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Review

Consequential Life Cycle Assessment of Grain and Oilseed Crops: Review and Recommendations

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Abstract: The field crop industry in Canada is a source of both significant economic benefits and environmental impacts. Environmental impacts include land and energy use, as well as greenhouse gas (GHG) and other emissions. Impacts also accrue upstream of the field in the product supply chain, from the production of such inputs as fertilizers and pesticides. There are currently two types of environmental life cycle assessment (LCA)—attributional LCA (ALCA) and consequential LCA (CLCA)—that may be used to study the life cycle impacts of products such as field crops. ALCA is a retrospective methodology that presents a snapshot of average, "status quo" conditions. CLCA is a prospective methodology that presents the potential implications of changes in a product system, including any associated market-mediated changes in supply or demand in other product systems. Thus, CLCAs can be used to assess large-scale changes in the field crop industry, including its relationship to other sectors and processes, such as the production of biofuel or of food for both human and animal consumption. The aim of this paper is to review and curate the knowledge derived through published CLCA studies that assessed the impacts of changes to field crop production systems on the life cycle resource use and emissions associated with the agricultural products, with a focus on their relevance to temperate climate conditions. The current study also highlights how previous studies, including ALCAs and farm management recommendations, can be used to inform the changes that should be studied using CLCA. The main challenges to conducting CLCAs include identifying the system boundaries, marginal products and processes that would be impacted by changes to field crop production. Marginal markets and product systems to include can be determined using economic equilibrium models, or information from local experts and industry reports. In order to conduct ISO-compliant CLCAs, it is necessary to include multiple relevant environmental impact categories, and to perform robust data quality and uncertainty analyses.

Keywords: consequential life cycle assessment; field crops; temperate climate; ISO 14040/44

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Citation: Bamber, N.; Turner, I.; Dutta, B.; Heidari, M.D.; Pelletier, N. Consequential Life Cycle Assessment of Grain and Oilseed Crops: Review and Recommendations. *Sustainability* **2023**, *15*, 6201. https://doi.org/10.3390/su15076201

Academic Editors: Ramjee Ghimire and Lila Kumar Khatiwada

Received: 8 February 2023
Revised: 20 March 2023
Accepted: 29 March 2023
Published: 4 April 2023

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1. Introduction

Canada is a major producer and exporter of grains and oilseed crops [1]. Grains and oilseeds such as wheat, canola, corn, barley, and soy are among the most important staple crops grown in Canada. Estimates for the total yield of Canadian field crops in the 2020–2021 crop year are 99.75 million tonnes of production, with grains and oilseeds accounting for 91% of this production [2]. This is an increase from the 2019–2020 crop year production of 93.29 million tonnes, with associated increases in yield and hectares cultivated for both grains and oilseeds, along with pulses and other crops [3]. Together, these estimates fall in line with a general trend of growth in the Canadian field crop industry. Taken together, the GDP in the Canadian agriculture and agri-food industries increased at a rate almost double that of total Canadian GDP for the years 2012–2016 [4].

While the economic benefits of the Canadian field crop industry are clear, there are also potential environmental impacts associated with field crop production (see, for example, [5–7]). Field crop production contributes significantly to land and energy use,

while also producing greenhouse gasses (GHGs) and other emissions at the field level, including carbon dioxide (CO_2) and nitrous oxide (N_2O) [8–11]. Rising N_2O emissions are particularly concerning, due to their high radiative forcing effect relative to CO_2 [12,13]. In addition, these direct environmental impacts of field crop production may be further exacerbated by resource use and emissions embodied in the products originating upstream of the field in the product supply chain, such as those associated with production of the fertilizers and pesticides applied to fields [14]. For this reason, assessing and seeking to mitigate the environmental impacts of field crop production requires a holistic, systems-level perspective supported by analytical tools of commensurate scope [15]. Life cycle assessment (LCA) has become the leading methodological approach based on life-cycle thinking. LCA supports systematic analysis of all aspects of the life cycle of a product (i.e., including raw material extraction, processing, transportation, use, and end-of-life phases) in order to quantify the cumulative resource demands and emissions over its entire life cycle (ISO 14044 [16]).

There are currently two principal types of environmental LCA: attributional LCA (ALCA) and consequential LCA (CLCA). Use of one or the other of these two types is largely dependent on the goals of a given study. ALCA is a retrospective methodology that aims to present a snapshot of average, "status quo" conditions along a product supply chain at a specific point in time. ALCA is often used to provide a baseline assessment of the impacts of a product and to identify "hot-spots", areas along the supply chain with large contributions to impacts, or areas for improvement [17]. On the other hand, CLCA is a prospective methodology for evaluating the environmental implications of potential changes in a product system of interest, including any associated market-mediated product substitution effects that may arise that impact other related product systems [18]. This fundamental difference in scope between ALCA and CLCA studies necessitates different "system boundaries", i.e., the portion of the product supply chain(s) covered by the LCA study for the analysis [19]. Attributional LCAs include all direct inputs and outputs (and their upstream/downstream life cycles) in the product system for the product being studied. In contrast, the system boundaries in CLCAs are expanded to include the changes in inputs and outputs in other product systems that would be affected given a change in the product system being studied via changes in supply and demand [17]. It is through the inclusion of these market-mediated substitution effects that CLCA is able to assess potential additional impacts not normally considered in an ALCA. For example, in their study investigating the impacts of camelina-derived jet fuel production in Canada, Li and Mupondwa [20] were able to take into account potential reductions in impacts from decreased production of conventional jet fuel as a consequence of increased camelina cultivation and processing for biofuels (which is assumed to substitute for jet fuel in the marketplace); in an ALCA study, these associated reductions would not be captured.

In the context of Canadian field crop production, CLCAs are well suited (and, indeed, even necessary) for accurate assessment of any foreseen large-scale changes in the industry, due to the field crop industry's linkages with other sectors and processes, including the production of biofuel and food for both human and animal consumption. For example, if more of a grain was used for biofuel, then less may be available for animal feed. In this case, increased production of some other feed product would be necessary to keep the supply of animal feed constant. Different types of feed crops and different production methods require differing amounts of land to produce equivalent feed outputs. Direct and indirect land use changes are also included in CLCA [21].

While the utility of CLCA in bringing additional nuance to environmental impact assessment is clear, its application remains quite limited to date. A recent review of 2687 LCA studies published over the past five years indicated that only 6% were CLCAs while 94% were ALCAs [22]. The use of ALCA (or related assessments, such as carbon footprints) is quite common globally, including for the assessment of grain and oilseed crops (see, for example, [23–25]). Attributional LCA and carbon footprint studies have been reported for a wide variety of Canadian field crop products in recent years (for a

review of these studies, see [26]). For example, the Canadian Roundtable for Sustainable Crops has produced a series of reports estimating the carbon footprints of ten major field crops in Canada on a regional basis (e.g., [27]). For these and other attributional studies, the system boundaries included the direct inputs and emissions of agricultural operations, as well as those associated with the upstream activities that provide inputs to grain crop production, such as fuels, fertilizers, and plant protection products. These studies focused on estimating the impacts of current, or past, farm practices. While this is a very useful type of assessment, it does not enable the provision of recommendations with respect to alternative practices to adopt at a broad scale in the future in order to reduce impacts. Instead, such questions must be answered through application of CLCA. Indeed, there are many potential strategies to reduce GHG emissions and other impacts in grain crop production, including crop rotation [28], reducing nitrogen fertilizer inputs [29], adding biochar as a soil amendment [30], etc. Tillage practices and removal of residues, along with land use change associated with conversion of land to the production of energy crops also influence the SOC stock. CLCA is therefore a useful tool in assessing the potential changes in cumulative impacts across multiple linked industries that may result from the widespread uptake of these different management strategies.

On this basis, the aim of this paper is to review CLCA studies that assess the impacts of farm practices on life cycle GHG emissions, and other environmental resource use and emissions, associated with agricultural crop production. The focus is on cereals, oilseeds and specialty crops produced in temperate climate regions, with a focus on Canada, specifically. The following questions are addressed:

(1) For what agricultural crops and associated farm practices have consequential LCA studies been reported to date? What proportion have specifically addressed field crops, and which among these are specific to Canada?

(2) Among these studies, how were the system boundaries defined, including the definition of assumed market-mediated substitutions? Which production systems were included? What were the affected technologies? What modelling approaches were employed, and what commonalities and differences can be observed across studies?

(3) What were the reported influences of specific farm practices on estimated life cycle impacts (e.g., GHG emissions) for the field crops considered, as modelled using CLCA? Can any recommendations be made for sustainability best practices in field crop production on the basis of these studies?

(4) What research gaps can be identified with respect to CLCA research of field crops, in particular for Canadian conditions? Are there obvious opportunities to build on prior, related research—for example, ALCA or other studies that investigate the GHG mitigation potential of alternative technologies and management strategies for grain production?

2. Materials and Methods

The Web of Science Core Collection was used to search for peer-reviewed journal articles using the search terms ("consequential life cycle assessment" OR "consequential LCA") AND ("crop" OR "grain" OR "cereal" OR "oilseed" OR "wheat" OR "canola" OR "corn" OR "maize" OR "barley" OR "soy") searched in All Fields, in order to capture any mention of these topics in the keywords, titles, abstracts, or main text. This search was performed using the University of British Columbia library system; therefore journal articles were only included in the search if they were open access and/or if the University of British Columbia provided access to them. Relevant industry and government websites such as Fertilizer Canada and Agriculture and Agri-Food Canada were also searched using the same keywords. In addition, the advanced search function of the Google search engine was used (with the same search terms) to identify any remaining non-academic LCA studies. Studies were selected for detailed review if they (1) reported a consequential LCA (or carbon footprint) of an agricultural crop, and (2) were published from 2010 to 2021. This timeframe was selected because CLCA modelling is relatively new in terms of widespread adoption as an LCA methodology, as well as to ensure the relevance of the studies to

contemporary conditions. From this pool of articles, studies were selected for inclusion if they (1) performed a consequential LCA of an agricultural crop, and (2) reported the life cycle emissions (e.g., CO_2 equivalent) associated with a change in the supply chain of that crop.

For each CLCA study, information was extracted and tabulated regarding the type of agricultural crop, farm practices modelled, and geographical area represented. The total number of CLCAs for each crop type were calculated, as well as the number of Canadian-specific crop CLCAs for each crop type. The types of farm practices (i.e., prevalent and crop/region-specific strategies and technologies for cultivation, seeding, pesticide application, fertilizing, harvesting, and storage) were tabulated in the same way. Information was also extracted from each study regarding the system boundaries, affected product systems, marginal data used (processes assumed to change as a result of the intervention assessed), and use of different modelling approaches (e.g., partial equilibrium, general equilibrium models) to determine market-mediated substitutions. Any reported information was tabulated on the impacts of these methodological choices on the LCIA results of each study.

The impact assessment results of each CLCA were summarized in tables (one for each relevant impact category), highlighting the impact of the farm practice assessed (fertilizing strategies, tillage operation, using of precision agriculture, etc.) on the impacts of the change in management practices for each crop in each study. These results were grouped by intervention type (farm practices, different uses of crops, etc.), and conclusions were drawn about the environmental benefits of these interventions.

Based on the information resulting from this review, data gaps were identified with respect to current CLCAs of field crops, including selection of the system boundaries, identification of marginal data, etc. Major grain/oilseed crops or associated farm practices with few or no CLCA studies were also identified. Similarly, research gaps specific to CLCA studies of Canadian field crops were identified.

3. Results and Discussion

3.1. Representation of Crops and Geographical Regions in CLCA Studies

A total of 34 CLCA studies from the past 12 years were identified that met the criteria of (1) being a consequential LCA study of an agricultural crop, and (2) reporting the life cycle emissions (e.g., CO_2 equivalent) associated with a change in the supply chain of that crop (Table 1). However, of all the studies identified, only one [20] addressed a Canadian crop, in this case the use of camelina oil as biodiesel for jet fuel. There were three studies from the United States that may be relevant to Canadian conditions. These studies assessed a corn bioenergy policy change [31], the integration of grass-clover and livestock production with a biorefinery [32], and the inoculation of corn with the soil fungus *Penicillium bilaiae* [33]. The majority of studies (22) focused on European crop production systems. Of these European studies, 17 assessed the use of crops (miscanthus, maize, grass, canola, wheat, barley, beet, beans, oats, rye, willow, ryegrass, oilseed radish, alfalfa, flax, and sunflower) for bioenergy production (Table 1). Other than bioenergy production, the remaining European studies variously addressed the use of peas for gin [34], the introduction of genetically modified soy [35], the planting of willows as riparian buffers on cropland [36], an increase in demand for bananas [37], and the use of flax for polymers [38].

Two studies were specific to Asia, respectively addressing flax production for use in polymer formation [38] and cassava production for bioenergy [39]. Four studies focused on South American systems, specifically sorghum production for bioenergy [40], an increase in grape production for pisco [41], the use of bioethanol residue as fertilizer in sugarcane production [42], and the introduction of yield-enhancing inoculants to soybean production systems grown in rotation with corn crops [43]. There was one study that took place in Australia, which assessed the expansion or contraction of Australian cotton production by 50% [44].

There is clearly a large gap in the literature with respect to Canadian-specific CLCAs of field crop production systems. Not only are other countries not representative of Canadian agricultural and economic conditions, but the single location within Canada considered by a crop CLCA study to date is clearly not representative of all of Canada given its heterogeneity; nor is the single crop addressed representative of all crops grown in Canada and their potential market-mediated changes.

A total of 23 different crop types were included in the CLCA studies identified. Wheat, corn, grasses, and soy were the most highly represented crops with 10, 8, 7, and 4 studies, respectively (Table 1). There were three studies for each of beets, canola, and willow crops, and two studies for beans. Camelina, peas, oats, rye, oilseed radish, banana, sunflower, flax, cassava, sorghum, grape, cotton, and sugarcane crops were only assessed in one study each. The prevalence of wheat and corn studies is in line with the prevalence of grain crops grown in Canada, however there is not as extensive a representation of oilseed crops, which are also prevalent in Canada.

Table 1. List of consequential life cycle assessments of agricultural crops from 2010–2021 with the farm practices assessed and geographical location of each study.

Geographical Location	Crop	Farm/Processing Practice or Policy	Citation
Canadian prairies	Camelina oil	Used for biodiesel or jet fuel	Li and Mupondwa 2014 [20]
United States	Corn	Policy change: Renewable Fuel Standard and Volumetric Ethanol Excise Tax Credit	Bento and Klotz 2014 [31]
United States	Corn	Inoculation of corn with the soil fungus *Penicillium bilaiae*	Kløverpris et al. 2020 [33]
United States	Grass-clover	Integrated crop–livestock system with biorefinery	Parajuli et al. 2018 [32]
Europe	Miscanthus, maize, grass	Production of biogas with different crops	Styles et al. 2015a [45]
Europe	Maize, canola, wheat, barley	Production of biogas with different crops	Styles et al. 2015b [46]
Europe	Wheat, barley, beet	Bioenergy or animal feed from different crops	Van Zanten et al. 2014 [47]
United Kingdom	Beet, grass, maize	Anaerobic digestion for bioenergy	Styles et al. 2016a [48]
United Kingdom	Pea	1 L of gin produced from peas instead of wheat	Lienhardt et al. 2019 [34]
England	Corn	Different processing technologies for bioethanol production	Abiola et al. 2010 [49]
Germany	Wheat grains and straw	Different processing technologies for bioethanol production	Buchspies and Kaltschmitt 2018 [50]
Germany	Wheat straw	Change from straw incorporation to bioethanol and biomethane production	Buchspies et al. 2020 [51]
Sweden	Soy	Introducing of genetically modified soy meal for feed	Eriksson et al. 2018 [35]
Sweden	Faba beans	Switch from protein feed to either bioethanol or roughage feed	Karlsson et al. 2015 [52]
Sweden	Beans, oats, canola, wheat, rye	Self-sufficient bioenergy for farms compared to fossil fuel reference	Kimming et al. 2011 [53]

Table 1. *Cont.*

Geographical Location	Crop	Farm/Processing Practice or Policy	Citation
Sweden	Willow	Addition of fertilized and unfertilised willow on riparian buffer strips and drainage filtration zones of cropland	Styles et al. 2016b [36]
Denmark	Ryegrass, willow, miscanthus	Heat and electricity production using different technologies, replacing fossil fuels	Tonini et al. 2012 [54]
Denmark	Willow, miscanthus, ryegrass, sugar beet, maize, wheat, barley	Production of bioelectricity, biomethane, and bioethanol from different crops or residues	Tonini et al. 2016a [55]
Denmark	Wheat, natural grass	Bioethanol and biogas production from different crops and residues	Tonini et al. 2016b [56]
Denmark	Barley, oilseed radish, wheat	Different combinations of crop residue for bioenergy	Kloverpris et al. 2016 [57]
Denmark	Winter wheat straw, alfalfa	Bioenergy produced from standalone wheat, standalone alfalfa, and both integrated	Parajuli et al. 2017 [58]
Denmark	Banana	Increase in demand for bananas	Sacchi 2018 [37]
Switzerland	Canola	Bioenergy production replaces human or animal consumption	Reinhard and Zah 2011 [59]
Switzerland	Residues from cereals	Bioenergy from different crops, residues, and waste based on different policies	Vadenbo et al. 2018 [60]
Belgium	Maize	Bioenergy produced from different crops, residues, and waste	Van Stappen et al. 2016 [61]
Spain	Sunflower and canola	Optimize feedstock combination for biodiesel according to policy objectives to increase 2.58 Mt demand	Escobar et al. 2017 [62]
China and France	Flax	Switch from glass to flax fibres for polymers	Deng and Tian 2015 [38]
Thailand	Cassava	Different ratios of cassava and molasses for bioethanol	Prapaspongsa and Gheewala 2016 [39]
Western and Northern Uruguay	Grain sorghum and sweet sorghum	Introduction in multi-crop system for bioethanol compared to gasoline	Adler et al. 2018 [40]
Peru	Grapes	Increase in pisco demand	Larrea-Gallegos et al. 2018 [41]
Brazil	Sugarcane	Residues from bioethanol replace chemical fertilizer for sugarcane production	Moore et al. 2017 [42]
Argentina	Soybeans	Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Mendoza Beltran et al. 2021 [43]
Australia	Cotton	Expansion or contraction of Australian cotton production by 50%	Nguyen et al. 2021 [44]

3.2. Definition of System Boundaries and Marginal Technologies in Crop CLCA Studies

One of the most important methodological aspects of consequential life cycle modelling is identifying the product systems to include in the study. As defined by Weidema [17], a consequential LCA should include all products (and associated flows and environmental impacts) that would be affected by the change being assessed (e.g., the use of a crop for bioenergy instead of animal feed). This is an important choice to make, since different product systems have different impacts, thus influencing the overall impacts and conclusions of the CLCA study. The technologies that would change as a result of a change in supply or demand of a product or service are called the marginal technologies in CLCA modelling [17]. There are several methods that have been employed to identify the marginal technologies in a CLCA study, including economic models of markets, and using regionally specific industry data.

Since most of the crop CLCA studies assessed the use of crops for bioenergy, most studies included the product systems for the agricultural cultivation of the specific crop assessed, as well as the production of bioenergy from that crop (Table 2). Of the 34 crop CLCA studies identified, 23 assessed crop use in some form of bioenergy (electricity, heat, or fuel), and included the product system(s) for the bioenergy produced in addition to the crop cultivation system(s). In general, this increase in energy from crop biomass would replace energy from conventional sources (generally fossil fuels). Many studies specified what conventional energy source was assumed to be replaced, and these included gasoline, natural gas, diesel, jet fuel, fossil fuels in general, and the local electricity grid mix. Co-products of bioenergy production can also be returned to the field as fertilizer, thus substituting synthetic fertilizer inputs to crop production. These product systems were included in seven CLCA studies of crops used for bioenergy as well as one of peas used for gin, in which the co-products of distillation could similarly be used for fertilizer. These products may also be sources of soil carbon, which has benefits in terms of reducing atmospheric CO_2, as well as improving soil fertility and yields.

Crop use for animal feed was the second most commonly studied change (four studies). In these CLCA models, the product systems for animal feed from the studied crops, and the displaced marginal animal feed product systems, were included in addition to the cultivation of the crop itself (Table 2). Even in some studies that did not focus on a change in crop use for animal feed, these product systems were included since they would be impacted by a change in use or production of the crop being assessed, highlighting the interconnection of crop production, animal feed, bioenergy, and agri-food product systems. The most common marginal animal feed products were barley, soy, and maize (Table 2).

Common modelling approaches to identify the marginal technologies and products to be included in the CLCA models included numerical economic models, or more simply designing likely scenarios based on published literature and reports on country or product specific economic trends (Table 2). The economic model types used were general equilibrium models, which take into account an entire market [31], and partial equilibrium models, which only account for some aspects of a market [62]. Buchspies and Kaltschmitt [50] indicated that they used a deterministic model based on legislative frameworks, trade statistics, market environments, and past developments. In addition to economic models, economic information on cost, production, and trade dynamics (sourced from peer-reviewed and grey literature) were used to determine the marginal markets by 10 studies (Table 2).

Of the studies that used economic data to determine the marginal markets for the use of crops for bioenergy, the majority found variable increases and decreases in overall GHG emissions for different scenarios modelled [31,50,51,54,59] (Table 3). Two studies found an overall increase in GHG emissions with the use of crops for bioenergy [39,62], and one found an overall decrease [49]. See Sections 3.3 and 3.4 for a discussion of all impact assessment results.

Six studies used published literature on similar systems to inform the choice of marginal technologies (Table 2). Of the studies, three found either an increase or decrease in GHG emissions depending on scenarios [40,55,56], three found an overall in-

crease [20,32,52], and one found an overall decrease [47] (Table 3). Three studies used assumptions about land use and availability to define the system boundaries and inform the choice of marginal substitutions (Table 2). Four studies did not indicate how marginal products and technologies were identified, but did list processes that were excluded from the product system, namely co-products that were considered waste [42], manufacturing of capital goods and infrastructure that had been shown by previous literature not to influence LCA results [34,53], and domestic production of dedicated energy crops and imported liquid biofuels due to policy objectives to avoid these products [60].

The rest of the studies (six studies) either did not give an indication of how marginal processes and system boundaries were defined, or simply stated that the system boundaries included all processes that were affected by the change assessed, which is in the definition of a CLCA [17]. Of the studies that did not indicate any method of identifying marginal technologies for changes in bioenergy production, four found a net reduction in GHG emissions with the use of crops for bioenergy [42,48,53,60], three found either an increase or reduction [45,46,61], and one found an overall increase [63].

In general, bioenergy studies represented the majority of identified CLCA studies, and employed each of the methods described above for identifying marginal substitutions. There were no clear trends with any of the methods of identifying marginal substitutions in terms of the modelled impact mitigation potential of using crops for bioenergy. With economic modelling, or marginal substitutions identified using the literature, the results were mostly inconclusive, though often indicating a slight increase in GHG emissions when using the crops for bioenergy. Of the other studies that did not indicate a method for identification of marginal substitutions, there was a slight trend toward decreased GHG emissions from crop use in bioenergy, although many studies were also inconclusive.

Table 2. Definition of system boundaries, market-mediated substitutions, and product systems included in consequential life cycle assessments of agricultural crops from 2010–2021.

Product Systems Studied	Definition of System Boundaries and Market-Mediated Substitutions	Assumed Substitutions	Citation
Economic model used to define marginal markets			
Corn, bioethanol	Economic framework from policy change. Multi-market general equilibrium model: numerical static model of the economies of the US and the rest of the world	Other crops, gasoline	Bento and Klotz 2014 [31]
Sunflower, canola, sugar beet, biodiesel	Land use assumed to be the same, so only crop rotations change, increases from intensification. Partial equilibrium model, demand for non-energy crops assumed stable	Domestic vegetable oil production, diesel	Escobar et al. 2017 [62]
Wheat, biofuels	Inputs, outputs and substitutions based on literature and industry reports. Deterministic model for the identification of marginal suppliers based on legislative frameworks, trade statistics, market environments and past developments	Fossil fuels, electricity, animal feed, vegetable oil	Buchspies and Kaltschmitt 2018 [50]
Market trends used to define marginal markets			
Molasses, cassava, bioethanol, use in vehicles	Based on which product is the determining or dependent co-product—sugarcane not included because not driven by demand for molasses. Marginal substitutions based on cost, trading and production conditions, countries imports/exports—electricity delimited within regional boundaries, agricultural products traded internationally—capacity for increased production, cheapest sources, countries with largest increasing production trend	Fossil fuel, barley	Prapaspongsa and Gheewala 2016 [39]

Table 2. *Cont.*

Product Systems Studied	Definition of System Boundaries and Market-Mediated Substitutions	Assumed Substitutions	Citation
Market trends used to define marginal markets			
Flax, composite manufacturing	System expansion to include all co-products. Marginal markets based on global supply and production trends	Glass composite manufacturing	Deng and Tian 2015 [38]
Canola, barley, biofuel,	Economic value criteria, constrained or linked markets	Soy, sunflower animal feed, fuel	Reinhard and Zah 2011 [59]
Soy	Proportions of types of soy based on publications of country market	Palm oil, canola	Eriksson et al. 2018 [35]
Banana	Market trends of countries and specific products	Agricultural and food products from a trade matrix	Sacchi 2018 [37]
Wheat	Substitution based on observed market trends in the previous years	Natural gas, mineral fertilizer, gasoline, animal feed (corn)	Buchspies et al. 2020 [51]
Soy	Linear regression modeling of FAOSTAT data to determine countries with the largest recent increases in production. Based on the assumption that those markets with the largest recent increases in production would also be the first to decrease production levels in response to changes in markets	Market mix of 52% US soybean production and 48% Brazilian soybean production	Beltran et al. 2021 [43]
Cotton	Marginal market representing a global market mix of different displaced/expanded products. Commodity prices are assumed to be inelastic in response to production changes	Beef, rice, wheat, sorghum	Nguyen et al. 2021 [44]
Ryegrass, willow, miscanthus, bioenergy,	Substitutions based on the literature, rebound effects based on changes in market prices	Fertilizer, electricity, heat, barley	Tonini et al. 2012 [54]
Bean, bioenergy, animal feed	Marginal fuel technologies from literature, market information on products/countries. Excess arable land assumed to be available. Marginal protein feed assumed to be soymeal based on country with largest increase in exports. Marginal effects of changes in demand for feed grain from Schmidt (2008) [64]	Soymeal	Karlsson et al. 2015 [52]
Literature used to define marginal markets			
Bioethanol	System expansion for co-products, included use phase of bioethanol, experts define crop rotations (not economic models), bioenergy crops did not affect pasture—only prices and location influenced rotations. Marginal products based on evidence from similar systems	Soy, gasoline	Adler et al. 2018 [40]
Wheat, grass, Brewer's grain, beet, potato, whey, bioenergy	Products defined as waste, co-products or products—to determine if substituted by another product or decay. Substitutions based on demand trends/projections in literature	Waste disposal, animal feed (maize, soy), fossil energy	Tonini et al. 2016b [56]
Ref. [56] Wheat, barley, beet, bioenergy	Includes alternative processes for which co-products could be used. Assumed stable market (demands for products equal), co-products not the determining product, products to substitute displaced co-products determined from literature, feed products substituted based on energy content	Fossil energy, artificial fertilizer	Van Zanten et al. 2014 [47]

Table 2. *Cont.*

Product Systems Studied	Definition of System Boundaries and Market-Mediated Substitutions	Assumed Substitutions	Citation
Literature used to define marginal markets			
Miscanthus, willow, ryegrass, sugar beet, maize, wheat, barley, agro-industrial residues, other waste/residues, bioenergy, animal feed	Substitutions from literature	Electricity, heat, animal feed	Tonini et al. 2016a [55]
Corn	Assumed to displace an equivalent conventional corn production system (average data), or marginal market from literature	Corn	Kloverpris et al. 2021 [33]
Camelina, bioenergy	Substituted animal feed soy selected based on similar nutrient profile	Soy, diesel, jet fuel	Li and Mupondwa 2014 [20]
Grass-clover, livestock, bioenergy	Feed substitutions based on nutrient contents	Feed (soy, barley), natural gas, fertilizer	Parajuli et al. 2018 [32]
Land use assumptions used to determine marginal markets			
Grape, pisco	Water other than irrigation excluded because irrigation shown to be 99% of total. To be called pisco, it must be grown in a certain region so available land constrained	Cotton, corn, onion, watermelon, potato, tomato	Larrea-Gallegos et al. 2018 [41]
Barley, bioenergy	Available Danish cropland assumed not to change	Natural gas, gasoline, electricity	Kloverpris et al. 2016 [57]
Willow	Based on current land capacity	Displaced food crops	Styles et al. 2016b [36]
Methods of determining marginal markets not indicated			
Sugarcane, bioethanol, fertilizer	Co-products of sugarcane considered waste. System expansion for co-products of fertilizer production and substituted alternatives	Synthetic fertilizer and substitutable chemicals for fertilizer co-products	Moore et al. 2017 [42]
Rotation of: wheat, ley, rye, beans, oats, canola; bioenergy, liquid CO_2 refrigerant	Manufacturing of capital goods/infrastructure excluded because previously shown to have minor impact	Fossil fuel, HFC refrigerant	Kimming et al. 2011 [53]
Agricultural residues (cereal, other), woody biomass, municipal waste, bioenergy	Domestic production of dedicated energy crops and imported liquid biofuels omitted due to policy objectives to avoid these products	Animal feed (maize, soy), waste treatment	Vadenbo et al. 2018 [60]
Peas, gin	Infrastructure excluded from system boundaries	Wheat, fertilizer, animal feed (barley, soy)	Lienhardt et al. 2019 [34]
Maize, manure, other agricultural by-products, bioenergy	System expansion for alternative product systems impacted	Animal feed, fertilizer, electricity, heat	Van Stappen et al. 2016 [61]
Corn, ethanol	Not indicated	Fossil fuels	Abiola et al. 2010 [49]
Wheat, canola, barley, maize, heat, bioenergy	Not indicated	Food crops, UK electricity grid, heat from boilers, petrol, diesel, food waste management, animal feed, fertilizer	Styles et al. 2015b [46]
Wheat, alfalfa, lactic acid, bioethanol	Not indicated	Ethanol, electricity, fertilizer, animal feed (barley, soy)	Parajuli et al. 2017 [63]
Grass, maize, wheat, biogas, electricity	Not indicated	Food waste in landfill, electricity grid, heat, soy, palm oil	Styles et al. 2015a [45]
Beet, grass, maize, biogas, waste disposal, bioheat, bioelectricity	Not indicated	Fossil heat, fossil electricity, diesel	Styles et al. 2016a [48]

3.3. Impacts of Changes on GHG Emissions

The most common changes assessed in the CLCA studies was the replacement of conventional energy with bioenergy derived from crops. A total of 23 studies assessed the impacts, in terms of GHG emissions, as well as resource use and other emissions in some cases, of replacing conventional energy sources with bioenergy (Table 3). For bioethanol production, almost all studies found both an increase and a decrease in overall life cycle GHG emissions, depending on the specific scenario or modelling choice assessed. Overall, the changes in GHG emissions with the use of crops for different types of bioenergy ranged from −209% to +370% of the impacts of conventional energy production, with most studies reporting smaller estimated changes in impacts. Throughout the majority of studies, the most important drivers of the observed differences in results were the assumptions around system boundaries, substituted products, and marginal technologies.

Only one study found a definitive decrease in GHG emissions associated with the production of bioethanol from crops [49], due to the efficiencies in the alternative fermentation processes which require lower feedstock inputs for fermentation-pervaporation (the separation of liquids by partial vaporization through a membrane) systems and lower energy for separation. One study found an increase in emissions due to SOC changes and the increase in other grain production to meet the same nutritional needs as the beans used for bioethanol [52]. Studies have shown that the harvesting of crop residues for bioethanol, particularly from legume crops, can have negative impacts on soil carbon and fertility [52,65]. Karlsson et al. [52] used the Introductory Carbon Balance Model (ICBM) to model SOC dynamics based on inputs from the crops in rotation and manure, and outputs to yield and crop residues. They found that in the reference scenario where the crop residues were returned to the soil, and the faba bean crops were used as protein feed, the system had a net carbon sequestration of 18 kg CO_2 per ha. However, in their scenario in which both the faba bean grain and crop residue are harvested for use in bioethanol production, the system was emitting 1030 kg CO_2 per ha.

For other biogas and biofuel production systems, three studies found either an increase or decrease in GHG emissions with the use of crops as the fuel stock, depending on the scenarios modelled [45,46,55], and one found an overall increase [20] due to the impacts of agricultural camelina production for the fuel stock. Escobar et al. [62] also found an overall increase in emissions when assessing different scenarios to fulfill an increase in production of 2.58 Mt of biodiesel.

For heat and electricity production, five studies found either an increase or decrease in GHG emissions associated with the use of crops, depending on the scenarios modelled [31,54,55,59,61]. Five found an overall decrease, due to the displaced fossil fuels, changes in land use, soil carbon, and field emissions [47,48,53,57,60]. One study found an overall increase [32] in GHG emissions when producing animal feed from field crops, due to the energy requirements for the production and processing stages as well as direct emissions from agriculture and waste management.

Three studies assessed the use of crops for animal feed (Table 3). Eriksson et al. [35] assessed the difference in emissions from genetically modified (GM) and non-GM soy used for animal feed. They found that the production of GM soy emitted 5.5 times more life cycle GHG emissions (including all marginal product systems) than non-GM soy. This was due to the market effects of switching from more expensive non-GM to cheaper GM soy which lead to increased soy production. Karlsson et al. [52] assessed the change from crops used as protein feed for livestock to roughage feed. They found an overall increase in GHG emissions of 164% if used for roughage feed, compared to protein feed. Similar to the biorefinery scenario discussed above, this was due to the system expansion increase in demand for other grain production to fill the requirements of the beans used as roughage feed. Van Zanten et al. [47] implemented an optimization model for crop use in animal feed, and found that the optimal scenario yielded 0.24 kg fewer GHG emissions per kg biomass. Deng and Tian [38] found an 80–200% increase in GHG emissions from the use of flax crops for polymers (replacing glass). This was due to system expansion effects that

increased the production of flax in China, which had lower yields as well as coal-based electricity production, and thus had higher impacts on a per yield basis.

Four studies assessed changes in production or consumption values of crops (Table 3). An increase of 1 L of pisco production from grapes would result in an increase of up to 9.23 kg CO_2 equivalent emissions from the indirect land use change effects [41], and production of 1 L of gin from peas resulted in a reduction of 2.2 kg CO_2e due to the substitution of soybean animal feed with the co-products of pea-gin production [34]. An increase in production of 1 kg of bananas was estimated to result in an increase of up to 0.34 kg CO_2e from the production and transportation of the bananas [37]. A 50% increase in production of Australian cotton would result in a 7000-fold decrease in CO_2e per kg, and a 50% decrease would result in a 7000-fold increase in CO_2e per kg cotton, including indirect land use change impacts of avoided protein feed and palm oil production [44].

Four studies assessed the GHG implications of changes in farm management practices for crop production (Table 3). Moore et al. [42] evaluated the replacement of synthetic fertilizer with bioethanol residues as fertilizer, and found that sugarcane produced with the bioethanol residues produced 384 fewer kg CO_2e per 10.8 t ethanol than those produced with synthetic fertilizers (a 5% decrease in emissions). Styles et al. [36] assessed the addition of willow crops to cropland either on riparian buffer strips or cropland drainage filtration zones, and found an overall reduction of 9.5 to 14.8 t CO_2e per ha per year. Beltran et al. [43] and Kloverpris et al. [33] assessed the introduction of inoculants and found that this decreased GHG emissions by 45.6 kg CO_2e per tonne of soybeans [43], and by 12.5–15.2% per tonne of corn in rotation with soy [33]. Beltran et al. [43] and Kloverpris et al. [33] included SOC changes in their calculations, calculated using the DayCent model. The presence of inoculants can increase crop productivity and root length, and thus increase soil carbon sequestration [33,43].

Table 3. Impacts of modelled changes on the estimated GHG emissions of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on GHG Emissions	Citation
	Crops for bioenergy	
Different processing technologies for bioethanol production	Conventional: 1.03 kg CO_2e per kg ethanol Bioethanol: 0.66 kg CO_2e per kg ethanol (−35%)	Abiola et al. 2010 [49]
Introduction in multi-crop system for bioethanol compared to gasoline	Gasoline (conventional): 93 g CO_2e per MJ energy Bioenergy: ranged from −2.9 g CO_2e (−3%) to +44.8 g CO_2e per MJ (+48%), compared to conventional, depending on methods	Adler et al. 2018 [40]
Different processing technologies for bioethanol production	Gasoline (conventional): 94.1 g CO_2e per MJ energy Bioethanol: ranged from −12.1 g CO_2e (−13%) to +16.5 g CO_2e per MJ (+17.5%), compared to conventional, depending on methods	Buchspies and Kaltschmitt 2018 [50]
Different ratios of cassava and molasses for bioethanol	Gasoline (conventional): 1.8 kg CO_2e per L bioethanol Bioenergy: range from 2.2 (+18%) to 4.8 kg CO_2e (+62.5%) per L bioethanol	Prapaspongsa and Gheewala 2016 [39]
Bioethanol produced from standalone wheat, standalone alfalfa, and both integrated	Standalone wheat: 0.13 kg CO_2e per MJ bioethanol Standalone alfalfa: 0.39 kg CO_2e per MJ Integrated: 0.05 kg CO_2e per MJ (−87% from standalone alfalfa)	Parajuli et al. 2017 [63]
Switch from protein feed to either bioethanol or roughage feed	For bioethanol: +370 kg CO_2e per ha × yr (+25%) For roughage: +2420 kg CO_2e per ha × yr (+164%)	Karlsson et al. 2015 [52]

Table 3. *Cont.*

Farm/Processing Practice or Policy	Impact on GHG Emissions	Citation
	Crops for bioenergy	
Production of bioelectricity, biomethane, and bioethanol from different crops or residues	Bioelectricity: ranged from −400 g CO_2e to +4000 g CO_2e per kWh Biomethane: ranged from −100 g CO_2e to +600 g CO_2e per MJ Bioethanol: ranged from −600 g CO_2e to +620 g CO_2e per MJ	Tonini et al. 2016a [55]
Bioethanol and biogas production from different crops and residues	Ranged from −2.5 kg CO_2e to +2.0 kg CO_2e per kg biomass, compared to conventional, depending on methods	Tonini et al. 2016b [56]
Production of biogas with different crops	Ranged from −209% to +359%, compared to conventional, depending on methods	Styles et al. 2015b [46]
Production of biogas with different crops	Ranged from −637 g CO_2e to +509 g CO_2e per kg dry matter	Styles et al. 2015a [45]
Optimize feedstock combination for biodiesel according to policy for an increase in production of 2.58 Mt	Ranged from 2 Tg CO_2e per 2.58 Mt biodiesel to 4 Tg, with one scenario at 29 Tg	Escobar et al. 2017 [62]
Used for biodiesel or jet fuel	For biodiesel: 7.61–24.72 g CO_2e per MJ For jet fuel: 3.06–31.01 g CO_2e per MJ	Li and Mupondwa 2014 [20]
Bioenergy produced from different crops, residues and waste	Ranged from −0.222 kg CO_2e to +0.096 kg CO_2e per MJ electricity, compared to conventional, depending on methods	Van Stappen et al. 2016 [61]
Heat and electricity production using different technologies, replacing fossil fuels	Ranged from −45 t CO_2e to +250 t CO_2e per ha, depending on scenario	Tonini et al. 2012 [54]
Self-sufficient bioenergy for farms compared to fossil fuel reference	Wheat straw: reduced 9% compared to fossil fuel Ley: reduced 35% compared to fossil fuel	Kimming et al. 2011 [53]
Bioenergy or animal feed from different crops	−329 g CO_2e or −239 g CO_2e per kg biomass	Van Zanten et al. 2014 [47]
Different combinations of crop residue for bioenergy	Conventional: 590 g CO_2e per kg biomass Bioenergy: ranged from 35 g CO_2e (−94%) to 470 g CO_2e (−20.5%) per kg biomass	Kloverpris et al. 2016 [57]
Integrated crop-livestock system with biorefinery	19.6 kg CO_2e per kg live weight cows + kg live weight pigs	Parajuli et al. 2018 [32]
Bioenergy production replaces human or animal consumption	Conventional: 89 g CO_2e Replaces edible oil: ranged from 58 (+65%) to 329 g CO_2e (+369.5%) Replaces animal feed: ranged from −175 (−196.5%) to 197 g CO_2e (+221%)	Reinhard and Zah 2011 [59]
Anaerobic digestion for bioenergy	Reduced 551–775 Gg CO_2e for entire sector	Styles et al. 2016a [48]
Bioenergy from different crops, residues, and waste based on different policies	Optimized bioenergy scenario: −47%	Vadenbo et al. 2018 [60]
Policy change: Renewable Fuel Standard and Volumetric Ethanol Excise Tax Credit	Ranged from −16.1 g CO_2e to +24.0 g CO_2e per MJ bioenergy, compared to conventional, depending on methods	Bento and Klotz 2014 [31]
Change from straw incorporation to bioethanol and biomethane production	Biomethane: steady state −979 (−436 to −1654) kg CO_2e per t DM straw, annual change −955 (−220 to −1623) Bioethanol: steady state −409 (−107 to −610), annual change −361 (+57 to −603)	Buchspies et al. 2020 [51]

Table 3. *Cont.*

Farm/Processing Practice or Policy	Impact on GHG Emissions	Citation
	Crops for animal feed	
Introducing 1 tonne of genetically modified soy meal for feed	non-GM: 1.3 kg CO_2e per kg soy GM: 8.3 kg CO_2e per kg soy (+538%)	Eriksson et al. 2018 [35]
Switch from protein feed to either bioethanol or roughage feed	For bioethanol: +370 kg CO_2e per ha × yr (+25%) For roughage: +2420 kg CO_2e per ha × yr (+164%)	Karlsson et al. 2015 [52]
Bioenergy or animal feed from different crops	−329 g CO_2e or −239 g CO_2e per kg biomass	Van Zanten et al. 2014 [47]
	Crops for polymers	
Switch from glass to flax fibres for polymers	Flax 80% to 200% higher than glass, depending on scenario	Deng and Tian 2015 [38]
	Changes in production or consumption volumes of crops	
Increase in pisco demand	Up to 9.23 kg CO_2e per L pisco, depending on scenario	Larrea-Gallegos et al. 2018 [41]
1 litre of gin produced from peas instead of wheat	Wheat (conventional): 2.0 kg CO_2e per L gin Pea: −2.2 kg CO_2e per L gin (−110%)	Lienhardt et al. 2019 [34]
Increase in demand of 100 kg bananas	Up to 0.34 kg CO_2e per kg bananas	Sacchi 2018 [37]
Expansion or contraction of Australian cotton production by 50%	Baseline 0.0003 kg CO_2e per kg −2.1 kg CO_2e per kg expansion (−7000×), 2 kg CO_2e per kg contraction (+7000×)	Nguyen et al. 2021 [44]
	Changes in farm practices for crops	
Residues from bioethanol replace chemical fertilizer for sugarcane production	−384 kg CO_2e compared to conventional per 10.8 t ethanol	Moore et al. 2017 [42]
Addition of fertilised and unfertilised willow on riparian buffer strips and drainage filtration zones of cropland	Reduced 9.5 to 14.8 Mg CO_2e per ha per yr	Styles et al. 2016b [36]
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Reduced 45.6 kg CO_2e per tonne soybeans	Beltran et al. 2021 [43]
Inoculation of corn with the soil fungus *Penicillium bilaiae*	Minnesota: −14.1% in corn-corn, −15.2% in corn-soy North Dakota: −14.6% in corn-corn, −12.5% in corn-soy	Kloverpris et al. 2021 [33]

3.4. Other Environmental Impact Categories Included in Crop CLCA Studies

One of the benefits of LCA is that it is a multi-criteria analysis [19]. This means that within the LCA framework, it is possible to include a broad suite of environmental impact categories, thus distinguishing it from a footprint study such as a carbon footprint that only assesses GHG emissions. Indeed, compliance with the ISO 14044 [16] standard for LCA specifically requires that all relevant impact types be included in the analysis. Multiple impact categories were seen in approximately 70% of the studies reviewed. In addition to GHG emissions, the impact categories included were land (12 studies), energy (10), and water use (one), acidification (11), eutrophication (15), ecotoxicity (seven), photochemical oxidant formation (six), ozone depletion (four), ionizing radiation (three), and respiratory emissions (two) (Tables 4–13). By including these multiple impact categories, more information about the environmental impacts of the changes assessed in the CLCA studies was revealed. For example, Eriksson et al. [35] assessed the impacts of a change from

non-GM to GM soy for animal feed. They found a 543% increase in GHG emissions from non-GM to GM soy. However, they also found a 36% decrease in ecotoxicity. Based on GHG emissions alone, the conclusions would have been that GM soy clearly had higher impact than non-GM soy, but when taking the other impact categories into account, that conclusion becomes less clear. This underscores why it is important to consider all relevant impact categories when using LCA as a decision support tool for farm practices and policies.

There were six studies that assessed using crops for bioenergy that assessed land use impacts, six for energy use, seven for acidification, 10 for eutrophication, four for toxicity, and one for each of photochemical oxidation and ozone layer depletion (Tables 4–11). Land use impacts for replacing conventional energy sources with bioenergy from crops ranged from a 57-fold decrease to a 137-fold increase, with most other results varying between smaller increases and decreases, depending on the specific scenarios assessed. For land use, as well as most impact categories, many of the studies only reported net changes (not percentages or absolute impact values), therefore the percent changes could not be calculated. Energy use impacts ranged from −100% to +167%, with a variety of increases and decreases depending on the scenario. Acidification impacts ranged from a 248% decrease to a 500% increase, with most studies reporting increases in impacts when using crops for bioenergy, although there was still some variability depending on the scenario. Eutrophication results were also variable, ranging from a 100-fold decrease to a 45-fold increase in impacts. Toxicity impacts ranged up to a 20 thousand-fold increase [59], with reported decreases as well but no reported or calculable percentage changes for these decreases. Reinhard and Zah [59] was also the only study on using crops for bioenergy that reported photochemical and ozone impacts, ranging from a 770-fold decrease to a twofold increase, and from a decrease of 176% to a decrease of 126%, respectively. They do not provide a contribution analysis for the other impact categories, as they did for GHG emissions, but the differences between scenarios are due to land use change in different countries associated with the use different crops as feedstock for bioenergy. As with the GHG emissions results, the main drivers of the differences in impact results between studies and scenarios were the choices of system boundaries, substitutions, indirect land use change, and marginal technologies.

Three studies assessed a change in amounts of production or consumption of crops, included the introduction of 1 t GM soy meal for feed [35], the production of IL gin from peas instead of wheat [34], and the expansion or contraction of Australian cotton production by 50% [44]. Eriksson et al. [35] found a 320% increase in land use with GM soy compared to non-GM, a 65% increase in energy use, a 165% increase in acidification, an 11–191% increase in eutrophication (depending on the eutrophication category), a 36% reduction in toxicity, a 126% increase in carcinogens, a 102% increase in photochemical oxidation, a 76% increase in ozone depletion, a 100% increase in ionizing radiation and a 361% increase in respiratory emissions (Tables 4–13). Other than toxicity, GM soy had higher impacts than non-GM soy, which was mainly attributable to market effects such as its lower cost of production in Latin America, leading to larger production volumes and high impacts, particularly in land use and associated emissions. Lienhardt et al. [34] found a 17-fold increase in land use associated with the production of gin from peas instead of wheat, as well as a 13% reduction in energy use, a 75% reduction in acidification, a range in eutrophication impacts from −113% to +700%, a range in toxicity impacts from −38% to +470%, a 38% reduction in photochemical oxidation, a 2% reduction in ozone depletion, and a 3% reduction in ionizing radiation (Tables 4–12). Nguyen et al. [44] found 286% increase in land use per kg of cotton expansion and a 293% decrease per kg of cotton contraction (Table 4). Per kg of cotton expansion, they also found increases of 157% in energy use, 52% in water consumption, 81% in water stress, and 500% in eutrophication (Tables 5–8). Per kg of cotton contraction, they found decreases of 188% in energy use, 17% in water use, 48% in water stress, and 1000% in eutrophication. These changes in impacts were due to the direct increases and decreases in crop production, as well as indirect changes through substituted products and marginal production in various countries globally.

Two studies [33,43] assessed the land use, acidification, eutrophication, and photochemical oxidation of changes in management practices for crop cultivation, and Beltran et al. [43] also included results for energy use, toxicity, ozone depletion, ionizing radiation, and respiratory emissions (Tables 4–13). Kloverpris et al. [33] found a 3.1–3.4% reduction in land use from the inoculation of corn with the soil fungus Penicillium bilaiae, as well as a 2.8–3.4% reduction in acidification, a 9–12.8% reduction in eutrophication, and a 3% reduction in photochemical oxidation. Beltran et al. [43] assessed the introduction of inoculants to soybean production systems grown in rotation with corn crops, and similarly found reductions in emissions: 3.3% for land use, 3.8–4.1% for energy, 3% for acidification, 2.9–4.6% for eutrophication, 1.9–6.0% for toxicity, 2.1% for photochemical oxidation, 4.6% for ozone depletion, 4.7% for ionizing radiation, and 2.1% for respiratory emissions. These results show a consistent trend of small reductions in life cycle resource use and emissions from the use of inoculant, due to their impacts on yield and soil nutrient dynamics. By increasing yields, crop production elsewhere may be avoided, which reduces the overall impacts of the system [43].

Table 4. Impacts of modelled changes on the estimated land use of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Land Use	Citation
Crops for bioenergy		
Converting feedstock to ethanol by integrated continuous fermentation-pervaporation system (CF) as alternative to conventional batch fermentation (BF)	Land use (kha): 34.69 conventional, 29.87 alternative (+14%)	Abiola et al. 2010 [49]
Reference scenario beans as protein feed for dairy cows, compared to 2 alternatives: whole crop for ethanol, protein concentrate and fuel briquettes (biorefinery), or whole crop as roughage feed (roughage)	Biorefinery decreased 0.2 ha (−20%) land use compared to reference Roughage decreased land use 1.3 ha (−130%)	Karlsson et al. 2015 [52]
2 energy self-sufficient systems for organic arable farms—compared to reference fossil fuel	Straw scenario: 25% of land use Ley: 13% land use	Kimming et al. 2011 [53]
(A) standalone wheat straw producing 1 MJ bioethanol, (B) standalone alfalfa producing 1 kg lactic acid, compared to (C) both integrated	Land use (m^2a) (A) 0.02, (B) 1.99, (C) 0.11	Parajuli et al. 2017 [63]
Rapeseed for energy production substitutes rapeseed as edible oil or barley as animal feed	Land use: (dm^2): reference scenario 0.0035, edible oil replacement ranges 5–48, barley replacement ranges −20 to 38.	Reinhard and Zah 2011 [59]
Maize, manure, other agricultural by-products for bioenergy	Land use +0.13 m^2y to +0.43	Van Stappen et al. 2016 [61]
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Land use non: 32,200 kg soil organic carbon, GM: 135,000 (+320%)	Eriksson et al. 2018 [35]
IL gin from wheat or peas	Land occupation (m^2 × yr): wheat 0.1, pea 1.8 (17x)	Lienhardt et al. 2019 [34]
Expansion or contraction of Australian cotton production by 50%	Land occupation, m^2y: baseline 1.5, +4.3 expansion (286%), −4.4 contraction (−293%)	Nguyen et al. 2021 [44]
Crops for animal feed		
(1) wheat middlings for dairy cattle instead of pigs, (2) beet tails for dairy cattle instead of bioenergy	(1) reduced 169 m^2 land per ton wheat middlings, (2) reduce 154 m^2 land per ton beet tails	van Zanten et al. 2014 [47]
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Nature occupation: −44.1 PDF × m^2a (−3.3%)	Beltran et al. 2021 [43]
Inoculation of corn with the soil fungus Penicillium bilaiae	Land occupation, m^2y: −3.1% Minnesota (MN), −3.4% North Dakota (ND)	Kloverpris et al. 2021 [33]

Table 5. Impacts of modelled changes on the estimated energy and resource use of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Energy/Resource Use	Citation
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Mineral resource non-GM: 0.522 kg Sb e, GM: 0.863 (+65%)	Eriksson et al. 2018 [35]
Expansion or contraction of Australian cotton production by 50%	Fossil energy resources, MJ lower heating value: Reference 14.9, +23.5 expansion (157%), −28 contraction (−188%)	Nguyen et al. 2021 [44]
IL gin from wheat or peas	Fossil resource depletion (MJ eq): wheat 44.7, pea 39.0 (−13%)	Lienhardt et al. 2019 [34]
Crops for bioenergy		
Reference scenario beans as protein feed for dairy cows, compared to 2 alternatives: whole crop for ethanol, protein concentrate and fuel briquettes (biorefinery), or whole crop as roughage feed (roughage)	Biorefinery decreased 100% energy use. Roughage increased energy 167%	Karlsson et al. 2015 [52]
2 energy self-sufficient systems for organic arable farms—compared to reference fossil fuel	Straw: 45% energy Ley: 24% energy	Kimming et al. 2011 [53]
Camelina oil derived biodiesel and jet fuel	Non-renewable energy for camelina biodiesel: 0.40–0.67 MJ/MJ, camelina jet fuel: 0.13–0.52 MJ/MJ.	Li and Mupondwa 2014 [20]
Residues from bioethanol to replace chemical fertilizer for sugarcane production	Energy demand increased 3.16 GJ	Moore et al. 2017 [42]
(A) standalone wheat straw producing 1 MJ bioethanol, (B) standalone alfalfa producing 1 kg lactic acid, compared to (C) both integrated	Non renewable energy (A) 1.25 MJ e, (B) 14.63, (C) 0.38	Parajuli et al. 2017 [63]
Biogas sector: anaerobic digestion substituting composting, incineration, sewer disposal, field decomposition, animal feeding of waste	Fossil energy depletion reduced by 8.9–10.8 Pje	Styles et al. 2016a [48]
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Non-renewable energy: −148 MJ primary (−4.1%) Mineral extraction: −0.77 MJ extra (−3.8%)	Beltran et al. 2021 [43]

Table 6. Impacts of modelled changes on the estimated water use of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Water Use	Citation
Changes in production or consumption volumes of crops		
Expansion or contraction of Australian cotton production by 50%	Water consumption, L: Baseline 2462, +1290 expansion (52%), −423 contraction (−17%) Water stress, WSI L H_2Oeq: Baseline 1043, +847 expansion (81%), −498 contraction (−48%)	Nguyen et al. 2021 [44]

Table 7. Impacts of modelled changes on the estimated acidifying emissions of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Acidification Emissions	Citation
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Acidification non-GM: 9.22 mol H+ e, GM: 24.4 (+165%)	Eriksson et al. 2018 [35]
IL gin from wheat or peas	Acidification (mol H+ e): wheat 0.028, pea 0.007 (−75%)	Lienhardt et al. 2019 [34]
Crops for bioenergy		
Optimize feedstock combination for biodiesel according to policy (2.58 Mt increase in demand for bioenergy to meet targets)	Acidification increased 13–41 Gg SO_2 e to produce an additional 2.58 Mt bioenergy	Escobar et al. 2017 [62]
Residues from bioethanol to replace chemical fertilizer for sugarcane production	Acidification reduced 120 kg SO_2e	Moore et al. 2017 [42]
Rapeseed for energy production substitutes rapeseed as edible oil or barley as animal feed	Acidification (mg SO_2 e): reference scenario 226, edible oil replacement 227–485, barley replacement 241–1353	Reinhard and Zah 2011 [59]
Biogas from crops on dairy farm	Acidification increased 10%	Styles et al. 2015a [45]
Biogas production with crop rotation and digestate nutrient cycling. Default scenario of wheat, rape, and barley, changed to wheat and maize instead of barley, for use in heat, electricity, biogas	Acidification −248% to +97%	Styles et al. 2015b [46]
Biogas sector: anaerobic digestion substituting composting, incineration, sewer disposal, field decomposition, animal feeding of waste	Acidification increased 8.1–14.6 Gg SO2e	Styles et al. 2016a [48]
Maize, manure, other agricultural by-products for bioenergy	Acidification: +9.81 × 10^{-3} AE e per MJ to +36.64 × 10^{-3} (+37%)	Van Stappen et al. 2016 [61]
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Acidification: −5.56 m^2 UES (−3.0%)	Beltran et al. 2021 [43]
Inoculation of corn with the soil fungus Penicillium bilaiae	Acidification, g SO_2eq: −2.9% corn after corn (C-C) Minnesota (MN), −3.4% corn after soy (C-S) MN, −2.8% C-C North Dakota (ND), −2.8% C-S ND	Kloverpris et al. 2021 [33]

Table 8. Impacts of modelled changes on the estimated eutrophying emissions of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Eutrophication Emissions	Citation
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Freshwater eutrophication non-GM: 0.765 kg P e, GM: 0.846 (11%), marine eutrophication non-GM: 17 kg N e, GM: 30.1 (78%), terrestrial eutrophication non: 27.0, GM: 78.5 (191%)	Eriksson et al. 2018 [35]
IL gin from wheat or peas	Freshwater eutrophication (kg P e): wheat 4 × 10^{-4}, pea 2.1 × 10^{-4} (−90%). Marine eutrophication (kg N e): wheat 0.001, pea −0.008 (+700%). Terrestrial eutrophication (kg N e): wheat 0.08, pea −0.01 (−113%)	Lienhardt et al. 2019 [34]
Expansion or contraction of Australian cotton production by 50%	Freshwater eutrophication, kg P eq: Baseline 0.0002, +0.001 expansion (+500%), −0.002 contraction (−1000%); marine eutrophication, kg N eq: Baseline 0.005, +0.011 expansion (+220%), −0.02 contraction (−400%)	Nguyen et al. 2021 [44]

Table 8. *Cont.*

Farm/Processing Practice or Policy	Impact on Eutrophication Emissions	Citation
Crops for bioenergy		
Optimize feedstock combination for biodiesel according to policy (2.58 Mt increase in demand for bioenergy to meet targets)	Eutrophication range 2 Gg PO_4 e per increase of 2.58 Mt biodiesel to 30	Escobar et al. 2017 [62]
Bioenergy production—reference (1): barley, oilseed radish, 100% straw incorporation, 2: 50% straw removed, 3: wheat, 100% incorporation, 4: 50%, 5: early seeding, 50% straw removed, 6: wheat and oilseed radish, 50% straw removed	Eutrophication: lowest to highest emissions: 5, 6, 3, 4, 1, 2.	Kloverpris et al. 2016 [57]
Residues from bioethanol to replace chemical fertilizer for sugarcane production	Eutrophication reduced 4.02×10^{-2} kg P e	Moore et al. 2017 [42]
(A) standalone wheat straw producing 1 MJ bioethanol, (B) standalone alfalfa producing 1 kg lactic acid, compared to (C) both integrated	Eutrophication (A) 1.5×10^{-4} kg PO_4e, (B) -1.4×10^{-3}, (C) 1.3×10^{-5}	Parajuli et al. 2017 [63]
Rapeseed for energy production substitutes rapeseed as edible oil or barley as animal feed	Eutrophication (mg PO_4 e): reference scenario 41, edible oil replacement 167–1119 barley replacement −1669 to −209	Reinhard and Zah 2011 [59]
Biogas from crops on dairy farm	Eutrophication increased 9%	Styles et al. 2015a [45]
Biogas production with crop rotation and digestate nutrient cycling. Default scenario of wheat, rape, and barley, changed to wheat and maize instead of barley, for use in heat, electricity, biogas	Eutrophication ranged from −43% to +129%	Styles et al. 2015b [46]
Biogas sector: anaerobic digestion substituting composting, incineration, sewer disposal, field decomposition, animal feeding of waste	Eutrophication increased by 1.8–3.4 Gg PO_4e	Styles et al. 2016a [48]
Fossil fuel reference vs production of heat/electricity from 1 ha land with 3 crops via anaerobic co-digestion with manure, gasification, combustion in small-to-medium scale biomass CHP plants, co-firing in large scale coal-fire CHP plants	P-eutrophication ranged from −8 to +50 kg P per ha, N-eutrophication from −800 to +2300 kg N per ha	Tonini et al. 2012 [54]
Maize, manure, other agricultural by-products for bioenergy	Eutrophication from $+0.43 \times 10^{-3}$ kg PO_4e per MJ to +3.58	Van Stappen et al. 2016 [61]
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Aquatic eutrophication: −1.61 kg NO_3- eq (−4.6%) Terrestrial eutrophication: −23.9 m^2 UES (−2.9%)	Beltran et al. 2021 [43]
Inoculation of corn with the soil fungus Penicillium bilaiae	Eutrophication, PO_4^{3-} eq: −12.8% C-C MN, −9% C-S MN, −12.5% C-C ND, −10.4% C-S ND	Kloverpris et al. 2021 [33]

Table 9. Impacts of modelled changes on the estimated toxic or carcinogenic emissions of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Toxic or Carcinogenic Emissions	Citation
Crops for bioenergy		
Optimize feedstock combination for biodiesel according to policy (2.58 Mt increase in demand for bioenergy to meet targets)	Freshwater ecotoxicity: +1.5 to +17 Tg DCB per 2.58 Mt increase in demand	Escobar et al. 2017 [62]
Residues from bioethanol to replace chemical fertilizer for sugarcane production	Human toxicity reduced 190 kg 1,4-DBe, terrestrial ecotoxicity reduced 7.95×10^{-2} kg 1,4-DBe	Moore et al. 2017 [42]
Integrated mixed crop-livestock system with green biorefinery	−3.9 CTUe freshwater ecotoxicity	Parajuli et al. 2018 [32]

Table 9. *Cont.*

Farm/Processing Practice or Policy	Impact on Toxic or Carcinogenic Emissions	Citation
Crops for bioenergy		
Bioenergy production replaces human or animal consumption	Human toxicity (g 1,4-DB e): reference scenario 9, edible oil replacement ranges 14–40, barley replacement 13–37 terrestrial ecotoxicity (mg 1,4-DB e): reference 84, edible oil replacement 11,953–184,851, barley replacement 84,820–95,791	Reinhard and Zah 2011 [59]
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Freshwater ecotoxicity non-GM: 54,900 CTUh.m^3·yr, GM: 35,000 (−36%) Carcinogenic effects non-GM: 0.0000498 CTUh, GM: 0.000112 CTUh (126%)	Eriksson et al. 2018 [35]
IL gin from wheat or peas	Human toxicity cancer (CTUh): wheat 8.5×10^{-8}, pea 5.3×10^{-8} (−38%). Human toxicity non-cancer (CTUh): wheat 2.7×10^{-7}, pea -1×10^{-6} (−470%). Freshwater ecotoxicity (CTUh): wheat 6.8, pea −5.9 (−187%)	Lienhardt et al. 2019 [34]
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Human toxicity, carcinogens: −0.218 kg C_2H_3Cl eq per tonne soy (−4%) Human toxicity, non-carcinogens: −0.217 kg C_2H_3Cl eq (−1.9%) Aquatic ecotoxicity: −609 kg TEG eq (−6.9%) Terrestrial ecotoxicity: −52.4 kg TEG eq (−3.5%)	Beltran et al. 2021 [43]

Table 10. Impacts of modelled changes on the estimated photochemical emissions of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Photochemical Emissions	Citation
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Photochemical ozone: -74.1 $m^2 \times$ ppm $\times$ h (−2.1%)	Beltran et al. 2021 [43]
Inoculation of corn with the soil fungus Penicillium bilaiae	Photochemical ozone formation, C_2H_4 eq: −2.9% MN, −3% ND	Kloverpris et al. 2021 [33]
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Photochemical ozone creation non-GM: 7.60 kg ethylene e, GM: 22.2 (192%)	Eriksson et al. 2018 [35]
I L gin from wheat or peas	Photochemical ozone (NMVOC e): wheat 0.008, pea 0.005 (−38%).	Lienhardt et al. 2019 [34]
Crops for bioenergy		
Rapeseed for energy production substitutes rapeseed as edible oil or barley as animal feed	Photochemical oxidation (mg C_2H_4 e): reference scenario 14, edible oil replacement 10–45, barley replacement −41 to −25	Reinhard and Zah 2011 [59]

Table 11. Impacts of modelled changes on the estimated ozone depleting emissions of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Ozone Depletion Emissions	Citation
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Ozone depletion non-GM: 0.000132 CFC-11 e, GM: 0.000232 (76%)	Eriksson et al. 2018 [35]
IL gin from wheat or peas	Ozone depletion (kg CFC-11 e): wheat 5.2×10^{-7} , pea 5.1×10^{-7} (−2%)	Lienhardt et al. 2019 [34]
Crops for bioenergy		
Rapeseed for energy production substitutes rapeseed as edible oil or barley as animal feed	Ozone depletion (mg CFC-11 e): reference scenario 0.013, edible oil replacement ranges from −0.0095 to −0.0066, barley replacement ranges from −0.0099 to −0.0034	Reinhard and Zah 2011 [59]
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Ozone layer depletion: −0.00000123 Kg CFC-11 eq (−4.6%)	Beltran et al. 2021 [43]

Table 12. Impacts of modelled changes on the estimated ionizing radiation emissions of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Ionizing Radiation Emissions	Citation
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Ionizing radiation non-GM: 54.7 U235 e, GM: 109 (100%)	Eriksson et al. 2018 [35]
IL gin from wheat or peas	Ionizing radiation (kBq U235 e): wheat 0.39, pea 0.38 (−3%)	Lienhardt et al. 2019 [34]
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Ionizing radiation: −17 Bq C-14 eq (−4.7%)	Beltran et al. 2021 [43]

Table 13. Impacts of modelled changes on the estimated respiratory emissions of the production of crops in consequential life cycle assessment studies.

Farm/Processing Practice or Policy	Impact on Respiratory Emissions	Citation
Changes in production or consumption volumes of crops		
Introducing 1 t GM soy meal for feed	Respiratory inorganics non-GM: 0.818 kg PM2.5 e, GM: 3.77 (361%)	Eriksson et al. 2018 [35]
Changes in farm practices for crops		
Introduction of yield enhancing inoculants to soybean production systems grown in rotation with corn crops	Respiratory inorganics: −0.031 kg PM2.5eq (−2.6%) Respiratory organics: −0.0006 pers × ppm × h (−2.1%)	Beltran et al. 2021 [43]

3.5. Potential for Performance of CLCA Studies of Crop Management Practices

The results of this review indicate large gaps in the research of CLCA studies for Canadian crop production systems. However, there have been many other assessments (field-level or ALCA) and recommendations for farm management practices that may reduce GHG emissions from crop production, in Canada and elsewhere, which can inform

future Canadian crop CLCA studies. For example, there have been Canadian ALCAs that have assessed the production of canola, corn, pulse crops, and organic crops [5–7,66,67].

In order to use these studies as a basis to perform CLCA studies on similar systems, a number of methodological choices must be made, and additional data must be collected. The two main choices discussed here are the identification of marginal markets and the adherence to ISO standards. According to the International Reference Life Cycle Data System (ILCD) handbook [68], CLCA is best suited to address macro-scale changes, defined as a change in more than 5% of the production capacity, or more than the annual replacement capacity. Once it has been determined that CLCA is appropriate, it will be necessary to identify the marginal markets affected by changes in farm management practices in the field crop industry. This challenge may pose considerable difficulty, due to the numerous other industries linked to crop production. Additionally, the marginal market affected may differ depending on the intervention studied, and the effects of the intervention on the crop. For example, Paré et al. [69] performed a long term study to identify best management practices with respect to crop rotation and tillage for northern agriculture. In other studies, both of these management strategies have been shown to reduce GHG emissions at field level [70,71]. However, Paré et al. [69] found that crop rotation resulted in increased yields, while tillage practices did not affect yield. As such, a CLCA of these two management practices, though both resulting in decreased GHG emissions at farm level, would have different affected marginal markets. In the case of crop rotation, marginal markets would include those affected by an increase in grain supply, and any changes in the crops included in the rotation. In the case of tillage practices, marginal markets would include those affected by a decrease in tillage practices, for example fuel and machinery markets, but would not include an increase in grain supply since yield was not affected. Proper identification of marginal markets is absolutely essential to performance of a robust CLCA.

Currently, few detailed guidelines exist on how to identify marginal markets. The ILCD handbook provides a set of guidelines for identification of associated market processes [68]; however, these guidelines suggest that the inclusion of experts in many fields, including technology cost, development, and forecasting, scenario development, market cost and forecasting, and general and partial equilibrium modelling is necessary for proper identification. In order to decrease the financial and temporal burden of identifying marginal markets, Bamber et al. [72] instead chose to identify marginal markets through interviews with experts in the field. This may be a viable alternative to increase ease in identification of marginal markets. Regardless of the method used however, the ILCD guidelines are still useful in demonstrating the scope of the market effects needing to be considered, including market directions, secondary markets, as well as potential market constraints [68]. Further investigation into the nature of the substitutions taking place may also be necessary, as the assumption of one-to-one substitution of marginal market equivalents may not be valid [73].

In addition to the identification of marginal markets, any future CLCAs performed for the field crop industry should follow the guidelines for the four phases of LCA outlined in the ISO14040-14044 standards. Adherence to the ISO guidelines for LCA ensures consistent application of methodologies, allowing for comparison between studies. Previous ALCA studies that assessed field crop production in Canada are largely in compliance with the standards outlined by the ISO14040-14044 series [5–7,66,67], including all four phases (goal and scope, inventory analysis, impact assessment, and interpretation). The same cannot be said of the CRSC field crop carbon footprint studies, since these focused exclusively on GHG emissions. That being said, much of the information necessary for bringing those reports into compliance with the ISO standards is available, and would only require some additional data collection along with formalization into the format specified by ISO. A common downfall of all the aforementioned studies is the lack of quantitative uncertainty assessment. While many did make note of uncertainty in their life cycle inventory data, efforts were not made to propagate this uncertainty throughout the model in a formal uncertainty assessment. Additionally, data quality assessment was not undertaken in these

studies to assess the fitness of the models produced for the intended purposes. The lack of these additional uncertainty assessments reduces their robustness. Recent work by Bamber et al. [22] includes significant detail regarding current practices for uncertainty assessment in CLCA studies, along with best practice guidelines.

3.6. Crop Production Recommendations for Inclusion into CLCA Studies

The aforementioned ISO-compliant ALCA studies of Canadian crop production could potentially be used as a basis for performing CLCA studies by expanding the system boundaries to include likely market-mediated substitutions. For example, Pelletier et al. [5] assessed the implications of transitioning to organic production of canola, corn, soy and wheat. In this case, marginal products would need to be considered for any changes in yield and inputs associated with the transformation of conventional to organic farming for these crops. MacWilliam et al. [6] assessed the inclusion of pulses into crop rotations. For a CLCA, the alternative uses of those pulses should be considered, as well as the decrease in production of the crops that were substituted by the pulses in the rotation. Additionally, any changes in yield and inputs (such as fertilizers) would also need to be included.

In addition to previously conducted ALCA studies, there are currently diverse recommendations that might potentially support more sustainable crop production, which can also be assessed using CLCA to determine their sustainability at a larger scale, including the life cycle of all product systems affected by a change in farm management. The 4R Nutrient Stewardship framework from Fertilizer Canada [74] refers to applying fertilizer to crops from the Right source, at the Right rate, the Right time and the Right place. The framework is currently practiced in many different regions of Canada (including Alberta, Manitoba, Ontario, New Brunswick, and Prince Edward Island), making it an ideal candidate to assess using CLCA at a regionally-relevant scale within Canada. Nitrogen fertilizer best management practices under the 4R framework have been shown to reduce GHG emissions by at least 25%, while increasing yields up to 20% [74]. The 4R Research Network has identified 10 best management practices for Canadian wheat, canola, soybean and potato production. These are (1) applying nitrogen fertilizer as a band close to the seed row for wheat and canola in Alberta, (2) optimizing the rate of nitrogen application during seeding in wheat production in Alberta, (3) integrating sulphur fertilizer using Fertigation for wheat production in Alberta, (4) applying phosphorus fertilizer as an in-soil placement to reduce runoff and waste for wheat, canola and soybean production in Saskatchewan, (5) applying nitrification inhibitors in wheat production in Manitoba, (6) applying urea at the time of planting in wheat production in Manitoba, (7) applying urea/urea ammonium nitrate with nitrification and urease inhibitors at the eighth leaf growth stage in corn production in Ontario, (8) applying nitrification and urease inhibitors with nitrogen fertilizer as a soil injection in corn production in Ontario, (9) applying phosphorus fertilizer as a sub-surface band in corn production in Ontario, and (10) applying nitrogen fertilizer at an optimized rate in potato production on Prince Edward Island. These best management practices could be assessed using CLCA to determine the broader market impacts of differing yield, land use or agricultural inputs.

Alberta's Agricultural Carbon Offsets program gives recommendations for ways to decrease the carbon footprint of crop production [75]. These recommendations include aerobic composting to reduce methane emissions, aerobic landfill bioreactors to reduce methane emissions, implementing the 4Rs framework mentioned above, creating biofuels from crops/residues to avoid GHG emissions from petroleum based fuel, carbon capture and storage to remove carbon from the atmosphere, conservation tillage to increase soil carbon sequestration, generating renewable energy (wind, water and solar) to offset fossil fuel sources, implementing energy efficient technologies, improving the fuel efficiency of combustion engines and capture of methane, using air rather than natural gas in pneumatic power tools, landfill gas capture and combustion to covert methane to carbon dioxide for an overall reduction in GHG emissions, and waste heat recovery. These recommendations could similarly be assessed using CLCA to determine the impacts of their implementation,

including an increased or decreased demand for certain products used on farms, displaced alternative sources of energy, crops, or other products produced, changes in yield, emissions, land use, agricultural inputs, etc. This would do much to resolve the current gap in terms of CLCA studies of Canadian field crops and provide useful insight to farmers in support of determining the most sustainable farm practices. Similarly, country or region-specific recommendations could be incorporated into CLCA studies of field crops produced in any other national or regional production context. Table 14 presents some of the common methodological challenges to conducting CLCAs, and the proposed guidance to overcome these challenges, as discussed throughout this paper.

Table 14. Methodological challenges to conducting CLCAs and proposed guidance to overcome them.

Methodological Challenges to Conducting CLCAs	Guidance to Overcome Challenges
Definition of system boundaries	information on changes in yield, inputs, alternative uses of products; expert inputs from relevant fields
Definition of marginal markets	economic general/partial equilibrium models; regionally specific industry data; expert inputs from relevant fields
Choice of impact categories	multi-criteria to avoid burden shifting; include all relevant categories to adhere to ISO standards; common categories are GHGs, acidification, eutrophication, ecotoxicity, land use and energy
Data inclusion	previous ALCAs and farm management recommendations can be used to inform choice of changes to study with CLCA; expand to include all relevant product systems (see above); include changes in land use and soil organic carbon
Data quality and uncertainty	semi-quantitative data quality assessment using the pedigree matrix for LCA; quantitative uncertainty propagation using Monte Carlo simulation

4. Conclusions and Recommendations

Based on the results of the current review, the majority of consequential LCA studies on agricultural crops published in the past 12 years have assessed the use of crops in bioenergy production. Despite the relatively large sample size of 23 studies, there was no consensus on the mitigation potential of bioenergy. Most studies found that depending on the specific scenario assessed (within each study), there was either an increase or a decrease in impacts associated with an increase in production of bioenergy from crops. There were also some studies that found an overall increase in emissions in all scenarios assessed, and some that found an overall decrease. Therefore, it can be concluded that the current state of art is inconsistent with bioenergy's impact being beneficial or detrimental in terms of environmental impact mitigation, but overall, there is a large amount of uncertainty and variability in the mitigation potential of bioenergy production. These differences are linked to the different system boundaries, assumed substitutions, and marginal technologies used in each CLCA study. Therefore, these results are highly case-specific and should not be used to make general conclusions.

There were also a small number of CLCA studies of crops used for animal feed. Two of these found increased GHG emissions, and one performed an optimization to determine the best scenario, which resulted in decreased GHG emissions. Similar to crop use in bioenergy production, there is uncertainty and variability in the mitigation potential of crop use as animal feed. There were five CLCA studies that assessed different crop management practices, which found that GM soy generally had higher impacts than non-GM soy for use as animal feed [35], the replacement of synthetic fertilizer with residue from bioethanol production reduced most emissions [42], adding willow crops to crop fields can reduce GHG emissions [36], and inoculants can reduce all resource use and emissions [33,43].

However, none of these crop management CLCAs were geographically representative of Canada. In fact, there was only one CLCA out of all studies assessed that was representative of a Canadian field crop (camelina oil for biofuel). This clearly indicates the need for further research in the field of consequential life cycle assessment of farm management practices for Canadian crops.

There have been a variety of ALCA studies of Canadian crop production and management practices that can be used to inform the changes to be studied using CLCA studies. In order to use these previous studies to inform data collection and performance of CLCAs, the most important step is to identify the potential market-mediated marginal substitutions, or the processes that would be affected by the crop production change being assessed. This can be done using economic optimization models, or interviews with experts and local producers. Data on changes in crop yield, inputs, and alternative uses of products and co-products are necessary to determine these marginal processes. In order to avoid burden shifting between different types of environmental impacts, and to comply with ISO standards, it is recommended to perform a full multi-criteria CLCA of any potential broad-scale change in crop management practice, including the impacts of SOC changes due to land use change. This involves including all relevant environmental impact categories in the impact assessment. Common impact categories included in agricultural LCA studies are climate change (GHG emissions), acidification, eutrophication, ecotoxicity, land use, and energy use. ISO-compliant CLCA studies also need to include data quality and uncertainty assessment to provide more robust results.

Due to the lack of Canadian crop management CLCA studies, it is strongly recommended to conduct a representative suite of CLCAs to inform best management practices for sustainable crop production in Canada. The information gathered in this literature review can support the development of CLCA methodology with respect to the identification of potential crop management strategies, definition of system boundaries, identification of market-mediated substitutions, and inclusion of impact categories. These CLCAs will be useful tools for application to many policies such as low carbon fuel standards and agricultural management.

Author Contributions: Conceptualization, N.B. and N.P.; methodology, N.B. and N.P.; software, N.B. and I.T.; validation, N.B., B.D., M.D.H. and N.P.; formal analysis, N.B. and I.T.; investigation, N.B.; resources, N.B. and I.T.; data curation, N.B. and I.T.; writing—original draft preparation, N.B. and I.T.; writing—review and editing, N.B., B.D., M.D.H. and N.P.; visualization, N.B. and N.P.; supervision, B.D., M.D.H. and N.P.; project administration, N.P.; funding acquisition, N.P. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by the Canadian Roundtable for Sustainable Crops.

Data Availability Statement: All data are provided in the tables in this manuscript.

Conflicts of Interest: The authors declare no conflict of interest. The funders contributed to the design of the study.

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