Application of Industrial Ecology Principles in the Bioenergy Sector in California

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ABSTRACT

Bioenergy has long been a promising solution to reduce the greenhouse gas (GHG) emissions from the transportation, electric power, and natural gas sectors, and thus, it has been promoted by policies in the interest of climate change mitigation globally. In the United States, environmental policies such as the federal Renewable Fuel Standard (RFS) and state-level Low Carbon Fuel Standards (LCFSs) and Renewable Portfolio Standards (RPSs) are in effect to regulate transportation fuels and electricity generation. While purpose-grown or energy-dedicated biomass feedstocks seem to be important to reduce GHG emissions under these policies, at present they typically provide only incremental reductions. To achieve a significant reduction of GHG, utilizing waste or residual biomass feedstock is likely required. The field of industrial ecology as long promoted waste valorization as a pathway for sustainable production systems. This concept has also been popularized more recently as part of circular economy.

The goal of this dissertation is to analyze applications through the lens of circular economy in California based on life cycle assessment (LCA) and techno-economic analysis (TEA). Studies presented in this dissertation contribute to current knowledge by developing a methodology for biomass feedstock classification to support regulatory practices, identifying the circular economy systems around bioenergy pathways, and evaluating the systems in terms of environmental and economic impacts.

The dissertation is comprised of three primary studies. In the first study, a classification system was developed for biomass feedstocks. Two decision trees were developed which are different in inclusion of an economic threshold, and they were applied to classify four biomass feedstocks: corn stover, distillers corn oil (DCO), palm fatty acid distillate (PFAD), and wheat slurry. The results showed that all four tested feedstocks were classified as the same categories by both decision trees, but there was a chance that one feedstock may be classified differently depending on its economic value. A Strengths, Weaknesses, Opportunities, and Threats (SWOT) analysis was also performed for each feedstock case, and the results

showed that there could be a challenge applying these decision trees to other biomass feedstocks that have not widely been used for biofuel production previously or that have recently emerged as biofuel feedstocks.

In the second study, two electricity generation systems using different agricultural residues, rice residues and almond residues, as feedstocks were evaluated for their economic performance, and the primary solid waste stream from these generation processes, ash, was analyzed to examine opportunities for valorization as a supplementary cementitious material (SCM). According to the results, both almond shell and rice hull were economically feasible feedstocks for electricity generation, and rice hull was selected as a target feedstock due to the abundance of rice hulls in California, better economic performance, and higher ash production per unit of electricity generation. While concrete mixtures with rice hull ash (RHA) showed lower environmental impacts than Portland cement or cement mixtures with the most common SCM, fly ash, these mixtures also showed changes in performance including reduced strength and improved chloride permeability, indicating that RHA can be used as an SCM if performance requirements for the application are met.

In the third study, LCA and TEA were performed to quantify the economic and environmental impacts of a system utilizing a waste stream from an anaerobic digester as a nutrient source for microalgae and further generating energy from the microalgal biomass. Three pathways were analyzed, producing biodiesel, electricity, or together from microalgae, respectively, and the pathway for electricity generation showed better economic and environmental performance than the other pathways.

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CHAPTER 1 Introduction

1.1 Overview

1.1.1 Bioenergy and environmental policies

Recent economic growth and development of modern society have been highly dependent on the increase of fossil fuel consumption. In the United States (US), the combustion of fossil fuels for energy generation was responsible for 74% of total greenhouse gas (GHG) emissions and for 92% of total anthropogenic carbon dioxide (CO_2) emissions in our atmosphere in 2019, the leading cause of climate change [1]. In response to climate change and concerns over energy resource security, there have been efforts to find new sustainable, renewable, and carbon-neutral energy sources that can replace fossil fuels, and bioenergy is one of the candidates. Bioenergy refers to forms of renewable energy such as biofuel, electricity, and biogas, which are generated from biomass. They have long promised to play an important role in replacing fossil fuels and reducing GHG emissions from the transportation, electric power, and natural gas sectors [2]–[4], and over time, policies have attempted to establish or promote bioenergy in the interest of GHG mitigation [5]–[7].

In the US, with the enactment of the Energy Policy Act of 2005, congress established a Renewable Fuel Standard (RFS). According to the RFS, a minimum volume of biofuels should be used in the US transportation sector each year, and the Environmental Protection Agency (EPA) is responsible for establishing and implementing this mandate. The initial RFS, also referred to as RFS1, required that a minimum of 4 billion gallons of biofuels should be consumed in 2006, and the minimum requirement should be increased to 7.5 billion gallons by 2012. However, after two years, the RFS was changed. According to the Energy Independence and Security Act of 2007, the biofuel mandate was increased, and the EPA established an expanded RFS, also known as RFS2. The RFS2 required that 9 billion gallons of biofuels to be used in 2008 and increase the minimum to 36 billion gallons by 2022 [8]. In addition, different from the RFS1, under the RFS2, biofuels are categorized into four groups (total renewable fuels, advanced biofuels, biomass-based diesel, and cellulosic biofuels), and each group is assigned its own volumetric requirement annually. For example, in the final rulemaking for the RFS 2020 standards, EPA calls for 20.09 billion gallons of total renewable fuel and 5.09 billion gallons of advanced biofuels, and the statutes in 2022 for total renewable fuel and advanced biofuels are 36 and 21 billion gallons, respectively [9]. The RFS2 also requires each group of biofuels to meet a certain minimum threshold of life cycle GHG emissions to qualify [10], [11]. The RFS essentially is a volumetric target that includes carbon intensity (CI) thresholds for qualification, measured as GHG emissions per unit of energy (i.e., g CO_2e/MJ).

California has long been a leader in environmental policy development in the US. Assembly Bill 32 (AB 32), The Global Warming Solutions Act of 2006, is a landmark California state law establishing a comprehensive program to reduce GHG emissions. The Low Carbon Fuel Standard (LCFS) is a key element of AB 32 for reducing GHG emissions from California's transportation sector by regulating life cycle GHG emissions, or CI, of transportation fuels [12]. The LCFS regulates the life cycle CI of transportation fuels and established a target of reducing the CI of transportation fuels by 10% from 1990 levels by 2020. It also regulates the carbon intensity benchmarks for fossil fuels and their substitutes by decreasing the average carbon intensities gradually (Table 1) [5]. The LCFS requires that fuel sellers achieve a certain sales-weighted CI of fuels sold. Compared to the RFS which mandates total volume, the LCFS has a higher uncertainty in its design and implementation due to the uncertainty in CIs' calculation, and this is one of its main challenges [13], [14].

Table 1. LCFS carbon intensity benchmarks for 2021 to 2030 and subsequent years (grams of carbon dioxide equivalents per unit of energy, gCO_2e/MJ) [5]

Year	Gasoline and substitutes	Diesel and substitutes	Substitute for conventional jet fuel
2021	90.74	91.66	89.37
2022	89.50	90.41	89.37
2023	88.25	89.15	89.15
2024	87.01	87.89	87.89
2025	85.77	86.64	86.64
2026	84.52	85.38	85.38
2027	83.28	84.13	84.13
2028	82.04	82.87	82.87
2029	80.80	81.62	81.62
2030 ~	79.55	80.36	80.36

While the RFS and LCFS regulate biofuels used as transportation fuels, other parts of the energy sector are also subject to GHG-related policies. Renewable Portfolio Standards (RPSs) are state-level regulations which require electricity suppliers or producers to provide a certain amount of electricity from renewable sources such as wind, solar, geothermal, and biomass. Though RPSs do not include thresholds or CI targets, they indirectly affect GHG emissions by mandating renewables. More than half of the states in the US have their own RPSs, and each state has a different target. The target can be either absolute units of electricity or a percentage share of total electricity sales. For example, California is targeting to increase the portion of renewable sources to 60% of the total by 2030 and 100% of renewables and zero-carbon sources by 2045 and Hawaii is also targeting 100% of renewable electrical energy by 2045. On the other hand, Arizona is targeting to increase to 15% of renewables by 2025 [15]–[17].

As of 2019, about 36% of retail electricity sold in California was from renewable sources, and about 3% of California’s in-state generation was from biomass, digester gas, landfill gas, and municipal solid waste (MSW) [18], [19]. In California, generating energy using biomass and waste, including agricultural residues, can address two issues: transitioning from fossil energy to renewable energy and disposal of large quantity of organic wastes including agricultural residues. However, there are several research gaps which need to be filled to assess the actual benefits and cost for improved decision-making to manage the waste material flow in California.

1.1.2 Industrial ecology and circular economy

Industrial ecology is "the study of the flows of materials and energy in industrial and consumer activities, of the effects of these flows on the environment, and of the influences of economic, political, regulatory, and social factors on the flow, use, and transformation of resources" [20]. A key concept of industrial ecology is that industrial flows should be based on circular flows mimicking those of natural ecosystems. Based on this analogy, it has long been viewed as a framework to minimize waste production and pollution, and to maximize the economic value of wastes by recycling them as inputs to other processes [21]. It also has been used as an approach to develop a sustainable manufacturing strategy by optimizing the total material cycle from raw materials to final disposal [22]. Based on these ideas, various tools and methods have emerged in the field of industrial ecology such as material flow analysis (MFA), life cycle assessment (LCA), urban metabolism, and application of a natural ecosystem paradigm to industrial systems (e.g., industrial symbiosis) [23]. More recently, the concept of circular economy has gained noticeable attraction as a framework to capture the core objectives of industrial ecology and to integrate previous strategies and concepts. The goal of circular economy is to keep the values of products, components, and materials at their highest level all the time, regardless of their end-of-life-cycle [24].

Industrial ecology and circular economy have long been applied to the bioenergy sector to quantify environmental impacts and total material flows. The concept of waste-to-energy is one example of existing work in this field. By recycling or utilizing the waste, particularly biomass and organic waste, as an energy source, the waste-to-energy approach can reduce GHG emissions by reducing fossil energy consumption as well as displacing regular waste disposal processes.

1.1.3 Life cycle assessment (LCA) and techno-economic analysis (TEA)

Life cycle assessment (LCA) is a tool for measuring the system-wide, or cradle-to-grave, environmental impacts of products and processes. Carbon footprints or CI calculations are narrow applications of LCA focusing on GHGs. This narrow form of LCA has been used especially in the US and Europe as an evaluation and decision-making tool for bioenergy and relevant policies. It has been also used for hotspot analysis in biofuel pathways, which identifies crucial sources of emissions in a pathway. Although carbon footprints provide crucial information on the relative performance of bioenergy pathways (relative to each other and other energy generation pathways), they do not capture the complete environmental burdens associated with energy generation, nor do they capture the broader issues of "sustainability," which typically includes broader economic, social, and environmental interests. Social and economic aspects, as well as environmental impacts should also be assessed in order to obtain a comprehensive understanding before making decisions in technology development and policy making [25].

While the LCA analyzes environmental impacts of a product or a system, techno-economic analysis (TEA) can be performed to analyze the economic value or performance of a product or a system. It has been used to estimate economic performance of new technologies or to optimize existing technologies or systems by identifying a hotspot which requires high monetary inputs. TEA has been also used to quantify or analyze economic uncertainty by implementing a sensitivity analysis in the TEA model.

1.1.4 LCAs on bioenergy

There is a large body of research on biomass energy including agricultural residues and wastes; however, most work so far has focused on environmental impacts or economic analysis separately. Since society, environment, and economy are three pillars supporting sustainable development, it is important to consider these impacts together for achieving sustainability. One limitation of current research is that evaluation results of bioenergy pathways can mislead decision-making process. For example, scattered evaluation results of each bioenergy product with different system boundaries, co-product allocation, or impact categories may lead to a decision that is not the best use of a resource from environmental, economic, and social standpoints. In addition, an LCA study alone may neglect the relationship between environmental and economic impacts or the consequences of environmental policies affecting the system of interest. For example, Moreno et al. [26] performed LCA on gasification of different feedstocks such as pine, eucalyptus, almond pruning, and vine pruning. The results showed that using eucalyptus as a feedstock has the lowest carbon dioxide equivalent (CO_2e) emissions for the evaluated gasification technology, but this does not mean that using eucalyptus for gasification is the most optimal solution. For example, if the cost of eucalyptus is higher than other feedstocks or the availability of eucalyptus is limited, energy production from eucalyptus may not be feasible despite its lowest environmental impacts. Moon et al. [27] analyzed the economic feasibility of two bioenergy scenarios, direct combustion and gasification, under RPSs in South Korea, and showed that gasification is more profitable. However, this result does not show that gasification is more sustainable in terms of environment or social aspects since the cost analysis was not complemented with other sustainability assessments.

Selection of a final energy product or carrier is also important. In the US, different states have different policies, and different types of energy products are regulated by different policies such as RFS, LCFS, and RPS. Even though using the same core technology and the same biomass feedstock, the impacts may differ depending on the final energy product and the location. For example, gasification of biomass can produce hydrogen or electricity depending on how the process is designed, and the impacts may differ depending on the final product: hydrogen may be regulated and subsidized under LCFS as a transportation fuel and electricity may be under RPS. The different subsidies or incentives from different policies may affect different energy markets and related value chain players. In addition, the state-level policies such as the LCFS are specific to a particular state, and thus, the impacts may differ depending on where the location is [28], [29].

Policies may also affect the economic aspects of bioenergy pathways. For example, California's LCFS includes a credit, which plays an important role in evaluating the economic feasibility of a given fuel pathway, because it incentivizes investment in renewable transportation fuels for private capital and the credit serves as a major revenue source for some producers of fuels. Depending on the credits given under the LCFS, for example, renewable natural gas produced from dairy digester may or may not be economically feasible [30].

1.2 Research Goals and Objectives

The broad goal of this dissertation is to analyze the circular economy systems around the bioenergy sector in California utilizing both LCA and TEA methods. This research aims to identify preferable bioenergy pathways in California based on the availability, key issues, and uncertainties of various agricultural residues, and to investigate possible circular economy systems in the bioenergy sector under different conditions, such as selection of feedstock and applicable regulatory policies.

This study will contribute to current knowledge in the following ways:

1) It develops an integrated model for assessing environmental and economic impacts of bioenergy pathways.
2) It develops an approach to determine the status of feedstocks as waste or non-waste materials to support rigorous and consistent practices for CI calculation in a regulatory context.
3) It identifies preferable bioenergy feedstocks and pathways in terms of environmental and economic impacts.
4) It explores the circular economy systems in California and analyzes their environmental and economic values.

This dissertation is presented in the following order:

Chapter 2 presents a development of a process to classify biomass feedstocks into categories related to waste and non-waste for policy application. The designation of the biomass feedstock is important under environmental policies such as the LCFS, because it affects the CI of the energy product from the feedstock. The process is then applied to a variety of feedstocks to assess the robustness of the tool.

Chapter 3 presents a detailed model for circular economy in California embracing the agriculture, energy, and construction sectors. The model analyzes a system with energy production from agricultural residues and recycling of by-product or waste from the energy generation.

Chapter 4 presents a detailed model for a circular microalgal biofuel production system. The model analyzes environmental and economic impacts of a system which consists of waste treatment and energy production. Based on the analysis results, the best use of microalgal biomass is determined.

CHAPTER 2 Biomass Feedstock Classification for Policy Application

2.1 Introduction

2.1.1 Background

Assembly Bill 32 (AB 32), the California Global Warming Solutions Act of 2006, established a comprehensive program including regulations and market mechanisms to reduce greenhouse gas (GHG) emissions quantifiably and cost-effectively. Under AB 32, the California Air Resources Board (CARB) was designated as a responsible agency for adopting regulations and tracking GHG emissions reduction. The Low Carbon Fuel Standard (LCFS) is a key element in the GHG reduction strategy under AB 32. The LCFS regulates the life cycle GHG emissions, or carbon intensity (CI), of transportation fuels used in California by setting a statewide reduction goal [12]. The current goal of the state is to reduce the CI of transportation fuels sold in California by at least 20% by the year 2030, which has been strengthened from the original goal of 10% by the year 2020, and this goal applies to all fuel providers such as producers, refiners, blenders, and importers. Under the LCFS, biofuels and other alternative transportation fuels have long been considered as a solution to achieve the required CI reduction. The biofuels typically reduce or eliminate fossil-derived GHG emissions from the tailpipe of vehicles, but they emit GHGs during their production. Thus, the scope of CI calculations for biofuels and alternative fuels must be inclusive of the full life cycle, including land use, production of the fuel feedstock, conversion of feedstock to energy products, generation of co-products, etc. For this purpose, life cycle assessment (LCA) has been used as an essential tool to execute the LCFS.

Biofuel feedstocks may be purpose-grown feedstocks, may be co-products or by-products, or may be residues or wastes. While purpose-grown or energy-dedicated biomass feedstocks seem to be important in achieving LCFS targets, at present they actually provide only incremental reductions in CI. Thus, alternative fuels produced from waste or residual feedstocks are required to achieve ultra-low, neutral, or even negative CI. Utilization of waste or residual feedstocks as biofuel or alternative fuel enables CI to be lowered, because the calculation of CI varies depending on the feedstock designation.

In LCA, when more than one product or service is generated from a process, some method is required to determine how to assign environmental flows among the resulting products and services, referred to as allocation. According to the ISO 14040 [31], allocation refers to a method for partitioning inputs and outputs of a process or a system between the product system of interest and one or more other systems. When a co-product is generated from a production system, partitioning or dividing the environmental flows equitably and consistently among products or services is required. However, the ISO 14040 standard does not specify a more complete taxonomy for co-products, which means that there is no clear distinction among co-products, by-products, residues, or wastes.

In implementing the LCFS or any biofuel policy aiming to reduce the CI of fuels, distinguishing among feedstock categories is crucial. Feedstocks first need to be categorized as either waste and non-waste, and then for non-waste categories, a distinction between primary and secondary products is required as well. For example, waste or residual feedstocks used for biofuel production are assigned no environmental burdens from upstream production processes other than transportation and any additional pre-processing needed prior to conversion to the biofuel. These feedstocks may even be assigned avoided burdens as credits when they are diverted from a business-as-usual (BAU) disposal process with non-negligible GHG emissions.

Waste or residual feedstocks are also assumed to have no market mediated effects, which is important because when non-waste feedstocks including co-products and by-products are used for biofuel production, environmental burdens from upstream processes are allocated to the biofuel and the diversion of the feedstock from existing uses or markets will generate market adjustments that may cause changes to the production system or increase the demand for substitutes with effects on GHG emissions.

2.1.2 Goals and objectives

The purpose of this study is to provide a clear process for agencies implementing environmental policies including clarifying the methods of CI calculations; and to establish a taxonomy for feedstock classification with a focus on the designation of waste feedstocks. Although existing literature has attempted to define and differentiate among the designations of the feedstocks [32]–[35], there are some philosophical inconsistencies on classification which affect allocation decisions. Thus, there is a need to define a logic for feedstock classification. This study develops guidelines to assist regulatory agencies in determining feedstock designations, including improved categorization of waste and non-waste feedstocks, and the identification of conditions where market-mediated effects should be considered. The taxonomy developed in this research is then used to classify several feedstocks currently in the LCFS system or potential future feedstock candidates, including agricultural residues, food-processing residuals, organic wastes such as food wastes, etc. Since an enormous number of conditions can affect the designation, it is hard to develop a complete taxonomy to categorize biofuel feedstocks. Thus, a broader decision tree approach is proposed as a method to determine feedstock designation.

2.2 Materials and Methods

2.2.1 Literature review

Nearly all regulations related to biofuels that include assessment or classification of biofuel types have had to consider waste feedstocks. Thus, a number of reports have been produced over the years that have informed or evaluated feedstock classifications for biofuel-related policies in the Unites States (US) and European countries. For the purpose of this research, the most important elements from these reports are their definitions of feedstock classification options and allocation rules resulting from the classifications. In fact, the Renewable Energy Directive (RED) [34] in the European Union (EU) and the Renewable Transport Fuel Obligation (RTFO) [36] in the United Kingdom (UK) use a double-reward method to promote waste feedstock utilization. Under the RED and RTFO, the double-reward method is applied to biofuels produced from waste feedstocks so that the biofuels receive twice the credits per unit biofuel (either volume or mass). Thus, the allocation rule associated with each feedstock classification is the critical factor for promoting valorization of waste streams for biofuel production under the LCFS and other policies seeking to promote sustainable biofuels.

Although it is important to classify feedstocks, currently there is no well-defined and complete method across policies and widely distributed models with respect to the taxonomy used for feedstock classification. These classifications are broadly relevant to LCA practices, and thus there are taxonomy rules from other sectors that can provide insights for feedstock classification. Based on the literature review of related policies and other sectors, the following classifications were selected as relevant to biofuel feedstock taxonomy: primary products, which includes primary and co-products, and secondary products, which includes by-products, residues, and wastes.

The ISO 14040 [31] contains only definitions for 'co-product' and 'waste'. However, reports from other agencies often provide additional categories, including materials determined to be residues, which are materials of no value but also no cost to producers. In the European context, under the RED [34] and RTFO [36] biofuels from wastes or residues are eligible to receive double credits. Thus, both the RED and RTFO have definitions for residue and waste. According to both the RED and RTFO, residue is defined as "[a]gricultural, aquaculture, fisheries and forestry residues, and processing residues. A processing residue is a substance that is not the end product(s) that a production process directly seeks to produce. It is not a primary aim of the production process, and the process has not been deliberately modified to produce it" [34], [36]. Waste is defined as "any substance or object which the holder discards or intends or is required to discard", but "excluding substances that have been intentionally modified to count as waste" to prevent abuse of the double-reward method [34], [36] (e.g., mixing virgin cooking oil with used cooking oil to make the mixture waste may occur if the double credit is more valuable than the fresh cooking oil to someone). The RTFO has also a definition for primary products, which includes co-products. In a report published by ICF International in 2015 entitled *Waste, Residue and By-Product Definitions for the California Low Carbon Fuel Standard* [32], feedstock classification hinges on the elasticity of supply and the value of the feedstock compared to the other products and, therefore, an in-depth economic and supply chain analysis is necessary for the feedstock classification.

In Table 2, terminology and definitions from the literature review are shown. This table demonstrates the importance of a taxonomy system. For example, a product with a significant economic value may be considered as a by-product depending on the elasticity of supply according to the 2015 ICF International report, while the RTFO defines a primary product in a similar way. Since a classification of a product may result in a significant change to an allocation method, a consistent taxonomy is required for feedstocks.

Table 2. Terminology and definitions for the feedstock classification

	Primary Products		**Secondary products**		
Source	*Primary product*	*Co-product*	*By-product*	*Residue*	*Waste*
ISO 14040 [31]		"Any of two or more products coming from the same unit process or product system"			"Substances or objects which the holder intends or is required to dispose of"
Renewable Energy Directive (RED) [34]				"Agricultural, aquaculture, fisheries and forestry residues, and processing residues. A processing residue is a substance that is not the end product(s) that a production process directly seeks to produce. It is not a primary aim of the production process, and the process has not been deliberately modified to produce it."	"Any substance or object which the holder discards or intends or is required to discard", but "excluding substances that have been intentionally modified to count as waste"
Renewable Transport Fuel Obligation (RTFO) [36]	"Materials that represent a significant economic value in relation to the main product, and that have other uses than energy applications, are likely to be considered as products. Any material that has been intentionally modified to count as a waste will be considered as a product."			Same as the RED	
ICF International [32]	"Primary products are the dominant value products of the process with elastic supply with demand."		Products of a process that have inelastic supply with demand and have a significant economic value	Products of a process that have inelastic supply with demand and have little or no economic value	
Ecofys [35]	"The primary aim of the process is the material(s) to which the process is normally optimized. Such materials should be regarded as main product or co-product" "If a material from a process constitutes an essential/considerable outcome of the process and this material has other uses than for energy applications, it should be regarded as a co-product"			"A material could be considered a residue if it has an economic value of around 15% or less compared to the total value of main products, co-products and residues."	Same as the RED

2.2.2 Feedstock category and allocation method

The importance of classifying feedstocks into different categories is related to the selection of the allocation method for each category. According to the ISO 14040, allocation refers to a method for partitioning the input and output flows of a process or a production system to the target system and other systems [31]. When more than one product is generated from a production system, a rule is required to equitably and consistently partition or divide the environmental flows among products, and this partitioning of the flows may be based on physical or value bases. While the ISO 14040 described allocation as a partitioning process, system expansion methods are also widely recognized as an allocation approach in the field of LCA [37] and choices among these methods have been debated and studied within the broader LCA literature and for biofuels in particular (e.g., [38]). The displacement method, also known as a system expansion approach or a substitution, is one of the other methods which is used to isolate the impacts of a particular product. The displacement approach is essentially a consequential method, which considers the consequences of introducing or removing a product into or out of a market, or alternatively considers the consequences of diverting a material from its current use. The displacement method is frequently used in biofuel LCA studies to account for co-products generated with biofuels. In the context of by-products and residues, a similar approach is used to account for the diversion of materials from current uses to use as a biofuel feedstock, where the consequences of diversion may lead to substitution of other products in a market fulfilling the same service or may be modeled if the product does not have a market but serves a purpose (e.g., residues left on a field for soil health).

Both partitioning-based allocation and the displacement method may be applied to either direct or indirect emissions. Direct emissions are those generated by a process and its supporting supply chains, while indirect emissions are those generated indirectly such as market mediated effects or other causal effects that the feedstock system might have on another system. Depending on the feedstock designation, direct or indirect emissions may be allocated differently.

In this study, the following allocation decisions were assumed to be used for each of the feedstock classifications. The logic laid out here helps define which feedstock classifications are required for a complete taxonomy, and the distinctions that are required. In particular, the difference between primary products (primary products and co-products) and secondary products (by-products, residues, and wastes) determines the scope of emissions attributable to products.

- Primary Products (those designated as primary products and co-products): Upstream production emissions, process emissions and applicable indirect emissions such as indirect land use change (ILUC) are attributed to primary products.

- Secondary Products (those designated as by-product, residues, and wastes): No upstream production emissions are attributed to a secondary product, but any collection and processing emissions such as transportation and pre-processing plus displacement effects from the utilization of these materials for biofuel production are attributed to the feedstock. Examples of displacement effects of each classification are provided below.

- By-products and Residues: If a by-product is currently used as animal feed, the effect of diverting the feedstock must be included. This means the demand for substitutable feeds will be modeled, and the impacts of their production will be assigned to the by-product. Alternatively, if a residue is currently left in place, such as corn stover left in field after harvest, then the consequences of removal such as increased nutrient requirements or soil carbon losses will be included.

- Wastes: No upstream production emissions are attributed to wastes, but the emissions from collection and processing steps plus the consequential emissions of diverting these materials from the current waste stream will be attributed to the feedstock. In this case, the consequential emissions are likely to be credits due to avoiding BAU disposal-related processes such as landfill operations and landfill gas generation.

Implementation of these recommended allocation methods is potentially challenging. In particular, displacement calculations can lead to a seemingly infinite number of calculations if one by-product displaces another. To solve this issue, ICF International [32] recommends calculation of displacement credits based on the primary product that would eventually substitute for the product. For example, a non-edible oil used as feed would, in theory, eventually be displaced by a virgin vegetable oil with elastic supply, and thus the displacement calculation should just be made with the virgin oil, which means that the virgin oil and the waste oil that currently has a use have the same CIs. ICF International's recommendation seems to address two challenges, the challenge of circular logic and the challenge of identifying the myriad substitutable products that could be identified. On the other hand, if the cascading effect of diverting the waste oil away from its current use is to create demand for the virgin oil, a more precise understanding of the elasticity of demand should be known. The actual implementation of this recommendation could serve as a disincentive for using secondary products that have a current use for producing biofuels, since they will ultimately be attributed the impacts of a primary product.

2.2.3 Methodology for a taxonomy

The development of a taxonomy to classify biofuel feedstocks presents a challenge because of the enormous number of conditions that can affect the decision. To develop a clear process for designation, this study proposes a decision tree to determine feedstock classification that can be applied to any feedstock used to produce biofuel under the policies and regulations such as the LCFS. Given that existing bioenergy policies have struggled with this question already, previous reports and articles have identified critical questions and some proposed definitions. Several questions were developed based on those previous studies, and the decision trees consist of a series of these questions to classify the feedstocks to one of four designations: primary product, co-product, by-product (including residue), and waste.

As described in Table 2, an economic threshold of 15% has been proposed by the report from Ecofys [35]. In addition, the RTFO [33] also had provided an economic threshold of 15% until 2018 to inform decisions around designation of secondary versus primary products. The RTFO guidance had suggested that a material which has an economic value around 15% or more of the main product in £/tonne is likely to be considered as a primary product [39]. The difference between these thresholds is that the report from Ecofys calculates the threshold based on the total product value while the RTFO calculates the threshold based on the per-unit value. Depending on the use of either total value or per-unit value, there is a potential for unintended outcomes. For example, in a microalgae production system which produces small quantities of high-value products and large quantities of biomass or microalgal oil for fuel, the biomass or oil which has a low per-unit value may be significant in terms of total value. However, the biomass or oil may be considered as a waste or residue when based on per-unit value, even if it is a primary aim of the production system. One potential solution is that the economic threshold may be implemented with both total and per-unit value thresholds, and if either are met, the product is designated as a co-product. To satisfy this condition, two decision trees were developed in this study. The two decision trees were then applied to a set of case studies that help reveal the potential benefits and drawbacks of selecting each of these decision trees. Four agricultural feedstocks including corn stover, distillers corn oil (DCO), palm fatty acid distillate (PFAD) and wheat slurry were tested with the decision trees as case studies. For each feedstock, its current use and market were analyzed, and then it was tested with the decision trees for the designation. In addition, as a way of organizing the benefits and drawbacks for each proposed decision tree, a Strengths, Weaknesses, Opportunities and Threats (SWOT) analysis is provided for each feedstock. Strengths and weaknesses are defined as those that are internal to the system, while opportunities and threats are those that are external to the system. Strengths may include conformity with existing LCA methodology and simplicity, weaknesses may include classifications that are debatable and complex, opportunities may include encouraging innovation, and threats may include risk of creating loopholes which allow cheating the classification rules for the benefits. Table 3 shows an example of a SWOT analysis in a tabular form.

Table 3. An example of a SWOT analysis

S (Strengths)	W (Weaknesses)
Clear designation of feedstock	Debatable designation Complicated implementation (e.g., data-intensive validation)
O (Opportunities)	T (Threats)
Encourages valorization of new waste streams	Easily cheated or gamed by producers Discourages valorization of waste streams

2.3 Results and Discussion

Based on the series of questions developed from analysis of the literature review, two decision trees were created. These decision trees assign primary product, co-product, by-product, or waste designations based on the questions, and the difference of two decision trees is an implementation of two different steps considering an economic threshold. The questions and response outcomes are shown in Table 4, and the decision trees with seven decision steps for determining feedstock designation are shown in Figure 1 and Figure 2.

The two branches were developed because they determine the choice of allocation method: a partitioning approach plus consequential indirect land use change or co-product displacement credits for primary products, and only consequential effects for secondary products. Since by-products and residues are treated as identical, they are not distinguished in this taxonomy and only by-products are shown in the figure of decision trees.

The first decision tree does not include an economic threshold for co-product designation, and it simply classifies any product with an economic value as a co-product. On the other hand, the second decision tree includes an economic threshold with per-unit or total value to differentiate between low value materials that have a market but do not generate significant revenue for a producer and materials that generate more significant revenue. This difference is reflected in the fifth question of both decision trees.

For decision tree one, question five asks: “Does the material have an established market or identifiable market prices? In other words, can or does the producer sell it?” Question five of decision tree two asks: “Does the material represent a significant share of value (either >15% of total value for all products, or >15% per unit relative to the primary product)?” With this question, decision tree two captures the value more precisely with an economic threshold for distinguishing between a co-product and secondary products. The economic threshold tests with 15% were drawn from current standards as discussed in the previous section, however, the intent is not to suggest that this particular threshold is correct, but rather to test the effect of implementing an economic threshold test in this taxonomy.

Table 4. Questions and response outcomes for decision trees

Q1. Has the material reached end-of-life of its intended use? (e.g., a material that can no longer function as its intended use due to degradation or other changes to quality)
Yes, Go to Q6; ***No***, Continue to Q2
Q2. Is the material the primary aim of the production system? This may be ascertained by evaluating if the system is optimized for its production or if it comprises an overwhelming share of the system's value (e.g., 85% -100%)
Yes, Feedstock is designated a Primary Product; ***No***, Continue to Q3
Q3. Is supply elastic?
Yes, Feedstock designated a Primary Product; ***No***, Continue to Q4
Q4. The production process has been deliberately modified to produce more of the material or change the properties of the material, at the expense of the primary or other co-products?
Yes, Feedstock is designated a Co-Product; ***No***, Go to Q5
Q5-1. Does the material have an established market or identifiable market prices? In other words, can or does the producer sell it?
Yes, Feedstock is designated a Co-Product; ***No***, Go to Q6
Q5-2. Does the material represent a significant share of value (either >15% of total value relative to primary product, or >15% per unit (e.g., $/kg) relative to the primary product)?
Yes, Feedstock is designated a Co-Product; ***No***, Go to Q6
Q6. Does it have any value to the producer? Value may be monetized or not (e.g., an agricultural residue that provides soil carbon maintenance)
Yes, Product is designated as a By-Product; ***No***, Continue to Q7
Q7. In the absence of its use as a biofuel feedstock, does the producer incur a net cost to manage the material (e.g., for final disposal or removal from the production site)?
Yes, Feedstock designated as a Waste; ***No***, Product is designated as a By-Product.

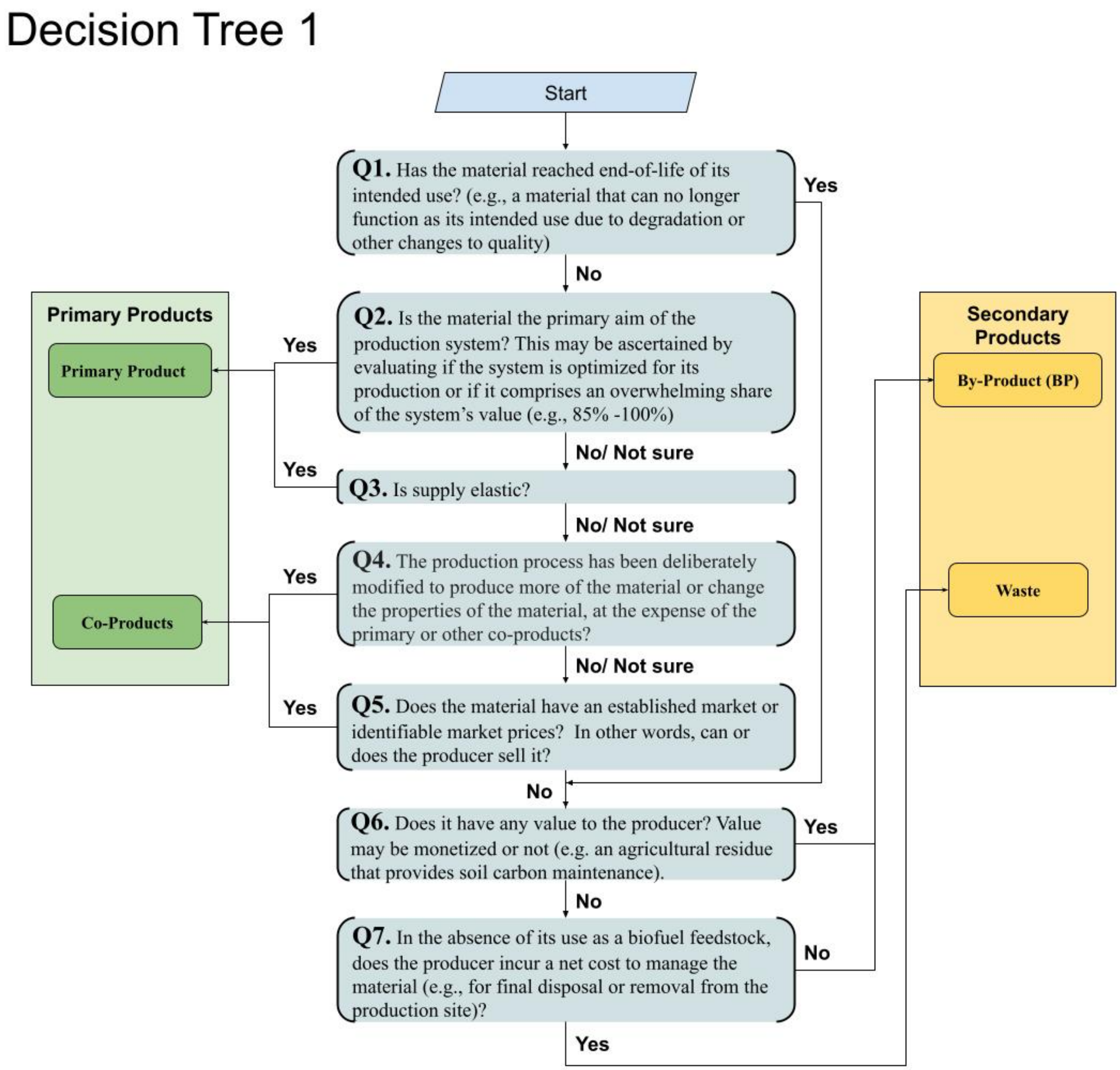

Figure 1. Decision tree 1. The existence of an established market or an identifiable market price serves as the critical distinction between co-product and by-product.

Decision Tree 2

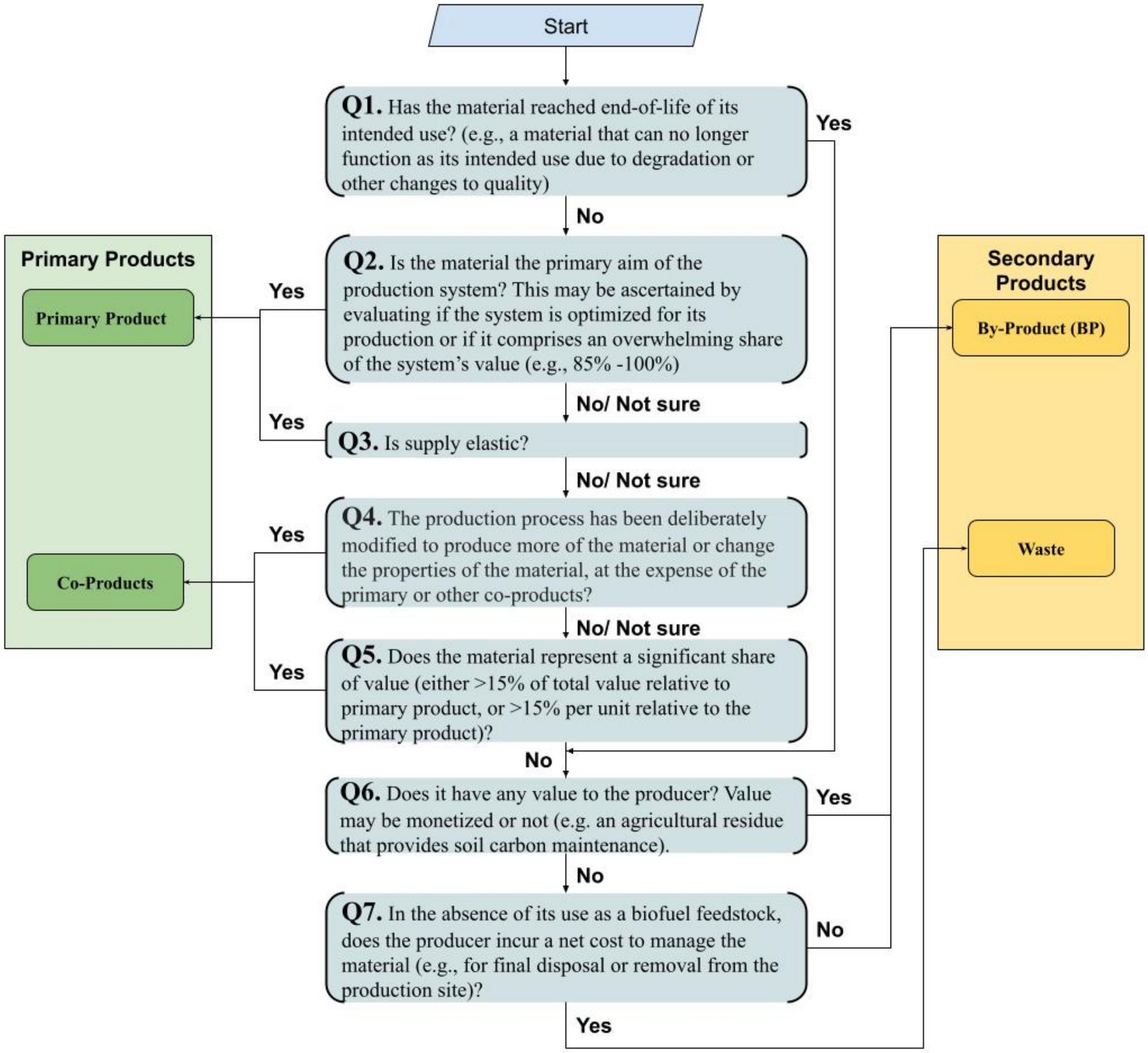

Figure 2. Decision tree 2. An economic threshold of >15% in either total or per unit value serves as the critical distinction between co-product and by-product

2.3.1 Corn stover

2.3.1.1 Background

Corn stover refers to leaves and stalks, and sometimes cobs, left on the field after corn harvest. Corn stover management decisions by farmers vary depending on many factors. It can be left on the field for soil and nutrient management, or it may be harvested and collected as well to be used for animal feed and bedding, packaging, or in mushroom compost. It can also be used as a feedstock for electricity and heat generation, chemical production, and biofuel production [40], [41].

2.3.1.2 Current market

While there are no consistently collected data on corn stover production volumes in the US or globally, corn stover production can be inferred from corn production [42]. Corn stover is generally produced at about a one to one mass ratio to corn [43]. According to the United States Department of Agriculture (USDA), the production of corn for grain was 360 million metric tons (MMt) in 2020 in the US, so it can be reasonably assumed that corn stover production was about 360 MMt in 2020 in the US [44]. The USDA also publishes the market price of corn stover weekly for several states (Iowa, Minnesota, Nebraska, South Dakota and Kansas) under the National Biomass Energy Report [45]. Since 2018, the average price for corn stover has fluctuated between \$47/tonne (field dry, unstated moisture content) and \$80/tonne, and it was \$53/tonne as of December 31st, 2020. At the same time, the average price of corn grain (wet basis) has fluctuated between \$123/tonne and \$164/tonne since 2018, and it was \$156/tonne as of December 2020 (Figure 3). Although it fluctuates, the price of corn stover is more than 15% of the price of corn grain in terms of per-mass economic value. Depending on the season, the ratio varies between 29% and 56% (Figure 4). However, in terms of total economic value, it can vary depending on the farmers' harvesting levels. In addition, the market prices shown in Figure 3 may not represent other markets outside of the listed states, because there is geographic variability in markets. Thus, there may be some markets with high demand for stover and others with little or no value.

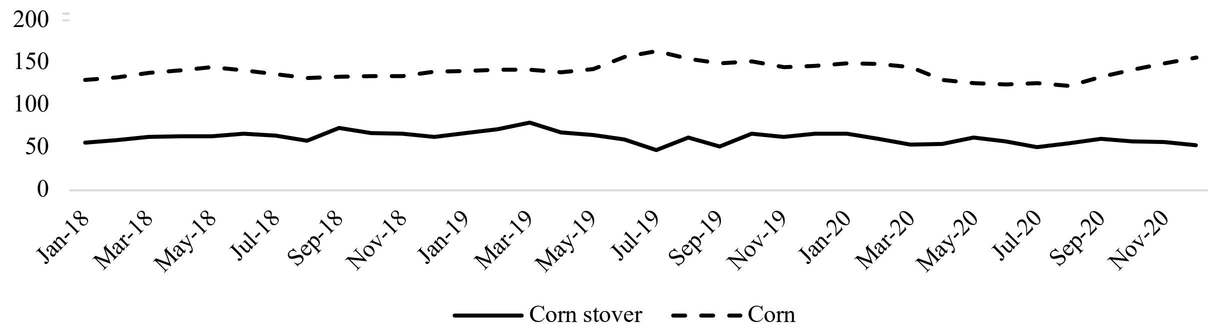

Figure 3. Weekly price change of corn and corn stover ($/tonne)

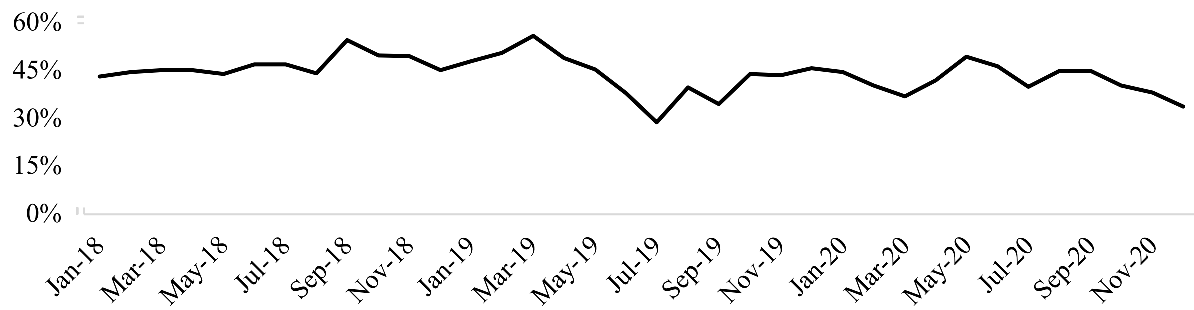

Figure 4. Ratio between corn price and corn stover price

2.3.1.3 Decision tree results and SWOT analysis

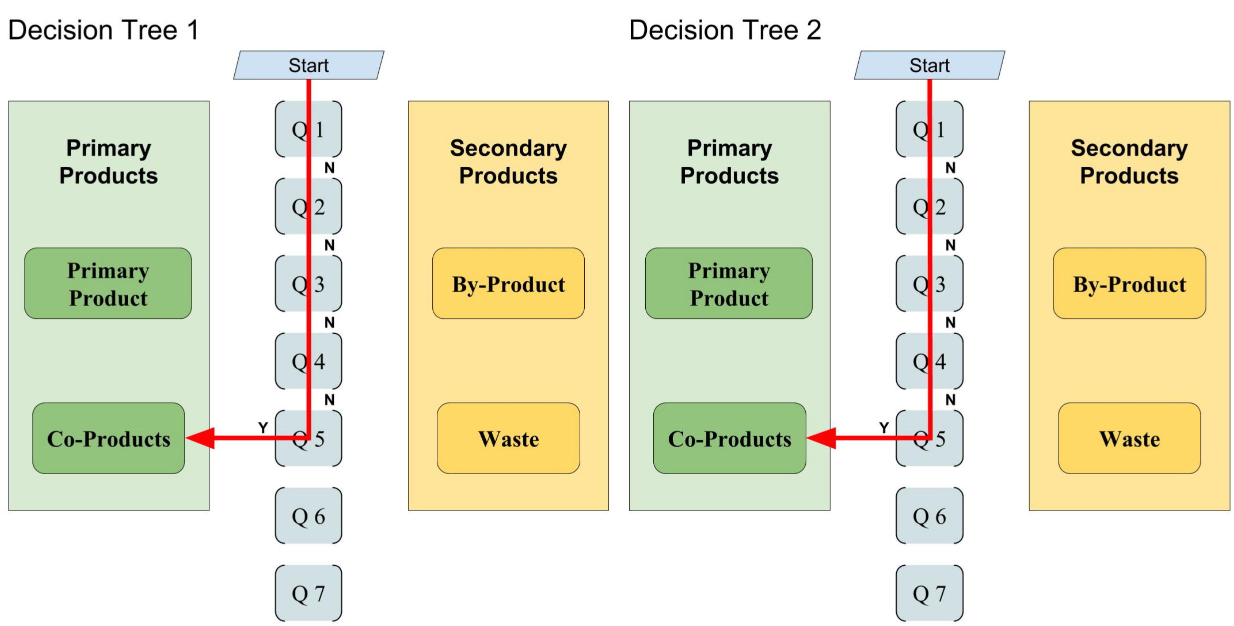

Figure 5. Decision tree results for corn stover

Corn stover is designated as a co-product in both decision trees (Figure 5). The questions and responses are detailed in Table 5, and the SWOT analysis results are summarized in Table 6.

Table 5. Detailed questions and response outcomes for corn stover

<table>
<tr><td colspan="2">Q1. Has the material reached end-of-life of its intended use? (e.g., a material that can no longer function as its intended use due to degradation or other changes to quality)</td></tr>
<tr><td>No</td><td></td></tr>
<tr><td colspan="2">Q2. Is the material the primary aim of the production system? This may be ascertained by evaluating if the system is optimized for its production or if it comprises an overwhelming share of the system’s value (e.g., 85% -100%)</td></tr>
<tr><td>No</td><td>The process is optimized for corn grain yield.</td></tr>
<tr><td colspan="2">Q3. Is supply elastic?</td></tr>
<tr><td>No</td><td>The supply of corn stover is inelastic because the level of crop planting is determined by the corn grain market, not corn stover market.</td></tr>
<tr><td colspan="2">Q4. The production process has been deliberately modified to produce more of the material or change the properties of the material, at the expense of the primary or other co-products?</td></tr>
<tr><td>No</td><td>There is no deliberate increase in stover production, and there has been no effort to change the quality or properties of stover.</td></tr>
<tr><td colspan="2">Q5-1. Does the material have an established market or identifiable market prices? In other words, can or does the producer sell it?</td></tr>
<tr><td>Yes</td><td>There are markets with identifiable prices for corn stover in Iowa, Minnesota, Nebraska, South Dakota, and Kansas. Not every region may have identifiable markets.</td></tr>
<tr><td colspan="2">Q5-2. Does the material represent a significant share of value (either >15% of total value relative to primary product, or >15% per unit relative to the primary product)?</td></tr>
<tr><td>Yes</td><td>Unit price of corn stover varies between 29% and 56% of the price of corn grain.</td></tr>
</table>

Table 6. SWOT analysis result for corn stover

S	W
If current practices are known (i.e., stover currently harvested and sold into the market, or stover is currently left on-field), designation is clear and identical for both decision trees.	Designation for corn stover may change based on geographic location or time (i.e., if a market is transient, or different conditions exist at the field level, which may make stover a waste in one place, and a by-product in another).
O	**T**
If corn and stover are treated as co-products, it could incentivize farmers or other stakeholders to optimize the harvesting system and residue management for co-production.	Possibility of gaming the system because of heterogeneity in current practices and markets for stover, meaning producers can argue for the designation that most benefits them.

2.3.2 Distillers corn oil (DCO)

2.3.2.1 Background

Corn grain is composed of more than 70% of starch and thus has been the most common feedstock for ethanol production by fermentation in the US. The remaining portion of the corn grain, comprised of proteins, oils, and fibers, are left over after ethanol distillation process, called distiller's grains with solubles (DGS). The oil portion, which is about 10% by weight of this remaining material can be separated out, and it is called distillers corn oil (DCO). DCO can be separated via centrifugation when the DGS is still wet, or it can be separated after drying of DGS to produce dried DGS (DDGS). Figure 6 shows the production routes for the DCO. To distinguish DCO from other types of corn oil products, it is also called industrial or technical corn oil, since it is inedible. Because DCO is inedible for humans, it is typically used as animal feed or feedstock for biodiesel production [46]–[51].

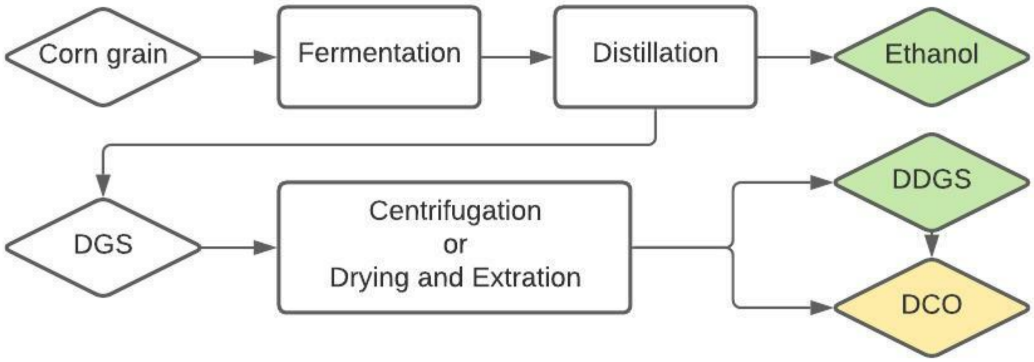

Figure 6. Production process of DCO from corn grain

2.3.2.2 Current market

DCO is currently traded in local markets such as Iowa, the Eastern Corn-belt, Nebraska, and South Dakota, and the USDA publishes the price of DCO in each market daily under USDA Daily Ethanol Report as well as the price of ethanol [52]. As of December 31, 2020, the average price of DCO was \$0.80/kg, and the average price of ethanol was \$0.34/liter.

2.3.2.3 Decision tree results and SWOT analysis

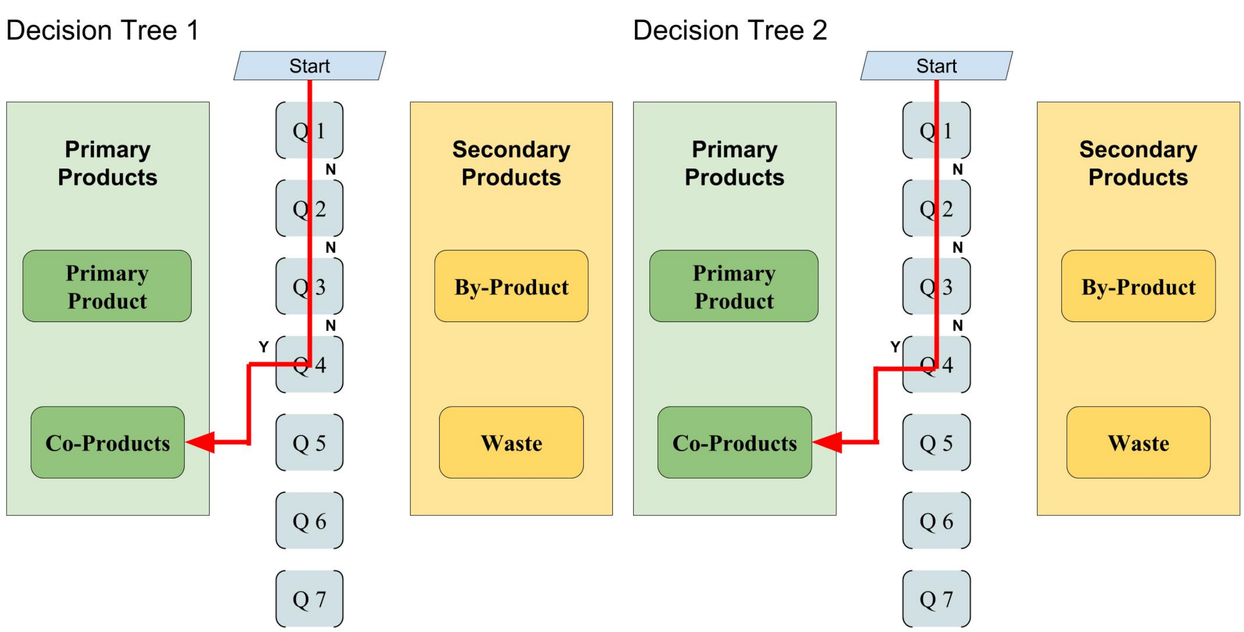

Figure 7. Decision tree results for DCO

DCO is designated as a co-product in both decision trees (Figure 7). The questions and responses are detailed in Table 7, and the SWOT analysis results are summarized in Table 8.

Table 7. Detailed questions and response outcomes for DCO

Q1. Has the material reached end-of-life of its intended use? (e.g., a material that can no longer function as its intended use due to degradation or other changes to quality)	
No	
Q2. Is the material the primary aim of the production system? This may be ascertained by evaluating if the system is optimized for its production or if it comprises an overwhelming share of the system's value (e.g., 85% -100%)	
No	The process is aimed at producing ethanol.
Q3. Is supply elastic?	
No	The supply of DCO is a function of the quantity of ethanol produced.
Q4. The production process has been deliberately modified to produce more of the material or change the properties of the material, at the expense of the primary or other co-products?	
Yes	The production process has been deliberately modified to produce DCO, and in so doing the quantity and characteristics of DGS are changed. DCO is a **co-product.**

Table 8. SWOT analysis result for DCO

S	W
Designation for DCO is clear and does not vary based on the selection of decision trees.	None identified.
O	T
None identified.	This finding goes against current practice of designating DCO as a by-product, and it affects CIs of DCO-based biofuels. It may discourage the use of DCO as biofuel feedstock.

2.3.3 Palm fatty acid distillate (PFAD)

2.3.3.1 Background

Palm fatty acid distillate (PFAD) is produced during the palm oil refining process. Crude palm oil (CPO) is refined to 'refined, bleached and deodorized palm oil (RBD PO)' and PFAD is the remaining portion of the deodorization step of the refining process [53], [54]. It is solid at room temperature and brown in color. CPO contains about 3-5% of free fatty acids (FFAs) by weight, and the FFA content of palm oil may vary depending on damages to fruit bunches and aging. From the CPO, more than 99% of the FFAs are removed in the form of PFAD. PFAD consists of 81.7% of FFA, 14.4% of glyceride, 0.8% of squalene, 0.5% of vitamin E, 0.4% of sterol and 2.2% of other substances [55]. Because PFAD contains significant quantities of FFAs, they can be converted to fatty acid methyl esters (FAMEs), which is also known as biodiesel [56]. PFAD is also used as livestock feed, as raw material for oleochemicals or soap, and as fuel for industrial boilers or engines [51], [57].

2.3.3.2 Current market

Although PFAD is produced at only 3-5% of CPO by weight, the size of the PFAD market is considerable due to the size of the palm oil market. However, there are no official production statistics about PFAD [55]. Thus, the market size of PFAD must be estimated based on the size of the palm oil market. Since CPO consists of 3-5% of FFA by weight, which is the main component of PFAD, it is reasonable to assume that the production of PFAD is approximately 4% of the production of CPO. Meanwhile, the yield of RBD PO from CPO is about 93% [58].

According to the Food and Agriculture Organization (FAO) of the United Nations, 71.7 million metric tons of palm oil were produced globally in 2018, which means that approximately 2.9 million metric tons of PFAD were produced in 2018. The major producers of palm oil are Indonesia and Malaysia which represent 56.6% and 27.2% of total production, respectively.

According to the Malaysian Palm Oil Board, the price of CPO was $502.04/tonne, the price of RBD PO was $542.25/tonne, and the price of PFAD was $436.36/tonne in 2019 (Figure 8). The price of

PFAD is about 80% of RBD PO, which is the target product of this process. However, even though the difference between the price of RBD PO and PFAD is declining, the total economic value of PFAD is still less than RBD PO due to the production size. The total economic value of PFAD compared to RBD PO remains less than 5% [59]–[65].

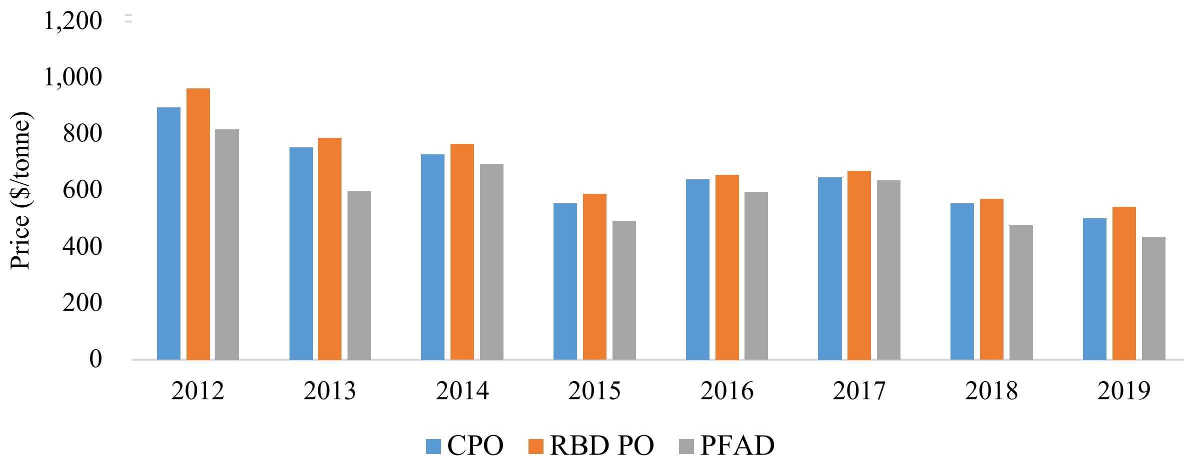

Figure 8. Price of CPO, RBD PO, and PFAD by year (CPO: Crude Palm Oil, RBD PO: Refined, Bleached, and Deodorized Palm Oil, PFAD: Palm Fatty Acid Distillates)

2.3.3.3 Decision tree results and SWOT analysis

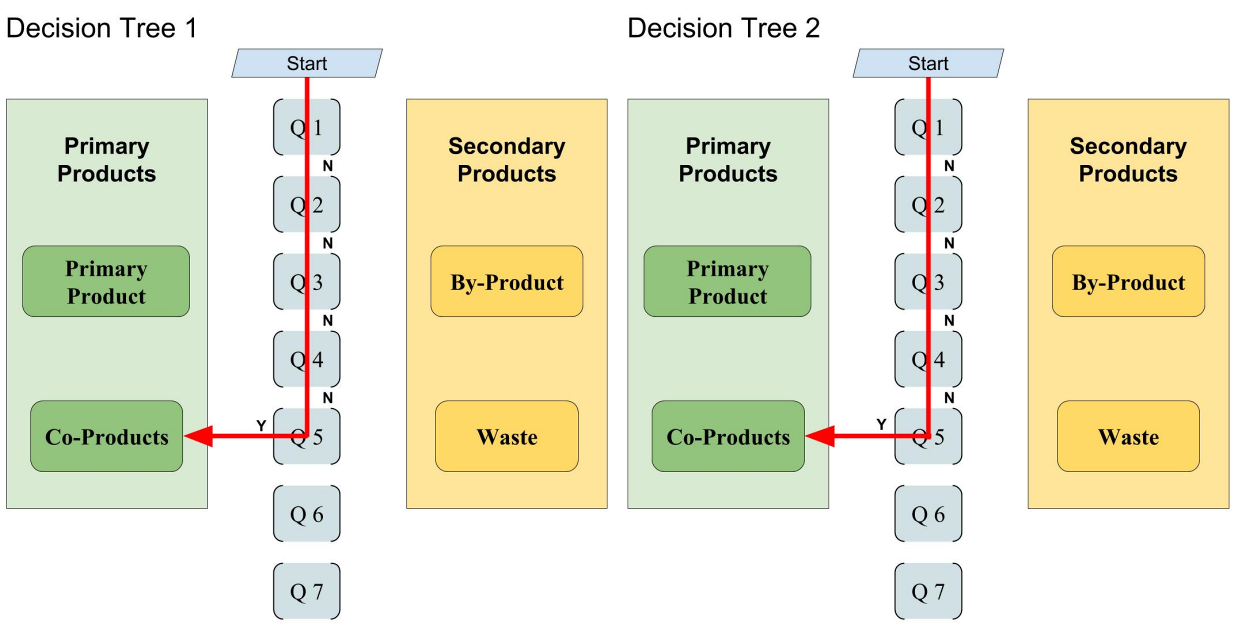

Figure 9. Decision tree results for PFAD

PFAD is designated as a co-product in both decision trees (Figure 9). The questions and responses are detailed in Table 9, and the SWOT analysis results are summarized in Table 10.

Table 9. Detailed questions and response outcomes for PFAD

Q1. Has the material reached end-of-life of its intended use? (e.g., a material that can no longer function as its intended use due to degradation or other changes to quality)	
No	
Q2. Is the material the primary aim of the production system? This may be ascertained by evaluating if the system is optimized for its production or if it comprises an overwhelming share of the system's value (e.g., 85% -100%)	
No	The process is optimized for RBD PO (Refined, Bleached, and Deodorized Palm Oil), and PFAD is the remaining portion of that process.
Q3. Is supply elastic?	
No	PFAD price changes will not induce greater production of PFAD. RBD PO (and thus CPO as a precursor) are the products with elastic response to demand. Because the price of PFAD is typically lower than the price of RBD PO (at least for the last 12 years) and the total production and economic value of PFAD is significantly smaller than the value of CPO and RBD PO, it will not be a driver for production volume.
Q4. The production process has been deliberately modified to produce more of the material or change the properties of the material, at the expense of the primary or other co-products?	
No	The process is modified to produce more RBD PO, not PFAD.
Q5-1. Does the material have an established market or identifiable market prices? In other words, can or does the producer sell it?	
Yes	PFAD can be used as livestock feed, as raw materials for other products, and as fuel, and there is a well-established market for these needs. PFAD is a **co-product.**

Q5-2. Does the material represent a significant share of value (either >15% of total value relative to primary product, or >15% per unit relative to the primary product)?	
Yes	While the total economic value of PFAD compared to that of RBD PO is about 3-4% due to the relatively small volume of PFAD the unit price of PFAD is about 80% of RBD PO.

Table 10. SWOT analysis result for PFAD

S	W
None identified.	The results for decision tree 2 hinge on the selection of an economic threshold based on unit price and not total economic value.
O	T
None identified.	None identified.

2.3.4 Wheat slurry

2.3.4.1 Background

Wheat slurry is a product generated from wheat processing facilities. During wheat starch and gluten production, the process has several washing steps for generating A-type starch and gluten. The liquid effluents from these washing steps are called wheat slurry. The target products for washing steps contain A-type starch and gluten, while wheat slurry contains some pentosane and high contents of B-type starch. Before the initial separation, the feed contains approximately 10% of water, 20% of pentosane, 35% of B-type starch (with gluten and fiber), and 35% of A-type starch by volume. Among these, normally A-type starch and gluten are packaged and sold in market, and wheat slurry which contains B-type starch and pentosane may be used for ethanol production through fermentation or may be land-applied [66], [67]. Figure 10 shows the production routes for wheat slurry.

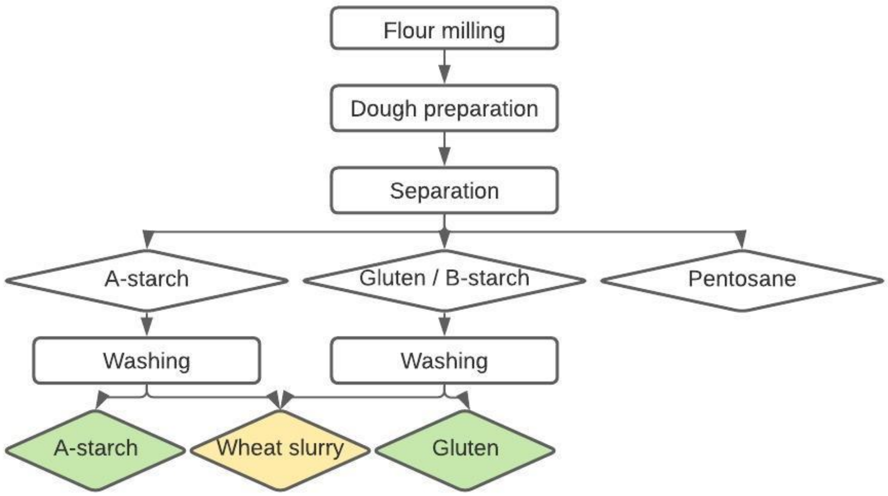

Figure 10. Production routes for wheat slurry

2.3.4.2 Current market

There is no clear or identifiable market and identifiable prices for wheat slurry.

2.3.4.3 Decision tree results and SWOT analysis

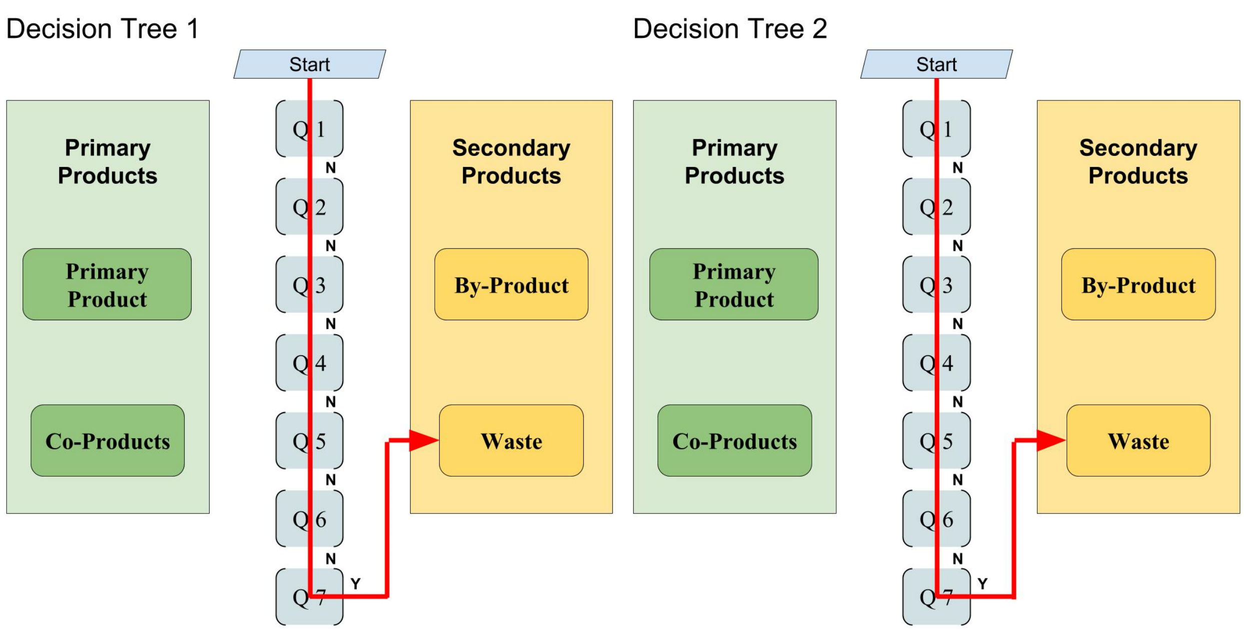

Figure 11. Decision tree results for wheat slurry

Wheat slurry is designated as a waste in both decision trees (Figure 11). The questions and responses are detailed in Table 11, and the SWOT analysis results are summarized in Table 12.

Table 11. Detailed questions and response outcomes for wheat slurry

Q1. Has the material reached end-of-life of its intended use? (e.g., a material that can no longer function as its intended use due to degradation or other changes to quality)	
No	
Q2. Is the material the primary aim of the production system? This may be ascertained by evaluating if the system is optimized for its production or if it comprises an overwhelming share of the system's value (e.g., 85% -100%)	
No	The process is optimized for A-type starch and gluten production.
Q3. Is supply elastic?	
No	The production of wheat slurry depends on the production of A-type starch and gluten.
Q4. The production process has been deliberately modified to produce more of the material or change the properties of the material, at the expense of the primary or other co-products?	
No	
Q5-1. Does the material have an established market or identifiable market prices? In other words, can or does the producer sell it?	
No	
Q5-2. Does the material represent a significant share of value (either >15% of total value relative to primary product, or >15% per unit relative to the primary product)?	
No	
Q6. Does it have any value to the producer? Value may be monetized or not (e.g., an agricultural residue that provides soil carbon maintenance).	
No	There are many cases where it is land-applied, and in theory could substitute for irrigation waters and provide nutrients to soils. However, it is often the least-cost solution for disposal, and it is unclear whether it benefits the land where it is applied.

Q7. In the absence of its use as a biofuel feedstock, does the producer incur a net cost to manage the material (e.g., for final disposal or removal from the production site)?	
Yes	The product is designated as a **waste**.

Table 12. SWOT analysis result for wheat slurry

S	W
Wheat slurry is clearly designated as a waste	None identified.
O	T
Potential for market development.	Possibility to cheat exists within washing process to increase the quantity of slurry.

2.3.5 Discussion

All four tested feedstocks in this study were classified as the same categories by both decision trees, but there was a chance that one feedstock (PFAD) may be classified as two different categories depending on its economic value (total versus per unit). In a case that a feedstock has an established market but has a lower value than the economic threshold in terms of both total value and relative per unit value, the feedstock may be classified differently either as co-product or by-product. In addition, a change of economic threshold has no effect on the results for PFAD unless the threshold becomes less than 3% or more than 80%. Depending on the change to either 3% or 80%, PFAD will be designated as a co-product or a by-product, respectively, regardless of its economic value. However, these values of 3% or 80% may only be applied to PFAD in 2019, because production and market prices vary continuously. Also, this conclusion is inapplicable to other feedstocks. For example, a designation of corn stover may change to by-product if a threshold at 35% is used, which would also vary depending on the market prices (e.g., 29%-56%). Thus, future research should determine the appropriate threshold for a particular policy (like the LCFS) if an economic threshold is adopted, and it requires two important data points: available feedstocks and market prices, because they are required to calculate total economic value ratio $(\frac{TV_t}{TV_p})$ and the unit price ratio (r_p) which are compared to an economic threshold as described by Equation 1.

Equation 1:

$$TV_p(\$) = M_p(tonne) \times P_p(\frac{\$}{tonne}), \quad TV_t(\$) = M_p(tonne) \times r_m \times P_p \times (\frac{\$}{tonne}) \times r_p$$

$$TV_x: Total\ economic\ value\ of\ X, M_x: Production\ of\ X, P_x: Unit\ price\ of\ X$$

$$(x = p: primary\ product, x = t: tested\ product)$$

$$r_m: Production\ ratio\ \left(\frac{M_t}{M_p}\right), r_p: Unit\ price\ ratio\ (\frac{P_t}{P_p})$$

There are many other available biomass feedstocks such as tallow, used cooking oil, rice straw, etc. that were not considered in this study, but which can be evaluated using the same approach. For example, investigation on their availabilities and production processes will provide M_p and r_m, and investigation on market prices will provide P_p and P_t in Equation 1. While a variation of r_m is limited (e.g., close to 1:1 mass ratio for corn grain and stover), M_p, P_p and r_p may vary continuously depending on market conditions. After collecting the data for available feedstocks, the appropriate threshold may be determined based on the ratios $(\frac{TV_t}{TV_p}, r_p)$.

2.4 Conclusions

The classification of feedstocks as secondary products (wastes and by-products) or primary products (primary products and co-products) plays an important role in determining their CI, so unbiased and effective methods for feedstock classification are required. This study developed two decision trees for feedstock designation and four feedstocks were tested by the decision trees. The critical difference between the two proposed decision trees is the inclusion of an economic value threshold to differentiate between co-products and secondary products (by-products and wastes). This threshold was tested at 15% of the value of the primary product, where percent value can be either be a unit basis (e.g., \$/kg) or based on the percent of total production value relative to the primary product. While developing both decision trees, a significant focus of the decision process is distinguishing between primary and secondary products, since this step determines the allocation method used to assess the impacts of feedstock production and utilization. Once classified as a secondary product, the difference between waste and by-product is a function of its current fate: either a low or no value use, neither use nor disposal, or disposal.

The results from SWOT analysis of four feedstocks show that there could be a challenge applying these decision trees to other feedstocks that are not widely used for biofuel production or that have recently emerged as biofuel feedstocks. Because of heterogeneity in current practice and markets for feedstocks, there is a possibility of cheating and gaming, which are the greatest weaknesses and threats. In these cases, initial research may be required to describe and provide evidence for current uses. For those feedstocks which are already used for biofuel production widely, data may be more available and thus the burdens of data collection may be reduced.

CHAPTER 3 Economic and Environmental Performance of Recycling Agricultural Residues in California

3.1 Introduction

California has achieved aggressive greenhouse gas (GHG) emissions targets as laid out in AB 32, the Climate Change Solution Act, achieving the state's 2020 target of 430 million metric tons of carbon dioxide equivalents (MMt CO_2e) well in advance of 2020. The greatest reductions have been in the electric power sector, the second largest contributor of GHG emissions in the state, where emissions have fallen from a peak in 2001 of more than 102 MMt CO_2e to less than 60 MMt CO_2e in 2019. The largest emitter of GHGs, the transportation sector, has also fallen, though not by nearly as much, and the third largest sector has fallen as well. At the same time, the fourth largest contributor, the agricultural sector has essentially remained unchanged over the last two decades, contributing around 30 MMt CO_2e per year [68].

While each sector can be examined on its own; in fact, these broad sectors are inexorably intertwined and interdependent. Effective solutions for reducing emissions may benefit from these connections across sectors. This chapter explores this very opportunity by examining a case study of how agricultural residues could be utilized as feedstocks for electricity generation while simultaneously producing downstream co-products that could serve as inputs to the industrial sector with the goal of reducing system-wide GHG emissions.

3.1.1 Background

3.1.1.1 Agriculture in California

California is the largest agricultural producer in the United States (US), producing about 13% of the nation's total in cash receipts [69]. California produces over a third of the country's vegetables and two thirds of the country's fruits and nuts [70]. Both nationwide and in California, bearing acreage devoted to agricultural products and the generation of agricultural residues have been increasing. With these increases, California agriculture and agricultural product processing is also becoming more energy- and resource-intensive, causing environmental issues such as an increase in GHG emissions and primary energy consumption [71], [72]. According to the California Air Resources Board (CARB), agriculture is the 4th largest economic sector in terms of GHG emissions in California, contributing 8% of total emissions in California. In the agricultural sector, GHG emissions can occur in various ways; not only from energy uses, but also from non-energy processes and activities.

GHG emissions can come from livestock enteric fermentation, livestock manure management, crop growing, and many other agricultural activities [73]. For example, for crop growing and harvesting, the emissions are from both energy-uses and non-energy-uses. Fossil energy is required for water pumping, farm equipment, transport, and processing, and as a result, lead to GHGs and pollutant emissions that affect air and water quality. At the same time, non-energy processes also emit GHGs and pollutants, such as fertilizer application or soil disturbance, and the current treatment of crop residues, which includes in-field burning with significant air pollution impacts and incorporation into the soil with release of methane and nitrous oxide. One solution to address both energy use and disposal of residues is to generate energy using agricultural residues.

The combustion of solid biomass such as agricultural residues has been widely used for renewable and low-carbon heat and electricity generation for decades. In California, about 3% of electricity generated is from biomass [19], [74]. Similar to coal burning, the combustion of solid biomass generates ash from the inorganic components of the biomass, and it has a potential to be used as a supplementary cementitious material (SCM).

3.1.1.2 Supplementary cementitious material (SCM)

The construction industry requires massive amounts of raw materials and energy, and subsequently, produces large amounts of GHG emissions. Concrete is one of the most widely used materials in the construction of infrastructure and buildings and is consumed at a rate of 1.4 cubic meters (m^3) per person annually [75]. The production of cement, the binding material in concrete, accounts for 5-8% of global carbon dioxide (CO_2) emissions, emitting between 0.66-0.82 kilograms of carbon dioxide equivalents (kg CO_2e) per kilogram of cement production [75], [76].

In the US, the cement production sector alone was responsible for 40.3 MMt CO_2e in 2018 [77]. Due to the high carbon intensity (CI) of cement, and the attendant impacts embodied in concrete, there have been efforts to decrease the GHG emissions and other environmental impacts from concrete, cement, and the construction industry more broadly [78], [79]. The use of SCMs such as coal fly ash or slag to displace some cement content in concrete is one widely used solution to decrease the environmental impacts and improve performance characteristics of concrete. However, the availability of such materials is becoming highly constrained in certain areas because of closures of coal power plants [80]. In California, which is the second largest producer of cement in the US, about 1 million metric tons (MMt) of fly ash are consumed as SCM annually [81], but all fly ash is imported since coal is not used for electricity generation in California. The demand for SCMs in California is expected to increase along with a statewide 65% increase in the demand for cement by 2050 [82].

3.1.2 Goals and objectives

The goal of this study is to address two important issues in California, utilizing agricultural residues as feedstocks to generate electricity and supplying SCM with reduced environmental impacts. Two objectives were set for the goal in this study: analyzing the performance of two feedstocks for electricity generation and analyzing ash utilization as SCM for one selected feedstock. A workflow for this study is shown in Figure 12. Among the available biomass resources, particularly agricultural residues, this study considers almond hulls/shells and rice hulls as potential candidates for electricity generation feedstocks, and one feedstock is selected as a potential candidate which provides ash to replace the conventional SCMs. Since California is the largest producer of almonds globally and the second largest rice-growing state in the US, the utilization of these agricultural residues can address the goal of this study. In addition, both agricultural residues have the advantage of being harvested and transported along with their respective primary products, rice grain and almond kernel, reducing the collection and transport burdens for their use as energy products.

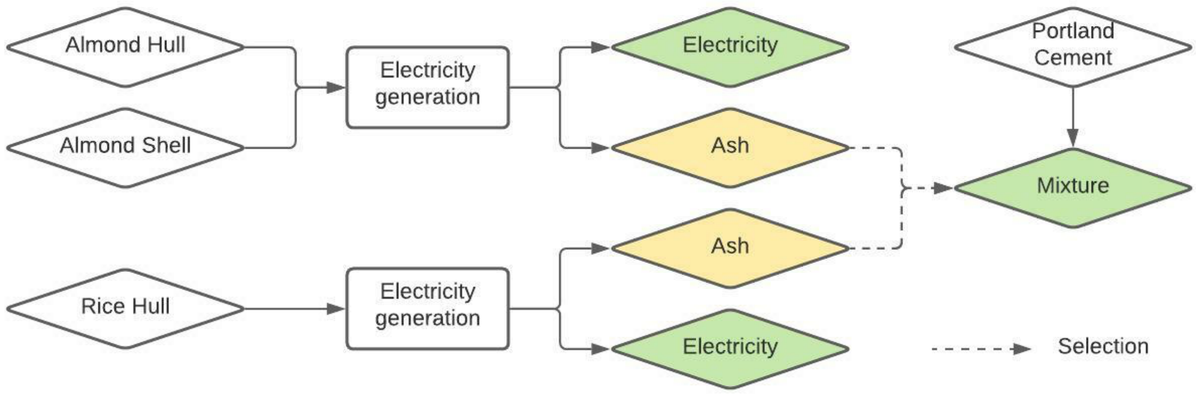

Figure 12. Workflow in this study. Two agricultural residues are evaluated as electricity generation feedstocks. One ash product from the electricity generation is selected as an SCM candidate and tested.

3.2 Materials and Methods

The approach for this research is to estimate the economic and environmental performance of a prospective system with TEA and LCA modeling that is referenced to commercial scale electricity generation. To achieve this, experimental results are used within a techno-engineering model that assumes full-scale operations and provides the backbone for TEA calculations and LCA modeling.

3.2.1 Feedstock selection

For this task, various agricultural wastes generated in California were investigated by considering both the availability and stability of the waste. The two potential waste streams that were identified and evaluated were almond hulls/shells and rice hulls.

Based on the statistics about the production amount and bearing acreage provided by the California Department of Food and Agriculture (CDFA), two agricultural products were selected based on the highest total acreage; almond and rice (Figure 13) [83]. For this study, agricultural residues, co-products, and by-products from these two crops are considered as bioenergy feedstocks: almond hulls/shells from almond production, and rice hulls from rice production. The selection of these two agricultural residues was based on both the availability and the stability of the residues as potential energy feedstocks. Although other agricultural products or residues can be potential feedstocks, some of them can be excluded from consideration depending on characteristics. For example, if the quantity of a feedstock generated at a particular site is small relative to the costs of transport and aggregation, it will not be considered a likely candidate.

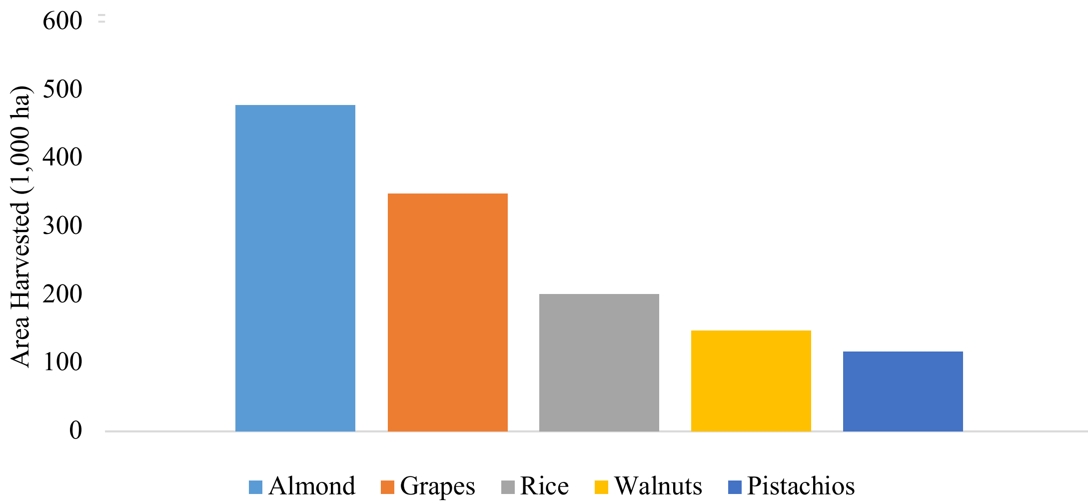

Figure 13. Harvested areas of the top 5 crops in California (2019) [83]

3.2.1.1 Almond hulls and shells

Almond processing after harvest begins with hulling and shelling and may then require refrigerated storage to preserve the quality of the raw almonds. Almond shells are highly stable and could easily be stored over the course of a year. On the other hand, almond hulls are less stable as they are the outer casing of the almond fruit with higher moisture content and have higher values than shells due to their use as animal feed.

California is essentially the only almond-producing state in the US, and the world's largest commercial almond producer, producing about 80% of total globally [84]. Almond orchards produce almond kernel, the primary product, along with co-products and residues/wastes of almond hulls and shells, which are all harvested from orchards. In 2016, production of almond kernels, hulls, and shells were 970,700, 1,941,400, and 679,600 metric tons, respectively [85]. As evident in these figures and in Figure 14, the co-products and residues/wastes comprise nearly three quarters of harvested material.

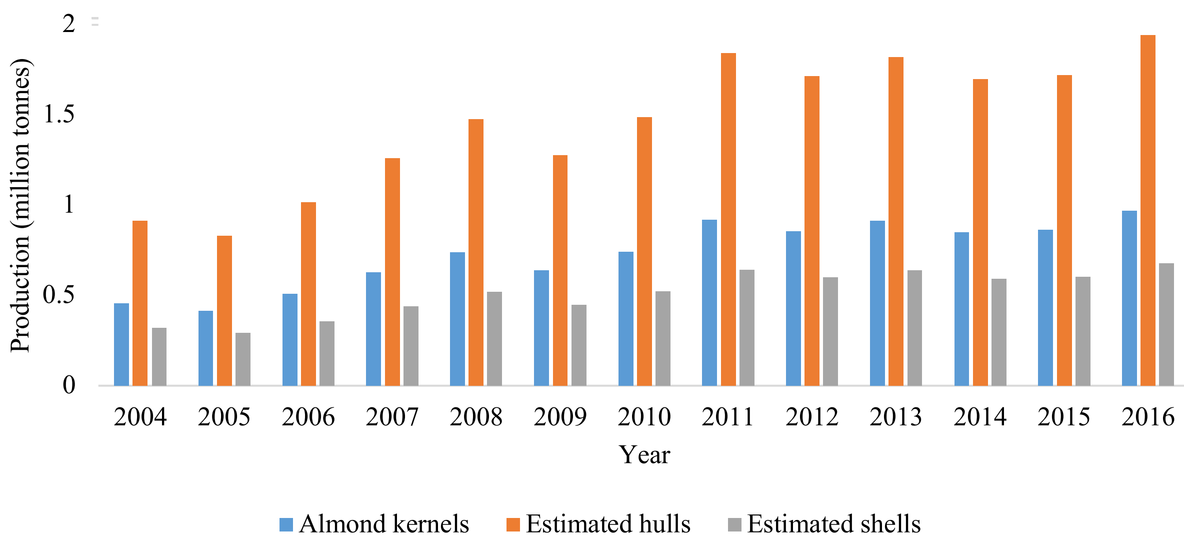

Figure 14. Annual almond, hulls, and shells production in California [71]

Currently, almond hulls are mostly used as dairy feed, and shells, when used, are typically used in very low-value applications such as livestock bedding material [86]. Use of almond hulls as dairy feed has a reasonably high value (e.g., \$140/tonne almond hulls [87]), but the supply of hulls has outstripped demand for their use in dairy feeds, meaning that material would be available for other uses such as energy production. The current low value uses of shells, and their stability, make shells an excellent potential target as a feedstock for energy recovery. The composition analysis of almond shell and hull are shown in Table 13. For each waste stream, compositions by element and component are shown [88].

Table 13. Almond shell and hull compositional analysis [88]

Component (wt%)	Shell	Hull
By component		
Moisture	3.3	11.3
Ash	0.6	3.6
Volatile Matter	80.3	71.2
Fixed Carbon	15.8	13.9

By element		
Carbon	52.55	50.53
Hydrogen	6.87	6.70
Nitrogen	0.22	3.85
Sulfur	0.006	0.009
Oxygen	40.30	38.90
Chlorine	0.05	0.01

The typical supply chain for almonds is illustrated in Figure 15 and begins with orchard cultivation [89]. Once harvested, almonds go to hulling and shelling sites, and then are ultimately transported to processors. Processors may either package and distribute raw almonds, or they may produce manufactured products which include roasted, blanched, sliced, milled, and other almond products. Almond processors use cold storage to preserve and maintain quality of almonds as they produce and distribute product over the year. Hullers and shellers also may have cold storage to keep almonds fresh.

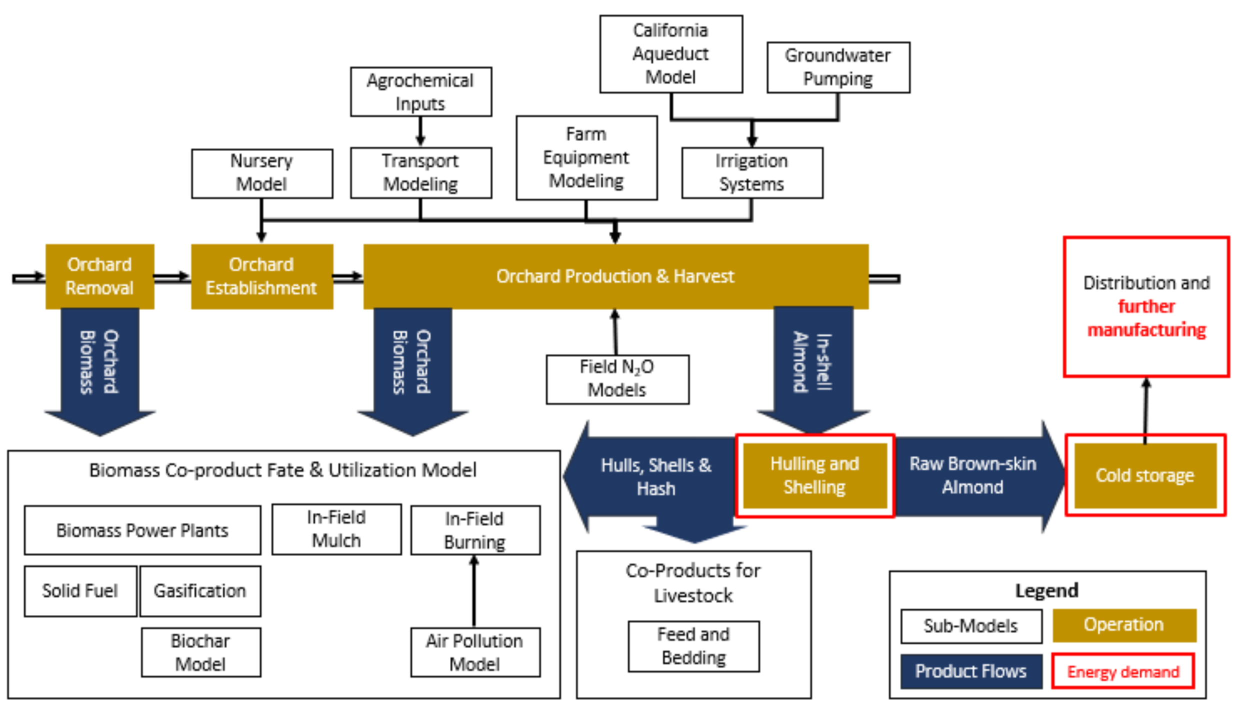

Figure 15. Almond cultivation and processing model [89]

There are several types of properties or facilities in the almond industry: orchards, hullers and shellers, and processors. This study assumes a facility with an integrated structure, whereby hulling and shelling as well as processing facilities are located together. The production of almond at orchards was excluded from the scope of this study [90]. Estimated total electricity demand for almond production and processing based on total production of almond kernels in California in 2016, 970,690 metric tons, was 197,438 MWh. The electricity demands per metric ton kernel and per unit are also shown in Table 14.

Table 14. Electricity demand for an almond facility [90] *and total demand for 2016*

Demanding unit	Electricity demand per metric ton almond kernel (kWh/tonne)	Total electricity consumption for 2016 (MWh)
Hulling and Shelling	110.7	107,455
Processing	92.7	89,983
Total	203.4	197,438

For the almond facilities, it is not required to have a cold storage by law, but it is recommended for food safety management [91]. The electricity demand for cold storage within a processing facility is about 3% of total demand [92].

3.2.1.2 Rice hulls

California is the second largest producer of rice in the US, following Arkansas (Figure 16), and in 2018, California produced 1.97 MMt of whole rice [93]. To perform a logistical and economic assessment of rice hulls, several assumptions were made based on reported statistics. The quantity of rice hulls produced was estimated based on the whole rice production. It was assumed that rice hulls are approximately 20% of whole rice by mass, and brans are approximately 10% of whole rice by mass [93], [94]. Since the whole rice is harvested and transported to rice milling facilities and the rice hulls are separated in the mill, collection and transportation steps of rice hulls from fields were not considered in this study. Since rice hulls are about 20% of whole rice by weight, 393,945 metric tons of rice hulls were produced in California in 2018. Annual estimates for the amount of rice hulls in California are shown in Table 15. Production of rice and rice hulls in California is generally consistent year-to-year as illustrated in Figure 17.

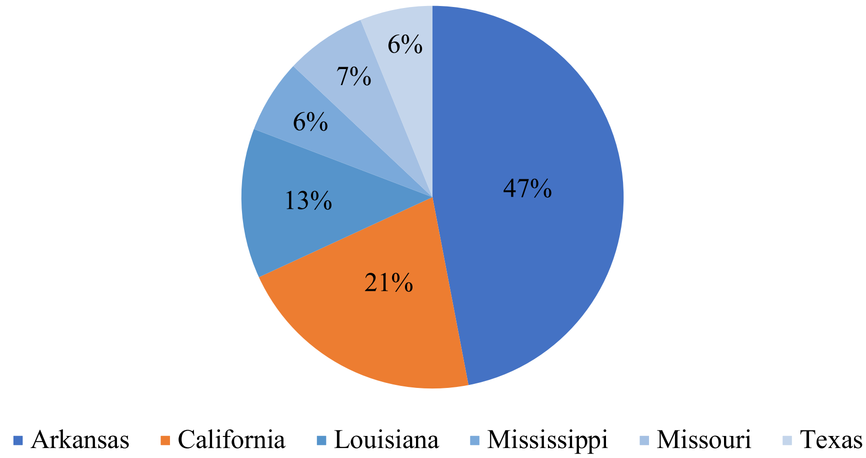

Figure 16. Rice production by state in the United States

Table 15. Rice hull availability in California [93]

	Unit	2014	2015	2016	2017	2018
Whole rice	tonne	1,720,749	1,718,072	2,149,756	1,690,857	1,969,725
Harvested area	1,000 ha	178.93	172.42	216.96	179.38	203.87
Rice yield	tonne / ha	9.62	9.96	9.91	9.43	9.66
Rice hull (est.)	tonne	344,150	343,614	429,951	338,171	393,945
Rice hull yield	tonne / ha	1.92	1.99	1.98	1.89	1.93

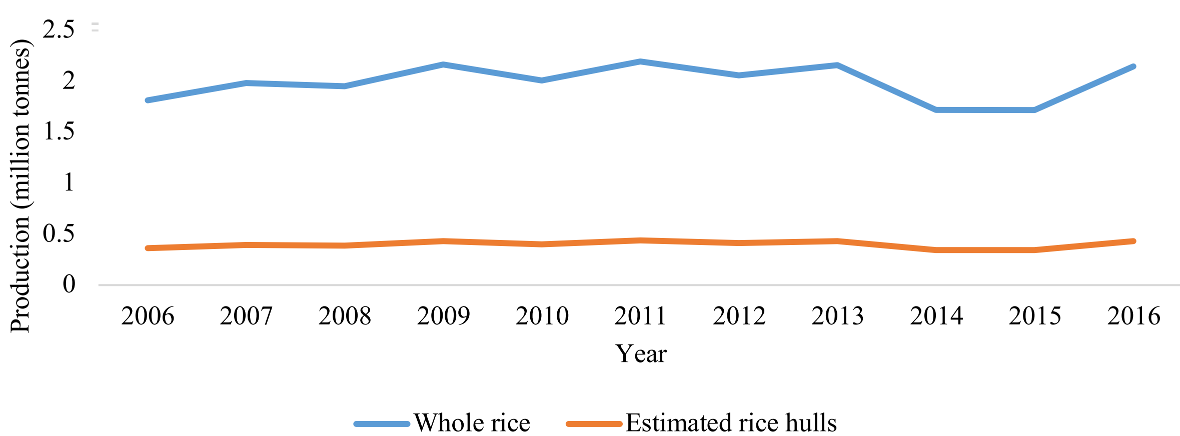

Figure 17. Annual rice and rice hulls production in California [93]

In California, rice is planted from mid-April to late-May, and harvesting season typically begins in August. As with many agricultural wastes and residues, rice hulls are generated with high seasonal specificity. Currently, about 40% of rice hulls generated in California are used for electricity generation at the Wadham facility in Williams, California [95]. The facility generates 200,000 MWh of electricity annually by burning rice hulls. It sells the electricity to Pacific Gas and Electric, and it also sells rice hull silica, a by-product, to the steel, cement, and photovoltaic industries. Feedstock compositional analysis is shown in Table 16 and as evident, rice hulls contain a notable portion of silicon dioxide (SiO_2). Because of the high silicon dioxide in rice hulls, silica is often produced as a byproduct of rice hull combustion, and this can present challenges including ash formation or fouling to combustion processes and the implementation of additional recovery processes for materials. However, on the other hand, it gives an opportunity to utilize the ash as SCM.

Table 16. Rice hull compositional analysis [96]

Main components	Content (wt. %)
SiO_2	18.8 – 22.3
Lignin	9.0 – 20.0
Cellulose	28.0 – 38.0
Protein	1.9 – 3.0
Fat	0.3 – 0.8
Nutrients after full digestion	9.3 – 9.5

Upon harvest, rice is hulled and dried, and this drying step requires energy, usually delivered by natural gas. While it might seem that using rice hulls as fuel for drying rice is an obvious match between waste flow and energy demand for preservation steps, the energy demand for drying occurs over a very short period. This means that the cost of investing in new equipment and the relative total displacement of fossil energy is limited.

3.2.2 Electricity generation

3.2.2.1 Almond hulls and shells

A gasification system was modeled for electricity generation from the almond residues based on a reference model provided by Sierra Energy using the Aspen Plus technology simulation software [97]–[99]. One benefit to using a simulation approach is that it was able to run scenarios of predicted performance based on varying feedstock characteristics.

The study considers a gasifier which can take about 60 metric tons of as-received feedstock per day (wet). There is a loss after the preparation step caused by recyclable or salvage materials recovery and moisture loss. The process is shown in Figure 18. The model assumes 347 days of operation in a year, and it takes 3 days for both start-up and cool-down. The gasification and polishing steps produce syngas and generate ash as waste daily. The raw syngas produced by the gasifier is then cooled, cleaned, and compressed, leaving clean syngas available to produce electricity while the remainder is recycled back to the gasifier and polisher. Electricity is generated by internal combustion of clean syngas (27.5% efficiency), and 19% of electricity is consumed on-site as a parasitic load for facility operations.

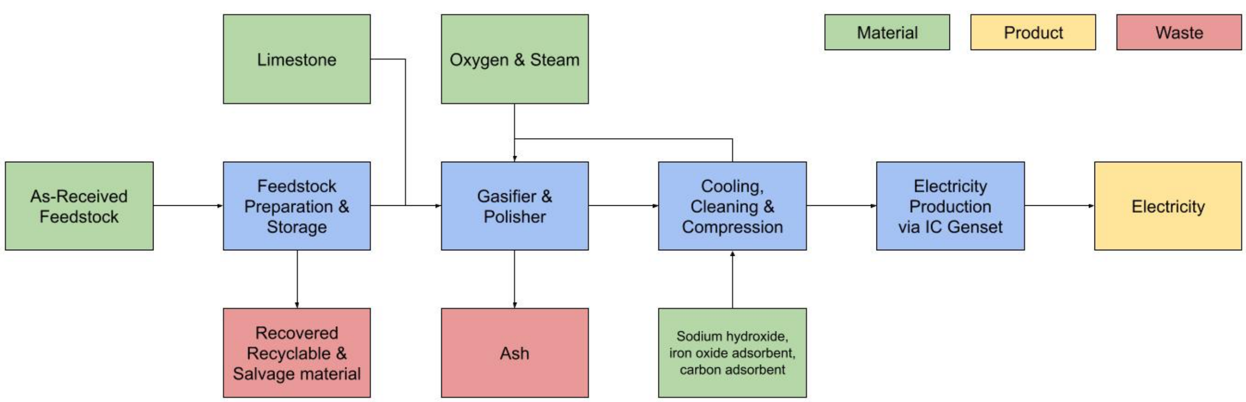

Figure 18. Gasification process flow diagram [98]

The model was run with the almond residues based on the selection of almond hulls and shells [97], [98]. The composition comparison between almond shell and hull is shown in Table 17. Composition of the shell and hull mixture is also shown in Table 17 as a weighted average, which is calculated based on mass ratio of shell and hull. In this study, weighted average by mass and shell-only cases were used as potential feedstocks. The shell-only case was considered because hulls currently do have a market as dairy feed, while shells have only low-value uses. In addition, shells contain less moisture than hulls, which means they are less perishable and can be easily stored for longer time than hulls.

Table 17. Composition comparison among target feedstocks and the reference case feedstock [88]

Component (wt%)	Shell	Hull	Weighted average
By component			
Moisture	3.3	11.3	9.26
Ash	0.6	3.6	2.84
Volatile Matter	80.3	71.2	73.52
Fixed Carbon	15.8	13.9	14.38
By element			
Carbon	52.55	50.53	51.09
Hydrogen	6.87	6.70	6.75
Nitrogen	0.22	3.85	2.84
Sulfur	0.006	0.009	0.008
Oxygen	40.30	38.90	39.29
Chlorine	0.05	0.01	0.02

3.2.2.2 Rice hulls

Currently, about 181,437 metric tons of rice hulls generated in California are used annually at the Wadham bioenergy facility in Williams, California, to generate electricity [95]. For the assessment of energy generation from rice hulls, the Wadham facility was used as a reference case.

The Wadham facility has a net capacity of 26 MW to the grid (gross capacity: 29.1 MW), and it generates about 200,000 MWh annually by burning rice hulls at 857°C and sells the electricity to the grid. Table 18 shows the properties of the Wadham facility.

Table 18. Electricity generation from Wadham facility [19], [100]

Parameters	2014	2015	2016	2017	2018
Gross Capacity (MW)	29.1	29.1	29.1	29.1	29.1
Annual electric power potential (MWh)	254,916	254,916	254,916	254,916	254,916
Gross generation (MWh)	190,440	171,432	218,376	208,728	186,264
Capacity factor (%)	74.71	67.25	85.67	81.88	73.07
Net generation (MWh)	170,418	154,054	196,433	187,149	166,216

Since the electricity generation from Wadham varies by year, the average of a 5-year period (2014-2018) was used for this study. The model parameters for the power plant in this study, which was modeled based on the publicly available information on the characteristics of the Wadham facility, are shown in Table 19.

Table 19. Parameters for a model power plant

Parameters		Unit
Plant capacity	29,100	kW
Capacity factor	76.5	%
Net station efficiency	22.5	%

3.2.3 Cost analysis

While estimated capital expenses (CAPEX) and operating expenses (OPEX) for almond case were provided by Sierra Energy [97]–[99], an economic analysis for rice case was performed based on the assumptions from the literature, interviews, and the Wadham case. In addition, it is assumed that feedstocks are available at no cost for both almond and rice cases.

In the almond case, CAPEX includes feedstock preparation units, an oxygen generator, a gasifier, a polisher, gas cleaning units, internal combustion electricity generator, and other system utilities. OPEX includes labor for thirteen operators for the facility, system maintenance, and other supplies and materials.

However, it does not include electricity costs in the OPEX, because it assumes that the electricity is directly supplied from the on-site generator.

In the rice case, CAPEX and OPEX from a United States Energy Information Administration (US EIA) report were used to model the economic assessment of the power plant [101]. Table 20 shows the CAPEX and OPEX for a biomass power plant with a capacity of 50 MW. The US EIA report provides CAPEX for both Northern and Southern California region, and their average is used in this study. For the OPEX, the report provides variable OPEX and fixed OPEX.

Table 20. Capital expenses (CAPEX) and Operating expenses (OPEX) for a model biomass power plant ($, 2019)

Parameters		Unit
CAPEX	4,104	$/kW
Variable OPEX	1.88	$/MWh
Fixed OPEX	125.19	$/kW-yr

In both almond and rice cases, there are two potential revenue streams from the process, net-generated electricity and ash. To analyze the economic feasibilities of electricity and ash, the prices of them were calculated based on different assumptions. The general price calculation method is shown in Equation 2. First, the price of electricity was calculated assuming that the value of ash is $0. In this case, the required revenue for electricity was calculated without considering revenue from ash, and the price of electricity was calculated from that required revenue. For the price of ash, revenue from electricity was included in the required revenue for ash as operating revenue. The average retail price of electricity was assumed at $0.15/kWh in California [102]. In Table 21, additional economic assumptions for the cost calculation are provided. To estimate the price of ash, the cash flow method known as “discounted cash flow rate-of-return” was used, assuming a straight line 30-year depreciation. For the cost of money, the weighted average cost of capital (WACC) for merchant-owned biomass facilities (7.21%) was used [103]. By using the WACC for biomass facilities as the cost of money for this study, the economic feasibility of this study compared to other biomass facilities can be determined.

Equation 2:

$$Price_{target} = \frac{Required\ revenue_{levelized\ annual}}{Total\ production}$$

$$Required\ revenue_{levelized\ annual} = \left(\sum_{year=1}^{Lifetime} \frac{(Required\ revenue)_{year}}{(1 + Cost\ of\ money)^{year}}\right) \times Capital\ recovery\ factor$$

$$Capital\ recovery\ factor = Cost\ of\ money_i \times \frac{(1 + Cost\ of\ money_i)^{lifetime}}{((1 + Cost\ of\ money_i)^{lifetime} - 1)}$$

$$Cost\ of\ money_i(inflation\ adjusted\ cost\ of\ money) = \frac{1 + Cost\ of\ money}{1 + General\ inflation} - 1$$

Table 21. Economic assumptions for cost analysis

Economic Indicator	Value	Unit
Federal tax rate	21	%
State tax rate (CA)	8.84	%
General inflation	2	%
Economic lifespan	30	year
Cost of money	7.21	%

3.2.4 Ash utilization

3.2.4.1 Candidate selection

After analyzing both cases (rice hulls and almond hulls/shells), one case was selected based on two criteria: availability and price. In California, cement consumption in 2016 was about 9 MMt [104], and thus, a corresponding quantitative demand for SCM is assumed to exist in the state. For concrete suppliers, it is important to obtain ash with consistent properties and have consistent access to ash. The ash should not exceed current SCM prices which is around $83 per metric ton for ground blast furnace slag from the iron industry and fly ash from coal electricity [105].

3.2.4.2 Experimental design for ash utilization

Rice hull ash (RHA) was selected for ash utilization based on the results[1]. The selection step is shown in the Results and Discussion section.

For the experiment, RHA was obtained from the Wadham bioenergy facility in Williams, California owned by the Enpower Corporation. ASTM Type II/V Portland cement (PC) obtained from Lehigh Southwest Cement Company in Stockton, CA. The concrete batched for this work used alluvial sand as the fine aggregate (with a 99.95% passing rate through a #4 sieve, 4.75 mm) and coarse aggregate was also used (with a 100% passing rate through a 25 mm sieve). Both fine and coarse aggregates were locally sourced from Esparto, California.

Concrete mixtures were made with two proportions: one control mixture containing no RHA and the other mixture containing 15% of RHA by mass replacing PC. Mixtures were batched using the same proportions of water, fine aggregates, and coarse aggregates (Table 22), and no chemical admixtures were used.

Table 22. Concrete mixture proportions

			Mixture proportions (kg/m^3)				
	Portland Cement (%)	Rice hull Ash (%)	Portland Cement	Rice hull Ash	Fine Aggregate (< 4.75 mm)	Coarse Aggregate (< 25 mm)	Water
100% PC	100	0	411	0	516	1141	193
15% RHA	85	15	349	61.6	493	1141	193

Concrete mixtures were produced to assess compressive strength, flexural strength, and chloride ingress. Experimental methods for each test are shown below. For compressive strength and chloride ingress tests, the mixture specimens were cast as cylinders approximately at 100 mm × 200 mm, and prismatic beams of 100 mm × 100 mm × 300 mm were cast for flexural strength testing. All specimens

[1] All experimental work was completed by project collaborators in the Miller Lab in Civil and Environmental Engineering at UC Davis.

were demolded 1 day after casting and then cured for the remaining time in a curing chamber at 25°C and 80% of relative humidity.

Compressive and flexural testing: Mechanical compression and flexure testing were conducted on mortar specimens after 28 days of curing. Compression tests were conducted on a SoilTest CT-950 load frame following an adaptation of ASTM C39 testing procedures [106] where cylinder specimens were capped on either end with a neoprene-padded aluminum cap and specimens were loaded under force control. Five specimens were tested for each mixture. Ultimate strength of the concrete mixtures was determined based on the maximum load reached before softening or failure occurred. Flexural strength was determined by performing three-point bending tests at 28 days age. Experiments were performed on a MTS Testline Component load frame managed by an MTS TestStarIIs controller following ASTM C293 testing procedures [107]. Three specimens were tested from each mixture and ultimate strength was determined based on the maximum load reached prior to failure.

Rapid chloride permeability testing: Chloride permeability reflects the ability for chloride ions to permeate into concrete and is a critical durability property for concrete used in certain regions. Chloride ingress in steel reinforced concrete is one of the largest contributors to the corrosion of steel, which is of interest in California because of our exposure to saltwater at the coasts and because most structural concrete in California requires use of reinforcement. To determine the effects of using ash on the ability for concrete to withstand chloride ingress, resistance of saturated concrete specimens with and without ash to chloride diffusion were measured. Concrete cylinders were cured for 90 days and then cut into disks following ASTM C1202 [108]. These experiments were performed using a Germann Instruments PROOVE-it control unit and testing cells with external cooling fins under ASTM C1202 [108] testing conditions.

3.2.4.3 Environmental impact assessment

To assess the potential environmental benefits of ash replacing conventional cement or SCMs, this study applies life cycle assessment (LCA). Since this study assumes that the ashes directly replace Portland cement (PC) or coal fly ash, the environmental impacts of each material were calculated and compared directly with setting the functional unit as one metric ton of each material.

The life cycle inventory (LCI) data for Portland cement was obtained from the Ecoinvent 3.5 database using the GaBi software tool. Since the LCI data specific to coal fly ash was not available, LCIs for cement production with 5 – 15 % pozzolana and fly ash and cement production with 15 – 40 % pozzolana and fly ash were collected instead. In addition, an LCI for electricity from solid biomass was used to build an estimated LCI for RHA. The LCIs are shown in Table 23. Based on the LCI data, environmental impact potentials were calculated using the life cycle impact assessment method, CML 2001 method developed by the Institute of Environmental Sciences at Leiden University [109].

Table 23. LCI data source (Geographic region: United States), obtained from the GaBi software tool. Unit output as shown in the database

Name	Unit output	Source
Cement production, Portland	1 kg	Ecoinvent 3.5
Cement production, pozzolana and fly ash 5-15%	1 kg	Ecoinvent 3.5
Cement production, pozzolana and fly ash 15-40%	1 kg	Ecoinvent 3.5
Electricity from biomass (solid) (West)	1 kWh	Ecoinvent 3.5

3.3 Results and Discussion

3.3.1 Electricity generation

3.3.1.1 Almond hulls and shells

The results are shown in Table 24. The results show that using shells-only is more efficient for electricity generation (e.g., higher electricity generation per unit feedstock), since the shell has lower moisture content and more volatile matter than the weighted average case. The almond shells generated 58.85 MWh of electricity per day using 60 metric tons of shells (as-received, wet), which means it requires 1.02 metric tons of shells to produce 1 MWh of electricity.

Table 24. Comparison for inputs and outputs per day

	Parameter	Weighted average	Shells-only	Unit
Inputs	Feedstock (As-received, wet)	60	60	tonne/day
	Water	54,500	54,500	liter/day
	Limestone	204	204	kg/day
Outputs	Electricity (Net)	51.19	58.85	MWh/day
	Heat	26.28	26.28	MWh/day
	Ash	1.92	0.41	tonne/day

The shells-only case also generates more than enough electricity to cover the electricity demand for almond processing facilities in California. Based on the almond production in California in 2016, 679,620 metric tons of almond shells were generated, and they can be converted to 666,294 MWh of electricity, which represents more than threefold the total electricity demand for the facilities.

3.3.1.2 Rice hulls

If all 393,945 metric tons of rice hulls (production in California, 2018) were used to generate electricity, it generates 390,006 MWh of electricity per year along with 79,814 metric tons of ash. It requires 1.01 metric tons of rice hulls to produce 1 MWh of electricity.

3.3.2 Cost analysis results

Table 25 shows the results of the cost analysis and ash price estimation for the almond case. While the weighted average and shells-only cases have the same CAPEX and OPEX, they showed different operating revenues per year due to the different electricity generation. Because the shells-only case generated more electricity than the average weighted case, it showed higher economic feasibility. Without considering the value of ash, the price of electricity was \$0.132/kWh for the shells-only case, which is lower than the average retail price of electricity in California, \$0.15/kWh [102] and the price from the weighted average case (\$0.276/kWh). With considering the revenue from electricity as operating revenue for ash, the price of ash was \$0/tonne for the shells-only case while it was \$3,382/tonne for the weighted average case, which is higher than the market price of fly ash. The price at \$0 of ash (shells-only) means that the required revenue for the system can be met by the selling electricity, and the ash may be available only at transportation cost from a power plant to a concrete production site.

Table 25. Cost analysis results (\$, 2016)

Economic parameter	Weighted average	Shells-only	Unit
Capital expenses (CAPEX)	15,960,000	15,960,000	\$
Operating expenses (OPEX)	1,432,000	1,432,000	\$/yr
Electricity price	0.276	0.132	\$/kWh
Operating revenues, electricity	2,665,000	3,063,000	\$/yr
Ash price	3,382	0	\$/tonne

Table 26 shows the results of cost analysis and ash price estimation for the rice case. The electricity price was found to be \$0.070/kWh without any revenues from selling ash. It is lower than the value of electricity assumed, \$0.15/kWh [102]. Thus, the electricity generation from rice hull is economically feasible without selling the ash from the process, which means RHA may be available only at transportation cost from a power plant to a concrete production site.

Table 26. Cost analysis results (\$, 2019)

Economic parameter		Unit
Capital expenses (CAPEX)	119,426,000	\$
Operating expenses (OPEX)	4,009,000	\$/yr
Electricity price	0.070	\$/kWh
Operating revenues, electricity	29,257,000	\$/yr
Ash price	0	\$/tonne

3.3.3 Results for ash utilization

3.3.3.1 Candidate selection

Table 27 summarizes key parameters for feedstock selection for the ash utilization step. Since the shells-only case showed better electricity generation and economic feasibility than the weighted average case, the parameters are selected based on the results from the shells-only case. For the almond, it was assumed that all almond shells generated in California in 2016 were used to produce electricity and ash, and for the rice, it was assumed that all rice hulls generated in California in 2018 were used to produce electricity and ash.

According to the results, rice hull was selected for the ash utilization model. Although the available electricity generation from almond shell is higher than the case of rice hull, the ash production is higher for the rice hull case due to the high ash content of rice hull. The electricity price in the rice hull case was also lower than the price in almond shells case. Since the required ash prices for economic feasibility of electricity generation were $0 for both cases, only the availability affected on the selection process.

Table 27. Summary for almond shell and rice hull results. Assuming all available feedstocks generated in California were converted to electricity and ash (almond shell in 2016 and rice hull in 2018)

Parameter	Almond shell	Rice hull	Unit
Total feedstock available in CA	679,620	393,945	tonne/yr
Total electricity generation	666,569	390,006	MWh/yr
Total ash production	6,666	79,814	tonne/yr
Ash production per electricity generation	10	205	
Electricity price	0.132	0.070	$/kWh
Required ash price	0	0	$/tonne

3.3.3.2 Concrete compressive and flexural strength

A 15% replacement of cement with RHA acquired from conventional bioenergy production methods resulted in a moderate change to 28-day compressive and flexural strength (Figure 19) [110]. Namely, approximately a 15% reduction in strength was noted for both cases. While a reduction in strength is not necessarily desirable, it must be noted that this 15% reduction is less than the reduction in strength that would have been found from removal of 15% of cement with no replacement. In addition, depending on proportions used, this 15% RHA mixture can exceed strength of a 15% fly ash replacement mixture [111]. The likely reason for these shifts in properties is that the rice ash could be acting as a pozzolan that contribute to later-age strength, but less so to early strength. With grinding, RHA has been shown to become more reactive [112], so if early age strength was desirable, post-combustion processing may contribute to desirably properties [110].

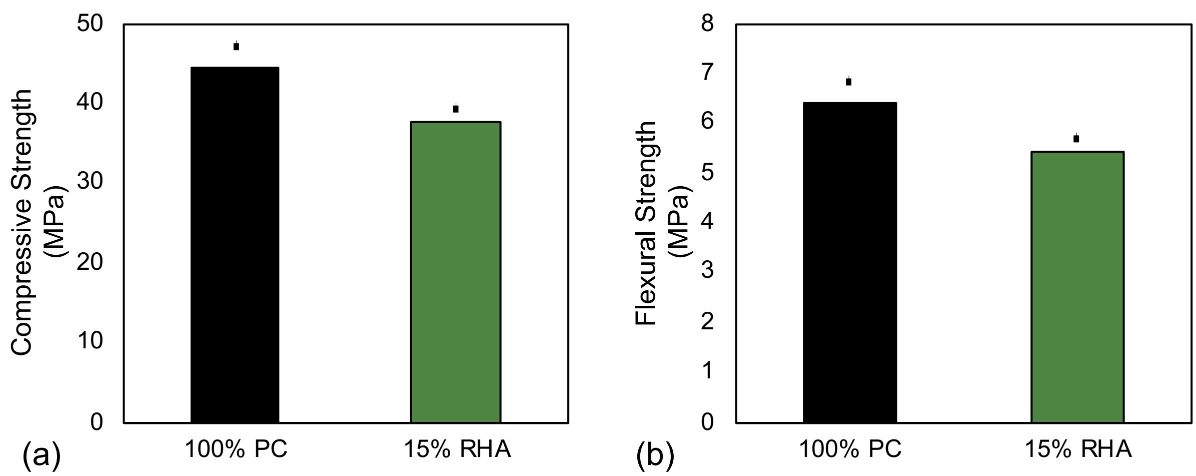

Figure 19. (a) Compressive strength and (b) flexural strength of concrete mixture (PC: Portland cement, RHA: Rice hull ash)

3.3.3.3 Chloride ingress

The use of RHA in concrete increased chloride permeability. In this case, the use of RHA led to almost a halving of the resistivity of the concrete and the improvement of Chloride Ion Penetrability Category defined by ASTM C1202 [108]. These results suggest that even with this less reactive rice ash, the permeability of concrete to chloride ingress could be substantially reduced from the "Moderate" permeability range (2000-4000 coulombs) exhibited by the control concrete to the "Low" permeability range (1000-2000 coulombs) exhibited by the RHA concrete. The near halving of resistivity and the change in permeability category agrees with the results reported in literature [113].

3.3.3.4 Environmental impact assessment

Allocation in LCA refers to the partitioning of the impacts among co-products generated from the system, and an economic allocation method, which is partitioning based on their economic values, is one of the widely used methods in the field of LCA [114]. In this study, electricity and RHA are the products from the rice hull power plant, and the environmental impacts should be allocated based on their economic values. Since the price of RHA from this study is $0 per metric ton, the total environmental impacts should be assigned to electricity and no impacts will be assigned to RHA. However, the price of $0 calculated in this study does not imply that the value of RHA is zero. Considering that RHA is replacing SCM, it is possible to assume that RHA has the same economic value as other SCMs such as fly ash from coal electricity or ground blast furnace slag from the iron industry. Thus, $83/tonne is used as the economic value of the RHA [105]. In Table 28, the total economic values of electricity and RHA are shown with assuming utilization of total rice hulls in California, and the assigned environmental impacts based on the economic values are shown in Table 29.

Table 28. Economic value estimates for electricity and RHA

	Electricity	RHA
Generation / Production	390,006 MWh/yr	79,814 tonne/yr
Unit price	$ 0.070/kWh	$ 83/tonne
Total economic value	$ 27,300,420/yr	$ 6,624,562/yr
Total economic value (%)	80 %	20 %

Table 29. Environmental impacts of electricity and RHA per unit output (ADP: Abiotic depletion, AP: Acidification potential, EP: Eutrophication potential, FAETP: Freshwater aquatic ecotoxicity potential, GWP: Global warming potential, HTP: Human toxicity potential, MAETP: Marine aquatic ecotoxicity potential, ODP: Ozone layer depletion potential, POCP: Photochemical ozone creation potential, TETP: Terrestrial ecotoxicity potential)

Impact category	Electricity	RHA	Unit
Unit output	1 MWh	1 tonne	
ADP elements	4.30E-05	1.08E-05	kg antimony (Sb) eq.

ADP fossil	380	95	MJ
AP	0.888	0.222	kg SO_2 eq.
EP	0.226	0.566	kg PO_4 eq.
FAETP	1.13	0.28	kg dichlorobenzene (DCB) eq.
GWP	33.2	8.3	kg CO_2 eq.
HTP	6.53	1.63	kg DCB eq.
MAETP	32,200	8,100	kg DCB eq.
ODP	1.07E-13	2.68E-14	kg chlorofluorocarbon (CFC)-11 eq.
POCP	0.163	0.041	kg C_2H_4 eq.
TETP	1.09	0.27	kg DCB eq.

As shown in Table 28, the total economic value of RHA is more than 15% if assuming the price as \$83/tonne. With the assumption applied, the RHA may be designated as a co-product based on the decision trees developed in Chapter 2 (Figure 20). Thus, upstream impacts from RHA production should be included to the environmental impacts of concrete mixture.

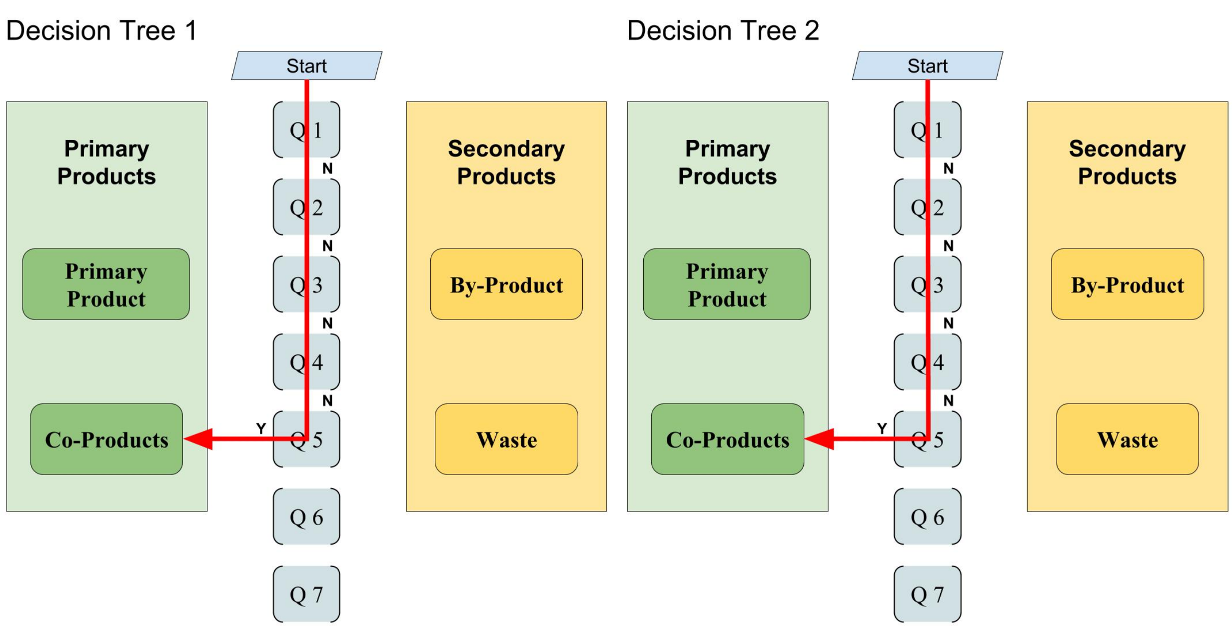

Figure 20. Decision tree results for rice hull ash (RHA)

In Table 30, environmental impacts of Portland cement, cement with 5-15% fly ash, cement with 15-40% fly ash, RHA, and mixture of 85% Portland cement and 15% RHA are shown. The results show that RHA has lower environmental impacts in all impact categories than other cement products and the mixture also has lower environmental impacts than Portland cement or cement with 5-15% fly ash. For example, the mixture of 85% Portland cement and 15% RHA, which was tested in this study, may lead to 15% reductions in the global warming potential (GWP) and 15% reductions in the abiotic fossil depletion compared to Portland cement. If 79,814 metric tons of RHA is used as SCM in lieu of Portland cement annually, a mitigated CO_2e emission is 70,771 metric tons per year, which is about 0.2 % of total CO_2e from the cement production sector in the United States.

Table 30. Environmental impacts of Portland cement, cement with 5-15% fly ash, cement with 15-40% fly ash, rice hull ash (RHA), and mixture of 85% Portland cement and 15% RHA (ADP: Abiotic depletion, AP: Acidification potential, EP: Eutrophication potential, FAETP: Freshwater aquatic ecotoxicity potential, GWP: Global warming potential, HTP: Human toxicity potential, MAETP: Marine aquatic ecotoxicity potential, ODP: Ozone layer depletion potential, POCP: Photochemical ozone creation potential, TETP: Terrestrial ecotoxicity potential)

Impact category	Portland cement (PC)	Cement, fly ash 5-15%	Cement, fly ash 15-40%	RHA	85% PC + 15% RHA	Unit
Unit output	1 tonne	1 tonne	1 tonne	1 tonne	1 tonne	
ADP elements	2.03E-03	1.93E-03	1.59E-03	1.08E-05	1.73E-03	kg Sb eq.
ADP fossil	4,020	3,810	3,120	95	3,430	MJ
AP	1.59	1.51	1.23	0.222	1.38	kg SO_2 eq.
EP	0.669	0.637	0.528	0.566	0.577	kg PO_4 eq.
FAETP	88.9	84.9	71.3	0.28	75.6	kg DCB eq.
GWP	895	848	687	8.3	762	kg CO_2 eq.
HTP	127	121	103	1.63	108	kg DCB eq.
MAETP	2.55E+05	2.44E+05	2.05E+05	8,100	2.18E+05	kg DCB eq.
ODP	2.51E-05	2.38E-05	1.94E-05	2.68E-14	2.13E-05	kg CFC-11 eq.
POCP	0.115	0.109	0.089	0.041	0.104	kg C_2H_4 eq.
TETP	1.79	1.70	1.40	0.27	1.56	kg DCB eq.

3.3.4 Discussion

In this study, both almond shell and rice hull show economic feasibility for electricity generation. Assuming that feedstocks are available at no cost and no revenue is generated from ash, the price of electricity from almond shell was \$0.132/kWh and it was lower for rice hull at \$0.070/kWh. Total available ash production was also larger for rice hull, which is a favorable condition for ash utilization. Based on the results, rice hull was selected as the feedstock for combined electricity and ash production, and thus, the following discussions are focused primarily on the rice hull case.

Acquisition of almond shell or rice hull may incur costs such as transportation cost or payment to providers. In this case, the costs should be added to the operating costs, and the price required for electricity should increase. When the price of electricity is assumed at \$0.15/kWh, the maximum cost to a power plant for almond shell is \$18/tonne and it is \$81/tonne for rice hull. If the costs become larger than these, electricity generation from such feedstock may be economically infeasible. In addition, with considering operating revenues from ash together, selling at \$83/tonne, the maximum cost for feedstocks increases to \$19/tonne and \$98/tonne for almond shell and rice hull, respectively.

The cost of feedstock may also affect allocated environmental impacts to RHA. Because only several pollutant emissions were available from Wadham facility (Table 31), this study used an LCI for a solid biomass power plant for environmental impact assessment. The LCI of the power plant was then allocated to electricity and RHA based on their total economic values. Under the assumption that defines the cost of feedstock (rice hull) as \$0 (e.g., no value to rice hull producer), the rice hull was designated as waste (Figure 21), and thus, it was eligible to avoid any upstream impacts from rice hull production. However, the designation may be affected by a change in cost or economic value related to rice hull, and the change in designation may cause addition of upstream impacts from rice hull production.

Table 31. Emissions factor (per MWh electricity) from Wadham facility [115]

Component	Emissions	Unit
GHG emissions	18.2	kg CO_2e/MWh
Particulate Matter	231.1	kg/MWh
SOx	154.7	kg/MWh
NOx	776.4	kg/MWh
Volatile Organic Compounds	3.3	kg/MWh

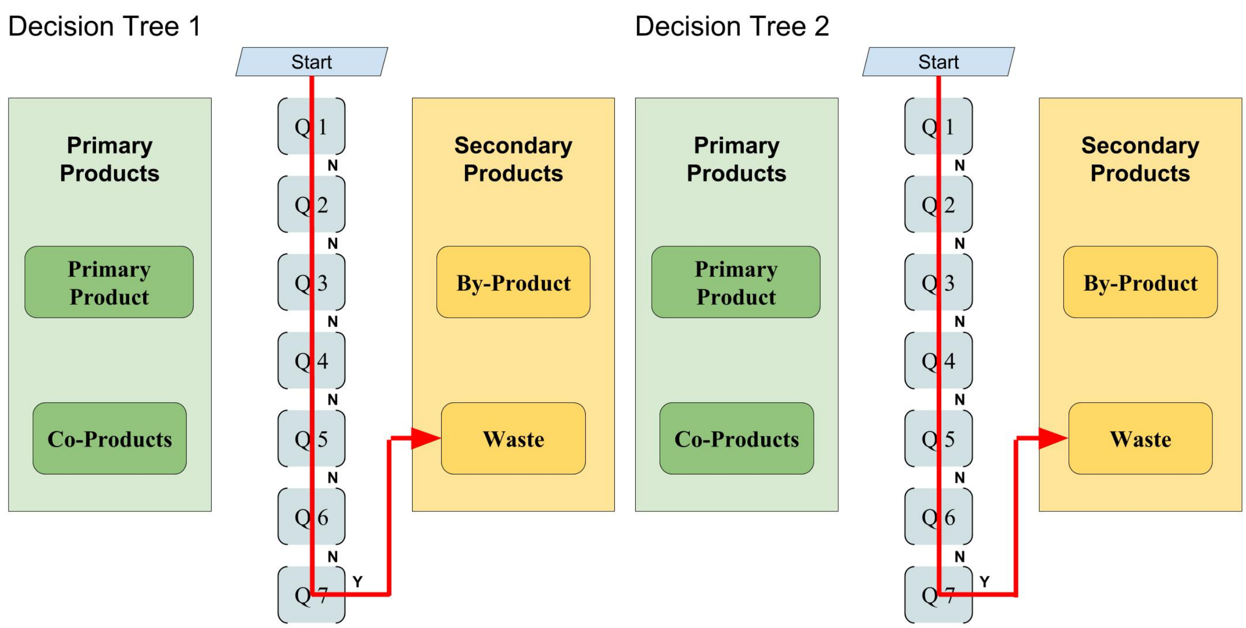

Figure 21. Decision tree results for rice hull

3.4 Conclusions

This study examined the performance of electricity generation systems using agricultural residues and utilization of ash from the electricity generation systems. Almond residues and rice residues were considered as candidate feedstocks. While rice hull was selected as a feedstock from rice residues, for almond residues, two scenarios were evaluated: using almond shells only and using a mixture of shells and hulls. The composition of the mixture was assumed to be a weighted average of shell and hull by their production mass ratio.

The cost analysis results showed that using rice hull is the most feasible among the three feedstock candidates. Without considering the cost of feedstock and the value of ash, two feedstocks, almond shell and rice hull, showed economic feasibility for electricity generation, and this means that ash from the generation may be available at no cost. Between two feedstocks which showed economic feasibilities, rice hull was selected as a final feedstock. The rice hull showed not only the best economic performance for electricity generation but also the most availability for ash. However, in practice, the cost of feedstock and the value of ash may not be zero. If the cost of rice hull increases to over \$81/tonne, electricity generation may be infeasible, and the limit for rice hull costs may increase to \$98/tonne assuming the value of RHA at its market price of \$83/tonne.

After testing showed that RHA can be used as an SCM, it was also shown that RHA has lower environmental impacts than Portland cement and a cement mix with fly ash. Compared to Portland cement, the mixture (85% Portland cement, 15% RHA) showed 15% less CO_2e emissions and 15% less abiotic fossil depletion, and it also showed less impacts than cement mixture with 5-15% fly ash.

There are several opportunities for future research, partly to improve this study. First, future research may include development of specific databases for the bioenergy sector. For example, CAPEX and OPEX used in this study were for a general biomass power plant, not specifically developed for the rice or almond residues case, and a LCI for a solid biomass power plant was used instead of the one specific to a rice hull plant. The assessment could be improved by using more specific databases and be further expanded by exploring additional feedstocks, as well as their location and availability.

CHAPTER 4 Recycling of Carbon Dioxide and Waste Nutrients from Anaerobic Digesters for Microalgae Cultivation

4.1 Introduction

4.1.1 Background

Among various types of alternative energy sources and biomass feedstocks, microalgae are considered one of the most promising feedstocks. Moreover, compared to terrestrial crops, microalgae have advantages such as higher oil content, higher areal productivity, and robustness to cultivation environments that cannot support terrestrial crops such as brackish water, salt water, or non-arable lands, thus avoiding land-based limitation and the food versus fuel problem [116], [117]. For example, while other feedstocks such as palm oil or soybean oil account for less than 5% of total biomass generated, some microalgal species are able to accumulate lipids up to 80% of total dry cell weight [118]. However, biofuels from microalgal biomass have not yet delivered on their promise of an abundant, low-carbon, and sustainable bioenergy pathway. The high cost of production, particularly during the cultivation step, which has high demands for nutrients, carbon dioxide and mechanical energy inputs to achieve high growth rates and biomass yield have prevented commercial scale production of low cost, low carbon intensity (CI) microalgal biofuels [119].

One strategy which has long been promoted for cost-effective and environmentally preferable production is utilizing waste flows such as wastewater and waste nutrients for the microalgal biomass production [120], [121]. Utilization of waste flows may avoid the use of high cost and high CI inputs including synthetic fertilizers, high quality fresh water and commercially produced carbon dioxide (CO_2). In the field of industrial ecology research, this approach, where waste streams are repurposed as valuable inputs to another system, is typically assumed to reduce the cost and improve the environmental performance of systems.

4.1.2 Goals and objectives

The objective of this study is to evaluate environmental and economic feasibilities of a closed loop system for microalgal bioenergy production coupled with a food waste anaerobic digester. To quantify the prospective economic and environmental performances of the system, techno-economic analysis (TEA) and life cycle assessment (LCA) are performed on the system assuming commercial-scale production conditions.

4.2 Materials and Methods

The approach for this research is to combine experimental results for commercial scale AD operation, pilot scale microalgae cultivation, and lab scale downstream processes with TEA and LCA modeling to estimate the cost and environmental performance of a prospective system. To achieve this, pilot and lab scale results are used within a techno-engineering model that assumes full-scale operations and provides the backbone for TEA calculations and LCA modeling. Based on the parameters collected from the experimental results and the literature, calculations for mass balance, energy inputs and outputs, and emissions were performed in spreadsheets on Microsoft Excel, as well as calculations for economic evaluations.

4.2.1 Scope and system boundary

Figure 22 shows the modeled system including all pathways. The evaluated system mainly consists of two parts; microalgae cultivation integrated with an AD system and energy generation pathways from the microalgal biomass. Biodiesel or electricity is generated as a final product depending on the energy generation pathway. The system begins with collection and transportation of food waste streams as feedstock for an AD. The production of food waste was excluded from the scope of this modeling as it is considered as a true waste; it is an unwanted flow generated from food services and it is a burden to its system, not an intentional product. It can also be determined by the decision trees developed in the previous study. The food waste is no longer fit to be consumed by people, and it is currently disposed of. In addition, the final disposal causes a net cost to manage the food waste. Thus, the food waste is classified as waste by both decision trees developed in Chapter 2 (Figure 23).

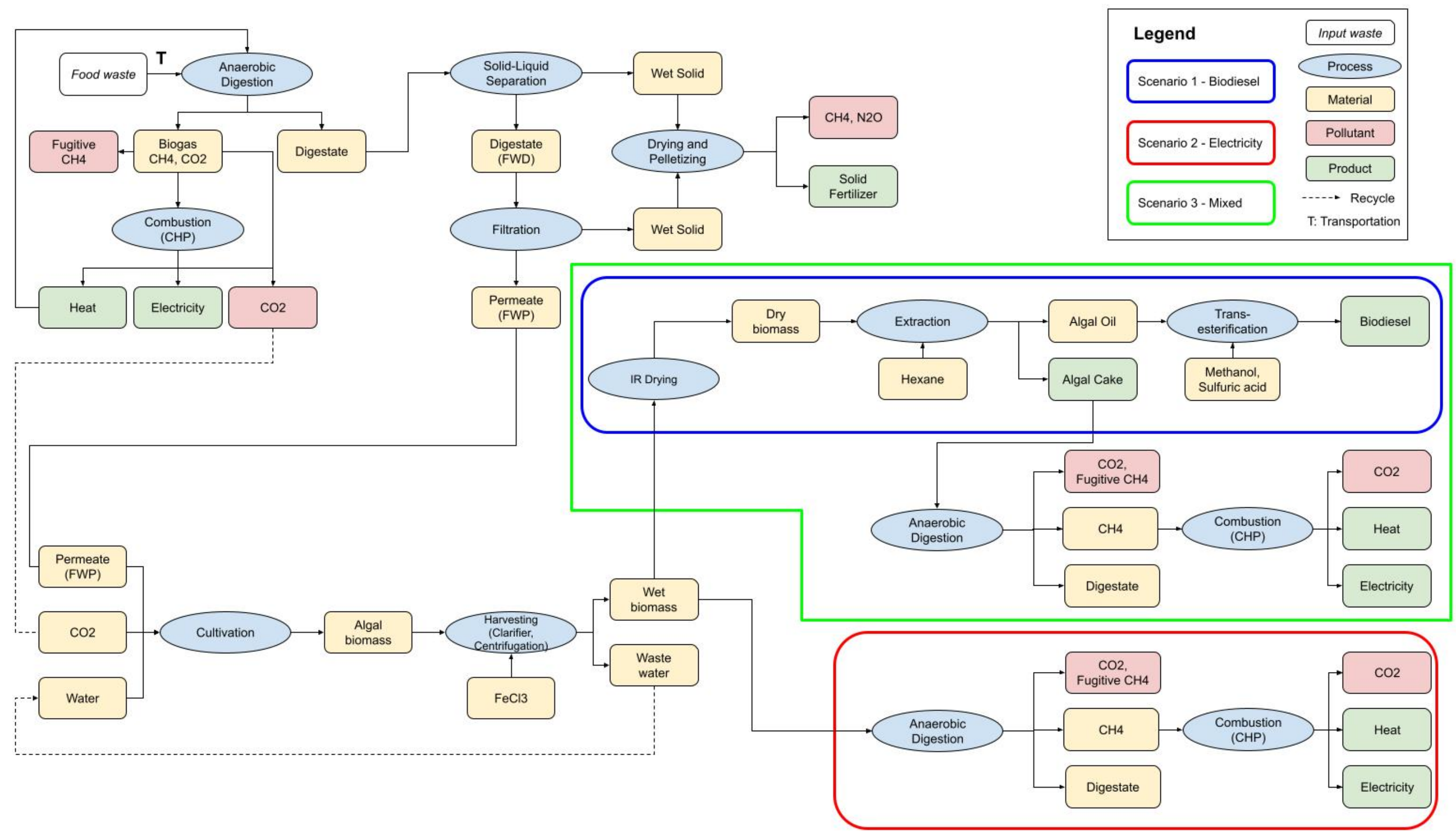

Figure 22. Flow diagram of the system in the study. Biodiesel or electricity is generated from microalgal biomass which is produced using waste streams from an anaerobic digester. Only major GHGs are shown as pollutants, and AD systems are shown separately for drawing purpose

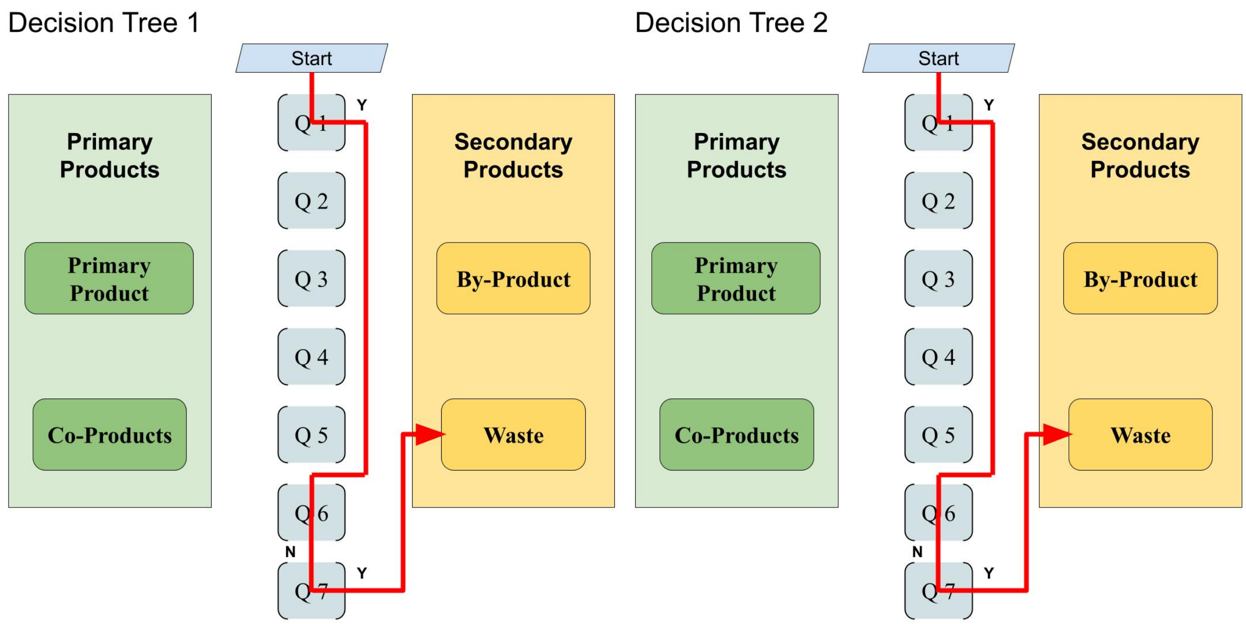

Figure 23. Decision tree results for food waste

Through the AD of food waste, biogas and digestate effluent are produced. The biogas is combusted in a combined heat and power (CHP) unit, and it provides electricity and heat for on-site demand as well as supplemental CO_2 to the microalgae pond. The digestate effluent is separated into a liquid fraction and a wet solid fraction with a screw press separation step. The liquid digestate is further separated using an ultrafiltration system, and the filtered liquid, called permeate, is used as a growth medium for the microalgae [122]. The solid fractions from the separation steps are dried and pelletized to make solid fertilizers.

The system produces two possible energy products, biodiesel from microalgae and electricity from biogas, and the processes included in the model vary depending on the final energy product. To produce biodiesel, microalgal biomass is harvested and dried with infrared radiation (IR), and then oil is extracted from dried microalgal biomass and converted to biodiesel through a catalytic upgrade. To generate electricity, harvested microalgal biomass is used as an additional feedstock for AD to generate biogas which is combusted in a CHP unit. In addition, microalgal residuals from the biodiesel production pathway are also considered as additional feedstocks for AD. Although the addition of microalgal biomass to the food waste digester may change properties of digestate or permeate, this study disregarded the changes because of low ratio (less than 1% by weight) of microalgae to food waste and insufficient experimental data. Future research may be performed to investigate the effects of adding microalgae to food waste for an AD operation.

In this study, the pathway for biodiesel production is called scenario 1, the pathway for electricity generation is called scenario 2, and the pathway including use of microalgal residuals for AD is called scenario 3.

4.2.2 Food waste collection and anaerobic digestion

The AD pathway was modeled based on the University of California Davis Renewable Energy Anaerobic Digester (UC Davis READ), which is a thermophilic digester using mixed food waste as a feedstock and designed to treat 45 metric tons of food waste per day [123]. The food waste is collected from UC Davis dining halls and nearby supermarkets and transported to the digester using a heavy-duty refuse truck running on compressed natural gas (CNG). At the digester, the food waste feedstock is loaded into the digester by a front-end loader using an internal combustion engine (ICE) with an efficiency of 42% [124].

The electricity and heat demand for the AD operation including pumping, mixing, and heating were modeled based on a report from the California Air Resources Board (CARB) [124], by extrapolating the values from the report to the scale of this study. The energy consumption for biogas purification was also estimated based on the CARB report. Biogas yield was assumed to be 7,190 m^3/day with 73% of methane, 26% of carbon dioxide and 1% of others [125]. This biogas is combusted in a CHP unit to generate electricity and heat. Fugitive methane emissions from this AD system are modeled as 4.9% of the generated methane [126]. Parameters for the modeled feedstock preparation, and AD and CHP systems are shown in Table 32.

Table 32. Parameters for feedstock preparation, anaerobic digestion, and CHP system

Description	Unit	Input	Data Source
Feedstock collection and Transportation			
Daily travel distance	km/day	25	modeled
Heavy-duty truck payload	tonne	19.7	[127]
Heavy-duty truck fuel economy	km/kg CNG	1.6	[127]
CNG consumption	MJ/day	1,692.9	[127]
Feedstock loading to digester			
Loading capacity	tonne/hr	22.7	[124]
Energy demand	kWh/day	325	[124]
Diesel consumption	kg/day	27.2	[124]
Digester operation			
Electricity demand	kWh/day	905.1	[124]
Heat demand	MJ/day	1,691.8	[124]
Heat exchanger efficiency	%	80	[124]
Biogas composition			
Methane	%	73	[125]
Carbon dioxide	%	26	[125]
Others (Oxygen, Nitrogen, Hydrogen sulfide, etc.)	%	1	[125]
Biogas purification electricity demand	kWh/day	2,305.3	[124]
CHP unit efficiency	%	39	[124]
Energy generation from the digester			
Electricity	kWh/day	19,318.5	modeled
Heat	MJ/day	78,462.9	modeled
On-site emission, methane	kg/day	169.0	[126]

4.2.3 Solid-liquid separation and ultrafiltration

This study assumes the digestate is utilized to produce solid fertilizers and as a medium for microalgae cultivation after the separation. The screw press and ultrafiltration were modeled for solid-liquid separation. The wet solids from the separation step were used as solid fertilizers after drying and pelletizing, and the liquid permeate was used for microalgae cultivation. In addition, energy consumption and emissions related to the transportation and the current treatment of the waste effluents were considered as avoided energy consumption and avoided emissions. Table 33 shows the parameters related to the separation and fertilizer production processes, and Table 34 shows the avoided energy consumption and emissions.

Table 33. Parameters for separation and fertilizer production processes

Description	Unit	Input	Data Source
Electricity demand			
Separation	kWh/m^3	0.55	[128]
Ultrafiltration	kWh/m^3	0.28	[129]
Drying and pelletizing	kWh/tonne pellet	46.8	[130]
On-site emissions			
Methane	kg/day	49.1	[130]
Nitrous oxide	kg/day	1.5	[130]

Table 34. Avoided energy consumption and emissions compared to transportation and disposal components of business-as-usual (BAU) case

Description	Unit	Input	Data Source
Transportation energy consumption, CNG	MJ/day	1,602.7	[127]
Emissions from transportation			
Carbon dioxide	kg/day	75.1	[127]
Methane	g/day	43.6	[127]
Nitrous oxide	g/day	0.27	[127]
Emissions from composting and landfill			
Methane	kg/day	52.3	[130]
Nitrous oxide	kg/day	2.43	[130]

4.2.4 Microalgae cultivation and growth

In this study, the liquid permeate from the ultrafiltration step is used for microalgae cultivation. The liquid permeate is diluted with freshwater to 10% by volume and supplied to an open raceway pond (ORP) for microalgae cultivation. The ORP and following downstream processing sites were in Davis, California, and the freshwater microalgal strain *Chlorella sorokiniana* is cultivated for its high lipid content and growth performance. Biomass density is assumed as 1.68 g/L (dry weight), and the lipid content is assumed as 30% of dry weight, which is reported for microalgae strain *Chlorella sp.* [131].

The total amount of the permeate generated from 45 metric tons of food waste treatment is 28.6 m^3 per day, and the total amount of media available after 10% dilution is 286.3 m^3 per day. The annual available media is 94,471.8 m^3. The model pond in this study has a total length of 100 m, width of 10 m, and depth of 0.3 m, and it is operated by circulation pumping and paddlewheel mixing. Carbon dioxide for microalgae was sourced from AD and methane combustion, and it was pumped into the ORPs using a regenerative blower with electricity demand of 1.14 kWh/m^3 CO_2 supplied [132] with assuming . The average residence time for each pond is 30 days, and the cultivation ponds are operated for 330 days per year. As a result, the total system for annual cultivation requires 30 ponds in total, and each pond is operated for 11 cycles per year. In this study, the model was calculated for one cycle of microalgae related operations, using the permeate from one batch of AD. This model was then scaled to represent the annual operation. Modeling annual operations means seasonality must be considered. In Davis, California, pond heating is only required for 3 months in winter, from December to February. It was assumed that heat energy from AD is used for the pond heating during the winter season. The calculations for heat demand are based on the specific heat of water and the assumption that the water must be heated from 8°C to 25°C during the winter months with assuming 80% efficiency. The energy required for heating is critical, and it may change significantly depending on thermal insulation and pond design. Table 35 shows the parameters related to microalgae cultivation system.

Table 35. Parameters related to microalgae cultivation system (Modeled: the values were assumed and put in the spreadsheets for related calculations)

Description	Unit	Input	Data Source
Pond structure			
Pond area	m^2	1,000	modeled
Pond volume	m^3	300	modeled
Operational parameters			
Residence time	day	30	modeled
Operation days	day/yr	330	modeled
Total media available	m^3/yr	94,472	modeled
Required number of ponds	counts	30	modeled
Number of cycles per pond	cycle/yr	11	modeled
Total number of cycles, all ponds	cycle/yr	330	modeled
Evaporation loss	m^3/day	0.43	modeled
Water recycle efficiency	%	75	modeled
Carbon dioxide supply	kg/day	2.27	modeled
Microalgal biomass characteristics			
Biomass density	g/L	1.68	[131]
Lipid content	%	30	[131]
Electricity demand			
Pumping	kWh/m^3	0.05	[132]
Paddle wheel mixing	kWh/m^3/day	0.02	[132]
Carbon dioxide supply	kWh/m^3 CO_2	1.14	[132]
On-site emissions			
Methane	g/cycle	8.24	[133]
Nitrous oxide	g/cycle	1.32	[133]

4.2.5 Downstream processes

The three scenarios in this study share all upstream processing through the harvesting step in common, but downstream processes vary depending on the final energy product selected. In scenario 1, the final energy products are biodiesel and electricity (biodiesel is the target product), and the downstream processes include IR drying, lipid extraction, and transesterification. In scenario 2, the final energy product is electricity, and the downstream process includes AD of the produced microalgal biomass. In scenario 3, the final energy products are biodiesel and electricity, and AD of microalgal residuals is added to the downstream processes in scenario 1.

In all scenarios, flocculation and centrifugation are assumed to dewater microalgal biomass. These steps are modeled based on the literature values. For flocculation, 125 mg of iron (III) chloride per 1-liter culture is used, and then the harvested culture is centrifuged at 5,000 rpm for 15 minutes [134]. The moisture content of the harvested microalgal biomass is assumed at 85.8%, and the energy requirement for harvesting process is 0.48 kWh/m^3 culture [135].

For scenario 1, IR drying at 100°C is modeled, and the microalgal biomass is dried from the moisture content of 85.8% to 20.5%. The electricity demand for IR drying at 100°C is 2,655.6 kJ/kg wet biomass. After drying, extraction and conversion are modeled for biodiesel production. Hexane, methanol, and sulfuric acid are required for these steps, and the requirements are 0.5 liter/kg dry algae, 0.05 L/kg dry algae, and 0.01 mL/mL methanol used respectively [136]–[138]. There is a hexane loss of 2.4 g/kg biomass, and the recycle efficiency of hexane is assumed as 95% [139]. Total energy requirement for extraction, conversion, and hexane recovery is 46 MJ per kg dry biomass [140]. After the extraction step, the residual biomass that is rich in protein and nutrients is assumed to be consumed as livestock or aquaculture feed.

For scenario 2, the microalgal biomass is directly pumped into the anaerobic digester for AD to generate electricity. The ratio of volatile solids (VS) to total solids (TS) (VS/TS) of microalgal paste is assumed as 83.1% [141], and the methane yield of microalgal biomass is modeled at 212 mL/g VS with a methane content of 65% [142]. The electricity generation from the microalgal biomass is calculated based on these parameters (Equation 3), and other operational conditions such as electricity demand for digester operation, electricity demand for biogas purification, and CHP unit efficiency are imported from the CARB report [124] as shown in Table 32.

Equation 3:

$$Electricity\ generation\ (MJ) = \left(\left(M_{TS} \times \frac{VS}{TS} \times Y_{methane}\right) - E_{methane}\right) \times LHV_{methane} \times \eta$$

$$M_{TS}\ (kg): Mass\ of\ biomass\ (total\ solids)$$

$$\frac{VS}{TS}: Ratio\ of\ volatile\ solids\ (VS) to\ total\ solids (TS)$$

$$Y_{methane}\ (\frac{L}{kg\ VS}): Methane\ yield$$

$$E_{methante}(L): Methane\ loss\ as\ fugitive\ emissions$$

$$LHV_{methane}\left(\frac{MJ}{L}\right): Lower\ Heating\ Value\ of\ methane$$

$$\eta: CHP\ unit\ electric\ efficiency$$

For scenario 3, electricity generation from the AD of microalgal residual is added to the analysis for scenario 1. The microalgal residual is directly pumped into the anaerobic digester, and the VS/TS ratio of microalgal residual is assumed at 89.9%. Depending on the microalgae species, the VS/TS ratios of lipid-extracted algae vary $\pm$ 10% from non-extracted algae. This study assumes 8% increase of the ratio, which is from an analysis of *Chlorella vulgaris* [143]. In addition, the methane yield of microalgal residual is modeled at 198 mL/g VS with a methane content of 65% assuming that the yield decreases as shown for *Chlorella vulgaris* case [143].

4.2.6 Life cycle assessment (LCA) model

The scope for the LCA starts with the transport of food wastes to the anaerobic digester and ends with the production of final energy products at the facility gate. Initial construction of the production facility and the anaerobic digester was excluded from the scope for this study, because previous biofuel LCA studies have shown that facility construction has a small effect on energy and CI estimates [144].

Defining the functional unit is often a critical step in LCA. The functional unit is usually based on the service provided by a system or a product for which the LCA results are reported. A common functional unit for biofuel is 1 MJ of biofuel. However, this study models a multifunctional system; the system generates several products including biofuel, electricity, and fertilizer. Thus, each product could have its own functional unit, and the results of this study were provided in terms of 1 MJ of biofuel or 1 kWh of electricity along with the results for all the products. It should be noted that the functional unit of LCA is different from the modeling unit of this study. The modeling unit of this study is based on one batch of AD operation, and it is one cycle of microalgae related operations (from cultivation to energy product generation) utilizing permeate from one batch of AD. Since the permeate from the digester was supplied daily, the total annual results were calculated by multiplying 330 (operating days/year) to the results from the modeling unit.

Since LCA is a tool for assessing potential environmental impacts of a product or a service throughout its life cycle including the full supply chain of inputs, the scope includes not only direct resource inputs and on-site emissions, but also resource consumption and emissions associated with the complete supply chain. These upstream resource inputs and emissions of the complete supply chain are modeled using reference life cycle inventory (LCI) datasets. The LCI refers to the accounting of direct and upstream resource inputs and emission outputs over the system. In this study, reference LCIs were obtained from the GaBi software tool which provides an access to reference LCI datasets such as GaBi Professional extension and Ecoinvent databases (Table 36). To maintain geographic consistency, LCI data for the United States were collected. However, global, or European LCI data were used where United States data were not available. These flows collected from the database were then processed into environmental indicators of interest using impact assessment characterization factors. As an example, all greenhouse gas (GHG) emissions are converted to carbon dioxide equivalents (CO_2e) during this step to facilitate summing of all GHGs into one indicator. For impact assessment, the CML method was chosen, which includes 11 different impact categories [109]. The impact categories and their characterization factor units are shown in Table

37. In addition to this impact assessment, the net energy ratio (NER), which is a ratio of produced energy to consumed primary energy, is included (Equation 4). A higher NER is desirable.

Equation 4:

$$Net\ Energy\ Ratio\ (NER) = \frac{Produced\ energy}{Consumed\ primary\ energy}$$

Table 36. LCI data source (Geographic regional code - GLO: Global, RER: Europe, US: United States) [145], [146]

Inputs	Name (Geographic regional code)	Source
Water	Process water from surface water (US)	GaBi Professional extension
Electricity	Electricity grid mix (US)	GaBi Professional extension
Natural gas	Market for natural gas, high pressure (US)	Ecoinvent 3.6
Diesel	Diesel, burned in agricultural machinery (GLO)	Ecoinvent 3.6
H_2SO_4	Market for sulfuric acid (RER)	Ecoinvent 3.6
Hexane	Market for hexane (GLO)	Ecoinvent 3.6
$FeCl_3$	Market for iron (III) chloride, without water (GLO)	Ecoinvent 3.6
Methanol	Market for methanol (GLO)	Ecoinvent 3.6

Table 37. Environmental impact categories and their characterization factor units

Category	Characterization factor unit
Abiotic depletion, elements (ADP elements)	kg antimony (Sb) eq.
Abiotic depletion, fossil (ADP fossil)	MJ
Acidification potential (AP)	kg SO_2 eq.
Eutrophication potential (EP)	kg PO_4 eq.
Freshwater aquatic ecotoxicity potential (FAETP)	kg dichlorobenzene (DCB) eq.
Global warming potential (GWP)	kg CO_2 eq.
Human toxicity potential (HTP)	kg DCB eq.

Marine aquatic ecotoxicity potential (MAETP)	kg DCB eq.
Ozone layer depletion potential (ODP)	kg chlorofluorocarbon (CFC)-11 eq.
Photochemical ozone creation potential (POCP)	kg C_2H_4 eq.
Terrestrial ecotoxicity potential (TETP)	kg DCB eq.

Allocation in LCA refers to the partitioning of the LCI or impacts among co-products generated from the system. The energy products and co-products generated in this study include biodiesel, electricity, solid fertilizer and microalgal residuals. While there is a large body of work debating the appropriateness of different allocation approaches, this study applies an economic allocation method which partitions environmental impacts among products based on their economic values. Economic values are taken from either current market values or estimated values from literature (Table 38).

Table 38. Economic values of the products from the system

Product	Value	Unit	Reference
Biodiesel	1.087	\$/kg	[147]
Electricity	0.15	\$/kWh	[102]
Solid fertilizer	5.5	\$/kg N-P-K	[141]
Microalgal biomass residual	0.175	\$/kg	[148]

4.2.7 Techno-economic analysis (TEA) model

The goal of techno-economic analysis (TEA) modeling is to estimate the cost-benefit of a production system to predict the economic feasibility and potential profitability of a project. Finding economic hot-spots and testing out the performance of different technologies or scales of production are also potential goals of TEA. In this study, the focus was on calculating economic viability of the products and identifying hot spots for cost.

TEA typically considers capital expenses (CAPEX) and operating expenses (OPEX). CAPEX was included in this assessment. Because of the novelty of the modeled system and the uncertainty of modeling a simulated system that requires design choices that might not reflect optimality for the system of interest, CAPEX estimates are uncertain. CAPEX estimates were drawn from other studies of commercial-scale microalgae cultivation systems that are larger than the modeled system. Cost estimates were drawn from a study from a 2012 National Renewable Energy Laboratory (NREL) study [149] and adjusted based on the conditions of this modeled system. Depending on the characteristics of a cost parameter, it was linearly scaled based on either pond size or microalgal oil production amount by using scaling exponential factor of 1. Although the scaling factor may vary for each unit process, this study assumed the same scaling factor as 1. For example, the pond liners cost was calculated using a size ratio of pond size, and the solvent extraction cost was calculated using a size ratio of oil production. The calculation method is shown in Equation 5. The dollar values were also adjusted to 2019 dollars based on the Consumer Price Index (CPI). Table 39 summarizes the cost parameters as reported by NREL.

Equation 5:

$$Scaled\ cost = Reference\ cost\ \times Size\ ratio^{Scaling\ exponential\ factor}$$

Table 39. Cost parameters from the National Renewable Energy Laboratory report (MGY: million gallons per year, MM: millions) ($, 2019)

Parameters	Value	Unit	Calculation method
Production Scale			
Algal oil production	10.4	MGY	
Facility size	4050	ha	
Pond size	405	ha	
Capital expenses (CAPEX)			
Ponds and paddle wheels	138.60	$MM	scaled by pond size
Pond liners	205.2	$MM	scaled by pond size
Flue gas delivery and distribution	38.7	$MM	scaled by pond size
Water delivery and distribution	3.7	$MM	scaled by pond size

Settling	47	$MM	scaled by oil production
Centrifuge	4.4	$MM	scaled by oil production
Cell rupturing	14.1	$MM	scaled by oil production
Solvent extraction	4.2	$MM	scaled by oil production
Land cost	35.9	$MM	scaled by pond size
Inoculum production system	50.2	$MM	scaled by pond size
Conversion process	6.0	$MM	scaled by oil production
Water pumps	13.2	$MM	scaled by pond size
Operating expenses (OPEX)			
Labor and overhead	8.2	$MM/yr	scaled by pond size
Maintenance, insurance	19.2	$MM/yr	scaled by pond size
Other costs			
Warehouse	1	%	of inside boundary limit CAPEX
Proratable cost	10	%	of total direct cost
Field expenses	10	%	of total direct cost
Office construction	10	%	of total direct cost
Contingency	20	%	of total direct cost
Other cost	5	%	of total direct cost
Working capital	5	%	of fixed capital investment

The cost of the chemicals and other input resources required for this system were included in operating costs. Table 40 shows the cost parameters of the chemicals and other input resources.

Table 40. Cost parameters of the chemicals and input resources

Element	Cost	Unit	Reference
Water	0.013	$\$/m^3$	[150]
$FeCl_3$	0.66	$/kg	[151]
Hexane	652.5	$\$/m^3$	[152]
Methanol	479.5	$\$/m^3$	[152]
Sulfuric acid	387.9	$\$/m^3$	[153]
Diesel	0.84	$/liter	[154]
CNG (transport)	0.56	$/liter	[155]

With the cost data, required revenues of the target energy products to reach the break-even were determined based on financial assumptions, and the prices of the target products were calculated based on the required revenues (Equation 2). The financial assumptions for TEA were imported from the NREL study [149] and the Energy Cost Calculator published by the California Biomass Collaborative [156]. Table 41 shows the financial assumptions for the cost calculation. As a cash flow method, a discounted cash flow rate-of-return was used, and depreciation was considered as a straight line for 20 years.

Table 41. Financial assumptions for techno-economic analysis

Parameters		Unit
Tax rate	30	%
General inflation	2	%
Debt ratio	50	%
Equity ratio	50	%
Interest rate on debt	5	%
Plant lifetime	20	year
Cost of money	10	%

4.3 Results and Discussion

4.3.1 Inputs and outputs analysis

Assuming 45.4 metric tons of food waste influent into the digester per day, 7,190 m^3 of methane was produced from the anaerobic digester, and 438.6 kg (dry weight) of microalgal biomass was harvested from the ORP using 10% diluted permeate from the digester after a 30-day cultivation period. The facility is modeled to operate for 330 days per year, and the annual results are calculated by multiplying 330 (days/year) to the results for one-day operation. The heating energy was only required for a quarter of a year (82.5 days/year, a quarter of 330 days/year), so the heating energy requirement was calculated separately and then added to the total annual results calculated without the heating requirement. During winter, it required 25,429 MJ of heat energy daily which was 2,097,893 MJ for the entire season.

In scenario 1 and 3, which consider the biodiesel as an energy product, 44,825 L/yr of biodiesel were produced from the harvested microalgal biomass. While the pathway in scenario 1 generated 101,050 kg/yr of microalgal biomass residuals, 623,514 kg/yr of solid fertilizers, and 1,083,738 kWh/yr of electricity (net) as co-products, the pathway in scenario 3 produced 623,514 kg/yr of solid fertilizers and 1,139,634 kWh/yr of electricity along with the biodiesel. In scenario 2, which considers the electricity as a target product, net electricity generation from the system was 5,166,490 kWh/yr, and 623,514 kg/yr of solid fertilizer was produced as a co-product. The inputs and outputs for energy, chemicals, and materials along with direct GHG emissions from the system are shown in Table 42. Since avoided emissions from the current practice of digestate treatment were considered as a part of direct emissions in this study, the negative values in Table 42 mean the avoided emissions.

Table 42. Energy, chemicals, and materials inputs/outputs and direct GHG emissions (net outputs for electricity and heat)

		Scenario 1	Scenario 2	Scenario 3	Unit
Inputs	Diesel	386,100	386,100	386,100	MJ/yr
	CNG	794	794	794	m^3/yr
	Water	43,186	43,186	43,186	tonne/yr
	$FeCl_3$	10,628	10,628	10,628	kg/yr
	Hexane	4,371	-	4,371	liter/yr
	Methanol	7,710	-	7,710	liter/yr
	Sulfuric acid	79	-	79	liter/yr
Outputs	Biodiesel	44,825	-	44,825	liter/yr
		1,598,891	-	1,598,891	MJ/yr
	Solid fertilizer	623,514	623,514	623,514	kg/yr
	Algae biomass residual	101,050	-	-	kg/yr
Net outputs	Electricity	1,083,738	5,166,490	1,139,634	kWh/yr
	Heat	23,236,608	23,576,264	23,498,936	MJ/yr
Direct GHG emissions	CO_2	29,860	29,860	29,860	kg/yr
	CH_4	1,378,757	1,399,453	1,393,313	kg CO_2e kg/yr
	N_2O	-80,364	-80,364	-80,364	kg CO_2e kg/yr

4.3.2 LCA results

Annual total primary energy consumption and total environmental impacts of the system are shown in Table 43. The system for scenario 1 requires 3,767 GJ/yr of total energy which is composed of 167 GJ of renewable energy and 3,600 GJ of non-renewable fossil energy, the system for scenario 2 requires 1,620 GJ/yr of total energy (146 GJ renewable and 1,474 GJ non-renewable), and the system for scenario 3 requires 3,767 GJ/yr of total energy (167 GJ renewable and 3,600 GJ non-renewable). The primary energy consumption for scenario 3 was unchanged from scenario 1, because the additional process in scenario 3, the AD of lipid extracted microalgal residual, requires no primary energy consumption. The process was operated by the electricity generated in the system, and thus, only net electricity generation was affected by the additional step. Similarly, the total environmental impacts of scenario 3 were unchanged from scenario 1 except for the GWP, which was changed by direct emissions from the AD process. Although the addition of microalgal residual to the digester may affect characteristics of permeate or AD operation and cause environmental impact changes, this study excluded these changes due to a lack of experimental results.

Table 43. Total primary energy consumption and environmental impacts per year for scenario 1,2, and 3 (ADP: Abiotic depletion, AP: Acidification potential, EP: Eutrophication potential, FAETP: Freshwater aquatic ecotoxicity potential, GWP: Global warming potential, HTP: Human toxicity potential, MAETP: Marine aquatic ecotoxicity potential, ODP: Ozone layer depletion potential, POCP: Photochemical ozone creation potential, TETP: Terrestrial ecotoxicity potential)

Category	Description	Scenario 1	Scenario 2	Scenario 3	Unit
Primary energy consumption	Fossil energy	3,600	1,474	3,600	GJ/yr
	Renewable energy	167	146	167	GJ/yr
	Total	3,767	1,620	3,767	GJ/yr
Environmental impacts	ADP elements	8.06E-01	6.20E-01	8.06E-01	kg Sb eq./yr
	ADP fossil	3.60E+06	1.47E+06	3.60E+06	MJ/yr
	AP	9.90E+02	6.83E+02	9.90E+02	kg SO_2 eq./yr
	EP	2.86E+02	1.33E+02	2.86E+02	kg PO_4 eq./yr
	FAETP	5.53E+04	4.60E+04	5.53E+04	kg DCB eq./yr
	GWP	1.62E+06	1.61E+06	1.63E+06	kg CO_2 eq./yr
	HTP	1.34E+05	1.10E+05	1.34E+05	kg DCB eq./yr
	MAETP	3.03E+08	2.71E+08	3.03E+08	kg DCB eq./yr
	ODP	4.07E-02	1.57E-02	4.07E-02	kg CFC-11 eq./yr
	POCP	4.86E+02	4.03E+02	4.86E+02	kg C_2H_4 eq./yr
	TETP	1.69E+03	1.33E+03	1.69E+03	kg DCB eq./yr

To understand the performance of each energy product and other co-products, the impacts were allocated to the products based on their economic values. The economic value share of each product is shown in Figure 24. While the fertilizer shows the highest total economic value in scenario 1 and 3, the electricity has a higher total value than the fertilizer in scenario 2. Based on the economic value shares, the impacts are allocated to each product normalized with the target product for each scenario (Tables 44 – 46). The results for scenarios 1 and 2 are allocated and normalized based on their target products, 1 MJ of biodiesel and 1 MJ of electricity, respectively. On the other hand, in scenario 3, the system has two energy products based on microalgal biomass, biodiesel and electricity, and biodiesel was selected (1 MJ of biodiesel) for normalization. According to the results, 2.44 MJ of electricity, 0.39 kg of fertilizer, and 0.06 kg of residual were produced with 1 MJ of biodiesel in scenario 1, 0.03 kg of fertilizer was produced with 1 MJ of electricity in scenario 2, and 2.57 MJ of electricity and 0.39 kg of fertilizer were produced with 1 MJ of biodiesel in scenario 3. With the additional AD step in scenario 3, 0.13 MJ of electricity was generated additionally in lieu of 0.06 kg of microalgal residual in scenario1. In addition, GWPs shown in Tables 44 – 46 represent GHG emissions attributable to products. The GHG emission of biodiesel from scenario 1 was 82.0 gCO_2e/MJ of electricity from scenario 2 was 62.7 gCO_2e/MJ, and of biodiesel from scenario 3 was 84.2 gCO_2e/MJ. The value of each product from this study was lower than its market competitor (94.71 gCO_2e/MJ of conventional diesel and 81.49 gCO_2e/MJ of California grid electricity [157])

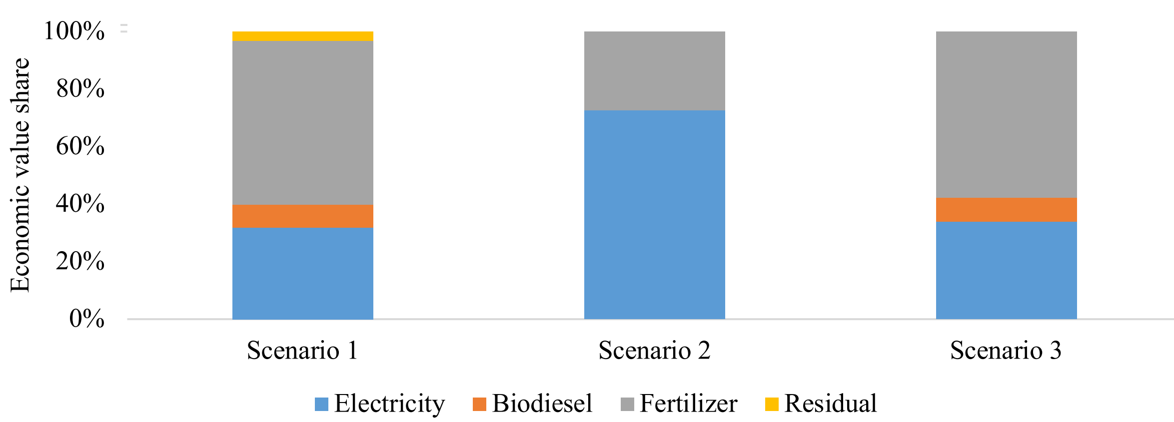

Figure 24. Economic value share of each product

Table 44. Allocated and normalized environmental impacts, Scenario 1

Impact category	Unit	Biodiesel	Electricity	Fertilizer	Residuals
		1 MJ	2.44 MJ	0.39 kg	0.06 kg
ADP elements	kg Sb eq.	4.09E-08	1.60E-07	2.86E-07	1.71E-08
ADP fossil	MJ	1.83E-01	7.14E-01	1.28E+00	7.63E-02
AP	kg SO2 eq.	5.02E-05	1.96E-04	3.52E-04	2.10E-05
EP	kg PO4- eq.	1.45E-05	5.66E-05	1.02E-04	6.05E-06
FAETP	kg DCB eq.	2.81E-03	1.10E-02	1.97E-02	1.17E-03
GWP	kg CO2 eq.	8.20E-02	3.20E-01	5.74E-01	3.42E-02
HTP	kg DCB eq.	6.80E-03	2.66E-02	4.76E-02	2.84E-03
MAETP	kg DCB eq.	1.54E+01	6.02E+01	1.08E+02	6.43E+00
ODP	kg CFC-11 eq.	2.07E-09	8.08E-09	1.45E-08	8.63E-10
POCP	kg C2H4 eq.	2.47E-05	9.63E-05	1.73E-04	1.03E-05
TETP	kg DCB eq.	8.60E-05	3.36E-04	6.02E-04	3.59E-05

Table 45. Allocated and normalized environmental impacts, Scenario 2

Impact category	Unit	Electricity	Fertilizer
		1 MJ	0.03 kg
ADP elements	kg Sb eq.	2.42E-08	9.11E-09
ADP fossil	MJ	5.76E-02	2.16E-02
AP	kg SO2 eq.	2.67E-05	1.00E-05
EP	kg PO4- eq.	5.18E-06	1.95E-06
FAETP	kg DCB eq.	1.80E-03	6.76E-04
GWP	kg CO2 eq.	6.27E-02	2.36E-02
HTP	kg DCB eq.	4.29E-03	1.61E-03

MAETP	kg DCB eq.	1.06E+01	3.99E+00
ODP	kg CFC-11 eq.	6.15E-10	2.31E-10
POCP	kg C2H4 eq.	1.57E-05	5.92E-06
TETP	kg DCB eq.	5.21E-05	1.96E-05

Table 46. Allocated and normalized environmental impacts, Scenario 3

Impact category	Unit	Biodiesel	Electricity	Fertilizer
		1 MJ	2.57 MJ	0.39 kg
ADP elements	kg Sb eq.	4.16E-08	1.71E-07	2.91E-07
ADP fossil	MJ	1.86E-01	7.64E-01	1.30E+00
AP	kg SO2 eq.	5.11E-05	2.10E-04	3.58E-04
EP	kg PO4- eq.	1.48E-05	6.06E-05	1.03E-04
FAETP	kg DCB eq.	2.86E-03	1.17E-02	2.00E-02
GWP	kg CO2 eq.	8.42E-02	3.46E-01	5.90E-01
HTP	kg DCB eq.	6.93E-03	2.84E-02	4.85E-02
MAETP	kg DCB eq.	1.57E+01	6.44E+01	1.10E+02
ODP	kg CFC-11 eq.	2.10E-09	8.64E-09	1.47E-08
POCP	kg C2H4 eq.	2.51E-05	1.03E-04	1.76E-04
TETP	kg DCB eq.	8.76E-05	3.60E-04	6.13E-04

4.3.3 TEA results

Tables 47 and 48 show the results from TEA. To calculate the prices of target products, microalgal biodiesel in scenario 1 and electricity from microalgal biomass in scenario 2, revenues from the other co-products were deducted from the operating cost, and it is called as net operating costs in this study. In scenario 3, the price of biodiesel was calculated assuming that the net electricity generation is a revenue source, which means revenues from the electricity were deducted from the operation cost. Like the LCA results, TEA results for scenario 3 were unchanged from scenario 1 except for other revenues and net operating costs. The additional AD process in scenario 3 required no supplementary CAPEX or OPEX, and it only affected the operating revenues from the electricity which was supplied to the process.

Based on the total capital costs and the net operating costs, total present value of required revenue was calculated and assigned to the target products. The price of biodiesel produced in scenario 1 is $6.47 per liter, and the price of the electricity generated in scenario 2 is $0.06 per kWh. Compared to the biodiesel in scenario 1, the biodiesel produced in scenario 3 requires higher price at $6.67 per liter. The difference in the prices was due to revenues from other products. While scenario 1 includes electricity, fertilizer, and microalgal residual as revenue sources, scenario 3 includes electricity and fertilizer. According to the prices shown in Table 38, 101,050 kg of microalgal residuals are more valuable than 55,897 kWh of electricity (net) generated from AD of the residuals, and thus, the other revenues decrease in scenario 3. However, the price of microalgal residuals as livestock or aquaculture feed may vary depending on temporal and regional variation or market conditions, and the change affects the price of biodiesel. For example, if the price of microalgal residual decreases by 52% ($ 0.085/kg), the price of biodiesel in scenario 1 will be equal to the price in scenario 3.

Table 47. Capital expenses ($, 2019)

	Scenario 1 & 3	Scenario 2
Direct capital costs		
Ponds with paddle wheels	1,274,000	1,274,000
Pond liners	1,886,000	1,886,000
Flue gas delivery and distribution	356,000	356,000
Water delivery and distribution	34,000	34,000
Settling	69,000	69,000
Centrifuge	6,000	6,000

Drying process	725	0
Cell rupturing	21,000	0
Solvent extraction	6,000	0
Land cost	330,000	330,000
Inoculum production system	461,000	461,000
Water pumps	121,000	121,000
Conversion process	9,000	0
Fertilizer process	1,250,000	1,250,000
Hexane recovery	454,000	0
Total installed depreciable capital	5,947,000	5,457,000
Total installed non-depreciable capital	330,000	330,000
Total installed capital	6,277,000	5,787,000
Warehouse	49,000	39,000
Total direct capital cost	5,995,000	5,495,000
Indirect capital costs		
Proratable cost	599,000	549,000
Field expenses	599,000	549,000
Office construction	599,000	549,000
Contingency	1,199,000	1,099,000
Other costs	300,000	275,000
Total indirect capital cost	3,297,000	3,022,000
Fixed capital investment	9,292,000	8,517,000
Working capital	465,000	426,000
Total capital cost (investment)	10,087,000	9,273,000

Table 48. Operating expenses ($/year)

	Scenario 1	Scenario 2	Scenario 3
Chemical input			
Flocculant (FeCl3)	6,974	6,974	6,974
Hexane	2,852	0	2,852
Methanol	3,697	0	3,697
Sulfuric acid	31	0	31
Water	570	570	570
Fertilizer production process	234,177	234,177	234,177
Hexane recovery process	45,384	0	45,384
Labor and overhead	73,000	73,000	73,000
Maintenance, insurance	171,000	171,000	171,000
Gross operating cost	537,685	485,721	537,685
Other revenues			
Electricity	165,487	(target)	174,022
Fertilizer	296,611	296,611	296,611
Microalgal biomass residual	17,684	0	0
Avoided cost			
Transportation cost for digestate	275,196	275,196	275,196
Wastewater treatment cost for digestate	584,560	584,560	584,560
Net operating cost (negative for revenue)	-802,000	-671,000	-793,000

4.3.4 Discussion

In scenario 1, GHG emissions attributable to biodiesel, represented by units of CO_2-equivalent (CO_2e), are 82.0 gCO_2e/MJ, and total energy required is 191.2 kJ/MJ biodiesel. The NER for biodiesel from scenario 1 is 5.2, meaning that consuming 1 MJ of primary energy can generate 5.2 MJ of biodiesel. In scenario 3, GHG emissions attributable to biodiesel are 84.2 gCO_2e/MJ, and the NER is 5.1. While results from other studies vary in large range as shown in Figure 25, the results from this study show relatively higher values than other studies for both GHG emissions and NER. This means that the microalgal biodiesel production modeled in this study emits more GHGs but requires less primary energy consumption compared to other studies. One of the key differences among the studies is a method for drying and extraction steps. The method for drying and extraction affects not only environmental impacts but also economic aspects [158]. While this study assumed dry solvent extraction, a method which extracts lipids from microalgal biomass after drying, there are other methods to process microalgal oil such as wet solvent extraction, hydrothermal liquefaction (HTL), and pyrolysis. Future research may consider implementation of these various methods into the model for the comparison study.

Meanwhile, the NER from this study was relatively high compared to other studies, which means that the system in this study required less energy than other systems to produce an equal amount of energy. The higher NER of this system may be attributed to the tie-in with the AD system. While other studies assumed typical microalgal cultivation with synthetic nitrogen and mineral phosphorous fertilizers, major contributors to primary energy consumption [159], [160], the system in this study avoided the use of fertilizers by utilizing waste streams from the digester. In addition, the electricity from the system decreased a demand for primary energy of the system, and it also decreased the allocated primary energy consumption to the biodiesel.

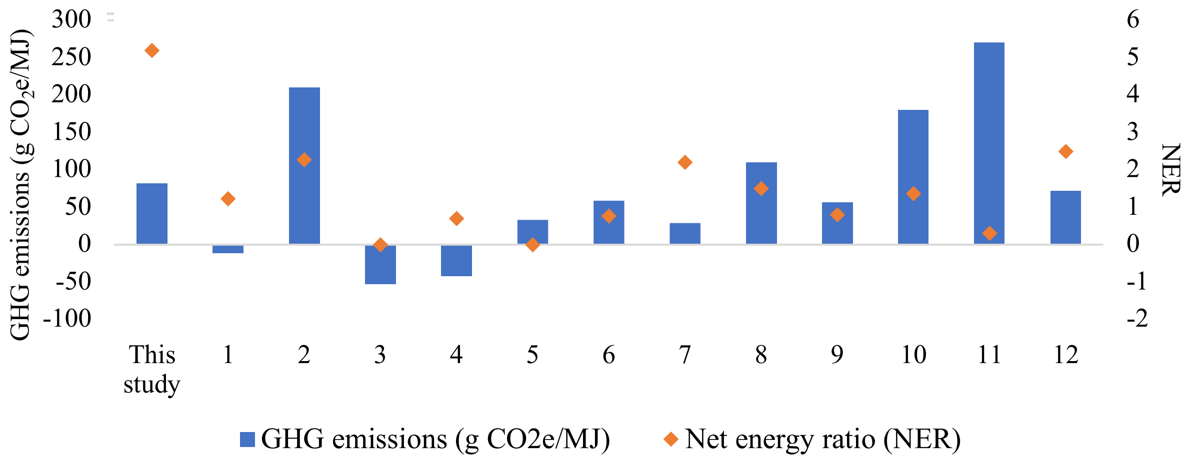

Figure 25. GHG emissions and net energy ratio (NER) results from other studies on microalgal biodiesel. Entries on the x-axis correspond with the following pathway and reference: Hydrothermal liquefaction: 1 [161]*, 3* [160]*, 5* [159] */ Pyrolysis: 2* [161] */ Wet solvent extraction: 4* [162]*, 7* [163]*, 8* [164]*, 10* [136]*, 12* [165] */ Dry solvent extraction: 6* [166]*, 9* [167]*, 11* [165]

The most notable issue, however, is the high price required for biodiesel for this system to simply break-even from a cost perspective. The prices for biodiesel calculated in scenarios 1 and 3 are \$6.47/liter and \$6.67/liter, respectively. These prices are both more than six-fold higher than the biodiesel market price (\$0.96/liter) and other results shown in various studies on microalgal biodiesel (e.g., \$2.60/liter [150], \$2.18 - 3.76/liter [168], \$0.42 – 0.97/liter [169]). The estimated price is high even though it is calculated using a scaling factor of 1 based on the assumptions of a study with larger scale production, which is shown for 38 million liters per year of biodiesel production with 4 ponds, the size of 405 ha each [149]. If a scaling factor smaller than 1 is used, the CAPEX will increase as well as the price of biodiesel.

The price is still high even including the low carbon fuel standard (LCFS) credit to the calculation. According to the Monthly LCFS Credit Transfer Activity Report provided by California Air Resources Board, the average price of a carbon credit in 2019 was \$192 per metric ton CO_2 equivalent. Since the carbon intensity of biodiesel from this study is lower than conventional diesel in the United States, it is eligible to receive credits from it. It decreases by 12.69 g CO_2e/MJ, so the total annual reduction is 20.29 metric tons CO_2e when replacing the conventional diesel with biodiesel from this study (on an energy basis) and the total annual revenue from the credit is \$3,895.68, or \$0.09/liter. The final biodiesel price including the LCFS credit is \$6.38/liter, still higher than the biodiesel market price and the results from previous studies. The results are shown in Table 49. For the LCFS credit analysis, the results for biodiesel from scenario 1 were used, because the biodiesel from scenario 1 showed a lower carbon intensity and a lower price than the biodiesel from scenario 3.

Table 49. GHG reduction and related LCFS credit for Scenario 1

		Biodiesel (Scenario 1)	Conventional diesel
Carbon Intensity (CI)	g CO_2e/MJ	82.02	94.71
Reduced CI	g CO_2e/MJ	12.69	
Annual production, this study	MJ/yr	1,598,891	
Annual CO_2 reduction	tonne CO_2e/yr	20.29	
Average credit price	\$/tonne CO_2e	192	
Annual revenues from credit	\$/yr	3,895.68	
Credit revenue per unit fuel	\$/liter	0.09	

In scenario 2, GHG emissions attributable to electricity are 62.74 gCO_2e/MJ, total energy required is 63.3 kJ/MJ electricity, and the required price for electricity is \$0.06/kWh. The NER for electricity from scenario 2 is 15.8, and it means that consuming 1 MJ of primary energy can generate 15.8 MJ of electricity. Compared to the average carbon intensity of California grid electricity, approximately 81.49 gCO_2e/MJ [157], the result from this study has a lower carbon intensity. The results also show a lower price than the average retail price of electricity in California, which is \$0.15/kWh [102].

According to the results, using microalgal biomass for electricity generation through AD, scenario 2 in this study, is a better option in terms of environmental and economic performance. Scenario 2 shows lower total environmental impacts for all impact categories, lower GHG emissions per unit energy produced (MJ), and higher NER value than the scenarios 1 and 3 in this study. It also shows a lower cost compared to market prices. In addition, the environmental impacts of common products (1 MJ electricity and 1 kg fertilizer) were also lower in scenario 2. Figures 26 - 28 show the relative percentage of scenario 1 and 2 compared to scenario 3 for each impact category, because the impacts were the highest in scenario 3.

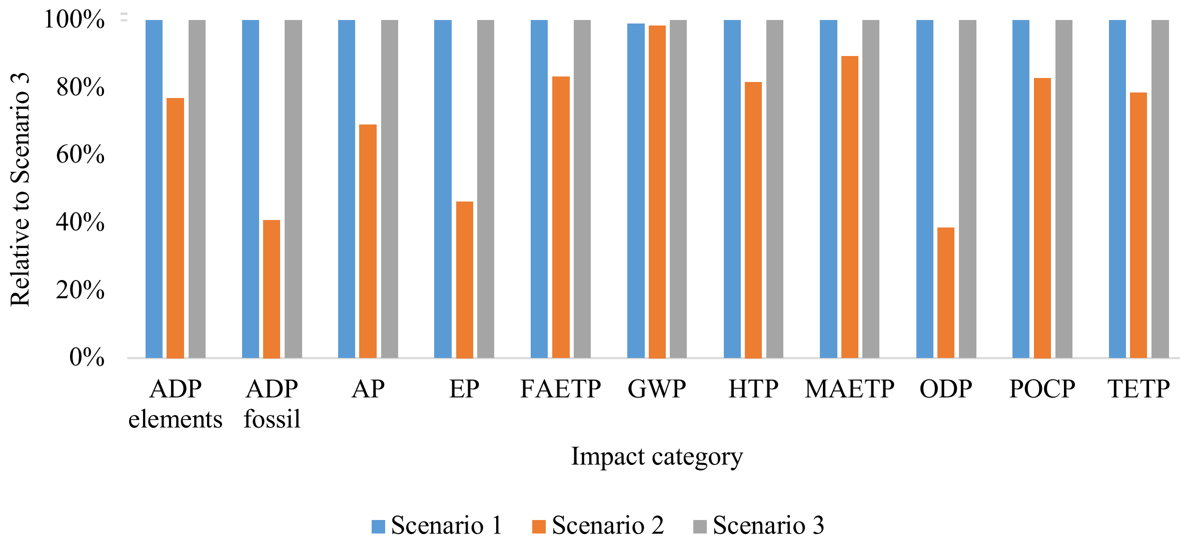

Figure 26. Comparison of total environmental impacts for scenario 1,2 and 3. Y-axis shows relative percentages of scenario 1 and 2 compared to scenario 3. (ADP: Abiotic depletion, AP: Acidification potential, EP: Eutrophication potential, FAETP: Freshwater aquatic ecotoxicity potential, GWP: Global warming potential, HTP: Human toxicity potential, MAETP: Marine aquatic ecotoxicity potential, ODP: Ozone layer depletion potential, POCP: Photochemical ozone creation potential, TETP: Terrestrial ecotoxicity potential)

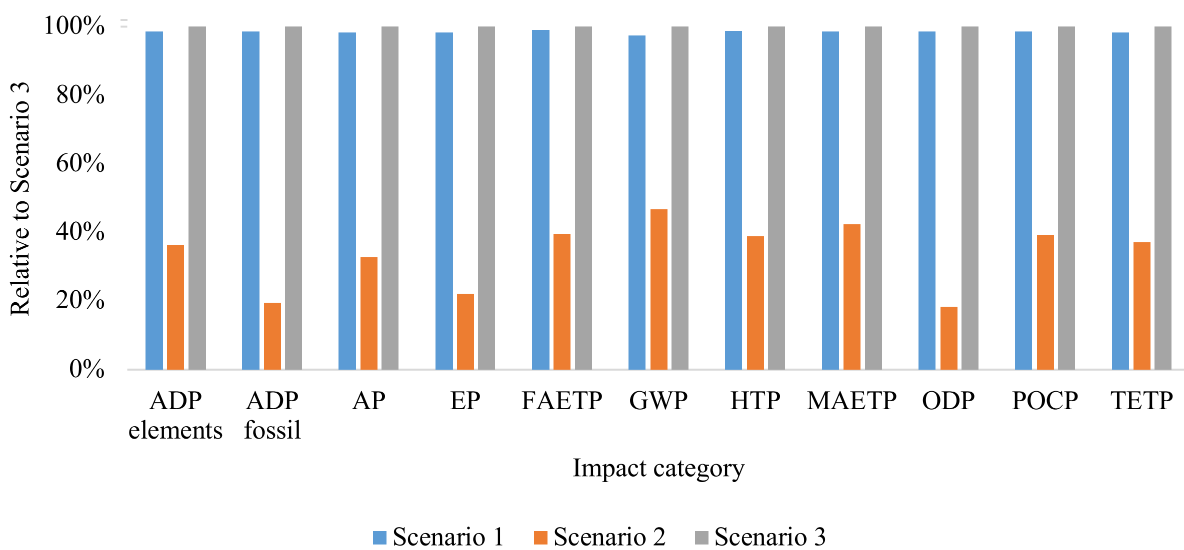

Figure 27. Comparison of environmental impacts of 1 MJ electricity. Y-axis shows relative percentages of scenario 1 and 2 compared to scenario 3.

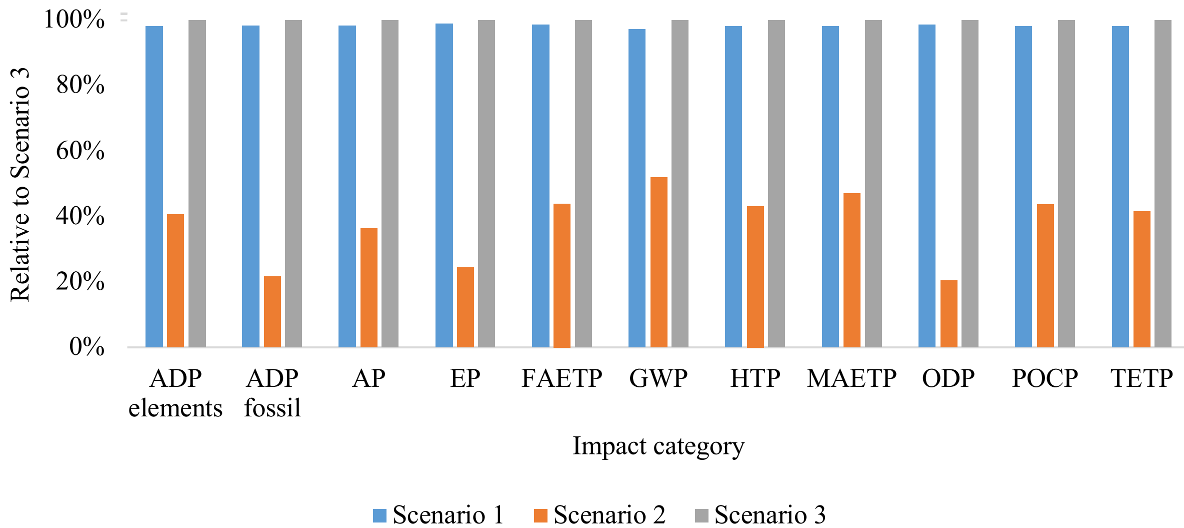

Figure 28. Comparison of environmental impacts of 1 kg fertilizer. Y-axis shows relative percentages of scenario 1 and 2 compared to scenario 3

To understand the cost reduction potential of this study, a sensitivity analysis was performed for several parameters: capital cost, gross operating cost, solid fertilizer selling price, microalgal residual selling price, and lipid content of microalgae. Capital cost and gross operating cost were varied by 10 percent from the baseline scenario (± 10%), and market price range (between \$3 and \$11) was used for solid fertilizer selling price. The fertilizer price used for the baseline scenario (\$5.5/kg N-P-K) is a break-even price for the fertilizer production capital and operating costs. The price of algae residual was varied by 50% from the baseline scenario (± 50%), and the lipid content was varied between 10% and 50%. Since the microalgal lipid was not utilized in scenario 2, the electricity price from scenario 2 does not vary depending on the price of algae residual and the lipid content.

As shown in Figures 29 and 30, an increase in the solid fertilizer selling price decreases both biodiesel and electricity prices the most, and the prices are more affected by capital cost changes than gross operating cost changes. If the fertilizer price is \$11/kg N-P-K, the biodiesel and the electricity will be available at no cost. The highest biodiesel price is \$19.71/liter when the lipid content of the microalgae drops to 10%, and the highest electricity price is \$0.08/kWh resulting from a low solid fertilizer price of \$3/kg and capital costs increased by 10%.

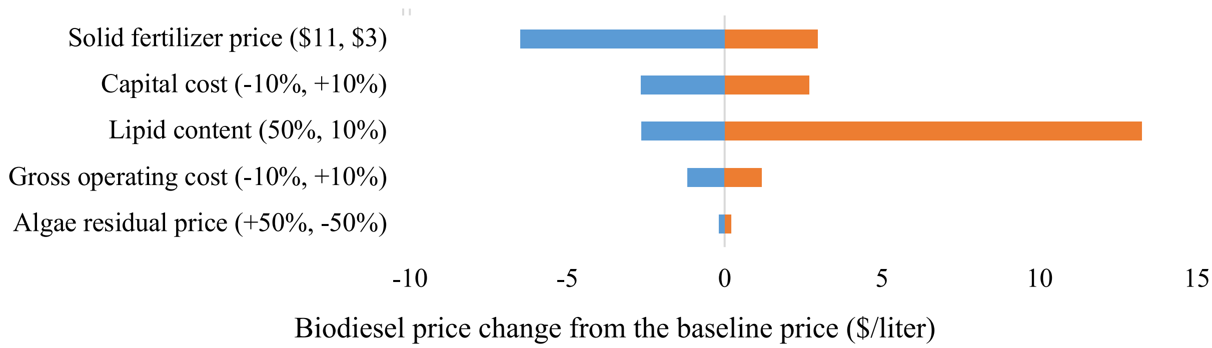

Figure 29. Scenario 1 sensitivity analysis results

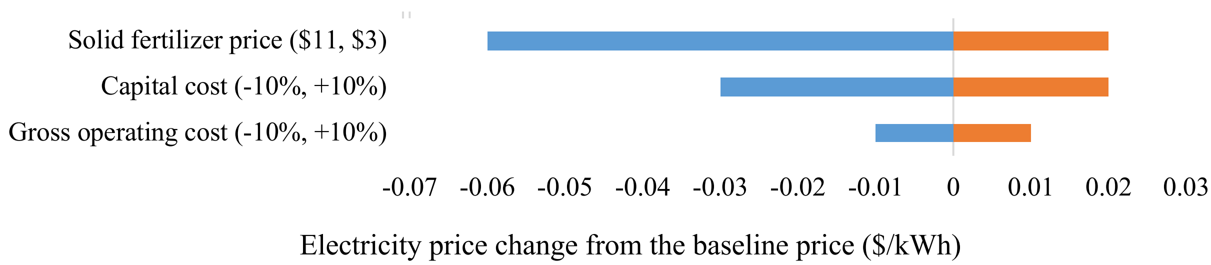

Figure 30. Scenario 2 sensitivity analysis results

4.4 Conclusions

This study examined the environmental and economic performance of a production system designed using the concept of circular economy. The system utilizes waste nutrient flows from AD of food waste coupled with microalgae cultivation to yield electricity and biodiesel, along with co-products including solid fertilizer. The study analyzed three scenarios, each of which produces biodiesel (Scenario 1), electricity (Scenario 2), and both biodiesel and electricity (Scenario 3) from microalgal biomass, and scenario 2 showed better results than scenario 1 and 3 in terms of environmental and economic performance. The total environmental impacts of the system in scenario 2 were lower than the others, and CI and price of the electricity were also competitive with the market (lower CI and lower price).

On the other hand, biodiesel produced from scenario 1 and 3 showed mixed results. In terms of environmental performance, biodiesel showed lower GHG emissions and higher NER than either conventional diesel or results from other studies, but the prices of biodiesel were higher than the market price and results from other studies. The better environmental performance in this study is due in part to the joining of microalgae system to AD system in several ways. First, the system required no chemical fertilizers, because of the waste nutrients in the permeate from the digester. Second, by joining to the AD system and using the electricity from it, primary energy dependency of the system decreased. Lastly, a business-as-usual disposal of wastewater from the digester could be avoided by joining it to the microalgae system. Meanwhile, the prices of biodiesel from scenarios 1 and 3 were higher than the market price and results from other studies, even with consideration of the LCFS credit. In addition, the prices were sensitive to changes of fertilizer price, and the biodiesel can be available at no cost if the fertilizer price increases to \$11/kg N-P-K.

There are several opportunities for future research. The future research may include exploring different pathways to produce microalgal biodiesel, such as HTL or wet solvent extraction. While there are studies exploring different microalgal pathways and technologies, there is a lack of studies which include a joint system of AD and microalgae process. Research on relationship between fertilizer market and energy market may also be investigated as future research. Since environmental and economic results of this study are related to fertilizers, the research on this relationship may contribute to understanding and implementation of the system in practice. For example, in the system of this study, use of fertilizer was avoided simultaneously with production of fertilizer, which may change the fertilizer market, and the price of biodiesel was sensitive to the change of fertilizer price.

CHAPTER 5 Conclusions and Future Research

The primary objective of this dissertation was to explore and analyze the potential contribution of circular economy principles applied to the bioenergy sector in California, and to identify opportunities and challenges. The research aimed to identify preferable bioenergy pathways in California focusing on organic waste streams as well as considering relevant policies. The selected pathways were evaluated using LCA and TEA to assess their environmental and economic impacts.

Three studies were presented in this dissertation. The first (chapter 2) developed a methodology for classifying feedstocks as waste or non-waste for bioenergy applications. This classification is critical for evaluating the life cycle carbon intensity and other environmental impacts of interest in LCA. The second and third studies (chapters 3 and 4) investigated the economic and environmental performance of wastes utilized as feedstocks or inputs to bioenergy systems.

In Chapter 2, two decision trees for feedstock designation were developed based on a series of questions, and four feedstocks were tested by the decision trees. Although the two decision trees had a different step for an economic value threshold (inclusion of 15% threshold and exclusion), all tested feedstocks were classified identically by both decision trees. However, there was a chance that one feedstock, PFAD, may be classified as two different categories depending on the basis of its economic value (e.g., 3% by the total value versus 80% by the per-unit value). Thus, future research may determine the appropriate economic threshold for a particular policy in terms of both percentage (e.g., 15%) and basis (e.g., total value, per-unit value) if an economic threshold is adopted. To determine the appropriate economic threshold for a particular policy, thorough market research for each feedstock is required. Since production quantity and market price of one feedstock varies depending on the time and location and each feedstock has a different range of variation, research is needed to collect data on available feedstocks and market prices. The proper threshold may be determined based on these data. The results from SWOT analysis also showed that there could be a challenge applying these decision trees to other feedstocks that are not widely used for biofuel production. In these cases, initial research for their current uses may be required to supplement the decision trees.

In Chapter 3, a circular economy based on symbiotic relationships between the bioenergy and construction sectors was investigated to find opportunities in California and measure the economic and environmental impacts. Rice residue (hull) and almond residues (shell and hull) were selected based on their harvested acreage and availability. Both residues are generated from the hulling or shelling sites which means they do not require additional collection or transport from the field. For the almond case, two energy feedstock candidates, shells and a mixture of shells and hulls, were analyzed for their electricity generation performance, and the shells-only case showed better performance in terms of generation efficiency and economics. For the rice case, electricity generation from rice hulls was analyzed, and the results showed its economic feasibility.

Between two feedstocks which showed economic feasibility, rice hull was selected as a final feedstock for electricity generation and ash utilization due to better economic performance and more availability. The electricity price required for break even on CAPEX and OPEX for rice hull was \$0.070/kWh, lower than the price required for almond shell (\$0.132/kWh) and the price in California (\$0.15/kWh). The prices of electricity from rice hull and almond shell remained below \$0.15/kWh with the increase of the feedstock cost to \$81/tonne (rice hull) and \$18/tonne (almond shell). In addition, assuming that the electricity from rice hull and almond shell is sold at \$0.15/kWh, the ashes from these feedstocks were available at no cost. The total ash production was higher for rice hull (79,814 tonne/yr) than almond shell (6,666 tonne/yr) due to higher ash content of the feedstock.

After testing showed that RHA can be used as an SCM as long as performance requirements for the application are met, it was also shown that RHA has lower environmental impacts than Portland cement and cement mixture with fly ash. By mixing RHA (15%) with Portland cement (85%), it may lead to 15% reductions in CO_2e emissions, and RHA substitution using California rice alone could mitigate 0.2% of total CO_2e from the cement production sector in the United States. Although rice hull and RHA are feasible for electricity generation and cement substitution, the total production is not enough to supply the cement production sector with needed SCMs, and thus exploring additional feedstocks is required for sustainability across the bioenergy and construction sectors. Future research may investigate additional feedstocks, considering their location and availability.

In Chapter 4, a circular economy around the traditional and emerging bioenergy sectors was investigated based on LCA and TEA. This study examined the environmental and economic performance of a combined energy production system which consists of an anaerobic digester and a microalgae-related processes. In this study, the waste flow from a digester (digestate) is used as a nutrient source for microalgae production, and three scenarios were developed, each of which produces biodiesel, electricity, or both as (a) final energy product(s) from the microalgal biomass. In particular, biodiesel, electricity, fertilizer and microalgal residuals were considered as products in scenario 1; electricity and fertilizer were considered as products in scenario 2; and biodiesel, electricity, and fertilizer were considered as products in scenario 3.

According to the results, the biodiesel from scenario 1 and 3 induced more GHG emissions compared to other studies' results, although it showed a higher NER. In addition, the high price required for biodiesel production, \$6.38/liter (scenario 1), prevents the biodiesel from being competitive in the market, even considering the incentive from the LCFS (\$0.09/liter). On the other hand, electricity production in scenario 2 showed better performance compared to the current electricity grid in environmental and economic terms. The carbon intensity (CI) of the electricity from this study (62.74 gCO_2e/MJ) was lower than the average CI of California grid electricity (81.49 gCO_2e/MJ), and the required price for the electricity (\$0.06/kWh) was also competitive with the market price in California of \$0.15/kWh [102]. Thus, producing electricity in lieu of biodiesel appears to be a better option to maximize the use of waste flows and supply lower-carbon energy. In addition, there are opportunities for future research. While this study included only one pathway for biodiesel production, future research may include different pathways to process the microalgal biomass. Production of biogas to substitute natural gas instead of electricity generation may be another possible scenario to consider since biogas used as transportation fuel and electricity may be regulated under different environmental policies.

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