THESIS

ECONOMIC APPROACHES TO ALLOCATION OF LIFE CYCLE ENVIRONMENTAL BURDENS BETWEEN BEEF PRODUCTION SYSTEMS AND ECOSYSTEM SERVICES

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ABSTRACT

ECONOMIC APPROACHES TO ALLOCATION OF LIFE CYCLE ENVIRONMENTAL BURDENS BETWEEN BEEF PRODUCTION SYSTEMS AND ECOSYSTEM SERVICES

Buck Island Ranch (BIR) is a cow-calf operation in central Florida that manages over 4,200 hectares of semi-native and improved pasture and produces over 2,000 calves each year. The operation has the unique distinction of being both a working ranch and a conservation site with extensive monitoring of everything from species diversity across taxa to nutrient dynamics in pastures and wetlands for the past 30 years. As a result of managing for profitable beef production and conservation, they provide key ecosystem services to their community through conservation-oriented management practices. The primary goal of this project was to perform a cradle to farm gate life cycle assessment (LCA) of environmental impacts and resource consumption in the production of BIR live weight (LW) sold from the ranch. In addition, reproducible methods were developed for multi-functional allocation of environmental impacts between beef and conservation benefits. The LCA was conducted using four approaches to economic allocation of emissions between beef and ecosystem services: (1) allocate all emissions to beef; (2) multi-functional allocation using payments for conservation management practices through the USDA Conservation Stewardship Program (CSP); (3) multi-functional allocation using the "highest and best use" (HBU) price based on real estate evaluation of BIR land; (4) multi-functional allocation using conservation easement prices set by the USDA Agricultural Conservation Easement Program (ACEP). The results of the life cycle impact assessment were as follows: 1 kg LW leaving the farm gate to be sold used 322.22 L of water consumption, 43.97 m^2 annual crop-eq, and 2.01 MJ energy surplus. The associated emissions were 12.27 kg CO_2-eq/kg LW and 36.97 g N-eq/kg LW. When emissions were allocated between beef and ecosystem services, the impacts for beef were reduced 2% using the CSP approach, 39% using the HBU approach, and 42% using the ACEP approach.

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DEDICATION

I would like to dedicate this thesis to my dogs Hank and Joey. You both were there, under my feet or on my lap, for this entire process.

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Chapter 1 - INTRODUCTION

Buck Island Ranch (BIR) is a cow-calf operation in central Florida that manages over 4,000 hectares of semi-native and improved pasture and raises over 2,000 calves each year. The operation has the unique distinction of being both a working ranch and a conservation site with extensive monitoring of everything from species diversity across taxa to nutrient dynamics in pastures and wetlands for the past 30 years. As a result of managing for profitable beef production and conservation, they provide key ecosystem services to their community through conservation oriented management practices. Their focus on conservation sets them apart from conventional practices, and they wanted to know how their management practices affected their environmental impact. Furthermore, they want to know how they compared to the industry standard.

After a thorough review, the inclusion of ecosystem services, provided as a result of conservation oriented management practices, in the estimation of environmental impacts of beef production was identified as a gap in the literature. Moreover, there was a lack of allocation approaches based on economic value that were reproducible for U.S. beef production. In agricultural life cycle assessments (LCA), the economic allocation of emissions between pastoral sheep production in the Mediterranean and ecosystem services was explored by Ripoll-Bosch et al. (2013). In Italy, an allocation approach based on the economic value of honey and pollination services provided by honeybees was created by Arzoumanidis et al. (2019). Weiler et al. (2014) surveyed small-holder dairy farmers in Kenya and proposed a livelihood allocation approach for allocating emissions between milk and the many other functions of cattle, like as dowry or as insurance.

The goal of this assessment is to provide scientific knowledge for evidence-based decision making by BIR personnel, create a baseline for continual monitoring of the ranch system's environmental impacts, and present a reproducible approach to multi-functional allocation of environmental impacts between beef and ecosystem services for production systems with quantifiable ecosystem service benefits.

Beef production's significant environmental impacts to air, water, and soil have been thoroughly studied and reported (Subak, 1999; Capper, 2011; Eshel et al., 2014). However, inclusion of the potential environmental benefits in life cycle assessment is not well studied. As some beef production systems are intentionally managed for the provision of ecosystem services (Teague et al., 2011; Rowntree et al., 2020), I propose that ecosystem services should be considered a co-product of beef production, and that emissions may be allocated using economic allocation methods.

The objectives of this study were to:

1. Quantify the environmental impacts from producing 1 kg live weight (LW) of sold cattle at a commercial cow-calf operation in Florida.

2. Develop three novel economic allocation approaches to allocate emissions between beef and ecosystem services.

3. Compare life cycle impact assessment (LCIA) results and demonstrate the variability of environmental impact footprints from beef production based on choice of economic allocation method.

Chapter 2 - LITERATURE REVIEW

This chapter is an extensive review of peer-reviewed literature covering: (1) a brief overview of the current state of the U.S. beef industry and more specifically beef production in Florida; (2) environmental impacts of producing beef; (3) ecosystem services and the methods used to assess their economic value; (4) life cycle assessment and allocation approaches for ecosystem services provided by multi-functional livestock systems.

2.1 Overview of the U.S. Beef Industry

Agriculture is an integral part of the United States' economy. With 2 million farms that cover over 364 million hectares, farmers and farming make significant contributions to the U.S. economy (USDA-ERS, 2020c). Agriculture and food related jobs represent 11% of total U.S. employment, employing 22 million people (USDA-ERS, 2020a). In 2017, the market value of agricultural products sold was $388 billion, with half of that amount coming from livestock alone (USDA-ERS, 2020a). Within the livestock industry, cattle and calves contribute the most to sales, adding almost $28 billion more than the next largest industry, poultry and eggs (USDA-ERS, 2020b). One driver of this contribution is beef, which is a staple in the American diet evidenced by the 26.5 billion pounds retail weight of beef consumed in 2017 (USDA-ERS, 2021b). Beef production systems can be found in all 50 states, but the greatest producing states are found in the Southern Plains, which include Kansas, Oklahoma, and Texas. Between 2013 and 2017, the Southern Plains produced an annual average of 7.5 million beef and dairy cows and calves, 10.1 million stockers/backgrounders, and 10 million finished cattle (Rotz et al., 2019). From cow-calf to seedstock to stocker operations and feedlots, there are many phases of beef production. While U.S. beef is mainly produced from dedicated beef herds, dual purpose systems are more common in other countries (Herring, 2014).

Florida is among the top producers of cattle and calves in the U.S., ranking 13^{th} with 914,000 animals in 2018 (USDA-NASS, 2019b). In that same year, cattle and calves were the 5^{th} largest agricultural commodity in Florida, representing over $507 million in farm receipts (USDA-NASS, 2019c). Buck Island Ranch—the subject of this study—is located in Highlands County, the second highest cattle producing county in the state (USDA-NASS, 2017). Cattle have been raised in Florida for almost 500 years, and pastures and rangelands cover close to 1.8 million hectares, representing over half of the state's total agricultural land (USDA-NASS, 2019a). Although there is a long history of cattle production in Florida, and in the broader United States, beef production is not without environmental impacts (Subak, 1999; Capper, 2011; Eshel et al., 2014). The environmental impacts from beef production are covered in the following section of this literature review.

2.2 Environmental Impacts of U.S. Beef Production

Globally, agriculture is a driving force behind environmental burdens, and beef cattle are the biggest contributor to the sector's emissions (Pelletier and Tyedmers, 2010; Foley et al., 2011; Gerber et al., 2013). Beef requires more land, irrigated water, and fossil energy, and produces more greenhouse gas emissions than all other livestock categories (de Vries and de Boer, 2010; Gerber et al., 2013; Eshel et al., 2014), partly because of the increased time spent on pasture compared to other livestock species (Stackhouse-Lawson et al., 2012; Gerber et al., 2013). Although beef accounts for 7% of all calories consumed in the U.S. (Eshel et al., 2014), reducing and/or redistributing consumption of livestock products, and transforming the way beef is produced have all been proposed as solutions to the increasing environmental footprints (Pelletier and Tyedmers, 2010; Foley et al., 2011; Eshel et al., 2014).

2.2.1 Global greenhouse gas emissions from beef

In 2005, the global cattle sector produced 61.4 million metric tons of beef and was responsible for emitting 2.83 billion metric tons of CO_2-eq, which is equivalent to 46.2 kg CO_2-eq per kg of carcass weight (CW) (Opio et al., 2013). Carbon footprints (CF) from a cradle-to-farm gate study of beef from dedicated beef and dairy herds were 67.8 kg CO_2-eq/kg CW and 18.4 kg CO_2-eq/kg CW, respectively (Opio et al., 2013). To develop those footprints, a specific model called GLEAM, the Global Livestock Environmental Accounting model, was designed to represent processes that consist of the necessary inputs and outputs of animal production (Opio et al., 2013). Using GLEAM, Opio et al. (2013) reported that enteric CH_4 made up 42.6% of the CF, N_2O from applied and deposited manure amounted to 18.1%, CO_2 from land use change (pasture expansion) made up 14.8%, CO_2 from feed production equaled 10%, N_2O from fertilizer and crop residues equaled 7.8%, and CO_2 and N_2O from manure management made up the final 7.1%. Emission intensities differ between countries, though a direct comparison of results is difficult due to differences in functional unit, system boundary, and production assumptions. For example, in a cradle-to-farm gate study of organic Swedish beef production, Cederberg and Stadig (2003) reported 22.3 kg CO_2-eq per kg of bone free meat. In 2005, Casey and Holden reported 11.26 kg CO_2-eq per kg of live weight in a one-year cradle-to-farm gate study of typical Irish suckler-beef production systems (Casey and Holden, 2006b). In a study of Japanese Black (Wagyu) beef production, including cow-calf and fattening systems, Ogino et al. (2007) reported 36.4 kg CO_2-eq per kg of beef gain.

Differences in emissions may also stem from different methods used to calculate enteric CH_4 emissions. Cederberg and Stadig (2003) calculated emission factors for enteric fermentation based on methane losses of about 7% of the cattle's gross energy intake (GEI), an Intergovernmental Panel on Climate Change (IPCC) standard value. Casey and Holden (2006b) estimated enteric CH_4 emissions using an IPCC methodology and a nutrition software package called RUMNUT, The Ruminant Nutrition Program. RUMNUT calculated the animal's required daily dry matter intake (DMI), metabolizable energy, and gross energy. That information was multiplied by a CH_4 conversion factor set by the IPCC and divided by the percentage of gross dietary energy lost as methane to determine the kg CH_4 per head per year (Casey and Holden, 2006b). Ogino et al. (2007) calculated enteric CH_4 emissions from DMI using a quadratic regression equation formulated from the results from 190 energy balance trials by Shibata et al. (1993). It can be difficult to gain a clear understanding of the carbon footprint of beef because the results are highly dependent on location, production system, choice of enteric CH_4 formula, choice of functional unit, etc.

2.2.2 United States greenhouse gas emissions from beef

Every year, the U.S. Environmental Protection Agency (EPA) publishes the national "Inventory of U.S. Greenhouse Gas Emissions and Sinks". Agriculture is responsible for 9.3% of total greenhouse gas emissions (GHG) in the U.S. when measured using the United Nations Framework Convention on Climate Change (EPA, 2020). Enteric fermentation is the largest anthropogenic CH_4 source in the U.S. and beef production accounts for 72% of those emissions with estimates of the contribution of the cow-calf phase ranging from 73-84% (Rotz et al., 2019; EPA, 2020). Out of all the beef industry segments, the cow-calf phase of production has the most significant environmental impact. This is largely because of the increased enteric fermentation and manure excretion from high-forage diets, longer duration spent on pasture/rangeland, and a higher rate of feed used to produce 1 kg of meat (Phetteplace et al., 2001; Nguyen et al., 2010; Mogensen et al., 2015; Rotz et al., 2015; Asem-Hiablie et al., 2019).

To better understand the environmental impacts of beef production across the U.S., the USDA Agricultural Research Service and The Beef Checkoff conducted the most comprehensive and detailed LCA of beef production to date (Rotz et al., 2019). The country was divided into 7 regions based primarily on climate and management practices. Cattle production data from surveys and site visits was compiled from over 2,200 responding operations across the country. Because of the wide range of cow-calf, stocker, backgrounder, and feedlot operations in each region, representative operations were simulated using a process-level simulation tool called the Integrated Farm Systems Model (IFSM). IFSM integrates farm processes (crop and pasture production, crop harvest, feed storage, grazing, feeding, and manure handling) with many years of weather data to predict long-term performance, economics, and environmental impacts from beef and dairy systems (Rotz et al., 2019; USDA-ARS, 2020). Footprints were calculated by region and were weighted according to each region's proportion of total cattle inventory and carcass weights produced to upscale to national emissions. The cradle-to-farm gate carbon footprints of U.S. beef, including beef from cull dairy cattle, is 21.3 ± 2.3 kg CO_2-eq/kg CW (Rotz et al., 2019).

Previous cradle-to-farm gate studies of beef production throughout the U.S. reasonably agree with the regional carbon footprints estimated in Rotz et al. (2019) study. Northeastern beef production systems included grass-fed beef and confinement dairy beef and resulted in 33.7 kg CO_2-eq/kg hot carcass weight (HCW) and 12.7 kg CO_2-eq/kg HCW, respectively (Tichenor et al., 2017). Upper Midwest beef production GHG emissions ranged from 14.8-19.2 kg CO_2-eq/kg live weight (LW) (23.5-30.5 kg CO_2-eq/kg CW, when assuming 63% dressing weight), with the feedlot production systems having the smallest footprint, followed by background/feedlot systems, and the largest footprint from pasture systems (Pelletier et al., 2010; Campbell, 2016). California beef production systems were reported to have carbon footprints of 10.7-22.6 kg CO_2-eq/kg HCW, with Holstein beef production systems at the low end and Angus beef production systems with stocker phase at the high end (Stackhouse-Lawson et al., 2012). Finally, Southern Plains beef production GHG emissions ranged from 13.8-25.8 kg CO_2-eq/kg CW across 28 simulated production systems, excluding Holstein cattle (Rotz et al., 2015). The differences between studies can be attributed to emission factors methodology, assumptions, system boundaries, modeling strategies, animal breeds, manure management, and lifetime of animals (Peters et al., 2010; Lupo et al., 2013).

2.2.3 Air and soil quality

Nitrogen is a critical component of agriculture. Crops require large amounts of reactive nitrogen to grow, which becomes the protein that contributes to animal growth. Nitrogen is then excreted by the animals and is used to fertilize crops, thus completing the cycle. Environmental footprints of U.S. beef calculated by Rotz et al. (2019) reported 155 ± 12 g N/kg CW in reactive N loss. Large quantities of N are lost in manure and loss through volatilization occurs soon after the manure has been deposited (Rotz, 2004). The excreted manure can quickly transform into ammonia and then diffuses into air (Han et al., 2001). Atmospheric ammonia emissions can happen anywhere manure is exposed to air and can contribute to ecosystem fertilization and acidification (NRC, 2003). While large quantities of N from manure become atmospheric ammonia, there are also losses from nitrogen leaching into the soil and as runoff from rain (Rotz, 2004). In cattle, urine N is primarily in the form of urea. Urea N can also quickly transform into ammonia, with transformation happening within 2 hours of deposition at its maximum rate (Monteny and Erisman, 1998). Leaching of N into the soil can be higher under grazing conditions with concentrations being as high as an equivalent application rate of 300 to 1,000 kg N/ha under urine patches (Rotz, 2004). Nitrogen emissions, in the form of nitrous oxide, also result from nitrification and denitrification because of natural microbial processes, manure storage, or from small amounts of enteric N_2O (Hamilton et al., 2010). Nitrous oxide is the most potent of all greenhouse gas emissions with a global warming potential 298 times higher than CO_2 (IPCC, 2014). While N_2O emissions are not insignificant, the greatest loss of reactive N in animal production is through ammonia volatilization resulting in air pollution (Rotz, 2004).

2.2.4 Water quality

In addition to N deposition from animal waste, synthetic fertilizers containing nitrogen and phosphorous are applied to fields and pastures to promote optimum plant growth. Yet, when fertilizers are applied in rates that are too high for plant utilization, excess nutrients can pollute surface water and groundwater through runoff and leaching. High levels of N and P in bodies of water cause eutrophication, which is the process of nutrient enrichment of a body of water which leads to increases in algal biomass (IPCC, 2019). The increase in algae decreases aquatic biodiversity and can cause hypoxia (NC State Extension, 2017). Phosphorus is essential for plant growth and its primary function is the storage and transfer of energy throughout the plant, along with many other important functions. However, the use of fertilizers has become increasingly prevalent in agriculture production, and erosion via surface runoff adds phosphorus, nitrates (a water-soluble form of N), and sediment to water resources (NRCS, 2003; NC State Extension, 2017). In beef cattle production, nitrogen, in the form of NO_3^-, is primarily lost through leaching to groundwater, while phosphorus is primarily lost through surface runoff (Rotz et al., 2019).

2.2.5 Resource consumption

When considering other footprints, like blue water use, fossil energy use, and land use, the U.S. regions vary based on their environment and climate adapted management practices (Rotz et al., 2019). Fossil energy use process contributions include: feed production, animal feeding, animal housing, manure handling, transport energy, and upstream processes. Feed production, dust control, drinking, and purchased feed contribute to the blue water use footprint, but blue water does not include precipitation. Similar to methane production, the cow-calf phase of production is responsible for over half of the fossil energy use and water use footprints. Some of the biggest blue water consumption footprints were in the Southern Plains, Northwest, and Southwest. That includes states like Texas, Colorado, California, and Wyoming, where the climate is drier and crop production relies heavily on irrigation. However, regions like the Midwest and Southeast have more annual precipitation and thus lower blue water use, but they tend to have higher reactive N loss due to increased runoff and leaching (Rotz et al., 2019). Total fossil energy use was highest for the Southern Plains, but that region also had the largest average annual cattle numbers and carcass weights. The average U.S. beef footprints for resource consumption were: 50.0 ± 4.7 MJ/kg CW in fossil energy use and 2,034 ± 309 L/kg CW in blue water consumption (Rotz et al., 2019). Globally, agriculture takes up almost 5 billion hectares of land, with approximately two-thirds of that amount consisting of meadows and pastures for grazing animals (FAO, 2021). In Europe, suckler production systems required an average of 49 m^2 of land to produce 1 kg of beef, whereas dairy beef systems, where calves are a by-product of milk production, only required 27-31 m^2 per kg beef (de Vries and de Boer, 2010). Eshel et al. (2014) reported that of all livestock species, beef cattle were the least resource efficient and required approximately 15 m^2 of land to produce 1 Mcal of beef.

2.3 Ecosystem Services and Their Economic Valuation

It is well documented that livestock production systems have a sizeable effect on their environment, both positive and negative. For these systems to remain sustainable, they must be managed in a way that maintains ecosystem functions to support current, as well as future, human needs (FAO, 2017). In the U.S., cow-calf operations, like the one operated at BIR, are typically located on land that is not suitable for crop production. Beef cattle graze rangelands, or pastures, with minimal feed supplementation (USDA-ERS, 2020a). Rangelands are intricate and dynamic natural landscapes that have the capacity to provide ecological goods and services (Havstad et al., 2007). Those services are called ecosystem services (ES) and have been defined by the Millennium Ecosystem Assessment as “the benefits people obtain from ecosystems” (MEA, 2005). They are classified into 4 types: (1) provisioning services, (2) regulating services, (3) cultural services, and (4) supporting services. Provisioning services include food (e.g., crops, livestock, aquaculture), fiber (e.g., from timber, cotton, hemp), water, genetic resources, biochemicals, and natural medicines. Regulating services provide regulation of water, climate, erosion, air quality, disease, pests, and natural hazards. Pollination, water purification, and waste treatment are also regulating services. Cultural services include spiritual and religious values, recreation, aesthetic values, and ecotourism. Supporting services serve as the basis for all other services and include nutrient cycling, soil formation, primary productivity, carbon sequestration, and more (MEA, 2005).

The connection between grazing livestock and rangeland ES is complex and can be highly dependent on land and animal management practices. A study conducted on Mediterranean rangelands by Divinsky et al. (2017) hypothesized that biodiversity, with species richness used as a proxy, would decline as grazing intensity increased. Surprisingly, they found that their theory only held true for heavy grazing, which was seasonally grazed at a stocking rate of 2.2 cows per ha. Species richness was greatest in the moderate grazing alternative with a stocking rate of 1.1 cows per ha, followed by the light grazing alternative with a stocking rate of 0.55 cows per ha, and then the control alternative, which was not grazed. However, when they considered other ES like biomass, pollination, and aesthetic value, the control alternative had the highest values, meaning any intensity of grazing had a negative effect on those services (Divinsky et al., 2017). In a review of 742 peer-reviewed publications, Pogue et al. (2018) found similarly conflicting results when examining ES and Canadian beef production. Beef cattle on pasture had a large positive influence on services like cultural heritage, biodiversity, and habitat maintenance; a moderate positive influence on food production, non-food goods production, and air quality regulation; and a slight positive influence on soil quality regulation and recreation/tourism. Yet, beef production on pasture also had a slight negative influence on water supply, water quality regulation, and disease regulation, and a moderate negative influence on climate regulation. Pogue et al. (2018) concluded that appropriate grazing management is vital to the health of grasslands, but many knowledge gaps still exist regarding the overall ecological impacts of beef production, and more studies are needed on the long-term effects of management practices (Pogue et al., 2018). From 2013 to 2015, Gomez-Casanovas et al. (2018) used the eddy covariance (EC) technique and EC towers scattered throughout plots at BIR to study the impact of grazing on carbon fluxes from pastures in subtropical and tropical regions. EC towers measure the exchange rate of gas emissions over ecosystems and agricultural fields, and they are used to estimate heat, water vapor, carbon dioxide, and methane fluxes. The plots were moderately grazed with a stocking rate of 0.4 AU/ha in 2013, 0.9 AU/ha in 2014 and 0.3 AU/ha in 2015 (AU = Animal Unit; 1 AU is equivalent to a 1,000 lb cow with calf) (Gomez-Casanovas et al., 2018). After taking measurements over several wet-dry season cycles, Gomez-Casanovas et al. (2018) found that grazing increased the net storage of C and decreased the Global Warming Potential (GWP) associated with C fluxes of pasture by increasing its net CO_2 sink strength.

In order to manage for ES, there needs to be a reliable way to measure changes in the ecosystem. This is often accomplished with ecological indicators, or metrics (Leh et al., 2013; Divinsky et al., 2017). Even so, measuring changes in ES with metrics is difficult because the metrics must be sensitive enough to detect changes in the ecosystem, but robust enough to filter out the effects of normal agricultural disturbances like crop tilling, cultivation, and rotation (Leh et al., 2013). In a case study on a dairy farm in northwest Arkansas, Leh et al. (2013) used metrics to examine the impact of land use change on ES. Changes were measured on a watershed-level and at the dairy farm field-level using an ES model called InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs). Results from InVEST showed that biodiversity was reduced by 50% in the watershed, but there was a 70% increase in carbon storage. At the field-level, the overall impact on ES loss was reduced compared to the watershed. Leh et al. (2013) suggested that ES provided by agricultural production systems might help offset the damage from urbanization and that there is value in monitoring changes in ES to manage the impacts.

With evidence that livestock production system, under the appropriate management, can contribute positively to ES and metrics to monitor change, the next step is determining methods to assign a monetary value to ES. While it is true that "estimates of economic value thus reflect only the current choice pattern of all human-made, financial and natural resources given a multitude of socio-ecological conditions" (TEEB, 2010), nonetheless, the application of economic valuation tools and methods can create markets for production systems that practice conservation and provide ES. One such method, described by Swartzentruber (2019), is the estimation of the public's willingness to pay (WTP) for a non-market good. There are two approaches to estimating WTP: stated preference, which directly asks people through surveys how much they would pay for a good or service; or revealed preference, which derive the value through people's actions, such as park entrance fees and travel costs. However, the hypothetical nature of the surveys, people's over or underestimate of how much they would pay, and the fact that not all ES are related to an "action" are limitations to the usefulness of WTP (Swartzentruber, 2019). Another common method for valuing ecological goods and services is benefit transfer. A benefit transfer study uses estimates of ecosystem service values from similar locations and applies an average value to the site of interest (Seidl et al., 2018). Unfortunately, the margin of error in these studies can be quite large when the benefit transfer is applied incorrectly by generalizing a few characteristics of the initial study site and assuming a 1-for-1 match at the site in question (Plummer, 2009). A third method is the use of government programs that pay producers to preserve ecologically significant lands or to adopt management practices that promote conservation and biodiversity. In an analysis of alternative compensation methods for private land conservation, Seidl et al. (2018) found that when agri-environmental programs incentivize conserving certain land types, it can change the land type mix in the region, thereby increasing the ecosystem benefits. While none of these methods are without limitations, three different approaches, derived from the third method above, were analyzed in this study for their suitability as economic allocation strategies in a life cycle assessment of beef production systems.

2.4 Life Cycle Assessment and Allocation Approaches

Consumers and companies are increasingly concerned about the environmental footprint and sustainability of products. Risk assessment, cost-benefit analysis, ecosystem service valuation, and integrated assessment models are just a few of the tools available in the sustainability assessment toolbox. Another tool, life cycle assessment (LCA), is a methodology for systemically assessing environmental impacts associated with a good, service, or process. Impacts are evaluated at each stage of the life cycle, from raw material extraction through processing, manufacturing, distribution, use, and disposal or recycling. This is referred to as "cradle to grave" or "cradle to cradle" analysis (McDonough and Braungart, 2002). Most of the environmental impacts of agricultural outputs occur in the production phase, therefore, LCAs of agricultural systems often define their system boundary as "cradle to farm gate", meaning only the impacts from raw material extraction until the time the product leaves the farm are considered. In addition to assessing life cycle environmental impacts of products, LCAs are also used to identify "hot spots" or significant contributors to the environmental impact of a product. It is reported that the first LCA was designed for industrial purposes in 1969 to answer a question posed by Coca-Cola: what are the trade-offs between packaging their product in glass or plastic bottles (Hunt and Franklin, 1996; Matthews et al., 2014)? Nearly 30 years later, the process was adapted for application to livestock production systems (Cederberg and Darelius, 2000).

2.4.1 Life Cycle Assessment

The International Organization for Standardization (ISO) developed a set of standards that govern how an LCA should be conducted. ISO 14040:2006 and ISO 14044:2006 establish the four main phases of LCA (ILCD, 2010):

1. Goal and Scope Definition
2. Life Cycle Inventory Analysis (LCI)
3. Life Cycle Impact Assessment (LCIA)
4. Interpretation

In the Goal and Scope Definition phase, the product (or service) of interest and the motivation for the study of that product is stated. It is also in this phase that the system boundary, functional unit (FU), and allocation method, if applicable, is defined. A FU is a reference unit to which all inputs and outputs are related (e.g., 1 kg of live weight or 1 filled plastic bottle). In the LCI phase, data collection begins, and an inventory of inputs and outputs is created. Extensive data collection is necessary to fully develop flows and processes. The amassed data is then used to estimate relevant environmental impact indicators in the LCIA phase.

There are two levels of impact indicators: midpoint and endpoint. Midpoint level indicators assess emissions that contribute to a single environmental concern. For example, carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are combined to assess their impact on global warming. Endpoint level indicators focus on much broader concerns like damage done to human health or to ecosystems. Even though endpoint indicators attempt to assign a "single score" for broad areas of concern, midpoint indicators are more commonly reported because of the increased uncertainty that is inherent in endpoint level indicators. The final phase is interpretation, where the results from the LCIA are evaluated in the context of the scope of the study and conclusions are drawn. An LCA can be extraordinarily complex, and the process is often iterative with adjustments made at any phase (ILCD, 2010).

There are two types of LCA: attributional or consequential. Attributional LCAs evaluate supply chains as they currently exist, modeling the inputs and outputs as static. Consequential LCAs attempt to model how environmental flows will change in the background system as a result of changes made in the foreground system; or viewed another way, consequential LCAs try to describe how environmental flows change in response to possible decisions (Finnveden et al., 2009; ILCD, 2010). Consequential LCAs require modeling of both the foreground and background systems, making the data collection phase even more extensive and potentially increasing the uncertainty of the results. Because of their already complex nature and limited primary data availability, most LCAs of livestock production systems are attributional.

The impact categories most commonly assessed in livestock production systems are GWP, acidification potential, eutrophication potential, energy use, land use, and water use (de Vries and de Boer, 2010). Other impacts that have been studied include pesticide use, ecological footprints, biodiversity, landscape image, and animal welfare, among others (Cederberg and Darelius, 2000; Haas et al., 2001; Pelletier et al., 2010). Global Warming Potential, as defined by the IPCC, is "calculated as the ratio of radioactive forcing of one kilogramme greenhouse gas emitted to the atmosphere to that from one kilogramme CO_2 over a period of time (e.g., 100 years)" (IPCC, 2019). The accepted unit of measurement for GWP is kilograms of CO_2-equivalents. The IPCC has also provided GWP values relative to CO_2 for a 100 year time horizon: CO_2 = 1, CH_4 = 28, and N_20 = 265 (IPCC, 2013). Eutrophication potential can be reported in kg N-equivalents, kg PO_4-equivalents, or kg P-equivalents. Acidification potential is the estimate of emissions that increase acidity of water and soils. In agricultural practices, sources of acidification include N fertilizers and plant removal due to harvesting or grazing. Acidification potential is measured in kg SO_2-equivalents. Livestock production systems are also significant consumers of natural resources such as land, water, and fossil fuels. Consequently, it is important that the impacts to those resources are evaluated and reported.

The ability to identify trade-offs between impact categories and evaluate entire systems makes LCA a powerful tool. Life cycle assessment has been used to assess beef production systems at local, regional, national, and global scales (Opio et al., 2013; Rotz et al., 2013, 2015; Asem-Hiablie et al., 2019) and to explore different farming intensities (intensive and extensive) (Willers et al., 2017), production systems (conventional, organic, and grass-fed) (Casey and Holden, 2006a,b; Lupo et al., 2013), phases of production (cow-calf, backgrounder/stocker, and feedlot) (Ogino et al., 2004, 2007; Beauchemin et al., 2010; Pelletier et al., 2010), and diet types (concentrate based and roughage based) (Tichenor et al., 2017).Some of these studies have explored modeling a production system that is multi-functional and used various methods to allocate emissions between multiple products, while others limited the system boundary to focus on a single output.

2.4.2 Multi-functional Livestock Systems and Allocation in LCA

Application of LCA to agricultural systems is not without its challenges; in particular, rarely does a livestock production system produce a single product. For example, dairy cows are not only used for their milk producing ability, but surplus calves and culled dairy cows also become part of the beef supply chain. Beef cattle produce meat and by-products like leather, tallow, blood, organs, and bones. The multi-functionality of these systems affects how environmental impacts are allocated in LCA. The ISO has a step-wise procedure to determine how to proceed with allocation in multi-functional systems. ISO 14044:2006 states:

> **Step 1:** Allocation should be avoided, wherever possible, by either dividing the process to be allocated into sub-processes or by expanding the product system to include additional functions related to the co-products, also known as system expansion.
>
> **Step 2:** If allocation cannot be avoided, the environmental impacts should be partitioned to reflect the physical relationship between the products (e.g., mass).
>
> **Step 3:** If a physical relationship cannot be established, other methods that reflect a relationship between the products should be considered (e.g., economic-based).

The allocation method chosen can cause the outcome of the assessment to vary significantly, and there are drawbacks to each method. System expansion is not clearly defined by the ISO and as a consequence there are conflicting interpretations of its meaning. As explained by Pelletier et al. (2015), a literal interpretation is that the system boundary of the study is expanded and impacts are analyzed and reported at the individual co-product level, but some LCA practitioners have interpreted system expansion as being comparable with substitution. An example of system expansion + substitution from an attributional modeling approach is determining the impacts from a dairy farm that also produces beef from surplus calves and cull cows. System expansion requires the LCA practitioner to model the dairy farm (the "foreground" system) and a beef herd proxy (the "background" system), and then subtract off the emissions of the beef herd from the dairy farm to quantify the emissions solely from dairy production. However, having enough data to fully model the foreground and the background systems is not always possible, or feasible, and the use of proxies may be a poor representation of the system and its environmental burdens (Pelletier et al., 2015).

Allocation Methods

A physical relationship between the products can be represented in many ways: mass-based, energy-based, or protein-based, to name a few. Each physical allocation method can produce varying results as demonstrated in Table 2.1 (Cederberg and Stadig, 2003; Thomassen et al., 2008; Rice et al., 2017). Economic allocation is based on the market value of each product, but value

Table 2.1: Comparison of Global Warming Potential (GWP) results based on allocation methods between different studies. Includes the type of production system and functional unit (FU).

Study	Country[a]	System	FU[b]	Allocation Method[c]	GWP (kg CO_2-eq)
Cederberg and Stadig, 2003	SE	Organic dairy & suckler	kg ECM	None	1.05
				Economic	0.97
				Biological	0.89
				System Expansion	0.63
Thomassen et al., 2008	NL	Average conventional dairy	kg FPCM	Economic	1.61
				Mass	1.56
				System Expansion	0.9
Rice et al., 2017	IE	Average grass-based dairy	kg FPCM	Economic	1.06
				Mass (LW)	1.21
				Mass (CW)	1.22
				Protein Content	1.15
				Energy Content	1.11
				Physical Causality	1.04
				Emergy	1.15

[a] NL = The Netherlands; SE = Sweden; IE = Ireland.
[b] ECM = Energy corrected milk; FPCM = Fat and protein corrected milk
[c] LW = live weight; CW = Carcass weight; Emergy (sel) = Energy (J) * Transformity (seJ/J)

can fluctuate from year to year or even month to month. Even though there are disadvantages associated with economic allocation, Mackenzie et al. (2017) argues that the justifications for basing allocation factors on an underlying physical relationship between co-products in complex agricultural systems are “essentially arbitrary”, because evidence of a causal relationship between the inputs and outputs is often lacking and the criteria used to decide if a mass flow is classified as a co-product or not is based on its economic value (Mackenzie et al., 2017). Cederberg and Stadig (2003) demonstrated how allocation factors differ based on the chosen allocation method when partitioning burdens between dairy production and beef production from culled dairy cows and surplus calves. In that study, with no allocation, 100% of the greenhouse gas emissions were attributed to dairy. When economic, ‘biological’, or system expansion methods were applied, 92%, 85%, and 63% of greenhouse gas emissions (GHG) were attributed to dairy, respectively. Although system expansion is the ISO-preferred method, biophysical or economic methods are more commonly used for livestock production systems based on the authors’ review of literature as shown in Table 2.2 below.

Allocation between provisioning, supporting, regulating, and cultural ecosystem services

Allocation of emissions between beef and dairy products is the most common among studies. These systems are highly intertwined and justifying an allocation method and allocation factors to show the relationship between these two systems can be fairly straightforward. In LCA, it becomes more problematic to quantify the relationship between a product and a co-product that is not tangible, like ecosystem services. There are very few LCA studies that have included ecosystem services as the “co-product” of agricultural production systems and have argued that emissions should be allocated between the products and services (Ripoll-Bosch et al., 2013; Weiler et al., 2014; Kiefer et al., 2015; Garg et al., 2016; Arzoumanidis et al., 2019).

To the knowledge of the author, Ripoll-Bosch et al. (2013) were the first to explore the allocation of environmental impacts between livestock production systems and ecosystem services in a peer-reviewed LCA. The study evaluated three sheep production systems in Spain for their respective carbon footprints (CF). The first was an extensive, pasture-based system where the sheep could graze freely, and feed was only supplemented when necessary (Pasture-Based). The second was a mixed sheep-cereal system where the sheep grazed daily and were supplemented with grains

Table 2.2: Studies included: type of production system and method of allocation (physical, economic, system expansion, or avoided)

Study	System	Physical	Economic	System Expansion	None or Avoided
Asem-Hiablie et al. (2019) (USA)	Suckler	X	X		
Beauchemin et al. (2010) (Canada)	Suckler				X
Bedoin and Kristensen (2013) (Denmark)	Suckler				X
Capper (2012) (USA)	Suckler	X			
Casey and Holden (2005) (Ireland)	Dairy	X	X		
Casey and Holden (2006a) (Ireland)	Suckler	X			
Casey and Holden (2006b) (Ireland)	Suckler				X
Cederberg and Mattsson (1999) (Sweden)	Dairy	X			
Cederberg and Darelius (2000) (Sweden)	Dairy & Suckler	X	X	X	
Dick et al. (2015) (Brazil)	Suckler				X
Haas et al. (2001) (Germany)	Dairy				X
Iepema and Pijnenburg (2001) (The Netherlands)	Dairy		X		
Lupo et al. (2013) (USA)	Suckler				X
Mogensen et al. (2015) (Denmark, Sweden)	Suckler				X
Nguyen et al. (2010) (European Union)	Dairy & Suckler				X
Ogino et al. (2004) (Japan)	Suckler				X
Ogino et al. (2007) (Japan)	Suckler				X
Opio et al. (2013) (Global)	Dairy & Suckler	X	X		
Pelletier et al. (2010) (USA)	Suckler				X
Peters et al. (2010) (Australia)	Suckler	X	X		
Phetteplace et al. (2001) (USA)	Dairy & Suckler				X
Rotz et al. (2010) (USA)	Dairy & Suckler		X		
Rotz et al. (2013) (USA)	Suckler				X
Rotz et al. (2015) (USA)	Dairy & Suckler	X			
Rotz et al. (2019)) (USA)	Dairy & Suckler	X			
Stackhouse-Lawson et al. (2012) (USA)	Dairy & Suckler		X		
Stewart et al. (2014) (Canada)	Suckler				X
Subak (1999) (USA, Africa)	Suckler				X
Thoma et al. (2013) (USA)	Dairy & Suckler	X	X		
Thomassen et al. (2008) (The Netherlands)	Dairy	X	X	X	
Tichenor et al. (2017) (USA)	Dairy & Suckler	X	X		
Willers et al. (2017) (Brazil)	Suckler	X			

(Mixed). The third was an industrial, no grazing system where the animals were confined indoors and fed a total mixed ration (Zero-Grazing). The authors go on to explain that in Spain most sheep farming systems provide not only meat, but also cultural ecosystem services such as biodiversity and landscape conservation when animals are allowed to graze. Ripoll-Bosch et al. (2013) argued that the ecosystem services provided should be considered “co-products” of sheep farming and thus should be allocated a portion of the GHG emissions. Economic values for sheep production were obtained by multiplying the number of sheep by the average price of lamb at farm gate and by the agri-environmental payments subsidized to Spanish farmers by the Common Agricultural Policy (CAP) of the European Union. CAP agri-environmental payments compensate for extensification of livestock production, conservation of habitats with endangered flora and fauna, and producing in less favorable areas with harsh conditions, among other things (EEA, 2004). As these payments represent the costs associated with conservation, or the willingness of society to pay for such services, Ripoll-Bosch et al. (2013) considered them as a proxy for the value of ecosystem services. When no allocation for ecosystem services was applied, Zero-Grazing CF was lowest at 19.5 kg CO_2-eq/kg live weight (LW), followed by Mixed at 24.0 kg CO_2-eq/kg LW and Pasture-Based at 25.9 kg CO_2-eq/kg LW. However, when ecosystem services were accounted for, the Pasture-Based CF was lowest at 13.9 kg CO_2-eq/kg LW, with CFs of 17.7 kg CO_2-eq/kg LW and 19.5 kg CO_2-eq/kg LW for Mixed and Zero-Grazing, respectively. The results show that it is important to consider integrating ecosystem services into the standard framework of agricultural LCA (Ripoll-Bosch et al., 2013).

Similarly, Weiler et al. (2014) proposed that smallholder livestock systems in Kenya raised dairy cows for more than just milk and meat and therefore emissions should be allocated among all pertinent co-products. For Kenyan farmers, dairy cows are also used as dowry, as a sign of wealth, as insurance, or as a source of emergency cash (Weiler et al., 2014). These functions are co-products of raising livestock and as such should be allocated environmental impacts according to the guidelines set by the ISO for LCA. Weiler et al. (2014) explored three methods for allocating GHG emissions: (1) Economic allocation between milk and meat, or ‘food allocation’; (2)

Economic allocation between all products that have a market value, such as milk, meat, manure, and cattle as a means of finance, or 'economic function allocation'; and (3) allocation based on the farmers' perspectives and their assessed value of the roles cattle play in their livelihood, or 'livelihood allocation'. The average CF of milk using 'food allocation' was 2.0 kg CO_2-eq/kg milk. When 'economic function allocation' was applied, the average CF of milk was 1.6 kg CO_2-eq/kg milk. Finally, when 'livelihood allocation' was applied, the average CF of milk was 1.1 kg CO_2-eq/kg milk. The authors acknowledge that CF is only one aspect of a sustainability assessment and that other environmental impacts should be studied in future research, but their results demonstrate how disregarding multi-functionality of cattle can result in a higher CF for milk (Weiler et al., 2014). A similar study was conducted by Garg et al. (2016) for smallholder dairy systems in Western India. This study also quantified the CF of dairy when co-products, such as meat, manure, cattle as insurance, and finance, were economically allocated.

More recently, an LCA of honey considered pollination a "co-product" of beekeeping and honey production in Italy (Arzoumanidis et al., 2019). To the knowledge of the author, this study is also the first cradle to grave agricultural LCA to use multi-functional allocation and consider impacts beyond the CF. The authors proposed that assigning an economic value to pollination was a potential basis for addressing multi-functionality in LCAs of honey production. There are many methods of assigning an economic value to pollination, from evaluating consumers' willingness to pay (WTP) for environmental improvements to calculating a dependence ratio based on the total value of the crop and the dependence of the crop on pollination. In the end, the dependence ratio method was chosen because there was insufficient data surrounding consumers' WTP and orange tree pollination services in Italy. Results from the study showed a decrease in all environmental impacts when pollination was considered a co-product and economic allocation was applied. In this "cradle to grave" analysis, the production phase was the largest contributor to most of the impact categories, due to the use of glass jars for packaging and the electricity consumption of the refrigerated rooms for storage. The transportation of the honey around Italy and abroad resulted in the distribution phase being the next largest contributor to the impact categories. For example, honey production contributed to 88% of the impact to freshwater eutrophication, and the distribution phase contributed to 65% of the impact to natural land transformation (Arzoumanidis et al., 2019).

These studies laid the groundwork for the inclusion of ecosystem services as co-products in multi-functional livestock production systems and for the allocation methods proposed in this study. The multi-functional allocation methods proposed in this study are described in the materials and methods chapter.

Chapter 3 - MATERIALS AND METHODS

3.1 Goal and Scope Definition

3.1.1 Goal

The primary goal of this study was to quantify the environmental impacts of 1 kg live weight from beef calves produced by a conservation-focused commercial cow-calf operation in Florida. A secondary goal was to develop a reproducible method for multi-functional allocation of environmental impacts between beef and conservation benefits. The environmental burdens were economically allocated using an average market value of weaned calves and four distinct methods determining the monetary value of ecosystem services. The allocation methods consisted of: 1) no allocation, 2) highest & best use real estate evaluation, 3) Conservation Easement payments, and 4) Conservation Stewardship Program payments. The intended audience includes the managers at Buck Island Ranch (BIR) and Archbold Biological Station and other agricultural LCA practitioners. The data and results will be published in a peer-reviewed academic journal.

3.1.2 Product System

BIR is a commercial cattle ranch and ecological field station in Lake Placid, FL with over 3,000 head of cattle. The ranch covers 4,249 hectares and is located in Highlands County, where the predominant agricultural activities are growing citrus trees and rearing cattle (USDA-NASS, 2017). The climate is subtropical with 2 seasons: hot and wet or cool and dry. They receive an annual average rainfall of 1,300 mm on fine, sandy soils and the mean annual low and high temperatures are 61 and 83 °F, respectively (Florida State University - FCC, 2020). Many endangered or threatened animal species reside in or pass through the property. Crested Caracaras, Burrowing Owls, and even Florida Panthers have been spotted on game cameras located throughout the ranch. The plentiful pastures and secluded wetland hammocks also attract large populations of nonnative species such as feral hogs and elk. Hunting leases provide additional revenue, as well as the selling of bahiagrass sod in certain years.

Most of the property is split between what is identified at BIR as "improved" pastures and "semi-native" pastures. There are 1,925 hectares of improved pasture that have been seeded with bahiagrass (*Paspalum notatum*) and 2,281 hectares of semi-native pastures that are a mix of bahiagrass and native grasses, such as carpetgrass, bluestem, and maidencane (Gomez-Casanovas et al., 2018). The cattle are Brahman-Angus crosses that are rotationally grazed between the two pasture types. Improved pastures have an average stocking density of approximately 4 head per hectare, while semi-native pastures average about 1.5 head per hectare due to the limited nutrient availability of native grasses.

In order to create a model of BIR within an LCA framework, the ranch was broken down into individual processes that represent the important phases of the operation. The two pasture types have different management styles and thus were separated into a "Grass Pastures, Improved" process and a "Grass Pastures, Semi-native" process. The off-farm feed consumed by the cattle was represented by a process called "Cattle Feed". Finally, a process called "Calf Production" was created to represent the production of livestock and their emissions, transportation and fuel use, ranch-wide energy use, and the outputs from the pasture and feed processes. The specific inputs and outputs from each process are summarized in subsection 3.3.7.

3.1.3 System Boundary

The system boundary for this analysis was "cradle to farm gate" at BIR. The environmental impacts associated with average inputs and outputs required for the cultivation of grass pastures, production of off-farm resources (e.g., energy, feed, mineral supplements, etc.), and raising of beef cattle for one year are included in this assessment, as shown in Figure 3.1. Impacts from feedlots, slaughter facilities, processing, distribution, and disposal are out of scope and were not considered. This assessment also does not take into consideration the impacts that are associated with the

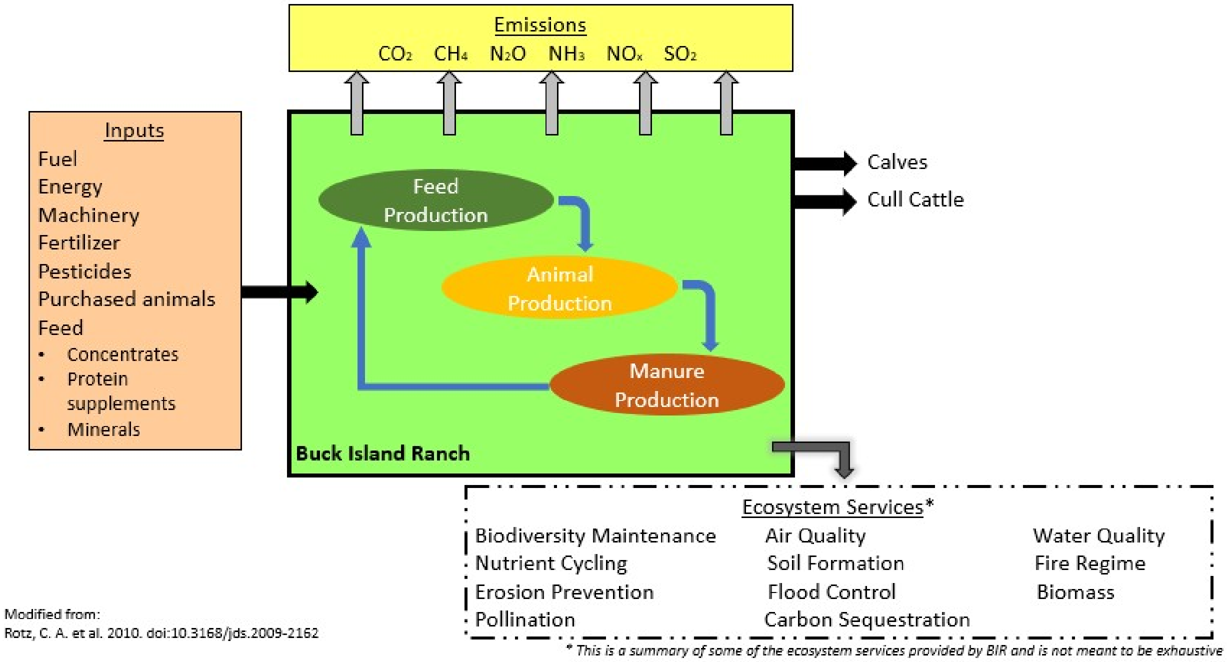

Figure 3.1: System boundary diagram of Buck Island Ranch.

manufacturing of machinery or the construction of on-site buildings because their contributions to the environmental impacts are minimal (Beauchemin et al., 2010; Pelletier et al., 2010).

3.1.4 Functional Unit

The functional unit was defined as one kilogram of live weight (LW) sold. All inputs and outputs are representative of one year of cattle production at BIR. All outputs and emissions have been normalized to one kg LW leaving the farm gate to be sold.

3.2 Modeling

3.2.1 Software

openLCA version 1.10.3. openLCA was used to complete the LCA. openLCA is an open source software developed by GreenDelta and is used for sustainability and LCA. openLCA provides detailed insights into calculations and analysis results as well as visualizations of results (GreenDelta, 2020).

3.2.2 Limitations

Life cycle assessments are robust tools for assessing environmental impacts, but they do not guarantee sustainability, and they do not replace other types of analyses like an Environmental Impact Assessment (EIA) or cost-benefit analysis (Finnveden et al., 2009). Life cycle assessments were not initially developed to account for agricultural sites, and it can be challenging to fully represent such a complex system.

As with most LCA studies, there are limitations to this study that may have influenced the results. For example, it was necessary to make assumptions about animal feed and water consumption, growth rate, and methane production. There were also assumptions made about the productivity, composition, and nutritional value of the improved and semi-native pastures. The data reflects three-year average values, but there can be significant differences from day to day, month to month, or year to year that are not captured by yearly averages. This study was also limited by the lack of specific data surrounding economic values of ecosystem services. The values are highly dependent on location and the public's willingness to "pay" for such services (Bohlen et al., 2009). Uncertainty and sensitivity analyses were carried out to quantify the effects of the decisions made on the final results.

3.3 Life Cycle Inventory

3.3.1 Assumptions

Cow-calf operations are complex systems with multiple processes occurring on different spatial and temporal scales and obtaining the quantity and quality of data required to complete an LCA can be challenging. As such, it was necessary to make assumptions about ranch inputs, outputs, and emissions in order to model the production system.

While significant primary ecological data were provided by BIR, animal emissions were not available as primary data and had to be estimated using the Integrated Farm System Model (IFSM) version 4.6 (USDA-ARS, 2020). IFSM is a process-level simulation tool that can assess the performance, environmental impacts, and economics of beef, dairy, and crop production systems (Rotz et al., 2013, 2018). IFSM simulates animal and crop production, feed storage and use, machinery operation, and the flow of nutrients to and from air, soil, and water over many years of weather. Briefly, IFSM predicts animal requirements, ration nutrient supply, and animal performance for beef or dairy herds. Animal responses to the nutritive value of feeds and supplements is simulated by animal group (e.g., heifers, mature cows, etc.) for beef or dairy herds. Protein and energy requirements are derived from the National Research Council's (NRC) nutrient requirements for cattle and were based on the characteristics of an average animal in each animal group. Cattle dry matter and water intake were estimated based on age, weight, sex, and weather (National Research Council, 2000).

Nutrient flows were simulated to estimate soil accumulation or loss to the environment, including emissions from manure, ammonia, enteric CH_4, CO_2, denitrification, erosion, leaching, phosphorus, and volatile organic compounds (Rotz et al., 2018). The model also accounts for any upstream emissions relating to inputs from off-farm resources and losses from animals, crops, excess feed, or manure leaving the farm. The model relies on three parameter files: weather, farm, and machine. Weather files contain 25 years of recent historical weather for cities throughout the U.S. Farm files contain crop area, pasture and grazing area, soil type and characteristics, equipment used, numbers of animals by group, harvest, tillage, manure handling strategies, and prices for various farm inputs and outputs. Machine files contain data regarding machine use for each type of farm. For this study, the parameter files used were a weather file for Orlando, FL, a typical Florida cow-calf operation farm file developed for the U.S. Beef Sustainability Project (Rotz et al., 2019), and a ranch machinery file. Pasture area was adjusted to match the total area of BIR. Pasture quality was adjusted to match the seasonal nutritional value of bahiagrass. Grazing management was adjusted to represent a year-round grazing period and a 68% pasture utilization efficiency. Breed and herd information was adjusted to match the cattle production table provided by BIR. Breed characteristic values were adjusted using the breed maintenance requirement multipliers, birth weights, and peak milk production table from Fox et al. (2004) for Brangus. Finally, the ration constituents and feed characteristics were adjusted to represent cottonseed meal (CSM) as a proxy for range cubes, and corn grain (CG) as a proxy for rations. All adjustments made to the "FL cow-calf" farm parameter file can be found in Table A.13 through Table A.17 in the appendix. No adjustments were made to the weather parameter file or machinery parameter file. Once all adjustments were made to the farm file, the model was run for a 25-year analysis of BIR. Annual emissions of important gaseous compounds predicted by IFSM were used to supplement primary data in this study were farmland N_2O (to estimate N_2O emissions from semi-native pasture at BIR), animal CH_4 (enteric and manure), animal N_2O, ammonia, hydrogen sulfide, and volatile organic compounds (VOC).

Greenhouse gas emissions from land can vary depending on the weather, temperature, or location within the property (Chamberlain et al., 2015). The CH_4 and CO_2 pasture emissions in this study were measured over a 2.5 year period at BIR (Chamberlain et al., 2017). Their results showed that improved pastures within BIR emit considerable amounts of CH_4 (up to 23.5 ± 2.1 g CH_4-C per m^2 per yr), but also that they are net CO_2 sinks, sequestering up to 163 ± 54 g CO_2-C per m^2 per yr. At the time of this analysis, measured data for pasture N_2O emissions at BIR were not available, so a "high" and a "low" N_2O value were assumed based on data from the literature and IFSM output in order to evaluate their effects on the GWP results reported in this study. The "high" value was 5 kg N_2O per ha per year and was based on a three-year study of N_2O emissions from planted and fertilized bahiagrass pastures at the University of Florida Range Cattle Research and Education Center (UF/IFAS RCREC) in Ona, FL (Lu et al., 2020). The study conditions were similar to improved pastures at BIR, produced comparable values to those measured at BIR, and represent an upper limit of potential N_2O emissions from semi-native pastures at BIR. However, as semi-native pastures at BIR are a mix of native grasses and bahiagrass, and are not fertilized, it is likely that N_2O emissions occur at a lower rate than those observed in the Lu et al. (2020) study. While N_2O emissions in the semi-native pastures would likely occur at a lower rate than in the improved pastures, N_2O emissions from agricultural lands can fluctuate from year-to-year and are difficult to predict accurately (EPA, 2020). Due to the lack of measured data, IFSM was used to predict an N_2O value for semi-native pastures. To represent semi-native pasture management at BIR, the farm parameter file was modified to exclude all fertilizer use on pastures. The simulated farmland N_2O emissions equaled 0.37 kg/ha/yr and was chosen to represent the "low" N_2O emissions value.

In addition to evaluating GWP using scenarios with high and low N_2O estimates, the effect of carbon sequestration on GWP was also explored. Carbon sequestration has generally been excluded from LCAs because of a lack of data and the assumption that soils reach long-term equilibrium soil carbon (Rotz et al., 2019; Rowntree et al., 2020). Although measured BIR data Chamberlain et al. (2017) showed that pastures at BIR are a net CO_2 sink, GWP for BIR was calculated both with and without the inclusion of carbon sequestration to facilitate comparison with the literature. Sensitivity and uncertainty analyses were performed to assess the influence of uncertainty in these values on the final results.

3.3.2 Data Sources and Collection

The life cycle inventory (LCI) phase of an LCA relies on collecting data from many different sources. Primary data were collected from BIR researchers and staff through in-person interviews, video conferencing, and email communications. A spreadsheet was used to share data pertinent to creating the flows, products, and processes necessary for this study. Values for all flows and processes were averaged from years 2014 through 2017 to represent a typical year of production at BIR. When primary data were unavailable, values were estimated using IFSM or collected from literature. In the model, primary data, IFSM output, and values from literature were used to create product flows and processes that were representative of operations at BIR. The ecoinvent 3.7 (cut-off system model) (Center, 2021) database was used when a suitable proxy for flows or processes was available. Electricity data from the National Renewable Energy Laboratory, provided through a joint effort by the USDA, NIST, NREL, NETL, EPA, and others (USDA et al., 2021), were imported from the LCA Commons to supplement missing energy data. Data relating to the annual market price per kg of live weight was collected from the USDA Economic Research Service (USDA ERS) database and the "2018 Florida Livestock, Dairy, and Poultry Summary" published by the Florida Department of Agricultural and Consumer Services and the

USDA National Agricultural Statistics Service (NASS) (Florida Department of Agriculture and Consumer Services, 2019; USDA-ERS, 2021a). The USDA NRCS website provided information and estimated values for the Conservation Stewardship Program payments and the Agricultural Conservation Easement Program payments (USDA-NRCS, 2021a,b).

3.3.3 Cut-off Criteria

A cut-off threshold of 1 x 10^{-4} was set for each product system. Processes that contributed less than 0.01% were excluded from the analysis. This threshold was chosen because it provides enough background information to each process to make the emission results meaningful, but still permits for the completion of 1,000 iterations of a Monte Carlo Simulation for each product system.

3.3.4 Data Quality and Uncertainty Analysis

An uncertainty analysis was performed for this study because in an LCA it is impossible to produce an exact value for every input and output. Uncertainty affects the reliability of LCA results and research conclusions. In LCA, sources of uncertainty can include: data quality, incompleteness in the sample, random error, appropriateness of the model, or uncertainty associated with the impact assessment methodology (Weidema et al., 2013). The uncertainty associated with the results of this study was evaluated using Monte Carlo Simulations (MCS), a statistical method that predicts the probability of different outcomes using repeated assessments of random input values selected from a specified range. In this study, 1,000 runs were simulated. The MCS provided a distribution of the results and calculated an average probabilistic mean value, median, standard deviation, min and max values, and 95% confidence intervals. Most often a lognormal distribution is used in the ecoinvent database because most parameters in reality are positive and a lognormal distribution fits a skewed distribution with low mean values and large variance (Weidema et al., 2013).

The ecoinvent database quantifies two types of uncertainty for each exchange (Weidema et al., 2013). Base uncertainty reflects the uncertainty associated with a lack of precise knowledge about the data or the use of averaged values. Additional uncertainty can be added to the lognormal distribution using data quality indicators provided by the pedigree matrix adapted from Weidema (1998). Each flow is assessed using the pedigree matrix and the indicator scores are broken down into five data quality categories: reliability, completeness, temporal correlation, geographical correlation, and further technological correlation. Each category is ranked 1-5, with 1 representing highest quality/lowest uncertainty and 5 representing lowest quality/highest uncertainty. After the ranking for each category has been selected, the uncertainty distribution, geometric mean, geometric standard deviation are calculated and applied to the flow. The pedigree matrix is provided in the appendix.

In this study, all of the values provided by BIR were averaged over a period of four years. This span of time was chosen for a few reasons: (1) that time period represents BIR "business as usual" management practices according to the ranch managers; (2) the data available were reliable and complete compared to other time spans; and (3) the average values from that number of years evened out the normal fluctuations in resources and production. Based on recommendations from the IPCC, predictions of GHG emissions from IFSM were given a base uncertainty value of ±20% for CH_4, ±50% for N_2O, and ±20% for CO_2 emissions related to fuel combustion (IPCC, 2006). Fossil energy use and feed production emission predictions from IFSM were given a base uncertainty of ±20%, ±40% for transportation, ±20% for drinking water, ±30% for water used in feed production, and ±20% for reactive N components based on expert opinion Rotz et al. (2019). All other inputs, outputs, and emissions not listed above were given a base uncertainty value of ±15%. In addition to the base uncertainty, all exchanges were assigned a data quality indicator score calculated by the pedigree matrix. All emissions from IFSM were assigned a pedigree uncertainty score of (4;1;3;3;3). Emission values based on primary data from BIR were given a score of (1;1;4;1;1). Indicator scores for all other flows were assigned based on available information relating to temporal, geographical, and technological correlation in the documentation of the process provider.

3.3.5 Sensitivity Analysis

A sensitivity analysis is different from uncertainty because it's a procedure that estimates the effects of choices made regarding data and methods on the outcome of the study rather than the actual values used (ISO, 2006a). If small changes in the assumptions result in large changes in the outcome, then it can indicate that the model is highly sensitive to a particular input or process. In order to complete the sensitivity analysis in openLCA, parameters were created and added into the process inputs and outputs. For this study, parameters that affect the calculation of GWP, including enteric CH_4, pasture N_2O, and pasture CO_2, were used to test the sensitivity of the GWP result to changes in GHG. A sensitivity index based on Rotz et al. (2019) was used to evaluate the percent change in the GWP footprint relative to a 10% change in each GHG. Because CO_2 and N_2O were broken out by pasture type, a 5% change was applied to semi-native CO_2 or N_2O and a 5% change was applied to improved CO_2 or N_2O to total an overall 10% change. As stated in Rotz et al. (2019), a sensitivity index score close to 0 indicated that the footprint was not sensitive, while a score close to or greater than 1 indicated a high level of sensitivity.

3.3.6 Unit Processes

Grass Pastures, Improved (IM)

Improved pastures at BIR cover approximately 1,925 ha. Every year the pastures are seeded with 696.6 kg of seeds and forage yield is 4.8 Mg DM/ha. An average of 909 ha of improved pastures were fertilized with N (as N), urea at rate of 31 kg/ha and 50 kg/ha K as (KCl). Urea ammonium nitrate (as N) was applied to 738.5 ha at a rate of 28 kg/ha and P (as P_2O_5) was applied to 312 ha at the same rate. In an average year of production, herbicides included glyphosate, fluoroxypyr, and triclopyr. Glyphosate was applied at a rate of 0.4 kg a.i./ha to 68 ha, fluoroxypyr was applied at a rate 0.6 kg a.i./ha to 16 ha, and triclopyr was applied at a rate of 1.7 kg a.i./ha to 16 ha. Fuel used in machinery equaled 28.6 l/ha/yr, as well as 0.069 l/ha/yr lubricant. Finally, solar panels produced 23.104 MJ/ha to pump water into troughs located throughout the improved pastures. Table A.10 contains all improved pasture values provided by BIR. Table A.5 and Table A.6 are tables of the life cycle inventory used in the model. All three can be found in the appendix.

Grass Pastures, Semi-native (SN)

Semi-native pastures at BIR cover approximately 2281 ha with a forage yield of 4.5 Mg DM/ha. Semi-native pastures at BIR are not seeded or fertilized. Triclopyr was applied to 53 ha with an application rate of 2.2 kg a.i./ha. Similar to improved pastures, 28.6 l/ha of fuel and .069 l/ha of lubricant were used. Solar panels produced 9.5 MJ/ha to pump water to troughs. Table A.9 contains all semi-native pasture values provided by BIR. Table A.7 and Table A.8 are the life cycle inventory used in the model. All three can be found in the appendix.

Cattle Feed

While most of their nutritional requirements are satisfied by grazing bahiagrass and native forages, the cattle were also supplemented with other feeds and minerals. Hay was provided at an average rate of 143.5 tons/yr. Range cubes with 18% protein equaled 67.3 tons/yr, and 24% protein range cubes equaled 89.3 tons/yr. Feed and supplement tags were provided, when available. Based on the ingredients listed in the range cube feed tags, cottonseed meal was the most common first ingredient and was used as a proxy for range cubes in the LCI. The cattle were also provided with 3.4 tons of protein tubs, 315.1 tons of molasses, and 4 tons of salt blocks annually. Rations totaled 239.7 tons/year and were divided between bull feed, heifer feed, weaning feed, and finishing feed. Similar to the range cubes, the most common first ingredient in the rations was cracked corn; thus, maize grain (feed) was used as a proxy for the rations in the LCI. A total of 115.7 tons of assorted minerals and 72 kg of pesticides, anthelmintics, and other cattle medications were used in an average year. In addition to feed, minerals, and medications, the herd drank just under 69 million liters of water, which was calculated from an average of 57 liters/head/day, as estimated by the ranch manager. Table A.11 contains all animal input values provided by BIR and Table A.2 is the life cycle inventory used in the model. Both can be found in the appendix.

Calf Production

The net live weight sold in an average year of production at BIR was 534,143.1 kg. This value is calculated using the herd composition, performance, and mortality rate data. The average annual cattle production numbers and rates are displayed in Table 3.1 below. During that time, the herd consumed over 1,000 tons of feed, minerals, and medications, drank approximately 69 million liters of water, and grazed 4,206 ha of improved and semi-native pastures. It required 77,781 kWh of electricity from the network to power the lights in the barns and around the ranch, but that value does not include electricity used in the research buildings, offices, or cabins. There are seven diesel powered trucks that drove an estimated combined total of 31,858 km every year. There are nine diesel powered agricultural tractors and pieces of heavy equipment machinery that drove a combined total of 10,137 km per year. The ranch also owns and operates gasoline powered vehicles: four trucks, six four-wheelers, and two swamp buggies. Those vehicles drove an estimated combined total of 84,311 km per year. There is also a swamp buggy that drove, on average, 1,900 km per year giving tours of the property. All of the vehicles consumed a total of 34,975.2 kg of diesel, 63.3 kg of lubricant, and 7,847.8 gal of gasoline. Table A.3 and Table A.4 contain the life cycle inventory used in the model and can be found in the appendix. The key cow-calf production measures used to estimate the average cattle production numbers can be found in Table A.1.

3.3.7 Calculation Procedures

Some of the values provided by BIR were given on a per ha basis or in different measures of weight or mass. Calculations were used in order to obtain appropriate units of measurement for the model. If an input for pastures was given in kg/ha, the value was multiplied by the total area of each pasture type to obtain an annual total of kg, unless noted otherwise. Fossil fuel and oil inputs provided in gallons were converted to kilograms or MJ/kg using the ecoinvent default values for gross or net calorific value and density of common fuels (see appendix). The total LW sold from BIR was calculated by multiplying the number of animals sold by their cull weight or weaning

Table 3.1: A summary of cattle production during an average year at BIR, averaged from 2014-2017.

Cattle Production Information	Value	Unit
Number of cows exposed to bulls	2,978	head
Number of bulls	164	head
Percent calf crop	74	%
Calf Mortality	19	head
Number of Calves (Survived until Weaning)	2195	head
Post-weaning Mortality for All Cattle	1.5	%
Cow Replacement Rate	15	%
Bull Replacement Rate	16	%
Replacement Bulls	26	head
Replacement Heifers	442	head
Purchased Bulls	22	head
Purchased Cows	1	head
Cull Bulls Sold	26	head
Cull Cows Sold	447	head
Calves Sold	1,727	head
Cull Bull Weight	717	kg/head
Cull Cow Weight	450	kg/head
Weaning weight	182	kg/head
Net live weight sold from the ranch[1]	534,143.1	kg

[1] Calculated from (weaning weight x calves sold) + (cull cows sold x cull cow weight) + (cull bulls sold x cull bull weight).

weight using inputs from Table 3.1. Transportation values in the life cycle inventory are measured on a "ton per km driven" basis. The vehicle fuel type, fuel quantities, and number of kilometers driven by vehicle type were provided by BIR, but not on a "per ton" basis. The transportation values used in the life cycle inventory were estimated using the conversions provided in the process provider.

3.4 Allocation

This study uses the economic value of the product and co-products to allocate the environmental burdens. ISO standards recommend that allocation of impacts be avoided, if possible, but if it is not possible, then system expansion should be considered, or allocation based on a physical or economic relationship (ISO, 2006b). As explained previously in the review of the literature, system expansion is not possible because this study proposes that a co-product of beef production is the provision of ecosystem services, and at this time it is not feasible to adequately model an entire ecosystem. Establishing a physical relationship between product and co-product is also not possible, because ecosystem services cannot be measured in units of mass or energy. Each product of this study does, however, have an estimated monetary value; therefore, an economic-based approach was chosen. In order to develop a reproducible method for multi-functional allocation between beef and ecosystem service, four economic allocation approaches were considered:

1. No allocation
2. Agricultural Conservation Easement Program (ACEP)
3. Conservation Stewardship Program (CSP)
4. "Highest and Best Use" (HBU)

3.4.1 Methods

The first method of economic allocation in this analysis was no allocation, or "all emissions were allocated to beef". The next method demonstrated was determining a value of the ES of BIR through the easement prices set by the USDA Agricultural Conservation Easement Program (ACEP). The ACEP provides financial assistance for protecting lands used for grazing, agriculture, and for wetland preservation (USDA-NRCS, 2021a). The land does not have to be taken out of production to receive the payments. Specific monetary values for different easement options can be found in the Florida Wetland Reserve Easement (WRE) Geographic Area Rate Cap (GARC). The easement rates vary by time commitment (i.e., 30 years, term, or permanent). A third method evaluated was the payments provided by the Conservation Stewardship Program (CSP) through the USDA Natural Resources Conservation Service (NRCS). The CSP is the largest conservation program in the U.S. with more than 28 million hectares enrolled and provides financial assistance to producers expanding conservation practices on agricultural operations. (USDA-NRCS, 2021b). Payments are made on a per acre basis with no minimum number of acres needing to be enrolled, but there is a maximum payment of $200,000 paid out over five years. Finally, a fourth method demonstrated was determining the worth of the land using Florida Statute 193.011 (2) "Highest and Best Use", which must be evaluated by qualified appraiser.

> "The highest and best use to which the property can be expected to be put in the immediate future and the present use of the property, taking into consideration the legally permissible use of the property, including any applicable judicial limitation, local or state land use regulation, or historic preservation ordinance, and any zoning changes, concurrency requirements, and permits necessary to achieve the highest and best use..." (Florida State Legislature, 2020).

However, it can be assumed that the "highest and best use" price for BIR would be far below its true value because the land is close to its natural state and includes wetlands, hammocks, semi-native pastures, etc. Theoretically, this would make the property less attractive to developers who would consider purchasing it for use as a retail space, housing development, etc.

3.4.2 Allocation Calculation Procedures

To determine the economic allocation percentages, the annual revenue from cattle sales was compared to the annual payments from each of the allocation methods developed for this study. The calculated annual payments for each allocation method represent the economic value of ES as a co-product of beef production at BIR in an average year of production.

To determine the economic allocation percentages the following formula was used:

$$Allocation\% = \frac{(approach)\$/kgLW}{((cattle)\$/kgLW + (approach)\$/kgLW)} * 100$$

No Allocation

With this economic allocation approach, all emissions were allocated to the production of LW leaving the farm gate. The average market value for cattle (cows, steers, heifers, and calves) between 2014-2017 was $3.78 per kg LW. The data was calculated using Table A.18 with the annual average price received by farmers for beef cattle and calves in the U.S. from the "2018 Florida Livestock, Dairy, and Poultry Summary" report (Florida Department of Agriculture and Consumer Services, 2019). The mean cattle prices over the four years were calculated as avg $/CWT, then converted to represent avg $/kg LW. The average annual revenue from cattle was $2,019,558.07 or $3.78 per kg LW.

Agricultural Conservation Easement Program

The ACEP approach was based on using the USDA NRCS payments provided to ranchers who set aside land for conservation purposes as a proxy for the value of ecosystem services. The value of the easements is determined by an appraisal conducted by the NRCS, then the NRCS contributes 50-100% of the fair market value. The percentage contributed is determined by the easement term. The NRCS pays 100% of the fair market value for permanent easements (the land is under conservation in perpetuity) and 50-75% of fair market value for 30 year or term easements (USDA-NRCS, 2021a). Because of the challenges associated with calculating ACEP payments in perpetuity, the assumption was that the NRCS paid 75% of the fair market value for easements at BIR over 30 years. The market value was calculated using the same real estate estimations as the HBU approach, except in this approach the higher average Florida value was used instead of the lower Highlands County value. The average FL value was used because the NRCS wants to protect agricultural lands in their natural state and is not concerned with future development potential. With these assumptions, the value of the easements was $43,143,941.25. The estimated value was then spread out over 30 years and divided by the total kg of LW produced in an average year to equal $2.69/kg LW.

Conservation Stewardship Program

The CSP approach was based on payments made to BIR by the USDA NRCS for their conservation practices. Similar to the other approaches, the payments served as proxy for the value of ES provided as a co-product of beef production. Once enrolled in the Conservation Stewardship Program, ranchers agree to implement conservation practices for the next five years and the NRCS provides financial and technical assistance (USDA-NRCS, 2021b). The payments will not exceed \$18/acre with a total cap of \$200,000 spread out over five years, or \$40,000 per year. When calculating CSP payments for BIR, it was assumed that the max payment of \$40,000 was received every year for five years (equaling \$3.85/acre, well below the payment cap of \$18/acre). The annual payment was divided by the total kg of LW sold in an average year to equal \$0.07/kg LW.

Highest and Best Use

The HBU approach was based on using the real estate value of the ranch as a proxy for the value of the ES provided. The value of the ranch was calculated using the "USDA Land Values 2019 Summary" report and the USDA NASS database (NASS-USDA, 2019). Between 2014-2017, the average asset value of agricultural land (including buildings) in Florida was \$5,535/acre. In Highlands County, FL agricultural land is valued approximately 33% lower than the state-wide average resulting in an average asset value of \$3,708.45/acre between 2014-2017. Ideally, the ranch would have been professionally appraised to determine its value, however that information was not available. For the time span considered in this study, BIR would have been worth \$58,117,500 using the average FL value or \$38,938,725 using the average Highlands County value. Because of the assumption that the natural state of the ranch would be less attractive to real estate developers and thus undervalued, the lesser value was chosen to represent BIR's HBU price. The calculated real estate value was then spread out over 30 years and divided by the total kg of LW sold in an average year to equal \$2.43/kg LW. The time frame of 30 years was chosen to make this approach comparable to the ACEP allocation approach.

3.5 Life Cycle Impact Assessment

3.5.1 Life Cycle Impact Assessment Methods

The database used to provide life cycle impact assessment methods was openLCA LCIA methods V2.1.0. The LCIA methods used in this study were IPCC 2013 GWP 100a, TRACI 2.1, and ReCiPe 2016 Midpoint (H) (Bare, 2012; IPCC, 2013; Weidema et al., 2013). The IPCC 2013 GWP

100a contains the climate change factors developed by the IPCC with a timeframe of 100 years. TRACI stands for "Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts" and is primarily a midpoint approach. TRACI 2.1 methodology is consistent with the US EPA guidelines and the category characterization factors are specific to the US (US EPA Office of Research and Development, 2012). ReCiPe has a broad set of midpoint impact categories and uses impact mechanisms that have global scope, when possible. “H” represents the hierarchist perspective and is considered the default model (PRé-Sustainability, 2021).

3.5.2 Impact Categories and Category Indicators

The impact categories and indicators most relevant to cow-calf production systems are global warming potential (kg CO_2-eq), eutrophication (g N-eq), fossil fuel depletion (MJ surplus), water consumption (m^3, converted to liters), and land use (m^2a crop-eq).

Chapter 4 - RESULTS

Without economic allocation, environmental footprints were as follows: 322.2 L of water consumed, 44.0 m^2 annual crop-eq in land use, and 2.0 MJ energy surplus in fossil fuel depletion per kg LW leaving the farm gate. Global warming potential with carbon sequestration was 12.27 kg CO_2-eq/kg LW when assuming the high N_2O value, and 7.03 kg CO_2-eq/kg LW when assuming the low N_2O value. BIR's GWP with carbon sequestration and high N_2O value will be called the "actual GWP" from this point forward. This "actual GWP" represents the value that BIR can use to compare their carbon footprint with other ranches in Florida and in the United States. For the purpose of comparison with the literature, GWP was also calculated without carbon sequestration. Without carbon sequestration and assuming the high N_2O value, GWP was 23.82 kg CO_2-eq/kg LW and 18.58 using a low N_2O emissions value for semi-native and with carbon sequestration.

4.1 Economic Allocation

When emissions were economically allocated between beef and ecosystem services, the footprints were reduced 2% using the CSP approach, 39% using the HBU approach, and 42% using the ACEP approach, as shown in Table 4.1. The reductions occurred equally across all impacts because this is a simple percent reduction based on unchanging economic values.

Table 4.1: Environmental impacts of beef with ecosystem services as a co-product by impact category and allocation method[1].

	GWP	**Eutro-phication**	**Fossil Fuel Depletion**	**Land Use**	**Water Consumption**
	kg CO2-eq/ kg LW	g N-eq/ kg LW	MJ surplus/ kg LW	m2a crop-eq/ kg LW	L/ kg LW
No Allocation	12.27	36.97	2.01	43.97	322.22
CSP	12.02	36.25	1.97	43.12	315.97
HBU	7.48	22.51	1.22	26.77	196.16
ACEP	7.12	21.59	1.17	25.68	188.20

[1] All results are reported including carbon sequestration and the high SN N_2O assumption.

4.2 LCIA Results

4.2.1 Global Warming Potential

Global warming potential refers to the relative warming that can occur as a result of increased greenhouse gas emissions. The IPCC 2013 GWP 100a impact assessment method calculates the potency of greenhouse gases relative to CO_2 over a 100 year time span. The GWP values are CO_2 = 1, CH_4 = 28, and N_20 = 265 (IPCC, 2013). (Bare, 2012; IPCC, 2013).

The total amount of each GHG emitted in one year of production is required to calculate GWP. As described in section 3.3.1, values for CH_4 and CO_2 were available, but the total N_2O emissions for each pasture type was unknown at the time of this study. A value from literature was used for the IM pastures because of the similar pasture conditions reported in Lu et al. (2020), but that value was not representative of the SN pastures at BIR which are composed of a mix of bahiagrass and native grasses. Because N_2O emissions are so variable and the literature on emissions from native grasses in subtropical climates is spares, it was decided that the Lu et al. (2020) value would also be used to estimate SN pasture emissions, but it would be considered a "high" value and is most likely an overestimation of the total SN N_2O emissions. A "low" value for SN N_2O emissions was estimated from a simulated cow-calf operation in south-central Florida using IFSM.

Figure 4.1 displays the top six contributors to the actual GWP when including carbon sequestration and the high SN N_2O emissions value. As expected, cattle production is the biggest contributor to GWP. With a value of -1.06 kg CO_2-eq/kg LW, SN pastures contribute the least. Figure 4.2 also displays the top six contributors to GWP per kg LW but includes carbon sequestration and the low SN N_2O emissions value. When comparing Figure 4.1 and Figure 4.2, the contributions of the top five are exactly the same, but the SN pastures value is more than four times lower the GWP calculated using the high value. This is because of the increased potency of N_2O relative to CO_2. In considering the overall GWP total for these figures, the value of carbon sequestration in SN pastures offsets the contribution of petrol (burned in machinery) in both figures. That suggests that pastures at BIR are drawing down more carbon than they are emitting through the burning of fossil fuels.

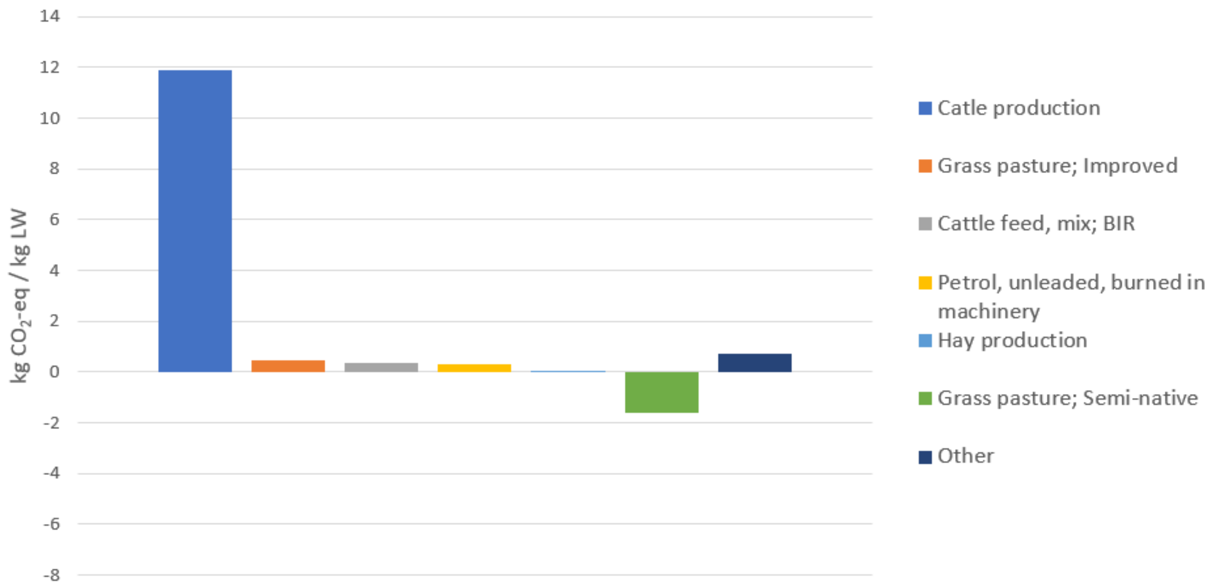

Figure 4.1: Cattle production, IM pastures, Cattle Feed, petrol used in machinery, hay production, and SN pastures are the top six contributors to GWP using the high SN N_2O value and including carbon sequestration.

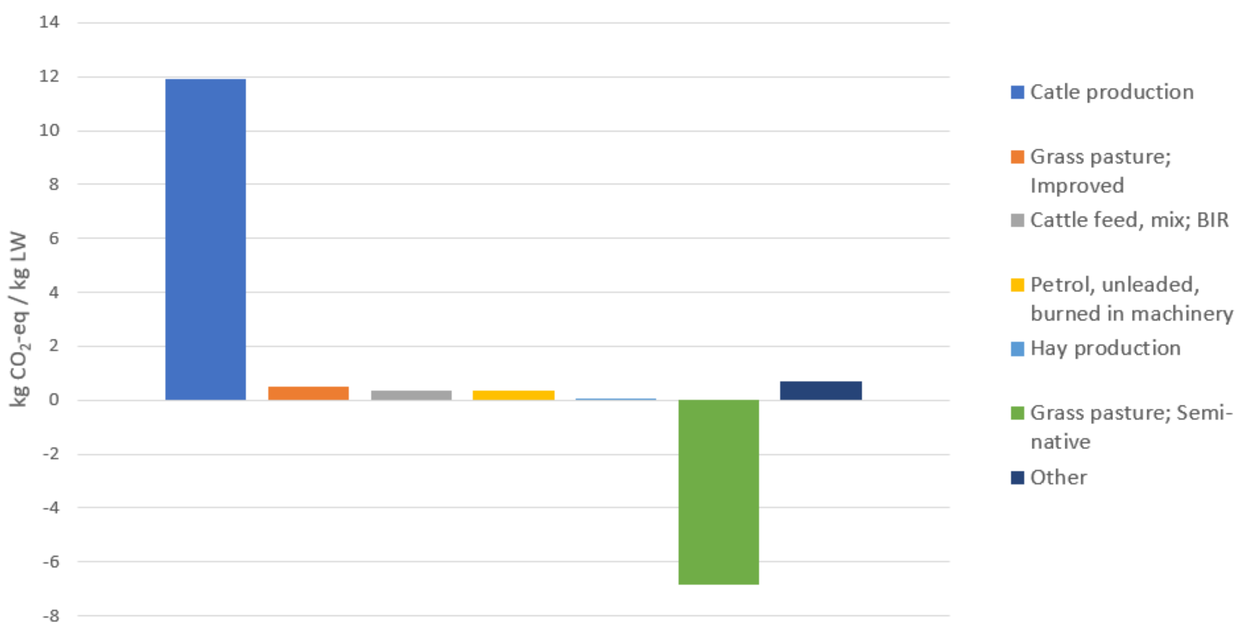

Figure 4.2: Cattle production, IM pastures, Cattle Feed, petrol used in machinery, hay production, and SN pastures are the top six contributors to GWP using the low SN N_2O value and including carbon sequestration.

The GWP results, without allocation, for four GWP scenarios is summarized in Table 4.2. For this study, the allocation approach results reported the GWP with high SN N_2O and including carbon sequestration because it was considered a conservative estimate. Presenting the GWP with carbon sequestration and low SN N_2O gives a lower bound to the range of scenario results. The inclusion or exclusion of carbon sequestration in LCA is still up for debate Rotz et al. (2019); Rowntree et al. (2020), therefore GWP results are also presented with and without carbon sequestration. When soil organic carbon is assumed to be at equilibrium and includes the high SN N_2O, the GWP per functional unit almost doubles to 23.82 kg CO_2-eq. There is a 43% difference between the scenarios that both assume carbon sequestration but differ in SN N_2O emission amounts, while the difference between the scenarios that both assume soil carbon equilibrium is only 22%.

Table 4.2: Global Warming Potential of 1 kg LW based on four scenarios: With carbon sequestration and high SN N_2O value, no carbon sequestration and high SN N_2O value, with carbon sequestration and low SN N_2O value, and no carbon sequestration and low SN N_2O value.

GWP	kg CO_2-eq/kg LW
Carbon sequestration High SN N_2O	12.27
No carbon sequestration High SN N_2O	23.82
Carbon sequestration Low SN N_2O	7.03
No carbon sequestration Low SN N_2O	18.58

The contribution of each GHG to the overall GWP in different scenarios is displayed in Figure 4.3. In all four scenarios, CH_4 is the largest contributor to GWP and that is being driven by enteric CH_4. The percent contributions of CH_4 and N_2O are the same for both high SN N_2O scenarios, regardless of carbon sequestration. The same is true for scenarios with low SN N_2O emissions. This shows that the choice of high or low N_2O is the driving force behind the differences in this study's GWP results.

The CSP allocation approach decreased the actual GWP from 12.27 kg CO_2-eq/kg LW to 12.02 kg CO_2-eq/kg LW. Applying the HBU approach for economic allocation resulted in a GWP of 7.48 kg CO_2-eq/kg LW for beef. Using the USDA NRCS ACEP payments as a proxy for the economic

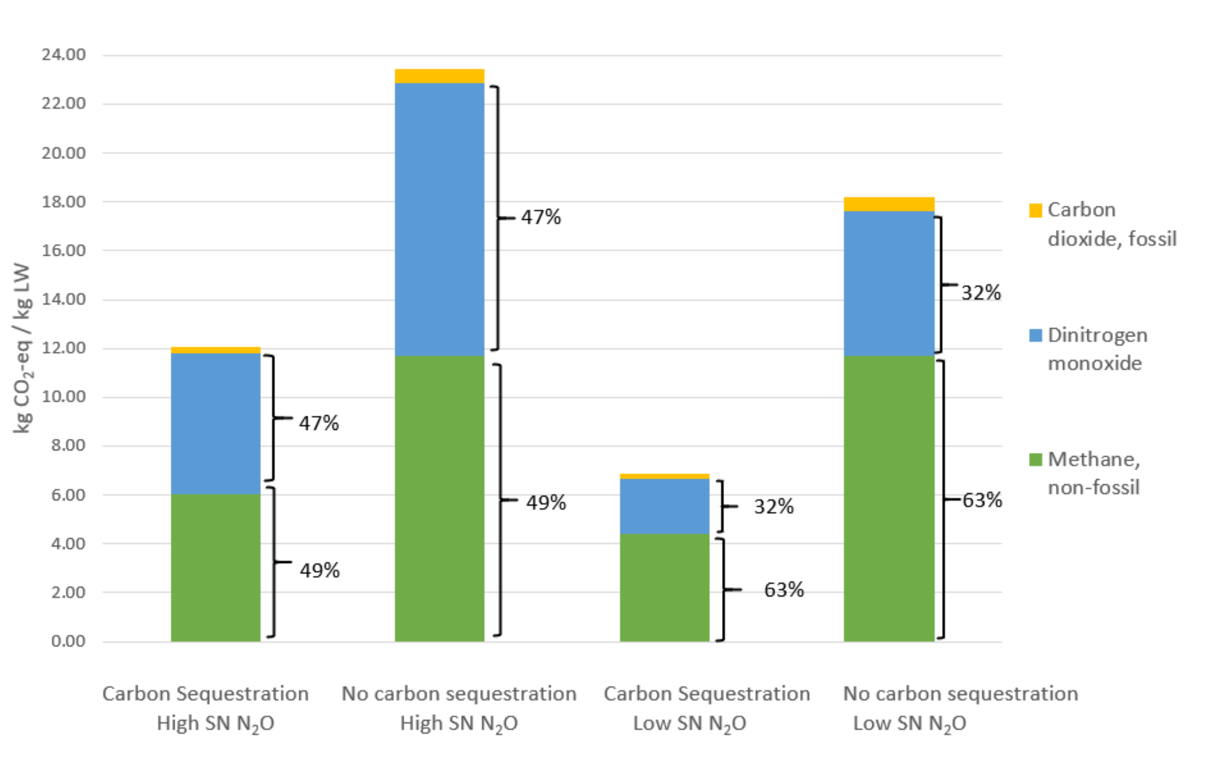

Figure 4.3: Global Warming Potential contribution analysis with high or low N_2O values and with or without carbon sequestration.

value of ecosystem services resulted in 58% of emissions allocated to beef for a GWP of 7.12 kg CO_2-eq/kg LW.

4.2.2 Eutrophication

Eutrophication refers to the runoff or leaching of excess nutrients, primarily nitrogen and phosphorus, into bodies of bodies of water subsequently causing algae blooms and hypoxic conditions. In TRACI 2.1, nitrogen equivalents are used to measure the eutrophication potential of coastal environments (Bare, 2012).

As seen in Figure 4.4, eutrophication potential without allocation was 36.97 g N-eq/kg LW. Using the different allocation approaches resulted in 36.25, 22.51, and 21.59 g N-eq/kg LW for CSP, HBU, and ACEP, respectively. The largest individual contributor to this impact category were the improved pastures (37.5%). This is most likely because improved pastures were regularly fertilized, contributing to greater potential nitrogen loss (Rotz, 2004). Semi-native pastures contributed 15.7% and cattle production accounted for 11.6%, mostly from urine and feces excretion (Rotz, 2004). The final 37.5% is attributed to off-farm resources, primarily the production of cottonseed, hay, and corn grain feed.

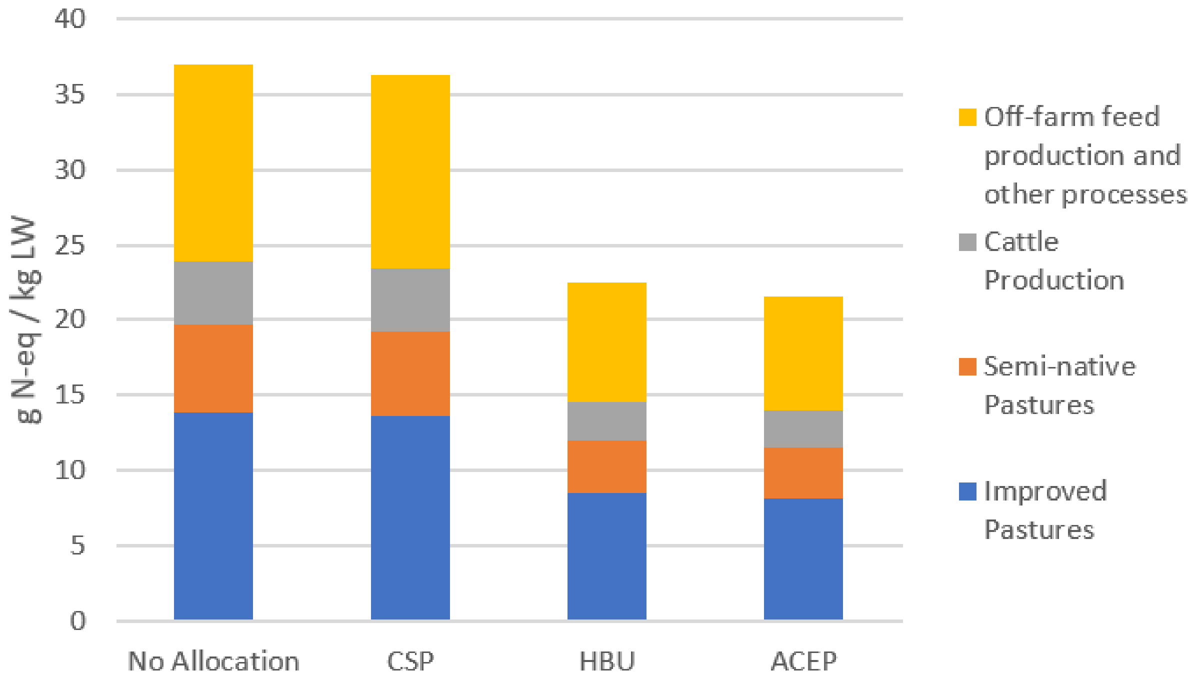

Figure 4.4: Contribution analysis of eutrophication measured in g N-eq by allocation method.

4.2.3 Fossil Fuel Depletion

Fossil fuel depletion refers to the depletion of non-renewable energy resources, like oil, gas, and coal. The TRACI 2.1 impact assessment method for resource use is based on the Eco-indicator 99 damage-oriented method and uses the concept of "surplus energy" (Bare, 2012). Surplus energy is defined as the difference between the energy needed to extract material now and the expected increase of extraction energy per kg of material in the future (Goedkoop and Spriensma, 2001).

The total fossil energy depleted was 2.01 MJ surplus per kg LW, without allocation, as displayed in Figure 4.5. With economic allocation using the approaches developed for this study, the overall fossil fuel depletion attributed to beef production was decreased by 0.04 MJ for CSP, 0.78

MJ for HBU, and 0.83 MJ for ACEP. The production of petroleum was by far the largest contributor to the overall footprint. This includes the gasoline and diesel consumed by the trucks, tractors, 4-wheelers, swamp buggies, and other machinery at BIR. The remaining 20% of the impact is mostly attributed to natural gas production, followed by hard coal operations.

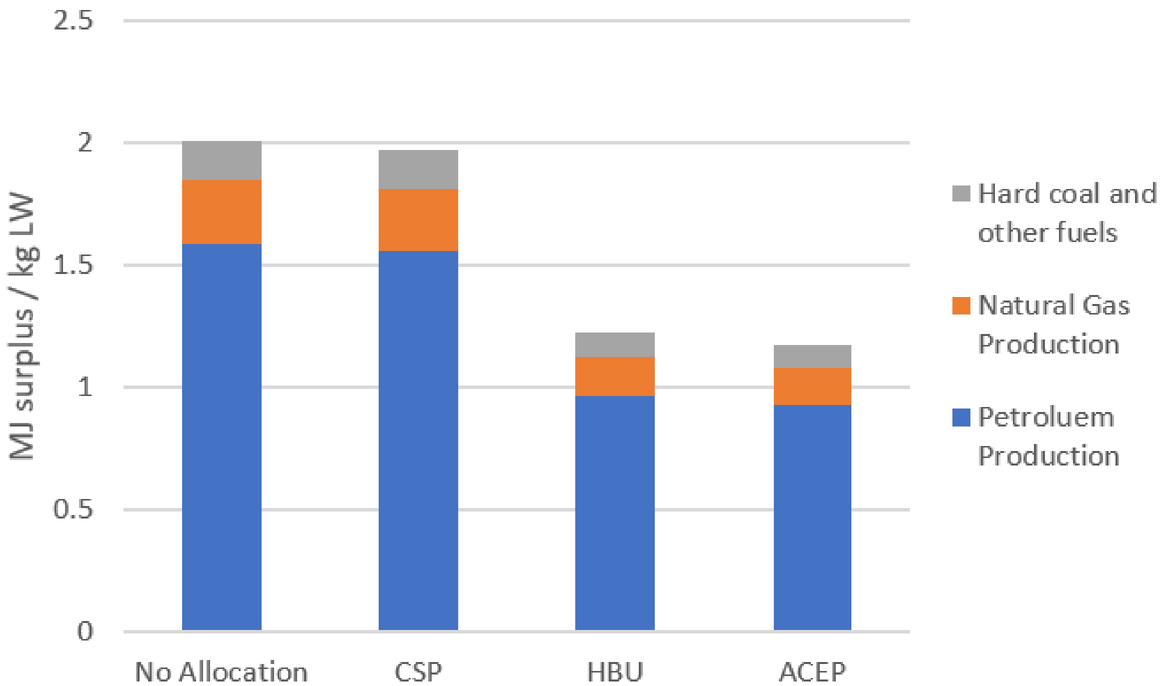

Figure 4.5: Contribution analysis of fossil fuel depletion measured in MJ surplus by allocation method.

4.2.4 Water Consumption

Water consumption refers to water that has been extracted from ground or surface waters and is evaporated, transferred, incorporated, or disposed and thus is no longer available to be used by humans or ecosystems (Huijbregts et al., 2017). In the ReCiPe 2016 Midpoint (H) impact assessment method, water consumption is defined as the amount of water that the watershed of origin is losing (Huijbregts et al., 2017). It is an estimate of blue water use, or water extracted from surface or groundwater. Reductions in water availability can lead to less water for irrigation, drinking, or other household and industrial uses, a reduction in plant diversity, and changes to river discharge. The impact category unit of measurement is m^3 but was converted to L to facilitate comparison with literature values.

In Figure 4.6, water consumption was 322.22 L/kg LW without allocation, or 315.97, 196.16, or 188.20 L/kg LW with economic allocation using CSP, HBU, or ACEP, respectively. Almost 64% of the total was attributed to the use of irrigation in the production of off-farm feeds (*e.g.*, cottonseed meal and corn grain). Drinking water for cattle consumed almost 91 L/kg BW, or 28.2% of the total footprint. The remaining approximately 8% is water attributed to the production of inputs like purchased feed, fertilizers, pesticides, or anthelmintics.

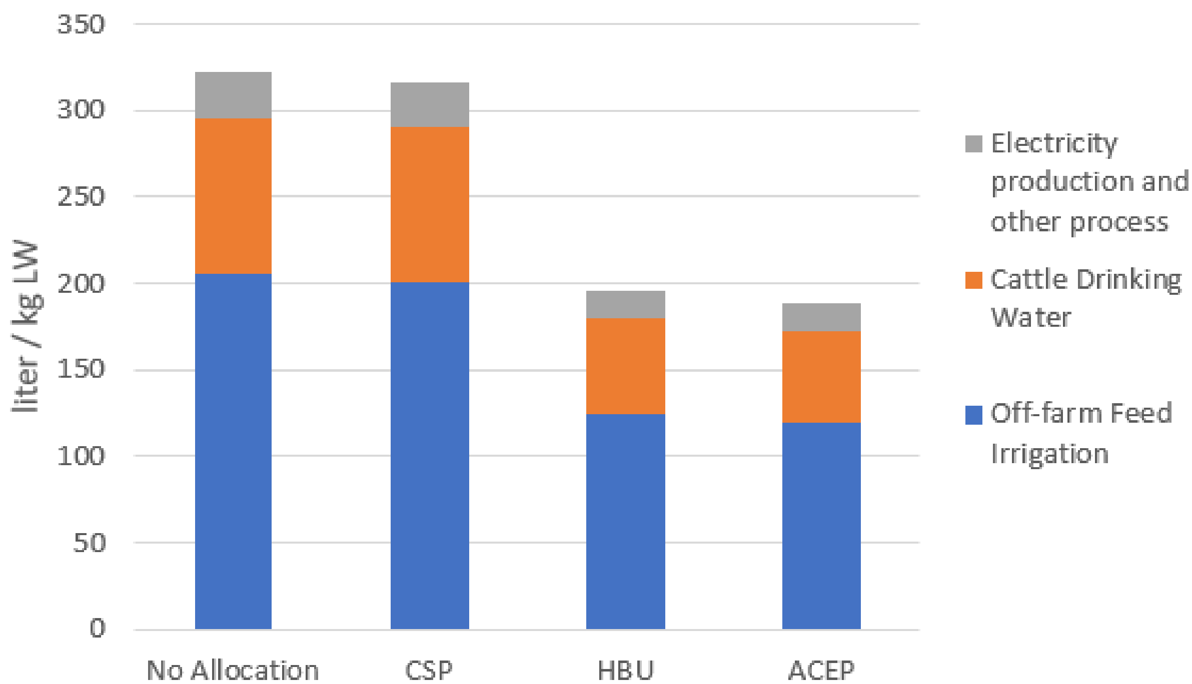

Figure 4.6: Contribution analysis of water consumption measured in liters for each allocation method.

4.2.5 Land Use

Land use refers to the impact on species richness from the transformation and occupation of the land from its natural state to its current use. Biodiversity is indirectly affected by changes to land use, either because of changes to land cover or through land use intensification (Huijbregts et al., 2017). In ReCiPe 2016 Midpoint (H), land use is calculated based on the relative species loss caused by different land use types, proportionate to the relative species loss resulting from annual crop production (Huijbregts et al., 2017). This study only reported land occupation in the LCI and not transformation, because the land has not been recently converted from another land use type.

Figure 4.7 shows that the production of 1 kg LW beef occupied 43.97 m^2 crop-eq annually. The total area per functional unit decreased by 0.85, 17.20, or 18.29 m^2 when considering the CSP allocation approach, the HBU approach, or the ACEP approach, respectively. As expected, the semi-native pastures account for just over 53% of the total land use and the improved pastures occupied 45%. The last 3% is attributed to the production of off-farm feeds and hay.

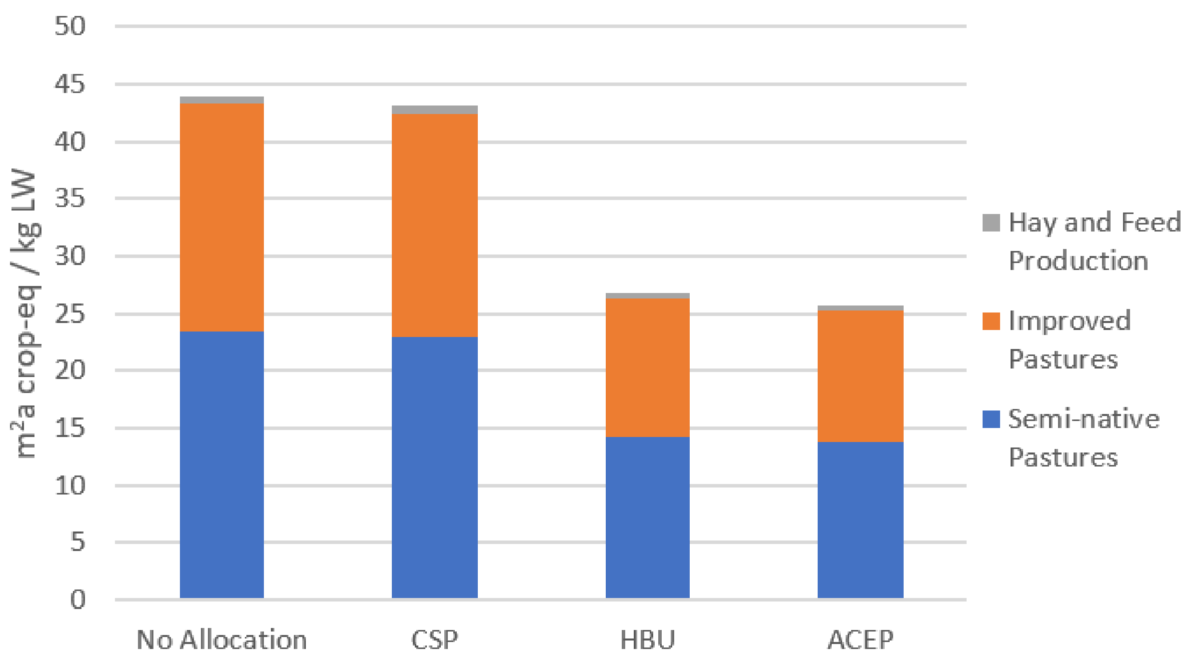

Figure 4.7: Contribution analysis of land use measured in m^2a crop-eq for each allocation method.

4.3 Sensitivity Analysis

Figure 4.8 demonstrates the sensitivity of GWP of beef production to 10% increases in GHG amounts. The sensitivity indices are presented for animal CH_4, pasture CO_2, and pasture N_2O and are broken down by whether the high or low value for SN N_2O emissions was used and whether carbon sequestration was included, or soil organic carbon was assumed to be at equilibrium. As the figure shows, GWP is most sensitive to changes in the amount of CH_4, particularly when a low value for SN N_2O is used and carbon sequestration is included. Animal GWP is 48% less sensitive to CH_4 when soil organic carbon is assumed to be at equilibrium, but only when the high N_2O value is used to represent SN N_2O emissions. Compared to CH_4 and CO_2, GWP is least sensitive to increases in pasture N_2O, but the sensitivity changes based on which criteria are applied. The sensitivity of GWP decreases by 49% when equilibrium of soil organic carbon is assumed, but the index decreases an additional 18% to 0.14 when the low value of SN N_2O emissions is assumed.

4.4 Uncertainty Analysis

Table 4.3 and Table 4.4 display the results of the uncertainty analysis using Monte Carlo Simulations (MCS). Using MCS, the parameters were sampled 1,000 times and the results were averaged to determine the average probabilistic value. The standard deviation, min, max, median, and 95% confidence bands were also reported. The relative standard deviation (RSD) was calculated for each impact category and the deterministic cumulative results (the reported impact category results) were compared to the average probabilistic values. The RSD is a standardized measure of dispersion of a probability distribution where a lower value indicates that the data is tightly grouped around the mean.

When considering all of the impacts reported, the land use impact category had the least RSD percent and the smallest difference between the probabilistic mean and the deterministic result indicating a low level of uncertainty. For 1 kg LW, without allocation, the probabilistic mean for land use is 45.27 m^2a crop-eq with a 95% confidence band range of 35.08-56.49 m^2a crop-eq. The RSD is 14.5% and there is less than a 3% difference between the MCS mean and the

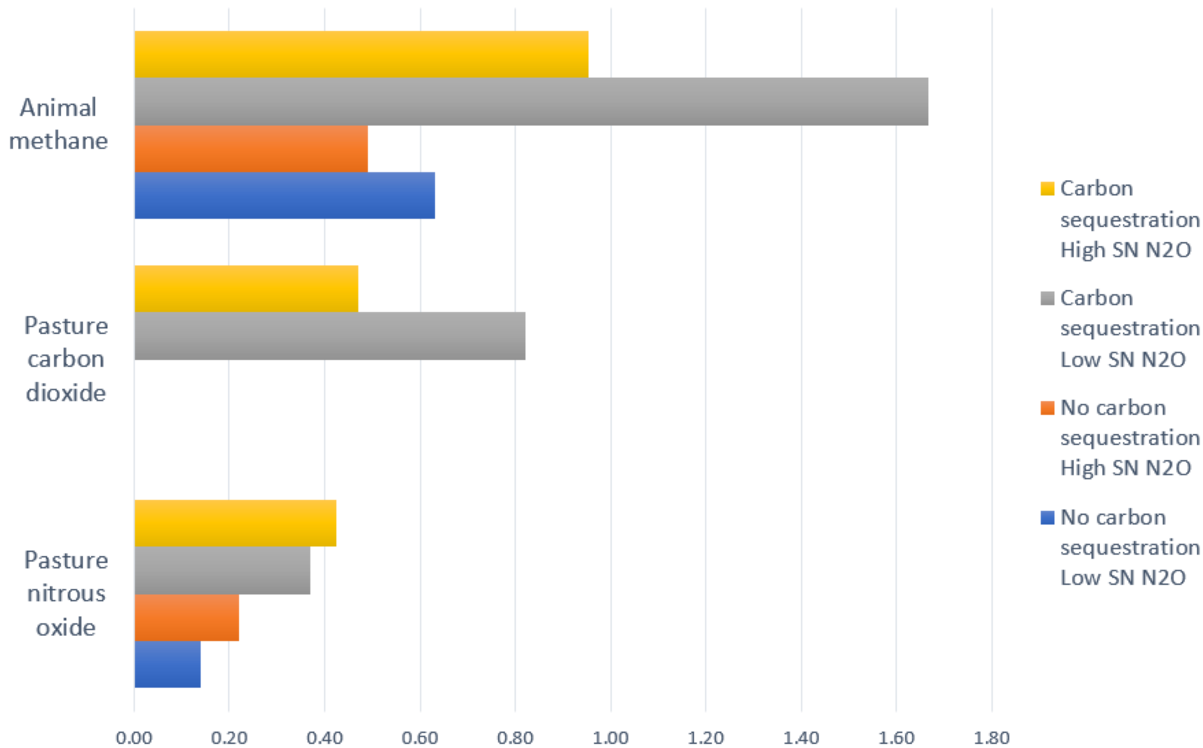

Figure 4.8: Sensitivity of GWP of beef production to changes in greenhouse gases, low or high SN N_2O values, and with or without carbon sequestration. The sensitivity index is modified from Rotz et al. (2019) and is the percent change in GWP compared to a 10% increase in GHG.

deterministic result. Conversely, the global warming potential scenarios have the highest RSD percent of the impact categories and the largest percent differences between the probabilistic means and the deterministic results. The probabilistic mean for the GWP of 1 kg LW that includes the measured carbon sequestration values from BIR and the "high" N_2O emissions value for semi-native pastures is 14.14 kg CO_2-eq with a 95% confidence band of 5.83-24.87 kg CO_2-eq. The RSD is almost 41% and the percent difference between the probabilistic mean and the deterministic result is over 15%, meaning that there is a large spread in the values and thus a higher level of uncertainty in the result. This is also demonstrated by the GWP of 1 kg LW that includes carbon sequestration but assumes the "low" N_2O emissions value for semi-native pastures. There is almost a 100% difference between probabilistic mean and the deterministic result and a 39% RSD.

Table 4.3: The results of 1,000 Monte Carlo Simulations for uncertainty analysis of Land Use, Water Consumption, Eutrophication, and Fossil Fuel Depletion.

Impact Category Uncertainty Analysis				
Descriptive Statistics (per kg LW)	**Land use (m^2a crop-eq)**	**Water consumption (L)**	**Eutrophication (g N-eq)**	**Fossil fuel depletion (MJ surplus)**
Result	43.97	322.22	36.97	2.01
Mean	45.27	342.65	40.41	2.24
Standard deviation	6.57	108.13	7.31	0.43
Minimum	26.31	122.58	23.66	1.33
Maximum	70.24	949.99	83.63	3.75
Median	44.94	321.60	39.37	2.19
5% Percentile	35.08	205.78	30.19	1.61
95% Percentile	56.49	553.33	52.77	3.03
RSD	14.52%	31.56%	18.08%	19.19%
% Difference	9.30%	2.96%	11.76%	6.34%

Table 4.4: The results of 1,000 Monte Carlo Simulations for uncertainty analysis of Global Warming Potential assuming carbon sequestration or organic soil carbon equilibrium and a low or high value of N_2O emissions for semi-native pastures.

Impact Category Uncertainty Analysis				
Descriptive Statistics (per kg LW)	**GWP Carbon Sequestration High N_2O**	**GWP No Carbon Sequestration High N_2O**	**GWP Carbon Sequestration Low N_2O**	**GWP No Carbon Sequestration Low N_2O**
Result	12.27	23.82	7.03	18.58
Mean	14.14	14.45	13.97	13.95
Standard deviation	5.76	5.71	5.45	5.56
Minimum	-1.32	0.13	0.71	1.74
Maximum	44.19	40.57	36.74	40.08
Median	13.55	13.87	13.42	13.42
5% Percentile	5.83	6.16	6.02	5.86
95% Percentile	24.87	25.12	23.56	24.36
RSD	40.74%	39.52%	39.01%	39.87%
% Difference	15.25%	-39.33%	98.68%	-24.92%

Chapter 5 - DISCUSSION

5.1 Discussion

Using the data provided by BIR, the cradle-to-farm gate GWP of 1 kg LW ranged from 7.03 to 12.27 kg CO_2-eq with carbon sequestration, or 18.58 to 23.82 kg CO_2-eq, without carbon sequestration depending on the assumption for SN N_2O. The results of this study compared to a cradle to farm gate study on the environmental impacts of the US beef value chain, where the cow-calf phase reported 28.51 kg CO_2-eq/kg consumed, boneless, edible beef in the USA, or 11.4 kg CO_2-eq/kg LW (assuming consumed, boneless beef is 40% of LW) (Asem-Hiablie et al., 2019). A cradle-to-farm gate study of California Angus beef production reported 21.2 to 22.6 kg CO_2-eq/kg HCW or 13.14 to 14.01 kg CO_2-eq/kg LW (assuming 62% dressing percentage) (Stackhouse-Lawson et al., 2012). This study's results are also similar to a cradle-to-farm gate LCA conducted throughout the U.S. in 2019 using data from U.S. Beef Sustainability Project to simulate representative regional cow-calf operations. In the Southeast, Rotz et al. (2019) reported 19.6 kg CO_2-eq/kg CW, or 12.15 kg CO_2-eq/kg LW (assuming a 62% dressing percent). That study assumed soil C levels are in long-term equilibrium, so while their result falls within the range of results reported for this study, it would more closely compare to the GWP without carbon sequestration and low SN N_2O result of 18.58 kg CO_2-eq. Even though the inclusion of carbon sequestration in LCA is a relatively recent development, BIR has data that suggests their pasture are net CO_2 sinks and thus the appropriate range for GWP in this study is 7.03 to 12.27 kg CO_2-eq/kg LW. Assuming the same dressing percentage, that study also reported 11.41 kg CO_2-eq/kg LW for the Midwest, 9.2 kg CO_2-eq/kg LW for the Southern Plains, and 8.43 kg CO_2-eq/kg LW for the Southwest (Rotz et al., 2019). A cradle-to-farm gate LCA of three beef production strategies in the Upper Midwestern United States reported 19.2 kg CO_2-eq/kg LW for pasture finished beef, but they also reported that the GWP decreased to 11 kg CO_2-eq/kg LW when carbon sequestration was included (Pelletier et al., 2010). A cradle-to-farm gate LCA study conducted in Georgia included carbon sequestration and reported 4.1 kg CO_2-eq/kg CW, or 2.54 kg CO_2-eq/kg LW. However, it is difficult to compare that study's results to this study because they reported the GWP for a multispecies pasture rotation production systems, which includes cattle, swine, poultry, sheep, goats, and rabbits (Rowntree et al., 2020). As shown by Figure 4.3 and Figure 4.8, GWP is most sensitive to changes in CH_4 and enteric CH_4 is responsible for the largest percent contribution to GWP. This study assumed enteric CH_4 emissions of 0.246 kg/head/day. In a study of CH_4 emissions of beef cattle (*Bos taurus*) grazing bahiagrass in Louisiana, enteric emissions ranged between 0.165 to 0.294 kg/head/day (DeRamus et al., 2003), which is similar to the assumption for this study. Rotz et al. (2019) reported CH_4 emissions of 360 g/kg CW for the cow-calf phase in the Southeast. Assuming a 62% dressing percentage, that equates to 0.267 kg/cow/day (Rotz et al., 2019). While the value for enteric CH_4 assumed in this study is outside of the IPCC range of 0.131 to 0.222 kg/head/day (IPCC, 2006), it closely agrees with studies that took place in conditions similar to BIR.

The results of the MCS demonstrated that there are varying levels of uncertainty surrounding the impact category results, with the carbon footprint scenarios having the most uncertainty. This agreed with results of the sensitivity analysis where the sensitivity of GWP depended on model parameter choices. Uncertainty analysis and sensitivity analysis were important to include in LCA because they provided support in the interpretation of results and trustworthiness of research conclusions (Finnveden et al., 2009; Cherubini et al., 2018; Weidema et al., 2013).

5.2 Key Points and Recommendations

One recommendation from this study is that the multi-functionality of cattle production should be considered when allocating emissions in future LCAs (Weiler et al., 2014). Gomez-Casanovas et al. (2018) found that grazing increased the net storage of C and decreased the GWP associated with C fluxes of pasture by increasing its net CO_2 sink strength. Chamberlain et al. (2017) found that pastures at BIR are, in fact, net CO_2 sinks, sequestering up to 163 ± 54 g CO_2-C per m^2 per yr. It can be argued that these ES would not occur at these rates, or possibly at all, without the addition of cattle. This study demonstrated the importance of including the multi-functionality of livestock production in an LCA. It is important to note that the chosen methods for allocation in this study are not paramount to other methods for dealing with multi-functionality. BIR offered up their data and management practices to be used as a case study to validate the importance of including ES as co-products of beef production in the context of LCA.

One of the interesting outcomes of using the allocation approaches developed in this study is the adaptability of the approaches to farms of various sizes, locations, and production types. This is important for small-holder, organic, and grass-fed systems that are actively managing for conservation. In the CSP approach, there is a cap of $40,000, or $18/acre annually. A ranch the size of BIR exceeds the payment cap on a per acre basis, but cow-calf operations of approximately 2,200 acres or less can take full advantage of enrolling as many acres as possible. When comparing the revenue from LW sold to the payments from CSP, the allocation percentage will increase and thus decrease the emissions allocated to beef production, assuming the total LW sold is scaled to match the size of the farm. The HBU and ACEP approaches are based on real estate appraisals and, as such, are subject to development pressure. In areas with high development pressure, the value of the land will increase and, using the logic of this study, so will the value of ES provided by beef production. This would increase the percentage of emissions allocated to the ecosystem and decrease the emissions allocated to beef. By assigning an explicit value to ES, this study provides quantitative data for preserving land and can be used to inform local policy on land use and development, especially in areas of high development pressure.

As reported in the results section, the overall environmental footprints of beef production change for each economic allocation approach. Instead of being solely allocated to beef production, a portion of the emissions are being allocated to the ecosystem. Nevertheless, the emissions are still there, and it might be asked how BIR could reduce their carbon footprint. Conventionally, the most efficient way to mitigate a system's carbon footprint was to increase efficiency. The larger the denominator, the smaller the fraction. One way to increase cattle production at BIR would be to convert more semi-native pastures to exclusively bahiagrass, which provides more nutritionally dense forage. However, as stated in the definition of the land use impact category, biodiversity can be negatively affected by changes to land use intensification. Decreasing biodiversity would go against BIR's mission to produce beef with conservation-oriented management practices. The conversion of semi-native pastures would also likely double their eutrophication impact as evidenced by Figure 4.4, where improved pastures are the largest contributor to eutrophication, mostly because of fertilizer use. Because of their proximity to large bodies of freshwater and the Everglades, reducing the amount of excess nutrients washed downstream is very important to BIR and the state of Florida. These are just two examples of the trade-offs that would be associated with increasing production to decrease their GWP. It can be argued that based on the results of this study, BIR is accomplishing their goal of managing for profitable beef production and conservation.

5.3 Limitations

This study was not without its limitations. Pasture-based production systems are extraordinarily complex with numerous factors interacting with each other in varying ways (Ripoll-Bosch et al., 2013). This added a layer of complexity and uncertainty to the data, the model, and the methodology. The results presented in this study were modeled to be representative of an average year of production at BIR, but more reliable quantitative results could be obtained using precise measurements. For example, using actual measured animal emissions from the cattle at BIR instead of simulated emissions from IFSM.

In this study, ecosystem services, as a whole, were assigned a monetary value. However, it fails to take into account that some services might be provided at higher rates than other based on management practices or just the environment in general. It also does not take into account that some services might be more valuable than others depending on the location. Soil water retention is important in areas with minimal precipitation, whereas flood control is important in areas that are regularly flooded and excess nutrients are washed downstream. Some of the socio-cultural services provided by cattle production, like a ranch that has been passed down through generations, are beyond economic value and therefore are difficult to measure their true worth (Weiler et al., 2014).

The economic allocation approaches developed for this study were based on government-funded programs and therefore were based on political decisions. The HBU approach excluded, the CSP approach and the ACEP approach are based on programs provided by the USDA NRCS. This means that funding for those programs or existence of those programs depends on the current political ideology. Thus, the methodology and the results of this study can change because they rely on policy rather than the product itself (Ripoll-Bosch et al., 2013). If the allocation approaches are applied to beef production systems outside of Florida (but still in the U.S.), the economic allocation percentages will change depending on the state. This is because each state has their own NRCS agency and decides independently, to a certain degree, the specific funding for each program USDA-NRCS (2021a); NRCS (2021).

5.4 Future Work

BIR has other revenue streams that were not incorporated into this study. They receive payments from the Northern Everglades Payment for Environmental Services (NE-PES) for water retention and they have over 1,600 hectares permanently enrolled in the Wetlands Reserve Program (WRP). They also sell hunting leases and grass sod when there is a demand. Future work could include these forms of revenue in the LCA and recalculate the allocation percentages based on these services provided to the community. The data for this analysis was based on management practices that are called "business as usual" at BIR. Starting around 2018, BIR changed some of their management practices and began calling the time period from 2018 going forward "aspirational" (ASP). Steps are being taken to complete an ASP LCA based on the changes in management practices.

Future research could include how the different economic allocation approaches change when policies change to incentivize other agricultural or conservation practices. The WRP was replaced by the Agricultural Conservation Easement Program (ACEP) in the Agricultural Act of 2014

(NRCS, 2014). This is not to say that wetlands were no longer protected by conservation programs, but to show that as policies and laws change so do the nature of conservation programs. Future research could also explore how the economic allocation percentages change for beef production in states other than Florida. The results could be used to make recommendations for allocation methodological choices in beef production LCAs across the U.S.

While this study was being completed, Liu et al. (2021) proposed that Global Warming Potential Star (GWP*) be reported alongside GWP, because GWP* takes into account the short atmospheric lifetime of CH_4 and provides a more accurate estimation of its warming impact. In the future, GWP* will be calculated for this study and reported with GWP.

Chapter 6 - CONCLUSION

Including ES as a co-product of conservation oriented beef production in the context of LCA decreased the environmental impacts of 1 kg LW sold by 2%, 39%, or 42% using the CSP approach, the HBU approach, or the ACEP approach, respectively. The economic allocation approaches developed for this study were either based on property valuation or payments offered by the USDA for conservation practices and, as such, are reproducible for any beef production system in the U.S.

The results of the LCA showed that 1 kg of LW sold from BIR, without allocation, required 322.22 L of water consumption, 43.97 m^2 annual crop-eq in land use, and 2.01 MJ energy surplus in fossil fuel depletion. The eutrophication potential was estimated to be 36.97 g N-eq/kg LW and the global warming potential ranged from 7.03 to 23.82 kg CO_2-eq/kg LW, depending on the assumption for the SN N_2O emissions and the inclusion, or exclusion, of carbon sequestration. The GWP (with carbon sequestration and high SN N_2O) decreased from 12.27 kg CO_2-eq/kg LW to 7.12 kg CO_2-eq/kg when emissions were allocated between beef the ecosystem using the ACEP approach. The uncertainty and sensitivity analysis indicated that GWP is most sensitive to changes in CH_4 emissions and is an important consideration when drawing research conclusions from these results. Further research should be completed to attain exact enteric CH_4 emissions from cattle at BIR.

This study demonstrated the importance of including the multi-functionality of livestock production in the context of LCA. BIR is actively managing for conservation and thus the ES provided should be considered a co-product of beef production. The results of this study will provide scientific knowledge for evidence-based decision making by BIR personnel and can be used to inform local policy on land use and development.

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APPENDIX

Table A.1: An amended SPA trend report for cow-calf key production measures 2014-2017.

SPA Cow-Calf Key Production Measures Trend Report				
	SPA Year 2014	SPA Year 2015	SPA Year 2016	SPA Year 2017
Exposed Cows to Produce Weaned Calves	3,011	2,834	2,900	3,165
Total number of calves at weaning	2,473	2,222	2,239	1,844
Pregnancy Percentage	83.33%	78.40%	83.70%	63.19%
Calf Crop or Weaning Percentage (number of calves/number of cows)	82.13%	72.79%	77.21%	58.26%
Loss pregnancy to weaning	1.87%	7.10%	6.10%	4.93%
Actual Average Weaning Weight per Calf (lbs)	418	407	355	420

Table A.2: Life cycle inventories for the production of feed for cattle during an average year (averaged from 2014-2017).

Inputs - Cattle Feed					
Flow	Amount	Unit	Uncertainty	Data Quality Score	Notes
hay	143.5	t	lognormal: gmean = 143.5 gsigma = 1.28	(1;1;3;3;3)	Hay
maize grain, feed	239.70	t	lognormal: gmean = 239.70 gsigma = 1.65	(4;1;5;3;3)	Proxy for rations
mineral supplement, for beef cattle	115.71	t	lognormal: gmean = 115.71 gsigma = 1.28	(1;1;1;5;3)	Minerals
molasses, from sugar beet	315.14	t	lognormal: gmean = 315.14 gsigma = 1.60	(1;1;5;4;3)	Molassess
protein feed, CSM, 100% crude	160.00	t	lognormal: gmean = 160.0 gsigma = 1.66	(4;1;4;5;4)	Proxy for range cubes
sodium chloride, powder	4.04	t	lognormal: gmean = 4.04 gsigma = 1.26	(1;1;2;3;3)	Salt blocks
Outputs					
Flow	Amount	Unit	Uncertainty		Notes
Cattle Feed, mix, BIR, BAU	978.10	t	lognormal: gmean = 978.10 gsigma = 1.06		Total cattle feed
Dinitrogen monoxide	756	kg	lognormal: gmean = 756 gsigma = 1.63	(4;1;3;3;3)	IFSM Indirect sources

Table A.3: Life cycle inventories for the production of cattle (inputs) during an average year (averaged from 2014-2017).

Inputs - Cattle Production					
Flow	Amount	Unit	Uncertainty	Data Quality Score	Description
Benzimidazole-compound	71.85	kg	lognormal: gmean = 71.85 gsigma 1.35	(2;1;4;3;3)	Antiparasitics
Cattle Feed, mix, BIR, BAU	978.095	t	lognormal: gmean = 978.10 gsigma 1.15	(1;1;2;1;1)	Total feed
Electricity, at grid	77781	kWh	lognormal: gmean = 77781.0 gsigma 1.30	(1;1;4;1;2)	Data from Glades Electric
Pasture; Improved; grazed DM; BIR	9240	Mg	lognormal: gmean = 9240.0 gsigma 1.15	(1;1;2;1;1)	Mg DM yield IMP
Pasture; Semi-native; grazed DM; BIR	10264.5	Mg	lognormal: gmean = 10264.5 gsigma 1.15	(1;1;2;1;1)	Mg DM yield SNP
lubricating oil	16.67	kg	lognormal: gmean = 16.67 gsigma 1.32	(2;1;3;3;3)	Oil for machinery
Water, well, in ground	68923888	l	lognormal: gmean = 68923888 gsigma 1.21	(2;1;2;1;1)	Drinking water

Table A.4: Life cycle inventories for the production of cattle (outputs) during an average year (averaged from 2014-2017).

Outputs - Cattle Production					
Flow	Amount	Unit	Uncertainty	Data Quality Score	Description
Ammonia	19371	kg	lognormal: gmean = 19371 gsigma 1.39	(4;1;3;3;3)	IFSM Grazing Ammonia
calves, LW, BIR, at farm gate	534143.08	kg	lognormal: gmean = 534143.08 gsigma 1.03		Net Sold LW
Dinitrogen monoxide	406	kg	lognormal: gmean = 406.0 gsigma 1.63	(4;1;3;3;3)	IFSM Animal Nitrous Oxide
Hydrogen sulfide	17	kg	lognormal: gmean = 17.0 gsigma 1.39	(4;1;3;3;3)	IFSM Grazing hydrogen sulfide
Methane, non-fossil	225474	kg	lognormal: gmean = 225474 gsigma 1.39	(4;1;3;3;3)	IFSM Animal Methane
VOC, volatile organic compounds	3222	kg	lognormal: gmean = 3222.0 gsigma 1.39	(4;1;3;3;3)	IFSM Field/ grazing VOC

Table A.5: Life cycle inventories for the production of improved pastures (inputs) during an average year (averaged from 2014-2017).

Inputs - Improved Pasture Production					
Flow	Amount	Unit	Uncertainty	Data Quality Score	Notes
diesel	15624.61	kg	lognormal: gmean = 15624.61 gsigma = 1.56	(1;1;1;4;4)	Diesel
Electricity, AC, 2300-7650 V	44475.2	MJ	lognormal: gmean = 44475.2 gsigma = 1.20	(1;1;2;3;1)	Solar panels
glyphosate	27.2	kg	lognormal: gmean = 27.2 gsigma = 1.54	(1;1;1;3;4)	Herbicide
grass seed, organic, for sowing	696.6	kg	lognormal: gmean = 12841.2 gsigma = 1.07	(1;1;1;3;4)	Seeds
lubricating oil	117.55	kg	lognormal: gmean = 117.55 gsigma = 1.32	(2;1;3;3;3)	Oil for machinery
Occupation, pasture, man made, intensive	1925	ha*a	lognormal: gmean = 1925 gsigma = 1.15	(1;1;1;1;1)	Total area
pesticide, unspecified	36.8	kg	lognormal: gmean = 36.80 gsigma = 1.54	(1;1;1;3;4)	Pesticides
petrol, unleaded, burned in machinery	1175768.45	MJ	lognormal: gmean = 1175768.45 gsigma = 1.58	(1;1;1;5;4)	Gasoline
potassium chloride	45450	kg	lognormal: gmean = 45450 gsigma = 1.34	(1;1;4;3;3)	K (as KCl)
triple super-phosphate	8736	kg	lognormal: gmean = 8736 gsigma = 1.34	(1;1;4;3;3)	P (as P2O5)
urea	28179	kg	lognormal: gmean = 28179 gsigma = 1.34	(1;1;4;2;3)	N (as N)
urea ammonium nitrate mix	20679.4	kg	lognormal: gmean = 20679.4 gsigma = 1.34	(1;1;4;2;3)	32-0-0

Table A.6: Life cycle inventories for the production of improved pastures (outputs) during an average year (averaged from 2014-2017).

Outputs - Improved Pastures					
Flow	Amount	Unit	Uncertainty	Data Quality Score	Notes
Ammonium/ammonia, as N	2.29845	Mg	lognormal: gmean = 2.29854 gsigma = 1.29	(1;1;4;1;1)	N total as NH4
Carbon dioxide	-2290.75	Mg	lognormal: gmean = -2290.75 gsigma = 1.1	(1;1;1;1;1)	Net sink
Dinitrogen monoxide	9.625	Mg	lognormal: gmean = 9.63 gsigma = 1.53	(3;1;1;4;2)	Lu et al. 2020
Improved; grazed forage, DM; BIR	9240	Mg	lognormal: gmean = 9240 gsigma = 1.03		Forage Yield
Methane, from soil or biomass stock	451.4125	Mg	lognormal: gmean = 451.41 gsigma = 1.1	(1;1;1;1;1)	Ecosystem
Nitrate	0.336875	Mg	lognormal: gmean = 0.34 gsigma = 1.29	(1;1;4;1;1)	N total as NO3
Nitrogen, total	12.67806	Mg	lognormal: gmean = 12.68 gsigma = 1.29	(1;1;4;1;1)	N total as N
Nitrogenous Matter, Kjeldahl, as N	12.6758555	Mg	lognormal: gmean = 12.68 gsigma = 1.29	(1;1;4;1;1)	N total as Kjeldahl N
Phosphate	3.081155	Mg	lognormal: gmean = 3.08 gsigma = 1.29	(1;1;4;1;1)	Phosphate as P

Table A.7: Life cycle inventories for the production of semi-native pastures (inputs) during an average year (averaged from 2014-2017).

Input - Semi-native Pastures					
Flow	Amount	Unit	Uncertainty	Data Quality Score	Notes
diesel	18514.15	kg	lognormal: gmean = 18514.15 gsigma = 1.56	(1;1;1;4;4)	Diesel
Electricity, AC, 2300-7650 V	21662.657	MJ	lognormal: gmean = 21662.657 gsigma = 1.2	(1;1;2;3;1)	Solar Panel
lubricating oil	139.29	kg	lognormal: gmean = 139.29 gsigma = 1.32	(2;1;3;3;3)	Oil for machinery
Occupation, pasture, man made, extensive	2281	ha*a	lognormal: gmean = 2281 gsigma = 1.15	(1;1;1;1;1)	Total area
pesticide, unspecified	116.6	kg	lognormal: gmean = 116.6 gsigma = 1.54	(1;1;1;3;4)	Triclopyr
petrol, unleaded, burned in machinery	1393209.075	MJ	lognormal: gmean = 1393209.08 gsigma = 1.58	(1;1;1;5;4)	Gasoline

Table A.8: Life cycle inventories for the production of semi-native pastures (outputs) during an average year (averaged from 2014-2017).

Outputs - Semi-native Pastures					
Flow	Amount	Unit	Uncertainty	Data Quality Score	Notes
Ammonium/ ammonia, as N	0.6760884	Mg	lognormal: gmean = 0.68 gsigma = 1.29	(1;1;4;1;1)	N total as NH4
Carbon dioxide	-3877.7	Mg	lognormal: gmean = -3877.7 gsigma = 1.1	(1;1;1;1;1)	Net sink
Dinitrogen monoxide	11.405	Mg	lognormal: gmean = 11.41 gsigma = 1.53	(3;1;1;4;2)	Lu et al. 2020
Semi-native; grazed forage, DM, BIR	10264.5	Mg	lognormal: gmean = 10264.5 gsigma = 1.03		Forage yield
Methane, from soil or biomass stock	117.4715	Mg	lognormal: gmean = 117.47 gsigma = 1.1	(1;1;1;1;1)	Ecosystem
Nitrate	0.1452997	Mg	lognormal: gmean = 0.15 gsigma = 1.29	(1;1;4;1;1)	N total as NO3
Nitrogen, total	7.871731	Mg	lognormal: gmean = 7.87 gsigma = 1.29	(1;1;4;1;1)	N total as N
Nitrogenous Matter, Kjeldahl, as N	7.8347788	Mg	lognormal: gmean = 7.83 gsigma = 1.29	(1;1;4;1;1)	N total as Kjeldahl N
Phosphate	1.2848873	Mg	lognormal: gmean = 1.28 gsigma = 1.29	(1;1;4;1;1)	Phosphate as P

Table A.9: A summary of semi-native pastures' inputs and emissions during an average year at BIR.

INPUT DATA	INPUT DESCRIPTION	VALUE	UNIT (per year)	NOTES
Land Use	Total area	2281	ha	
Forage yield	Forage yield	4.5	Mg DM/ha	2017 NIFA data
Seed	Seeding rate	0	kg seed/ha	No seeding in SN pastures
Fertilizers	N (as N), urea	0	kg/ha	No fertilization in SN pastures
	P (as P2O5)	0	kg/ha	No fertilization in SN pastures
	K (as KCl)	0	kg/ha	No fertilization in SN pastures
	Urea ammonium nitrate (as N)	0	kg/ha	No fertilization in SN pastures
Herbicides	glyphosate	0	kg a.i./ha	
	triclopyr	2.2	kg a.i./ha	53 ha sprayed
	fluoroxypyr	0	kg a.i./ha	
Utilities	Diesel and gasoline	28.6	l/ha	Includes fuel for burns
	Lubricant	0.069	l/ha	Estimate from diesel and gasoline quantities
Energy	Renewables bio-based	9.50	MJ/ha	Solar panels used to pump water to troughs
EMISSIONS DATA	**DESCRIPTION**	**VALUE**	**UNIT (per year)**	**NOTES**
Emissions to air	Ecosystem CO2	-1700	kg/ha	EC tower data Gomez-Casanovas et al. 2018
	Ecosystem CH4	51.5	kg/ha	EC tower data Gomez-Casanovas et al. 2018
	Soil N2O	0.0036938	kg/ha/min	Sparks data (2016-2017)
	Soil CO2	587.26	kg/ha/min	Sparks data (2016-2017)
Emissions to water	Phosphate as P	563.3	g/ha	Estimated from BIR CDEM watershed in 2008-2010
	N total as NH4	296.41	g/ha	Estimated from BIR CDEM watershed 2008-10
	N total as NO3	63.69	g/ha	Estimated from BIR CDEM watershed 2008-10
	N total as Kjeldahl N	3434.81	g/ha	Estimated from BIR CDEM watershed 2008-10
	N as total N	3451.39	g/ha	Estimated from BIR CDEM watershed 2008-10

Table A.10: A summary of improved pastures' inputs and emissions during an average year at BIR.

INPUT DATA	DESCRIPTION	VALUE	UNIT (per year)	NOTES
Land Use	Total area	1925	ha	
Forage yield	Forage yield	4.8	Mg DM/ha	2017 NIFA data
Seed	Seeding rate	2.7	kg seed/ha	Seeded in 2014 (259.8 ha) and in 2017 (257 ha)
Fertilizers	N (as N), urea	31	kg/ha	N applied to 909 ha
	P (as P2O5)	28	kg/ha	P applied to 312 ha
	K (as KCl)	50	kg/ha	K applied to 909 ha
	Urea ammonium nitrate (as N)	28	kg/ha	32-0-0 applied to 738.55 ha
Herbicides	glyphosate	0.4	kg a.i./ha	68 ha sprayed
	triclopyr	1.7	kg a.i./ha	16 ha sprayed
	fluoroxypyr	0.6	kg a.i./ha	16 ha sprayed
Utilities	Diesel and gasoline	28.6	l/ha	Includes fuel for burns
	Lubricant	0.069	l/ha	Estimated from diesel and gasoline quantities
Energy	Renewables bio-based	23.10	MJ/ha	Solar panels used to pump water to troughs
EMISSIONS DATA	**DESCRIPTION**	**VALUE**	**UNIT (per year)**	**NOTES**
Emissions to air	Ecosystem CO2	-1190	kg/ha	EC tower data Gomez-Casanovas et al. 2018
	Ecosystem CH4	234.5	kg/ha	EC tower data Gomez-Casanovas et al. 2018
	Soil N2O	0.0041584	kg/ha/min	Sparks data (2016-2017)
	Soil CO2	706.36	kg/ha/min	Sparks data (2016-2017)
Emissions to water	Phosphate as P	1600.6	g/ha	BIR 20, 29, 30,35 watersheds in 2008-2010
	N total as NH4	1194	g/ha	BIR 20, 29, 30,35 watersheds in 2008-2010
	N total as NO3	175	g/ha	BIR 20, 29, 30,35 watersheds in 2008-2010
	N total as Kjeldahl N	6584.86	g/ha	BIR 20, 29, 30,35 watersheds in 2008-2010

Table A.11: A summary of animal inputs and emissions during an average year at BIR.

INPUT DATA	DESCRIPTION	VALUE	UNIT (per year)	NOTES
Supplementary Feed	Hay	143.5	t	
	Range cubes 18-25%	156.65	t	Cottonseed Meal as proxy based on feed tag info
	Protein tubs	3.35	t	Cottonseed Meal as proxy based on feed tag info
	Molasses	315.14	t	
	Salt blocks	4.04	t	
	Ration, assorted feeds	239.70	t	Cracked corn as proxy based on feed tag info
	Mineral, assorted brands	115.711	t	
Antiparastics	Benzimidazoles	55.9483	kg	
	Piperonyl butoxide	15.9	kg	
Utilities	Drinking water	68923888	l	Calculated from #cow-days * 15 gal/cow/day
	Electricity from network	77780.5	kWh	Data from Glades Electric
	Diesel	11015.3	gal	
	Gasoline	7847.785	gal	
	Lubricant	18.834684	l	
EMISSIONS DATA	**DESCRIPTION**	**TOTAL ANNUAL**	**UNIT**	**NOTES**
Emissions to air (Animal emissions)	Ammonia	19371	kg	Estimated using IFSM 4.6 - Grazing
	Hydrogen Sulfide	17	kg	Estimated using IFSM 4.6 - Grazing
	Ozone Forming VOC	3222	kg	Estimated using IFSM 4.6 - Field/Grazing
	Methane	225474	kg	Estimated using IFSM 4.6 - Animal
	Nitrous Oxide	406	kg	Estimated using IFSM 4.6 - Animal
Emissions to air (Feed emissions)	Nitrous Oxide	756	kg	Estimated using IFSM 4.6 - Indirect Sources

Table A.12: A summary of vehicles and machinery used during an average year at BIR.

Vehicles & Machinery	# of vehicles	Type	Purpose	Fuel Type	Distance driven	Unit
Tractor	1	7610	Cattle	Diesel	1609	km/yr
Tractor	1	5210	Cattle	Diesel	1609	km/yr
Tractor	1	7110	Cattle	Diesel	1609	km/yr
Tractor	1	9310	Cattle	Diesel	1609	km/yr
Tractor	1	6115D	Cattle	Diesel	1609	km/yr
Truck	1	Silver Tundra	All purpose	Gas	28962	km/yr
Truck	1	Blue F150	Cattle	Gas	16090	km/yr
Truck	1	Silver F150	90% cattle	Gas	16090	km/yr
Truck	1	White F250	100% cattle	Diesel	4827	km/yr
Truck	1	White F250	90% cattle	Gas	17699	km/yr
Truck	1	White F450	90% cattle	Diesel	804.5	km/yr
Truck	1	Silverado 2500HD	90% cattle	Diesel	4827	km/yr
Truck	1	F350	90% cattle	Diesel	12872	km/yr
4-wheelers	6	Yahmaha/kabota	All purpose	Gas	4827	km/yr
Swamp Buggy	2	-	Cattle	Gas	643.6	km/yr
Swamp Buggy	1	-	Tours	Gas	1930.8	km/yr
Front End Loader	2	Caterpillar IT28F, 930	Cattle	Diesel	1609	km/yr
Dump Truck	1	Ford 8000 Dump Truck	Cattle	Diesel	1287.2	km/yr
Military Truck	1	M35	Cattle	Diesel	2413.5	km/yr
Military Truck	1	M925	Cattle	Diesel	4827	km/yr
TrackHoe	1	Kobelco	Cattle	Diesel	321.8	km/yr
Grader	1	John Deere 570A	Cattle	Diesel	160.9	km/yr

Table A.13: A summary of IFSM 4.6 pasture quality values.

Pasture Quality	Early Spring	Late Spring	Summer	Early fall	Late fall and winter
Crude Protein (% DM)	11.50	10.80	9.80	9.50	9.50
Degradable Protein (% CP)	61.00	61.00	61.00	61.00	61.00
Acid Detergent Insoluble Protein (% CP)	2.00	2.00	2.00	2.00	2.00
Net energy of maintenance (Mcal/kg DM)	1.22	1.10	1.08	1.15	1.09
Total Digestible Nutrients (% DM)	63.00	56.00	55.00	56.00	58.00
Neutral Detergent Fiber (% DM)	67.00	70.00	73.00	70.00	73.00
Phosphorus (% DM)	0.30	0.30	0.30	0.30	0.30
Potassium (% DM)	2.20	2.20	2.20	2.20	2.20

Table A.14: A summary of IFSM 4.6 feed characteristics.

Feed Characteristics	CSM	CG
Crude Protein (% DM)	43.60	10.00
Degradable Protein (% CP)	58.00	48.00
Acid Detergent Insoluble Protein (% CP)	3.00	8.00
Net energy of maintenance (Mcal/kg DM)	1.81	2.09
Total Digestible Nutrients (% DM)	78.00	85.00
Neutral Detergent Fiber (% DM)	27.00	10.00
Phosphorus (% DM)	1.25	0.29
Potassium (% DM)	1.35	0.37

Table A.15: A summary of IFSM 4.6 land and soil information and grazing management.

Land and Soil Information	
Total grass area (grazing)	4206 ha
Stand life including seeding	10 yrs
Yield adjustment factor	100%
Max annual irrigation	0 cm
Initial sward dry matter	112 kg/ha
Initial sward composition	95% Warm-season grass 5% legume
Predominant Soils	Medium loamy sand
Farm topography	Nearly level (A, 0-3% slope)
Farm soil phosphorus level	Optimum (30-50 ppm)

Grazing Management	
Grazed forage yield adjustment factor	48%
Pasture utilization efficiency	68%
Grazing period	12 months

Table A.16: A summary of IFSM 4.6 breed characteristics and herd information.

Breed & Herd Information	
Breed	Brangus
Number of cows and bulls	3142 head
Number of other stock	936 head
First lactation cows	15%
Calving month	March
Age at weaning	6 months

Breed Characteristics	
Mature cow shrunk body weight	450 kg
Peak milk yield	8 kg/day
Calf birth weight	33 kg
Genetic influence on fiber intake capacity	1.12
Genetic factor for carcass leanness	7.00
Grazed animal groups	Cows, Calves, Replacement heifers
Grain fed animal groups	Cows

Table A.17: A summary of IFSM 4.6 ration constituents.

Ration Constituents	
Protein feeding level (NRC recommendation)	97%
Phosphorus feeding level (NRC recommendation)	100%
Relative forage to grain ratio	High
Crude protein supplement	Cottonseed meal (CSM)
Undegradable protein supplement	None
Energy supplement	Corn grain (CG)
Grain feeding method	Hand feeding
Silage feeding method	No silage fed
Hay feeding method	Self-fed

Table A.18: Cattle and Calves Average Price Received by Farmers – U.S.: 2014-2017 (dollars per cwt)

Beef Cattle	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Avg
2014	138	144	148	148	146	147	156	158	157	161	167	164	152
2015	164	159	160	162	160	155	149	148	139	128	129	122	147
2016	130	132	135	131	128	125	119	117	108	101	104	111	119
2017	117	119	125	128	136	132	120	114	105	109	119	118	120
Steers and Heifers													
2014	140	145	150	150	147	148	157	159	158	163	169	166	153
2015	166	161	162	164	161	156	150	149	140	129	131	123	148
2016	132	134	137	133	129	127	120	118	109	102	106	113	121
2017	119	121	127	130	138	133	121	115	107	111	121	120	122
Cows													
2014	88.3	95.4	102	103	104	106	115	121	118	116	115	115	107
2015	112	110	114	113	114	113	113	110	104	89.5	82	74.8	103
2016	74.2	77.5	80	81.5	79.6	80.9	81.5	80.6	74.5	65.4	61.9	61.1	74.3
2017	64	64.9	69.5	72.2	73.3	76.5	77.3	76.3	69.9	65.4	63.4	62	69.1
Calves													
2014	208	209	216	222	229	249	257	271	279	307	305	303	261
2015	288	277	290	288	288	292	275	273	241	234	217	193	247
2016	193	192	197	180	170	157	145	153	139	134	143	148	158
2017	152	151	159	164	171	164	157	163	173	177	177	174	168
Source: Adapted from USDA NASS "2018 Florida Livestock, Dairy, and Poultry Summary" .													

	gross calorific value MJ/kg	net calorific value MJ/kg	Density kg/l
agricultural biogas	23.7	21.4	0.00113
crude oil	45.8	43.2	0.86
Diesel	45.4	42.8	0.84
gasoline	45.1	42.5	0.75
hard coal	30.4	28.9	
hard coal, briquette	32.4	31.4	
hard coal, coke	28.9	28.6	
heavy fuel oil	43.7	41.2	1.0
kerosene	45.6	43.0	0.795
light fuel oil	45.2	42.6	0.86
lignite, briquette	20.9	19.5	
lignite, hard	17.8	16.8	
lignite, soft	9.5	8.4	

Figure A.1: ecoinvent Table 5.1 - Default values for gross and net calorific values and density of some common fuels.

Indicator Score	1	2	3	4	5 (default)
Reliability	Verified data based on measurements	Representative data from all sites relevant for the market considered, over and adequate period to even out normal fluctuations	Less than 3 years of difference to the time period of the data set	Data from area under study	Data from enterprises, processes and materials under study
Completeness	Verified data partly based on assumptions or non-verified data based on measurements	Representative data from > 50% of the sites relevant for the market considered, over an adequate period to even out normal fluctuations	Less than 6 years of difference to the time period of the data set	Average data from larger area in which the area under study is included	Data from processes and materials under study (i.e. identical technology) but from different enterprises
Temporal correlation	Non-verified data partly based on qualified estimates	Representative data from only some sites (<< 50%) relevant for the market considered or > 50% of sites but from shorter periods	Less than 10 years of difference to the time period of the data set	Data from area with similar production conditions	Data from processes and materials under study but from different technology
Geographical correlation	Qualified estimate (e.g. by industrial expert)	Representative data from only one site relevant for the market considered or some sites but from shorter	Less than 15 years of difference to the time period of	Data from area with slightly similar production conditions	Data on related processes or materials
Further technological correlation	Non-qualified estimates	Representativeness unknown or data from a small number of sites and from shorter periods	Age of data unknown or more than 15 years of difference to the time period of	Data from unknown or distinctly different area (North America instead of Middle East, OECD-Europe instead of	Data on related processes on laboratory scale or from different technology

Figure A.2: Pedigree Matrix with data quality indicators and scores.