while mitigating negative environmental impacts and achieving sustainable development strategy. However, it is important to note that this study considered a limited set of potential end products and was based on the current technological state of the considered products. Future technological advances may reduce the cost and environmental impacts of these options and therefore introduce a level of uncertainty in the solution. The effect of such uncertainty may be tested using stochastic methods but was considered beyond the scope of this work. Furthermore, technological advances may also produce new products that may be more viable for in the future economy. Such discontinuous changes were not considered in this study as their predictability is not possible.

Additional research is required by including other valuable applications such as biochar product for soil protection, other bio-energy products, as well as other industrial utilisation of bio-oils such as pharmaceutical and bioplastics. Further research is also recommended to include social impacts (including social life cycle assessment S-LCA) to address the full scope of sustainable design and construction practices.

7.6 Conclusions

This study developed a multi-objective, multi-period linear programming optimisation framework for the management of low-value forestry by-products to minimise the environmental impact and the economic cost. The framework was applied to the Australian hardwood plantation mid-rotation thinning in South-east Queensland. Eight utilisation pathways were assessed including two structural use scenarios and six bioenergy production options. The functional unit for the LCC and LCA study was selected as the treatment of 1 Mg of green thinned logs on the forest floor. The costs of the displaced products by the output were also taken into consideration in line with the LCA analysis, to provide a more equitable comparison. The constraints included the total feedstocks availability, final product requirements, GHG emission reduction and economic targets. The MOO was solved in six periods with ten years for each. The different weights for the environmental and economic objectives were also tested for the analysis. The result showed that the LVL and WCG were the dominating solutions for the balanced weights of LCA: LCC (40:60% to 70:30%). However, the percentage of thinned logs allocated to each option varied in different periods. The solution favoured energy production with 85% allocation to WCG in the first 10 years. Nevertheless, the percentage allocated to energy production declined progressively over the subsequent periods. The LVL surpassed the WCG and became more dominant option starting from the fourth period. In the final period, 100% of thinned logs were allocated to the LVL option due to the increased demand. When the environmental weight exceeded 70%, WCG option became the dominant solution while the LVL option became less favourable with only 2% of the total feedstocks' allocation. On the other hand, the VBC option did not feature in the solution until the weight assigned to the economic objective reached 80%.

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Appendix

Appendix 1. Summary of the existing publications on the optimisation of forestry biomass supply chains

Authors	**Research focus**	**Objective functions**	**Parameters**	**Model**
You et al. (2012)	optimal design and planning of cellulosic ethanol supply chains	Minimising annualised total cost and GHG emissions, as well as maximising the number of accrued local jobs	Supply seasonality and geographical diversity, biomass degradation, feedstock density, diverse conversion pathways and by-products, infrastructure compatibility, demand distribution, regional economy, and government incentives	MILP
Santibañez-Aguilar et al. (2011)	optimal planning of a bio-refinery, considering the optimal selection of feedstock, processing technology, and a set of products	Profit maximisation and minimising overall environmental impact measured by Eco-indicator-99	Selection of feedstock, processing technology, and a set of products	LP
Kanzian et al. (2013)	supply network optimisation for bioenergy production	Profit maximisation and GHG emissions minimisation	Chipping location, transport mode and volume and terminals used.	MILP
Pérez-Fortes et al. (2014)	biomass waste supply chain optimisation for energy generation	NPV maximisation and Overall impact minimisation (impact 2002+)	Transportation, storage and change of properties (moisture content and hence dry matter, energy density and bulk density) through the use of different pre-treatments technologies	MILP
Čuček et al. (2012)	regional biomass supply chains for the conversion of biomass to energy	Profit maximisation, non-renewable energy use and pollution minimisation	Biomass harvesting and supply, collection and pre-processing, main processing and use	MINLP (mixed integer nonlinear program ming)

Gunnarsson et al. (2004)	forest residues converted into forest fuel	minimising the total supply chain cost	Supply of forest residues, transportation flows of products from sources to terminals and heating, as well as storing in harvest areas and at terminals	MILP
Ghaffariyan et al. (2013)	biomass supply chain optimisation for energy generation	Minimising total operating costs of the biomass supply chain	Energy demand, storage cost, chipping cost, transport cost, and energy content of woodchips	LP
Frombo et al. (2009)	woody biomass supply chain for energy production	Maximizing net annual profit	Annual quantity of biomass harvested from each supply area and the plant capacity for different conversion technologies	LP
Kanzian et al. (2009)	woody biomass supply chain for energy production	Minimising biomass supply cost to the heating plants (chipping, storing and transporting costs)	Volume of wood chips transported from each terminal to each plant and location of terminals and plants (Binary variable)	MILP
Keirstead et al. (2012)	woody biomass supply chain for energy production	Minimising system cost (biomass purchase, storage, transportation and conversion costs)	Optimal capacity of boilers and whether chipped forest biomass should be imported from neighbour area or non-chipped residues should be imported and then chipped within the area (Binary variable)	MILP

CHAPTER 8 – GENERAL CONCLUSIONS AND FUTURE WORK

In this chapter, general conclusions are drawn in response to the research questions. The main contributions and suggestions for further research are also outlined.

8.1 General Conclusions

This research study was conducted in response to the current situation of the Australian hardwood plantation thinning management. Low financial return and low utilisation rate of the hardwood logs from mid-rotation thinning caused of concern. The aim of this study was to investigate the optimal management strategy for the utilisation of logs produced during the second thinning operation to achieve better environmental performance and economic value.

A number of pathways were identified at the beginning of this research, including the production of engineered wood; solid fuel and biomass to liquid fuels (BTL). A series of studies were conducted to analyse the environmental and economic performance of the proposed scenarios. The studies were conducted following the international standardised methods for the Life Cycle Assessment (ISO14040:2006) and Life Cycle Costing (AS/NZ4536:1999 R2014). The OpenLCA 1.4.1 and SimaPro v.8.0.4.30 software were used to model the system. Primary data were used whenever possible to calculate the life cycle inventory. Additionally, eco-invent 3.3 and AusLCI databases were used in the cases when primary data were missing. The life cycle impact assessment followed the best practice guide for conducting LCA studies in Australia. Five different environmental impact categories were assessed: Global Warming (GWP); Eutrophication (EP); Acidification (AP); Fossil depletion (FDP) and Human Toxicity (HTP). The Excel spreadsheet was used to conduct the LCC analysis and calculate the present value of the future costs.

Veneer based composite (VBC) product was first assessed as a potential material for manufacturing utility poles. The average service lifespan of a VBC pole was assumed to be 25 years. Three different disposal scenarios for the end-of-life treatment were considered: landfilling; incineration; and recycle as particleboard. Landfilling and incineration options showed better environmental performance than the recycling scenario. Although recycling and reuse of VBC poles generate extra credits due to avoid the use of virgin material, these scenarios resulted in higher overall environmental burdens due to secondary manufacturing and transportation. The VBC pole manufacturing process was identified as the biggest environmental impactor. The long-distance transportation and fossil fuel consumption due to pre-conditioning, veneer drying, and compressing contributed significantly to the GWP, AP, and FDP. The production of PF resin is an essential component in the manufacturing of VBC, which is also a major contributor to HTP.

In addition, ACQ preservative used for VBC pole treatment has a significant effect on EP due to ammonia release. An extended study was conducted to compare VBC pole to other materials conventionally used for pole manufacturing (i.e., steel and concrete). The LCA was also integrated with Life Cycle Costing Analysis and included the carbon emissions cost. The results indicated that the VBC utility pole had outstanding performance on both environmental and economic grounds. Low energy intensity and utilisation of low-value material in the manufacturing of the VBC were the most significant contributing factors to the savings in both the life cycle cost and environmental impacts. In addition, using biomass fuel to substitute fossil fuel as an energy source during VBC pole manufacturing led to significant reduction in the overall environmental impacts of the VBC pole. Sensitivity analysis indicated that service life was the most sensitive factor affecting both the environmental and economic results. Transportation distances and fossil fuel consumption also had significant effects on the LCA result. Monte Carlo analyses further revealed that despite the high levels of uncertainties in the input parameters, the overall ranking of the options remained the same with VBC pole being the best performer option followed by steel and then concrete.

Another potential option is the manufacturing of Laminated Veneer Lumber (LVL) structural products that can be used in the construction and building industry. To assess the efficiency of this option, a representative four-storey frame residential apartment building, compliant with the Australian codes was designed. LCA and LCC were conducted to evaluate the environmental and economic performance of using different building materials for the construction of the building frame, including concrete, steel and LVL products. With regard to the LVL option, different timber sources were also assessed, such as LVL manufactured from hardwood plantation mid-rotation thinned logs, mature hardwood plantations and mature softwood logs from plantations final harvesting. The result showed that the LVL building frame options presented tangible benefits from both environmental and economic perspectives. Although chemical consumption resulted in environmental burdens in all impact categories, the use of relatively low impact raw materials and wood fuel during the manufacturing process, as well as the environmental credits generated during energy recovery stage at the end of life treatment, significantly contributed to offset the negative impacts. From the economic perspective, the low construction cost of the LVL options, particularly labour and production costs, led to significant savings compared to the steel and concrete options. Among the three LVL options, the LVL manufactured from plantation thinned logs had the lowest environmental impact and LCC. This is because of the use of economic allocation method, thus attributing most of the emissions from the plantation stage to the final harvest. Monte Carlo simulations were conducted to test the effect of the uncertainties in the environmental and economic analysis. The uncertainties in transportation distance, energy usage, chemical consumptions, inflation rate, discount rate, labour cost, maintenance cost, demolition cost, and carbon cost were all taken into account. The Monte Carlo analysis showed that the performance of the alternatives was stable and the overall ranking was not affected by the uncertainties in these variables.

The results from the two studies strongly indicated that wood products made from thinned logs outperformed traditional structural materials on both the economic and environmental fronts. However, comparing the two utilisations of engineered wood products (VBC and LVL), the result indicated that using LVL in multi-level building structural frame presented greater environmental and economic benefits than the VBC utility poles. The longer durability and higher material efficiency of the LVL (in a protected environment) compared to the VBC poles (exposed to the elements) were identified as the main reason for lowering the environmental impacts of the LVL option. Hence, the LVL structural element would yield better environmental benefits and less LCC per FU of thinned logs than the VBC option.

The use of thinned logs as a source of energy was also assessed. The environmental and economic performance of six bioenergy conversion scenarios were evaluated and compared. These options included: woodchips gasification in combined heat and power plant (WCG); wood pellets gasification in combined heat and power plant (WPG); wood pellet combustion for domestic water and space heating (WPC); pyrolysis for power generation (PyEl); pyrolysis with bio-oil upgrading to transportation fuels (PyLT) and ethanol production for transportation fuel mix (EthP). Almost all the bioenergy conversion options had noticeable environmental saving, particularly on the GWP. The WCG was found to be the best option among the assessed energy scenarios, due to high energy conversion ratio and the relatively low capital and operational and maintenance costs associated with establishing and running a biomass power plant. Although wood pellet has relatively higher energy density and it is easier to be transported and stored than woodchip, the extra energy requirements during the manufacturing and transportation processes outweighed the environmental benefits gained from the improved energy density of wood pellet and led to higher life-cycle cost, lower energy return and lower environmental benefits. The PyEl option has the advantage of avoiding the secondary upgrading process of the bio-oil as the oil can be directly used (after drying and cleaning); nevertheless, the low energy conversion ratio led to high environmental burdens. Additionally, compared to the solid wood fuel options, such as WCG, WPG and WPC, the liquid fuel generation options (i.e. EthP and PyLT) presented humble environmental performance. This is mainly because of the nature of the final displaced energy as well as the energy conversion efficiency.

In the final stage, an optimisation framework was designed to investigate the optimal combination to minimise the environmental impact and the life cycle cost of the thinned logs from a life cycle perspective given the market and resource constraints. The overall performance of each option was calculated and compared based on the same functional unit (treatment of 1 Mg of thinned logs). Nevertheless, the different outputs could not offer a valid comparison for the different options. Therefore, in order to overcome this limitation, the concept of 'substituted values' (SV) was introduced to account the life-cycle cost savings from the displaced products. The 'SV' borrows the concept of 'displaced emissions' from the LCA. The SV was calculated as the difference between the LCC of the final product (alternative) and the displaced (substituted) product. Among all options, WCG had the largest environmental saving, followed by the WPG and then WPC option. From the economic perspective, utilising thinned logs to produce engineered wood products was more likely to yield higher SV than bioenergy productions.

The LCA and LCC analysis presented a conflicting result regarding the minimisation of the environmental impact and life cycle cost. However, to solve this problem using supply and constraint values averaged over the long planning period (60 years) would result in an inaccurate solution. Therefore, the problem was modified to account for strategic planning through breaking the long-term plan into multiple strategic periods. A constrained multi-period linear programming multi-objective model was then constructed to identify the optimal solution for the plantation thinning utilisation. The linear programming was solved over six-time intervals (10 years for each). Normalisation was used to covert environmental and economic indicators on the same scale to simplify a MOO problem into a SOO problem. The normalised economic and environmental factors were then assigned a random weighting, while the sum of the environmental and economic weighting is always "100%". The environmental and economic objective functions were subjected to three constraints, including total feedstock availability, final products requirements and environmental & economic performance target.

The result indicated when the economic and environmental weightings were balanced; the WCG and LVL were identified as the only two options in the solution set. The WCG was identified as the most favourable option in the first three periods. However, there was a gradual decrease in the percentage of logs allocated to the WCG option as the time went on. After the 3rd period, the LVL surpassed the WGC and became the more favourable option and eventually turned to be the only favourable option in the solution set during the sixth period. This solution was stable for the economic function weight range from 40% to 70%. In addition, for this weighting range, the allocation of logs to the LVL option was limited by the constraint of market demand. However, when the environmental weighting approached 70%, the LVL option lost favour to the WCG but retained 2% allocation due to the economic target. On the other hand, when the economic weighting factor approached 80%, the amount of thinned logs allocated to the WCG were slightly reduced and moved to the VBC option along with the time. However, due to the market demand of utility poles, the share of the VBC option remained insignificant (<5.5%) up to the fifth period. After that, the LVL option became the sole solution in the last period because the LVL option has higher SV than the VBC option. The results also indicated that the weighting assigned to each objective function might affect the allocation of resources. Nevertheless, the effect of weighting was less prominent over the realistic weighting range (e.g. environmental weighting between 30% and 60%). Only for extreme environmental weights (≤20% or ≥70%), the final solution would be significantly affected or the solution set would be changed to include or exclude options form the balanced solution.

8.2 Novel Contribution

This research study is relevant nationally and internationally as it presents a novel method to integrate LCA with LCC analysis in a multi-objective optimisation framework to identify the optimal utilisation of forestry products including multiple utilisation pathways. This study also presents a novel method to simplify the complex optimisation problem by converting the multi-objective optimisation (MOO) question to a single objective optimisation (SOO) problem. The framework introduced two novelties: (a) introduced normalisation factors for better representation of the Australian context; and (b) incorporated the potential economic credits from alternative substitutions into the LCC mimicking the accepted accounting methods of the LCA which allow credits for offset emissions.

The developed method can also be applied to different industry sectors and locations. The outcomes of this research have implications for the Australian forestry sector, particularly on the hardwood plantation management. The developed optimisation management strategy can enhance the current economic profitability of the forestry sector by developing new markets for the low value thinned logs from timber plantation while increasing the utilisation rate of wood waste hence to satisfy the global rising timber demand and mitigating the GHG emissions. The results of the study are also relevant to other timber residues and low-grade products from the softwood and pulp-wood plantations. In addition, the results of this study can be potentially used by decision-makers to make informed choices to lower the environmental impacts and life cycle cost of utility infrastructures systems and buildings. Furthermore, this study has implications for the bio-energy generation sector. This study confirms the feasibility of using forestry residue as feedstock to substitute fossil fuel energy while mitigating negative environmental impacts and achieving sustainable development strategy. Globally, this study is a small yet significant contribution towards the achievement of the United Nations Sustainable Development Goals, specifically under the Affordable and Clean Energy; Climate Action and Responsible Consumption and Production.

8.3 Limitations

This study was limited to the goal, scope and system boundaries used in the LCA and LCC. The specific limitations of this entire research were presented in each paper. The simulation model was conducted based on data available for the South-east Queensland region. Availability and accuracy of collected data published data, and publically available databases might also impose limitations. The life cycle inventories completed in this study were assumed to represent the typical or average products on the market. However, the actual environmental and economic impacts from manufacturing or production may vary depending on the technology, scale, and location. Additionally, the data used for this research were based on the current state of the art, while the technology advancement was not considered, which may also affect future results.

In the analysis, the hardwood plantation system was assumed to be in a steady state with respect to carbon stocks and management operations. Therefore, all the carbon emissions released from the burning of the harvested biomass were assumed to be taken up by the next generation of the plantation. This assumption has been used either explicitly or implicitly in most other forestry LCIs. However, if carbon uptake rate of the next generation was slower than the rate of carbon released, then this may change the results because some of these emissions will now count towards the GWP. Furthermore, the GWP in the entire study was accounted using 100 years interval, which is commonly applied in the LCA studies. However, selecting a different time horizon can significantly affect the numerical values obtained for carbon dioxide equivalents.

What more, the forestry thinning management options were only limited to these identified products, while other applications were not taken into account because of the lack of data available on them, or because the technology is still in early stages, and it is not yet commercially viable or technically mature, or the lack of perceived market, etc. However, if these options become feasible, then they need to be considered.

The environmental impact categories considered in this study were limited to GWP, AP, EP, FDP, and HTP, which have been recommended for the Australian situation. Other environmental impact categories were not included. However, if applying this study to other regions, the selection of the assessed environmental impact categories should be reconsidered to suit their local characteristic.

The life cycle assessments conducted in this study were attributional LCA (aLCA), which might not effectively describe how environmental impacts would change in response to potential policy decisions or technology development, whereas, the consequential LCA (cLCA) does. However, the policy can be changed frequently. For instance, the "emissions trading scheme" was proposed by the former government, as part of its climate change policy and it was due to commence in

2010. However, this policy was replaced with the "carbon tax" policy by the subsequent government in 2011 as the Clean Energy Act 2011, which came into effect on 1 July 2012. It was later repealed in 2014 by the next government and replaced by the Emission Reduction Fund (Teeter and Sandberg, 2016). Therefore, the cLCA may also introduce a significant uncertainty factor into the system that could lead to erroneous conclusions due to policy instability. Nevertheless, in this research, we have identified these uncertainties and simulated in Monte Carlo Analysis in response to these limitations.

8.4 Future Study and Recommendations

In the optimisation model, the weighting of the objectives is an important factor which may affect the final allocation of resources (Islam et al., 2015b). In this study, the impact of weighting was tested by varying the weight assigned to each objective function and re-running the model. Nevertheless, it is recommended that a study solicit weights from relevant stakeholders to be conducted in the future.

Additional research is required by including other valuable applications such as biochar product for soil protection, as well as other popular bio-energy products. Further research also requires to include other relevant environmental impact categories, such as solid waste, biodiversity loss, and land use impacts; hence to generate a more comprehensive environmental assessment.

Due to the existing pulpwood market is facing a major downturn; as a result, many of the originally intended pulpwood plantations are now left unharvested. The trees from the pulpwood plantation would have similar mechanical properties to the mid-rotation thinning logs. Therefore, they also present an opportunity to increase the woody biomass supply. Nevertheless, the differences in the plantation operations, as well as the differences in the physical properties of the harvested logs, may affect the overall performance of the final product. Our preliminary calculations indicated that pulpwood has slightly higher emissions and LCC than the thinned logs but still offer substantial benefits compared to current practices. However, a detailed study is recommended to evaluate optimal utilisation of this resource. This can also be extended to include short-rotation hardwood plantation management practice.

This life cycle study had a techno-economic focus and did not address the social aspect. Furthermore, the effect of new technologies or new policies should also be considered. The social aspects are a crucial component of sustainability, in regards to the biodiversity and its social contribution to the development of local employment and occupational safety and health. Therefore, an expansion of the current analysis by combining social impact is recommended to be conducted through the evaluation of human health and direct employment potential, to address the full scope of sustainable design and construction practices.

In conclusion, the optimal combination of products is time-dependent with product demand constraint being the limiting factor. Energy utilisation is likely to realise highest utility value for the first three periods while structural components would realise better utility value at the later stages. The solution showed low sensitivity to the objective weightings. A summary of the results is presented in Figure 8.1.

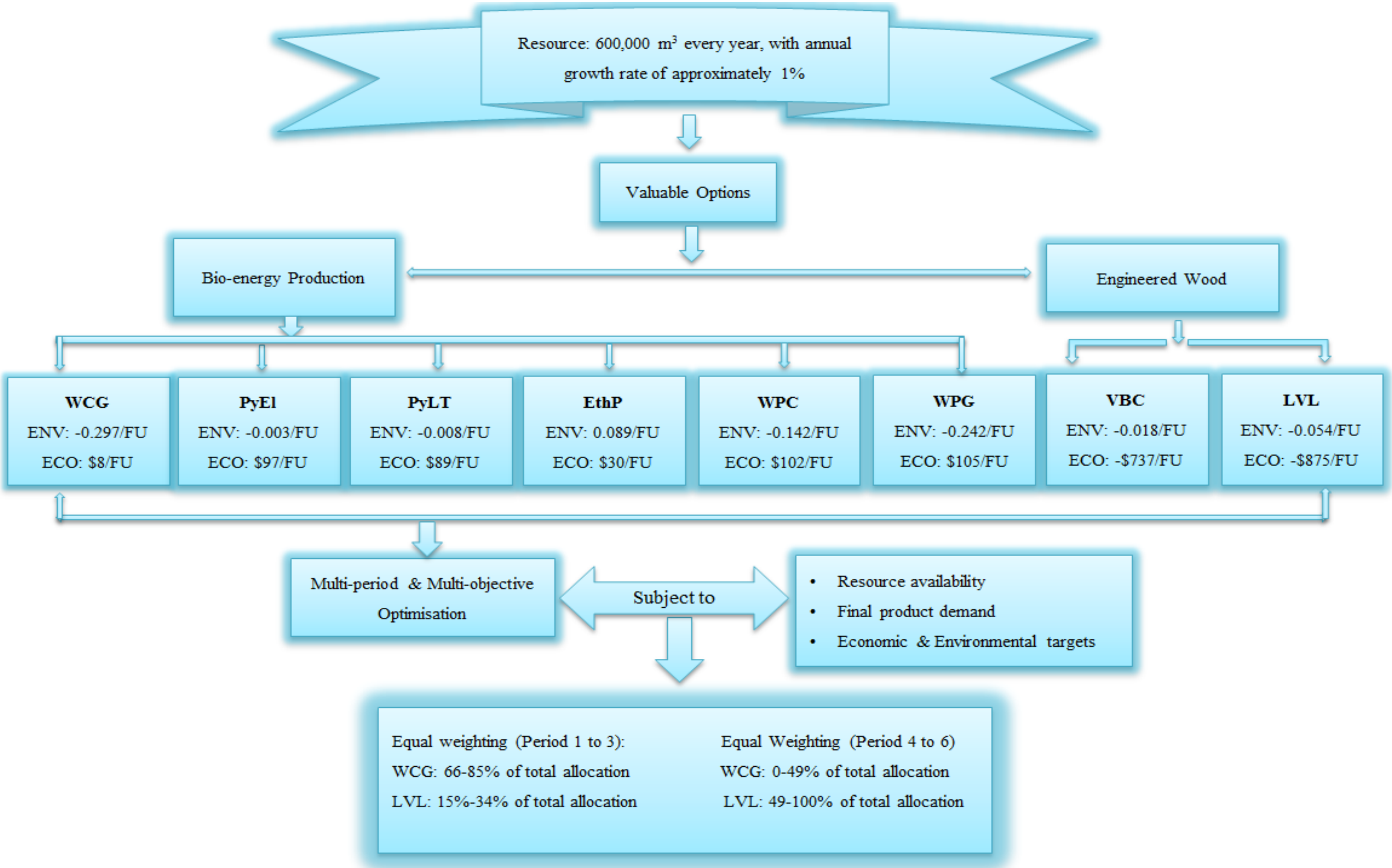

Figure 8.1 Summary of findings

CHAPTER 9 - REFERENCES

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