Assessing the Economic and Environmental Performance of Dairy and Paddy Field Agricultural Systems in Brazil

Dissertation

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Summary

Agricultural systems form the basis of the economy of many low- and middle-income countries, playing a pivotal role in providing food security, employment and safeguarding livelihoods in rural areas. Increasing urbanisation, income growth and shifts in dietary patterns will continue to drive the demand for agricultural products worldwide over the coming decades. Consequently, increasing even further the importance of agriculture and livestock husbandry for maintaining global food security and human well-being. Despite their importance, many agricultural practices and systems currently in use also generate environmental externalities that negatively affect humans, deplete natural resources and harm the environment.

Given the foregoing, globally agricultural policymakers face the challenge of designing policies that sustain rural development and national economies while mitigating or avoiding environmental externalities. This has become paramount in recent years, primarily for countries that assumed international environmental commitments, such as reducing deforestation and greenhouse gas (GHG) emissions. Generally, sustainable intensification and integration of agricultural systems are important strategies suggested to meet these goals. These practices are expected to promote more efficient use of scarce resources, reduce greenhouse gas intensity and increase biodiversity, as well as and the productivity of agricultural systems. Foreseeable, sustainable intensification and integration practices are at the core of the contemporary Brazilian agricultural policy and research agenda. On the ground, however, the effective implementation of environmentally friendly agricultural systems and practices is complex and context-specific. Therefore, a better understanding of the heterogeneity among farms and of the trade-offs and synergies involved in the process of moving farmers from the status quo could support the design and implementation of more sustainable agricultural systems.

The overall objective of this dissertation is to investigate the economic, environmental and joint performance of dairy and paddy fields agricultural systems in Brazil. The goals are to contribute to the interdisciplinary literature concerning the sustainability of agricultural systems in Brazil and propose improvements. The dissertation is presented in three papers, complemented by a general introduction and conclusion.

The first paper is dedicated to analysing the heterogeneity and carbon footprint among dairy farms and to identify key strategies and management practices that farmers can adopt to reduce the

carbon footprint[1] (CF) of dairy farming. Based on a sample of 911 dairy farms from Paraná state, southern Brazil, a farm-specific CF was calculated and farm groups were created using the k-means clustering algorithm. We also compared the traditional Global Warming Potential 100-year horizon (GWP_{100}) and the recently published Global Warming Potential Star (GWP*). Clustering the farms allowed to identify four distinct groups operating in Paraná. Farm structure and the CF per kg of Fat and Protein Corrected Milk (FPCM) varied significantly between the groups. The mean CF results ranged from 1.75 kg CO_2eq. (kg FPCM)$^{-1}$ in Group 1 to 3.27 kg CO_2eq. (kg FPCM)$^{-1}$ in Group 4. The results also confirmed the strong negative correlation between CF intensity and animal productivity in dairy systems. In summary, researchers, policymakers and extension services should strive to accelerate the adoption of intensification technologies and practices among less efficient farmers. However, they must recognise the heterogeneity among dairy farms operating in a certain region to guide targeted GHG mitigation strategies and surveil the rising of other environmental impacts associated with intensive dairy farming.

In the second paper, we advanced the analysis of dairy farming by integrating economic and environmental analysis under the same framework. We estimated the environmental efficiency of dairy farmers from Minas Gerais state, the leading milk producer in Brazil. We analysed a sample of 208 farms taking part in the Embrapa's Full Bucket programme. The programme was created to foster dairy farms' sustainable intensification in the country through technology transfer and participatory learning. Methane emissions was selected as the undesirable output for the analysis, and the calculation of shadow prices (i.e., abatement costs). Methane is the most important GHG emitted in dairy farming and has recently received much attention in the global environmental policy agenda. The results show that farmers can improve the environmental efficiency of their farms and contribute to the Brazilian commitments to decarbonise the economy. On average, farmers can increase farm production by 9.4% and reduce methane emissions by 8.7% without requiring further inputs. Moreover, we identified that climate types occurring in Minas Gerais influence the production frontier and should be considered when developing regional policies. By investigating key variables that contribute to farm (in)efficiency, we found that owning more productive cows, improving pastureland and adjusting the percentage of lactating cows in the herd

[1] In this dissertation Carbon Footprint, Global Warming Potential and CO_2eq. are used interchangeably to describe the warming potential of the diverse greenhouse gases according to the Global Warming Potential 100-year horizon.

positively influence the environmental efficiency of dairy farms. The mean shadow price per tonne of methane emitted was US $2,254, indicating a high abatement cost of emissions. Overall, policymakers should focus on supporting farmers in becoming more efficient since the high abatement costs of methane may hinder the effective development of other interventions under the present technology.

The last paper addresses the economic and environmental performance of lowland paddy field-based crop-livestock systems in the state of Rio Grande do Sul (RS), southern Brazil. RS is the largest paddy rice producer in Brazil, and a pioneer in producing beef cattle and soybeans. Rotation between beef cattle and rice has a long tradition in the lowland areas in this state. Recently, field experiments have been conducted in RS to improve the sustainability of paddy field-based cropping systems. We compared two multi-year field experiments (beef cattle-rice and beef cattle-rice-soybeans rotation) with a baseline system (traditional beef cattle-rice). We assessed a set of environmental indicators based on the Life Cycle Assessment approach. Namely, three environmental impacts (Global Warming Potential, Terrestrial Acidification Potential and Freshwater Eutrophication Potential) and three impacts related to natural resources usage (Fossil Depletion, Agricultural Land Occupation and Water Depletion). Additionally, we calculated Returns to Land and Management to represent the economic indicator. Overall, the results showed that improving the traditional beef cattle-rice rotation increased farm output and profitability. However, trade-offs emerged, and the results of the environmental metrics depended on the functional unit the decision maker evaluates. Production-related functional units (e.g., protein production) tended to favour improved systems, while area-related functional unit favoured the less intensive baseline system. Increasing farm inputs in the crop-livestock system without including soybean in the rotation increased production and profits but brought little advantages in terms of environmental performance.

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Everton Vogel

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Acronyms

ACT	Annual Climate Type
AFOLU	Agriculture, forestry and other land use
AI	Artificial Insemination
ALO	Agricultural Land Occupation
AP	Acidification Potential
APP	Permanent Preservation Area
Aw	Tropical with dry winter
BS	Dry Semi-arid
CAR	Environmental Rural Registry
CF	Carbon Footprint
CL	Crop-Livestock
CO2eq.	Carbon dioxide equivalent
COP	Conference of the Parties
CP	Crude Protein
Cs	Humid Subtropical With dry summer
CV	Coefficient of Variation
Cw	Humid Subtropical With dry winter
DDF	Directional Distance Function
DEA	Data Envelopment Analysis
DMI	Dry Matter Intake
EF	Emission Factor
Embrapa	Brazilian Agricultural Research Corporation
EP	Eutrophication Potential
FD	Fossil Depletion
FPCM	Fat and Protein Corrected Milk
FU	Functional Unit
GDP	Gross Domestic Product
GE	Gross Energy
GHG	Greenhouse Gas
GWP	Global Warming Potential
GWP*	Global Warming Potential Star
GWP_{100}	Global Warming Potential 100 years time horizon
HDF	Hyperbolic Distance Function
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
K	Potassium
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment

LMIC	Low- and Middle-Income Country
LR	Legal Reserve
LW	Liveweight
MC	Monte Carlo analysis
N	Nitrogen
P	Phosphorus
PCA	Principal Component Analysis
SDG	Sustainable Development Goal
SI	Supporting Information (Annex)
SLCP	Short-lived Climate Pollutants
SM	Supporting Material (Annex)
VS	Volatile Solids
WD	Water Depletion

Chapter 1

1 General introduction

Agricultural systems form the basis of the economy of many low- and middle-income countries, playing a pivotal role in providing food security, employment and safeguarding livelihoods in rural areas. Moreover, global societal trends (e.g., increasing population, urbanisation, income growth and shifts in dietary patterns) will continue to drive the demand for agricultural products over the next decades, increasing even further the importance of agricultural systems for maintaining global food security and human well-being. Notwithstanding, there are increasing concerns regarding the environmental externalities produced during farming and husbandry, and their effects on human health and the natural environment. Consequently, developing, improving and implementing more sustainable agricultural systems has become a major goal for policymakers, researchers and farmers around the world (Tilman et al., 2011; UN, 2015a; Willett et al., 2019). This dissertation presents empirical evidence to support the sustainable development of dairy and paddy fields production systems in Brazil.

The remaining of this chapter presents a brief overview of the development of the agricultural sector in Brazil and its actual status, focusing on how recent international environmental agreements influenced the national policy for the agricultural sector. In the sequence, we provide an overview of dairy and paddy field-based production in the country and close by presenting the research objectives.

1.1 Context of the research

The Brazilian agricultural sector evolved considerably since the 1950's, when the country was a net importer of agri-food products. The reliance on imported staple foods as well as the global political and economic instabilities during the cold war period led Brazil to invest in the national agricultural sector. Despite historically being an agricultural country, for the first time, the Brazilian government directed considerable investments to crop and animal sciences. Initially, these investments were concentrated in the states of São Paulo (SP) and Rio Grande do Sul (RS), later expanding to other states (Figure. 1) (Dias et al., 2016; Fishlow and Vieira Filho, 2020). The establishment of the Brazilian Agricultural Research Corporation (Embrapa) in 1973 is considered a milestone for expanding agricultural research to other regions (Fishlow and Vieira Filho, 2020).

Beneficial spillovers also occurred with investments in the creation of universities and the expansion of national advisory services institutions. Among others, the most influential developments were the genetic improvement of animal and plant breeds and the development of soil management practices, such as pH correction, no-tillage production and biological nitrogen fixation. These technologies allowed a significant increase in land productivity and the expansion of production into the savanna Cerrado and the Amazon biome. Thus, after a half-century, the agricultural sector has become a fundamental part of the national economy, accounting for 26% of the national gross domestic product (GDP) in 2020 (Cepea-CNA, 2022). Food prices reduced significantly over the years[2] and Brazil has become self-sufficient in primary foodstuffs to assure food security for the growing population. Moreover, Brazil is also a leading exporter of several agri-food products (e.g., orange juice, coffee, sugar, cellulose, meats) (Fishlow and Vieira Filho, 2020).

The need for regionalised and crop- and livestock-specific research resulted in the development of some state-of-the-art agricultural technologies. These developments improved the productivity of land and socioeconomic standard of farmers in some regions and production chains. However, it limited the availability and adoption of feasible technologies in other regions[3] and for non-prioritised cultures (Fishlow and Vieira Filho, 2020; Vieira Filho, 2013). In addition to the geophysical and climatic diversity, low technology diffusion, lack of proper funding and market access led to a large social disparity in the country – not only between regions but also within regions and production chains. For instance, in 2006, almost 70% (3.2 M) of the farms in the country produced a monthly gross value of production which classified the household as being in extreme poverty. Farms classified as *low income* and in *extreme poverty* were responsible for 15% of the annual gross value produced in the agricultural sector but composed 90.5% of the farms in the country (Fishlow and Vieira Filho, 2020).

[2] In the period between 1974-2012, food prices decreased by about 75% in the country (Fishlow and Vieira Filho, 2020).

[3] This could be expected given the geophysical and climatic diversity present in the Brazilian territory. Brazil has an area of 8,515,770 km², and is covered by six biomes, and at least 11 Köppen's climate types (Alvares et al., 2013; IBGE, 2021).

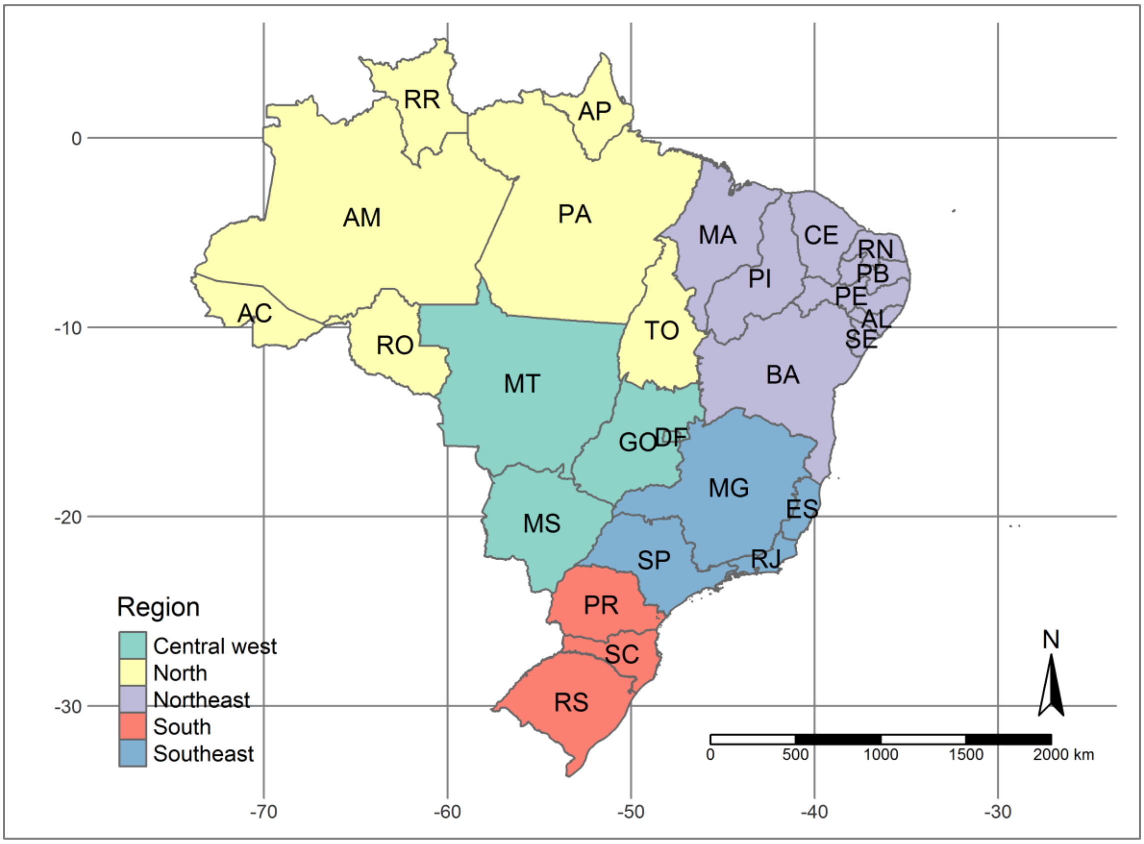

Figure 1-1. States and regions of Brazil.
States: AC: Acre, AL: Alagoas AP: Amapá, AM: Amazonas, BA: Bahia, CE: Ceará, ES: Espírito Santo, GO: Goiás, MA: Maranhão, MT: Mato Grosso, MS: Mato Grosso do Sul, MG: Minas Gerais, PA: Pará, PB: Paraíba, PR: Paraná, PE: Pernambuco, PI: Piauí, RJ: Rio de Janeiro, RN: Rio Grande do Norte, RS: Rio Grande do Sul, RO: Rondônia, RR: Roraima, SC: Santa Catarina, SP: São Paulo, SE: Sergipe, TO: Tocantins, DF: Distrito Federal.

According to the last national census (IBGE, 2018), currently, the country has more than 5.1 million farms which occupy 41% of the national territory (351,289,816 ha). From this area, 63 Mha (18%) are dedicated to crops and orchards, 159 Mha (45%) are dedicated to pastures, and the remaining 37% is dedicated to native vegetation, forestry and other uses. In terms of farm size, half of the producers (50.2%) in the country own 10 ha or less, and 31.3% own medium-size farms with areas between 10 and 50 ha. The national census also shows that only 15% of the farms in the country relied on external sources of financing, with half of them (53%) coming from governmental credit programmes. In terms of education, 7.3% of the farmers have technical, university or graduate course degrees. However, more than half of the farmers never went to school or only attended elementary school. In addition, 23% of the farmers declared being illiterate (IBGE, 2018).

An important concept in the Brazilian agricultural sector is the concept of Family Farm (*Agricultura Familiar*). According to the Brazilian Federal Law number 11,326[4] from 2006, to be classified as family farmer a person must possess all the following characteristics. First, the size of the farm must not be larger than four fiscal units (*módulos fiscais*), which varies in size across municipalities according to the land productivity capacity (e.g., in the Northern region a fiscal unit ranges from 90 to 110 ha, while in the southern region it ranges from 5 to 40 ha). Second, the main source of household income must be derived from agriculture. Third, Family Farms primarily use household labour for on-farm activities. Fourth, the farm must be managed by a member of the household (Brasil, 2006). This characterisation is important since it drives most governmental policies for the development and investments in rural areas.

Despite providing incentives to farmers to expand crop and livestock production over unexplored native areas in the past, the Brazilian government adopted a stringent sparing approach in terms of environmental policy. For instance, the Brazilian Forest Code from 1965 determined the adoption of Legal Reserve areas (LR) and Permanent Preservation Areas (APP). LR represents land that must be spared with native vegetation in every rural property (farm). The spared area should correspond to 20% of the farm, except for the Legal Amazon region, where the spared area should be 35% and 80% for farmers operating in the savanna Cerrado and Amazon Biome, respectively. APPs are areas of high ecological value and also should be conserved. These areas are represented by springs, water bodies and their surroundings, hilly areas, lowlands and mangroves (Brasil, 2012a). Unfortunately, lack of compliance by farmers and weak law enforcement led to some degree of exploitation of RL and APPs areas in the country. The Forest Code was improved in 2012, leading to several amendments (Brasil, 2012a). While the new Code maintained the regulation regarding LR and APPs, the strict sparing of such areas was relieved by the possibility of sustainable management under exceptional circumstances. Conversely, farmers that recently illegally exploited LR or APPs should compensate for it, either by restoring the area or acquiring conserved area with the same size. Moreover, the new Forest Code also included important features related to sustainable agricultural practices and ecosystem services preservation in the national territory. Brazil made notable progress through the new Code with the creation of the Environmental Rural Registry. (*Cadastro Ambiental Rural-CAR*). The CAR requires that all

[4] http://www.planalto.gov.br/ccivil_03/_ato2004-2006/2006/lei/l11326.htm

landowners in the country conduct and provide a detailed geospatial inventory of the farms, identifying LR, APPs, and all land use categories at the farms. The CAR should support strategical land use planning, law enforcement and environmental regularization[5].

In addition to deforestation, relevant externalities from the agricultural sector recently became evident in Brazil. Among the most critical issues is the degradation of cropland and pastureland. Despite recent advances in soil regeneration practices, it is estimated that at least 60 Mha of agricultural land in Brazil suffers from some level of degradation. Remarkedly, half of the area dedicated to pastures suffers from some degradation, with 18.2 Mha located in the Cerrado biome (Pereira et al., 2019). Soil degradation is a major issue since it leads to lower yields and increases production costs, consequently reducing farm productivity. Besides, agricultural soil degradation also leads to pollution of water bodies with nutrients and pesticides through runoff and leaching (Nemecek and Schnetzer, 2012). This raises particular concerns in microregions with high livestock density and intensive crop production. Moreover, the Brazilian agricultural sector is responsible for 80.7% of the country's demand for water (Embrapa, 2018). Water availability is an increasing issue in many agricultural regions of Brazil due to increasing drought periods and competition for water usage (Brazil, 2021a; Cunha et al., 2019; Theisen et al., 2017). Furthermore, greenhouse gases (GHG) emissions are currently among the most concerning externalities produced by the agricultural sector. In contrast to other externalities, which cause local or regional impacts, GHG emissions can cause global warming. Global warming affects the global climatic system, increasing the incidence of extreme weather events and accelerating climate change (IPCC, 2019a).

Increasing scientific evidence and awareness regarding the damage extreme weather conditions and climate change can cause to anthropogenic and natural systems[6] led to major international agreements and commitments to reduce GHG emissions and avoid global warming. The United Nations Framework Convention on Climate Change (UNFCCC) is the steering institution guiding global actions towards this common goal. Development of instruments and

[5] By January 2018, the Environmental Rural Registry of 4,819,574 farms were uploaded in the national CAR database, representing 95% of the farms in the country (Embrapa Territorial, 2020).

[6] …. "*Climate change is one of the greatest challenges of our time and its adverse impacts undermine the ability of all countries to achieve sustainable development. Increases in global temperature, sea level rise, ocean acidification and other climate change impacts are seriously affecting coastal areas and low-lying coastal countries, including many least developed countries and small island developing States. The survival of many societies, and of the biological support systems of the planet, is at risk.*" (UN, 2015a).

evaluation of the progress of the signatory countries (Parties) is conducted annually at the Conference of the Parties (COP) (UNFCCC, 2022). The decisions ruled in the conferences directly affect the agricultural sector, given that agriculture, forestry and other land use (AFOLU) are responsible for 23% of all anthropogenic emissions of CO_2eq.[7] (IPCC, 2019a).

In Brazil, however, the share of contribution from the agricultural sector to the national CO_2eq. emissions is much higher. Overall, Brazil is responsible for emitting 2.16 Gt CO_2eq. per year, representing about 2.9% of the global emissions from anthropogenic activities (Climate Watch, 2021; SEEG, 2020). Markedly, 72% of all emissions in the country are directly or indirectly related to the agricultural sector. About 28% of the emissions are directly linked to farming and husbandry, with cattle and paddy fields dominating as the most important source. Another 44% comes from land-use change activities, mainly illegal deforestation (SEEG, 2020). These values rank Brazil among the top ten contributors to CO_2eq. emissions in the world, and one of the countries with the largest share of emissions coming from AFOLU.

Negotiations and commitments[8] assumed by Brazil in the COPs have significantly influenced the national policy for sustainable development over the years. The commitments assumed by the Brazilian government in the COP 15, in 2009, resulted in the implementation of the National Policy on Climate Change[9] in the same year and the amendments to the Forestry Code in 2012, as already described. Specific actions for the agricultural sector was set by the publication of the national *Plan for Mitigation and Adaptation to Climate Change for the Consolidation of a Low Carbon Economy in Agriculture* (ABC Plan) in 2010 (Brasil, 2012b). The first phase of the ABC Plan, which ended in 2020 mainly focussed on fostering research of more sustainable and adaptative technologies, e.g., (Brazil, 2021b, 2021c). The funding and promotion of technologies, programmes and agricultural practices that were already recognised for having some synergic outcomes regarding farm socioeconomic and environmental benefits. Some of the main actions in the plan were: improvement of 15 Mha of degraded pastures; support of the implementation of 4 Mha of integrated agricultural systems (e.g., crop-livestock and crop-livestock-forestry); expansion

[7] If not stated otherwise, CO_2eq. refers to values derived based on the GWP 100-AR5 (Myhre et al., 2013). In this dissertation Carbon Footprint, Global Warming Potential and CO_2eq. are used interchangeably to describe the warming potential of the diverse greenhouse gases according to the Global Warming Potential 100-year horizon.

[8] In the first intended Nationally Determined Contribution (iNDC) Brazil committed to reducing its greenhouse gas emissions in 2025 by 37%, compared with 2005. Additionally, the country committed to reducing its emissions in 2030 by 43%, compared with 2005.

[9] http://www.planalto.gov.br/ccivil_03/_ato2007-2010/2009/lei/l12187.htm

of no-tillage crop production in 8 Mha; adoption of nitrogen 5.5 Mha; increase forestry areas by 3 Mha; implementation of manure managements systems to treat 4.4 Mm^3 of manure. Moreover, Brazil committed to reducing illegal deforestation by 80% in the Amazon and 40% in the Cerrado Biomes (Brasil, 2012b).

Another important step was taken during the COP 21 held in 2015 in Paris, where an international treaty on climate change was negotiated between the parties leading to the Paris Agreement. The Paris Agreement settled commitments to strengthen actions to avoid and adapt to climate change while supporting sustainable development and eradicating poverty[10] (UN, 2015b). The Brazilian commitments were reinforced with the ratification of the Paris Agreement in 2016. By the end of the first cycle of the ABC Plan in 2020, mitigation in the agricultural sector surpassed 55% of the initial mitigation target for the sector (Brazil, 2022). However, in some areas the advance was not as good as expected, e.g., in the reduction of illegal deforestation (SEEG, 2020) and in the adoption of sustainable practices in some regions (Cortner et al., 2019).

Recently during the COP 26, held in Glasgow in 2021, the Brazilian government confirmed its commitment to reducing GHG emissions in 2025 by 37% and increasing the commitment to reduce GHG emissions in 2030 by 50%, compared with 2005 (Brazil, 2022). The country also set the goal of reaching neutrality by 2050 (Brazil, 2022) and signed the Global Methane Pledge, which aims at reducing global methane emissions by at least 30 percent from 2020 levels by 2030 (EU, 2021). For the agricultural sector, Brazil presented the second cycle of strategies by launching the ABC+ Plan (Brazil, 2021d). Since the problems tackled in the first cycle of the ABC Plan are far from being completely solved, the ABC+ expanded the actions in these key areas[11]. Moreover, the

[10] More specifically in its article 2 the Paris Agreement sets the following objectives: "*(a) Holding the increase in the global average temperature to well below 2°C above pre-industrial levels and pursuing efforts to limit the temperature increase to 1.5°C above pre-industrial levels, recognizing that this would significantly reduce the risks and impacts of climate change;*
(b) Increasing the ability to adapt to the adverse impacts of climate change and foster climate resilience and low greenhouse gas emissions development, in a manner that does not threaten food production; and
(c) Making finance flows consistent with a pathway towards low greenhouse gas emissions and climate-resilient development."(UN, 2015b).

[11] The actions of the ABC+ were segmented into seven broad categories: "*1 – Maintaining stimulus for the adoption and maintenance of conservationist and sustainable agricultural production systems, with increased productivity and revenue, and greenhouse gas emissions control.*
2 – Strengthening technology transfer and diffusion, training, and technical assistance.
3 – Stimulate and support applied research for the development or improvement of sustainable production systems, practices, products, and processes focusing on increasing resilience, productivity, and revenue, and controlling greenhouse gas emissions.

ABC+ Plan also introduced some important new features. First, it promotes stronger actions towards adaptation and resilience building of farming systems against climate change. Second, the Plan reinforces the connection with the broader framework of sustainable development goals (SDG), mainly between SDG 13 (*Take urgent action to combat climate change and its impacts*) and SDG 2 (*End hunger, achieve food security and improved nutrition and promote sustainable agriculture*). And third, it promotes strategies that integrate a synergic outcome between production, rural landscape conservation, and enhancement of farm biodiversity (Brazil, 2021d).

1.1.1 The case of dairy farming

The Brazilian dairy farming is evolving fast and has become one of the main activities of the national agricultural sector. This evolution is marked by a reduction in the number of dairy farms at the country level, followed by an increase in herd size and productivity – a trend similar to that observed in most countries with developed dairy farming (Clay et al., 2020). According to the last agricultural census (IBGE, 2018), during the period 2006-2017, the number of dairy farms in the country decreased from 1.35 M to 1.17 M farms, while the number of milked animals reduced by 9%, from 12.7 M to 11.5 M cows; conversely, milk production increased by 70% in the same period. In 2020, The national milk production reached 36.5 Mt, generating around US $12 billion in value for farmers (Embrapa, 2021; Rocha et al., 2020). This volume placed Brazil as the third-largest dairy milk producer in the world (Embrapa, 2021; Rocha et al., 2020). Figure 2 shows the evolution of the number of milked cows and milk production in Brazil between 1974 and 2020.

The dairy sector is also relevant for generating employment in rural areas. According to the last agricultural census (IBGE, 2018), some 81% of the 1.17 million Brazilian dairy farms are Family Farms. Altogether, more than 4.7 M people are enrolled in milk production activities around the country. Nevertheless, the significant number of farmers leaving dairy farming over the last decade suggests many producers dissatisfaction with the activity. This high number of farmers

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4 – Create and strengthen mechanisms which make it possible to value products that adopt sustainable production systems, practices, products, and processes.

5 – Fostering, increasing and diversifying economic, financial, and fiscal sources and instruments linked to sustainable production systems, practices, products, and processes.

6 – Improving the ABC+ information management system to make effective monitoring, report and verification, and monitoring and evaluation of its action and results.

7 – Fostering landscape integrated agriculture, so as to incentivize environmental regularization of rural properties and sustainable production in agricultural areas." (Brazil, 2021d).

leaving the sector can be associated, to some extent, with price instability and the low profitability faced by the producer in the country (Beber et al., 2019; Embrapa, 2021).

Furthermore, dairy farming supports national food security. Overall, the national annual per capita milk consumption is ~166 L, higher than most countries (Siqueira, 2019). Milk has a particular contribution to food security in rural areas. For instance, more than one-quarter of the milk produced in the country does not enter the dairy processing industry, indicating that it is either consumed directly by the household or commercialised locally through short supply chains (IBGE, 2018).

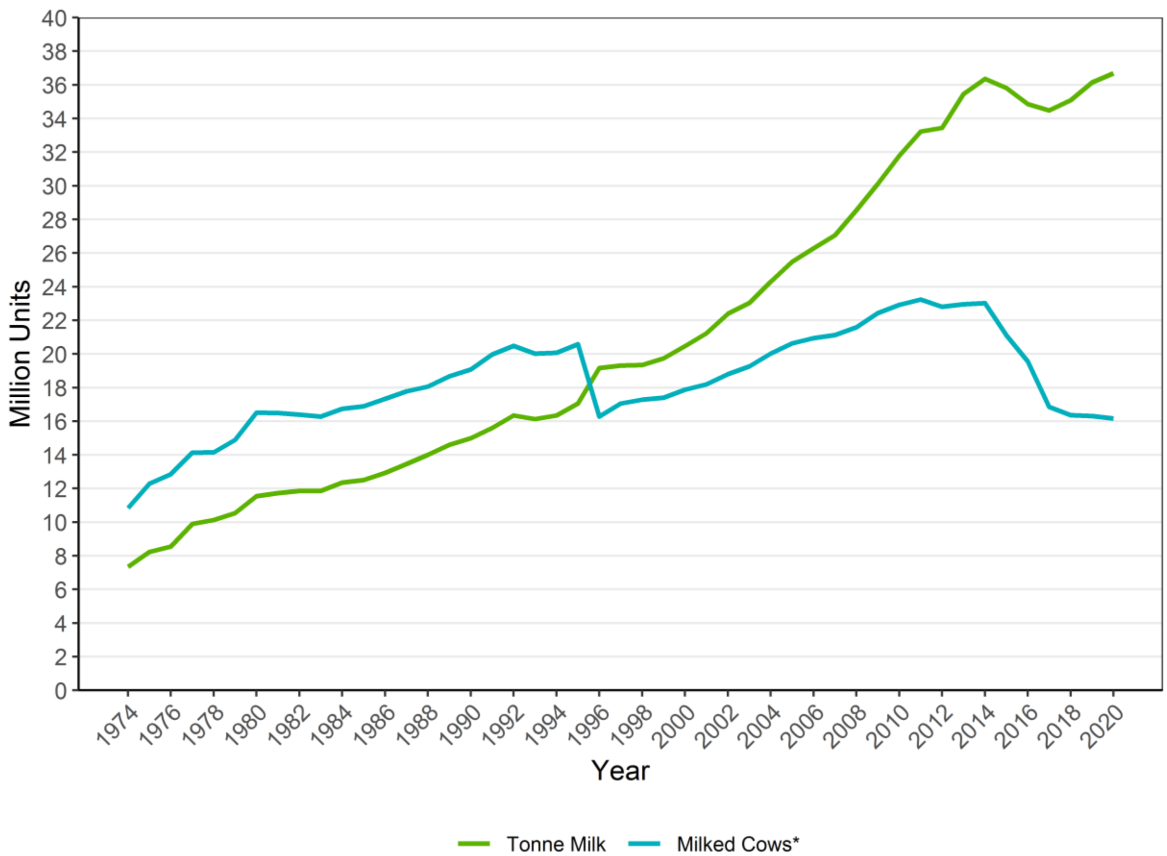

Figure 1-2. Evolution of the number of cows milked (million cows per year) and milk production (million tonnes per year) in Brazil.

Based on (IBGE, 2022a).
* In 1996 the dairy herd was updated based on the agricultural census conducted in that year, which explains the sharp drop in milked cows in that year.

The Brazilian dairy farming is heterogeneous, with factor endowment, production systems, and management practices varying considerably within and between regions and states. In terms of production, four states are responsible for 70% of the national production. Minas Gerais (MG) leads with 27% of the national share. MG is followed by Paraná (PR), Rio Grande do Sul (RS), Goiás (GO) and Santa Catarina (SC), with 13%, 12%, 9% and 8.8% of the national production, respectively (Figure 3). The most common production systems are pasture-based and semi-confined systems (Carvalho et al., 2021; de Léis et al., 2015). The country has a small share of producers who use modern technologies and high-productive breeds, allowing them to operate with yields similar to traditional milk-producing countries. However, the majority of the farms rely on non-specialised breeds and rather rudimentary technologies and management practices, thus presenting low productivity. This is clearly revealed by the average milk production per cow in the country (2.6 t yr^{-1}) (IBGE, 2018). Moreover, animal productivity also varies considerably across states (Figure 3). The southern states present the highest productivity per cow (> 3.5 t yr^{-1}). This region is benefited from its mild subtropical climate, which favours the use of European dairy breeds and temperate pastures in the winter season. Conversely, states in the tropical and Semi-arid regions (North and Northeast) face constraints in this regard, which contributes to the very low productivity per cow in these regions (< 1.5 t yr^{-1}) (IBGE, 2018; Rocha et al., 2020). Furthermore, the heterogeneity in dairy farming across the country is further magnified by the lack of a stable processing industry sector, which eventually compromises the competitiveness of the whole dairy supply chain (Beber et al., 2019). This heterogeneity is partially attributed to the lack of long term governmental strategic planning and incentives to the dairy sector as a whole (Beber et al., 2019, 2018; Novo et al., 2013).

Dairy farming is associated with several environmental externalities. Rearing dairy cows contributes to the depleting of scarce resources, such as water, fossil fuel and nutrients (e.g., Phosphorus). Moreover, the production and use of farm inputs, rearing animals, and related activities are responsible for the emission of nutrient rich compounds to soil, water and atmosphere (Bartl et al., 2011; De Boer, 2003). Depending on the intensity and fate of these compounds, environmental impacts may emerge. The most concerning externalities produced at dairy farms linked to such emissions are the surplus of Nitrogen (N) and Phosphorus (P) compounds as well as emissions of GHG, mainly methane, nitrous oxide, and carbon dioxide (IPCC, 2019a; Nemecek and Schnetzer, 2012). Leaching and runoff of N and P is a primary concern in regions with a dense population of livestock, since these nutrients may lead to groundwater contamination (e.g., nitrate), acidification of the environment and eutrophication of water bodies (Cederberg and Mattsson, 2000; De Boer, 2003; Mu et al., 2017; Palhares et al., 2012; Zumwald et al., 2018).

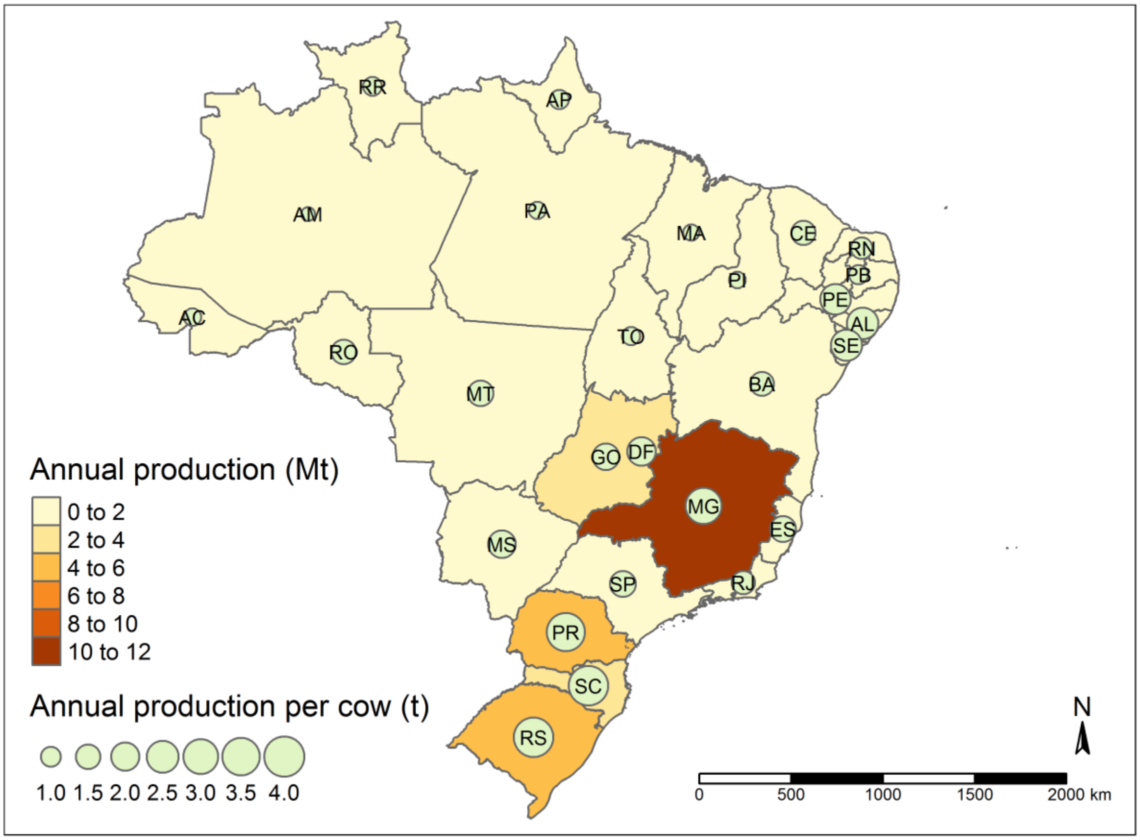

Figure 1-3. Milk production in Brazil in the year 2020 per state in million tonnes (Mt), and per cow, in tonnes (t).

Based on (IBGE, 2022a).

Furthermore, emissions of GHG along the dairy production chain are substantial (Herrero et al., 2016; IPCC, 2019a). This is particularly important for countries with systems presenting low productivity, which is the case for most dairy farms in Brazil (Gerber et al., 2013, 2011; SEEG, 2020). Gerber et al. (2011) show that the carbon footprint intensity of milk production is inversely related to animal productivity (Figure 4). Besides, there is also evidence that high CF intensity is also associated with farms with lower economic returns (Jayasundara et al., 2019; O'Brien et al., 2015).

The low productivity of the Brazilian dairy herd aligned to its expressive size makes the country a major contributor to the global emissions of GHG in the dairy sector. The national dairy herd emitted 53.8 Mt CO_2eq. in 2019, representing about 2.5% of the global emissions form dairy farming and 9.3% of the national emissions from agri-food sector (SEEG, 2020). Moreover, methane stands out as the most important GHG emitted in dairy farming, contributing with the largest share of the CO_2eq. emission from dairy farms. In Brazil, methane emissions account for almost three quarters of the total CO_2eq. emissions from the national dairy farms (SEEG, 2020).

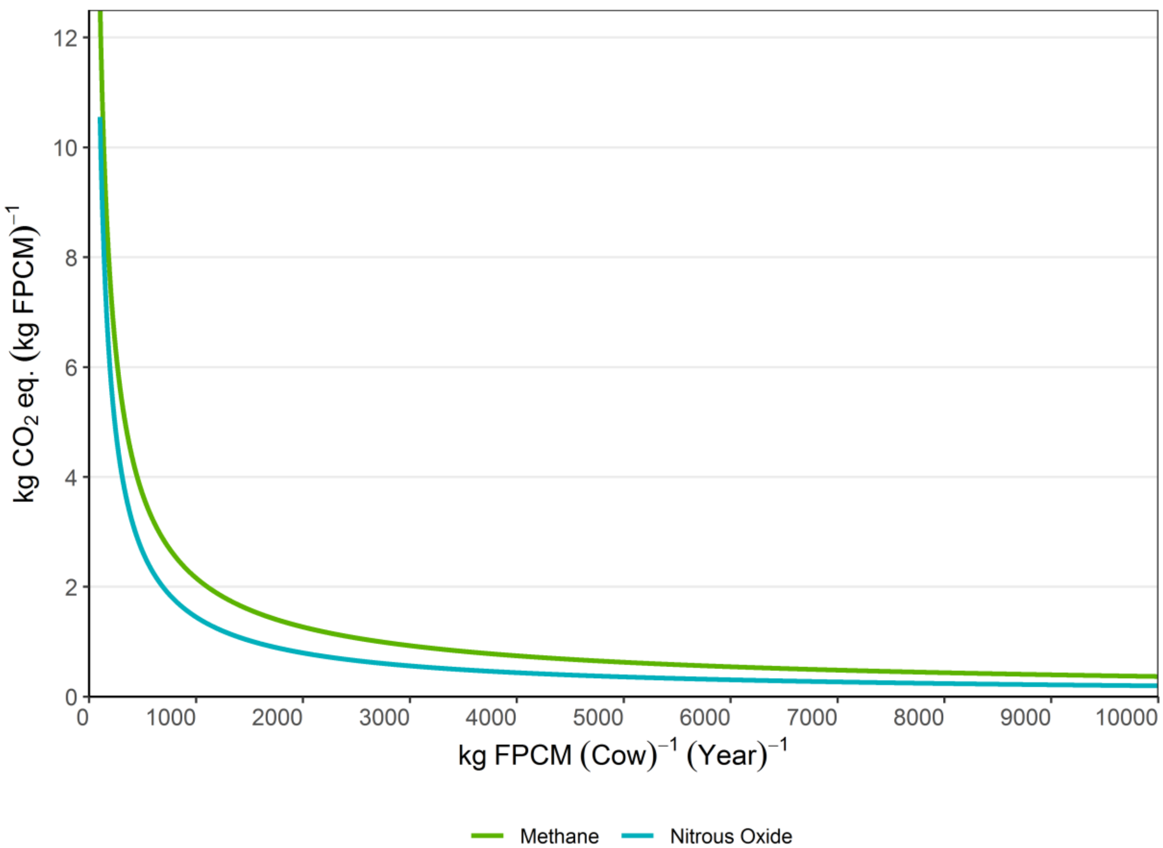

Figure 1-4. Relationship between cow productivity and CO_2eq. intensity from methane and nitrous oxide.

Adapted from (Gerber et al., 2011).

In general, the sustainable intensification[12] of low-productive dairy systems is suggested as a strategy to improve the profitability and reduce the carbon footprint intensity of dairy farms

[12] The most spread view is that sustainable intensification agriculture should increase yields without advancing over new land areas and without causing adverse environmental impacts (IPCC, 2019a; Rockström et al., 2017; Willett et al., 2019). Moreover, there is also a call to consider the broad view of sustainable governance in the sustainable intensification framework. Within this framework, sustainable intensification agriculture should focus on"... *managing natural capital for long-term productivity and social–ecological resilience at field, watershed, and*

(Gerber et al., 2013; IPCC, 2019a; Jayasundara et al., 2019; O'Brien et al., 2015). However, distinguishing the structure and carbon footprint intensity of farms operating in a region is crucial to implementing tailored sustainable intensification actions (Gerber et al., 2013; González-Quintero et al., 2021; Ortiz-Gonzalo et al., 2017). In the Brazilian context, there are case studies investigating the environmental impacts of dairy farming (Carvalho et al., 2021; de Léis et al., 2015; Ribeiro-Filho et al., 2020) and the structure of dairy farms (Balcão et al., 2017; Bánkuti et al., 2018; de Mendonça et al., 2020). However, studies considering the diversity of dairy farms at the regional level while accounting for their carbon footprint are still lacking in the literature.

Furthermore, environmental efficiency analysis has emerged as a robust approach to evaluate the joint production of desirable and undesirable outputs of agricultural systems. Moreover, environmental efficiency analysis also allows for the calculation of shadow prices, wich can be understood as abatement costs for reducing externalities in terms of foregone production. Therefore, this approach is suitable to evaluate and guide further actions for projects intended to improve farm sustainability, such as those promoted by the ABC+ Plan and other sustainable development programmes. Environmental efficiency analysis of dairy farming was applied in developed countries to investigate nitrogen (Adenuga et al., 2019; Mamardashvili et al., 2016; Skevas et al., 2018), phosphorus (Adenuga et al., 2020; March et al., 2016) and GHG (CO_2eq.) (Le et al., 2020; Njuki et al., 2016; Njuki and Bravo-Ureta, 2015; Wettemann and Latacz-Lohmann, 2017) emissions as undesirable outputs. However, environmental efficiency analysis of dairy farming under the Brazilian conditions still remains a gap in the literature.

1.1.2 The case of paddy fields agricultural systems

Rice is one of the most consumed staple foods in Brazil[13]. With almost all rice consumed in the country coming from the national harvest, Brazil stands out as the largest rice producer outside Asia and is among the top 10 global producers. The volume of production in the country increased modestly over the years, advancing from 6.8 Mt in 1974 to 11.1 Mt in 2020. Conversely, land use decreased significantly – the harvested area peaked at 6.6 Mha in 1976 and decreased gradually to 1.7 Mha in 2020. Thus, land productivity reached 6.6 tonne ha^{-1} in 2020, representing

regional scales, in agricultural systems that operate within planetary boundaries to safeguard Earth system."(Rockström et al., 2017).

[13] Daily rice consumption per capita in Brazil is around 131g, thus supplying the largest share of dietary energy ingested by the Brazilian population (IBGE, 2020).

a 450% increase over the last 46 years (Figure 5). Nevertheless, seasonal fluctuations in the harvested area occurred across the years, mainly explained by seasonality in the fallow of land and losses due to bad climatic conditions (IRGA, 2022; Theisen et al., 2017). Since 92% of the rice produced in the country is cultivated in flooded paddy fields concentrated in the southern region, losses due excess or lack of rain in the region significantly affect the national harvest.

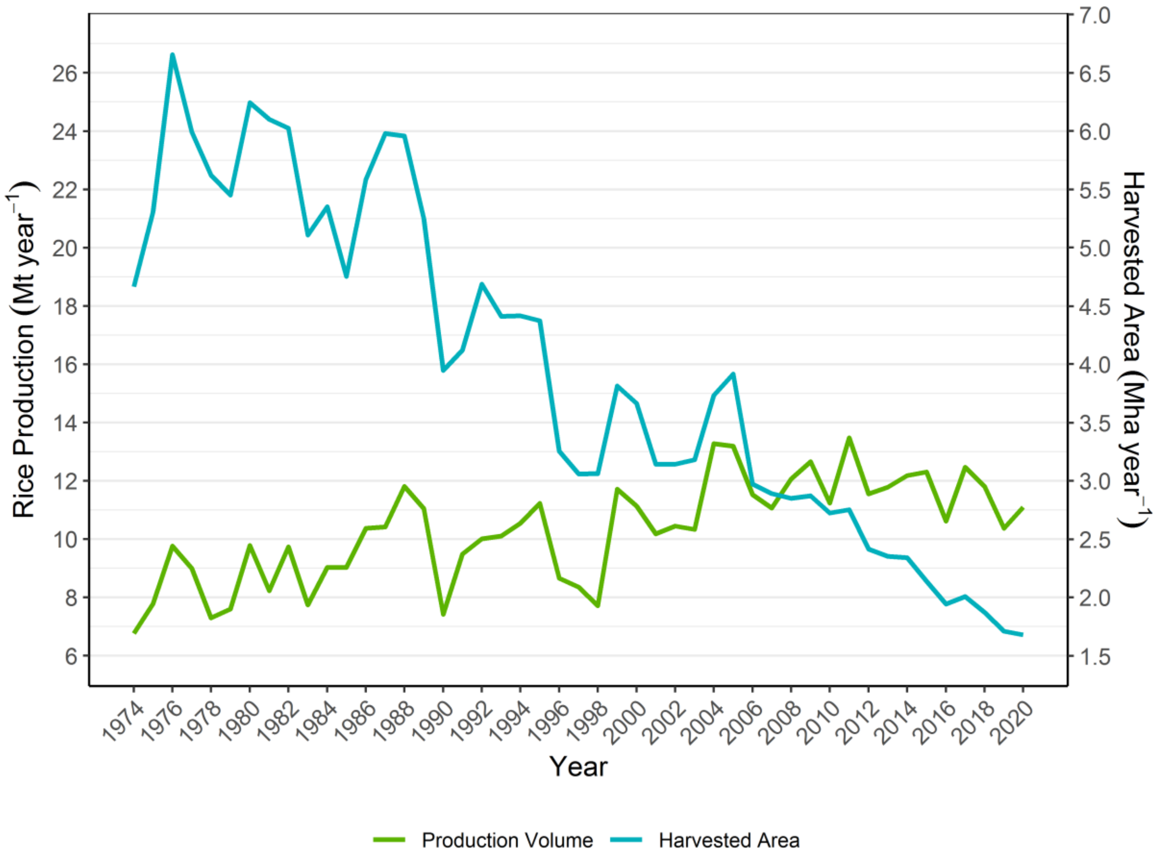

Figure 1-5. Evolution of rice production in Brazil in million tonnes (Mt) and harvested area in million hectares (Mha).

Based on (IBGE, 2022b).

The subtropical climate and availability of lowland areas in southern Brazil were decisive for the development of flooded paddy rice production in this region. However, the significant gain in productivity was mainly driven by long-term research and development of technologies. Research to improve rice productivity was pioneering in Brazil. Lowland rice research visioning dates back to 1938, with the creation of the Rio Grande do Sul Rice Institute (IRGA). IRGA was created to defend the interests, develop research, and provide advisory services to rice producers in RS (IRGA, 2022) [14]. Currently, the southern region is responsible for producing 82% of the national rice supply, with RS accounting for 70% of this total (IBGE, 2022b). The improved seeds produced by IRGA are planted in 64% of the cultivated area in RS. And the average yield in the region is among the highest in the world (>8 t ha^{-1}). Figure 6 presents the distribution of harvested area and productivity of rice in Brazil.

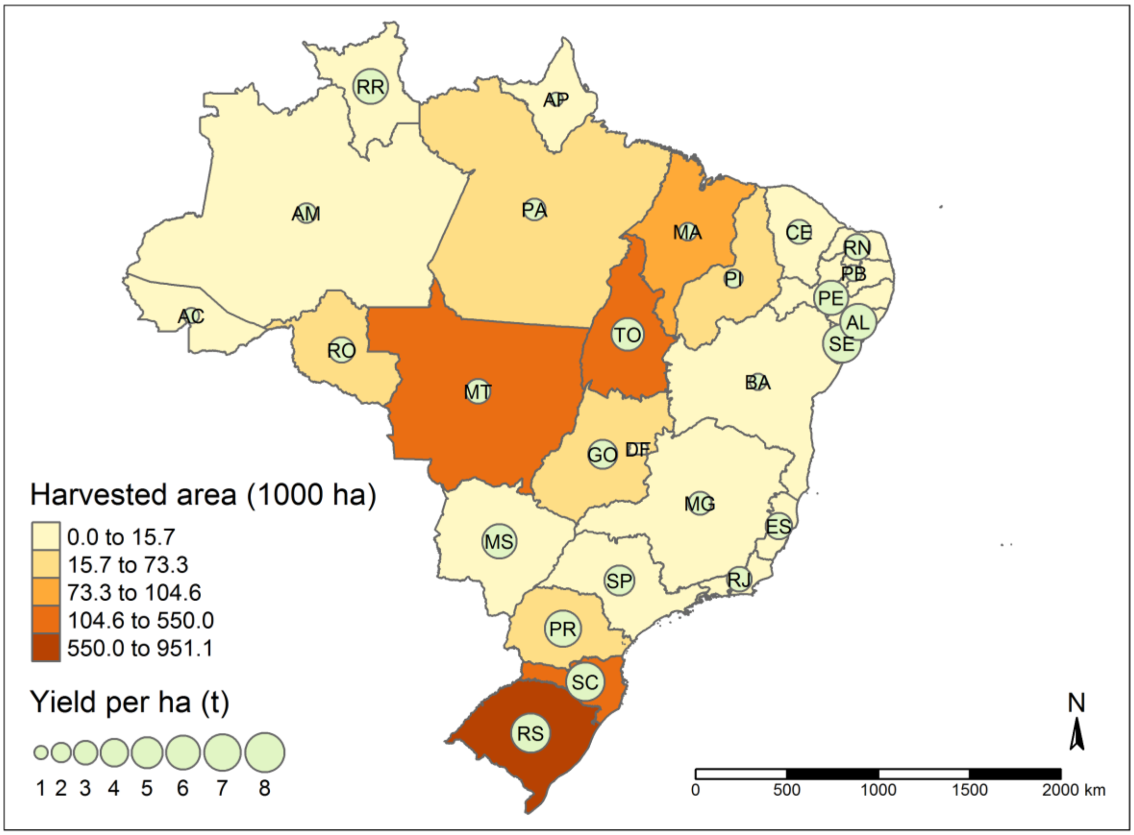

Figure 1-6. Rice area harvested by state in 2021, in thousand hectares, and yield per ha, in tonnes (average of crop seasons 2019-2021).

Based on (IBGE, 2022b).

Rice also plays an essential socioeconomic role in Brazil, with prominence to RS and SC. In 2021, the national rice production accounted for more than US $3.2 Billion in farm income for Brazilian producers. In RS, rice production represents 1.6% of the State's GDP and generates employment for around 372,000 people (SOSBAI, 2018). Similarly, rice is also an important

[14] As discussed earlier, research and actions developed by State-based research institutes and later Embrapa was fundamental to the development and diffusion of technologies for rice production in other states of the country.

source of income in SC. Moreover, in this state 90% of the producers are Family Farmers, and in total there are more than 30,000 people enrolled in rice production activities (SOSBAI, 2018).

Contrasting paddy field production systems were developed for rice cultivation in southern Brazil. The most common production systems used in RS are the minimal tillage, conventional tillage and pre-germinated systems, occupying 61%, 30%, and 9% of the cultivated area, respectively (SOSBAI, 2018). Contrastingly, in SC, almost all production is conducted in the pre-germinated system (SOSBAI, 2018). Production systems differ mainly based on soil management practices, sowing and irrigation strategies (Nunes et al., 2016; SOSBAI, 2018; Theisen et al., 2017). Regarding seasonal cycles, in the most traditional practice, rice paddies are cultivated for four to five growing seasons and then fallow to break the growing-cycle of competing weeds. Rotation between rice and extensive beef cattle production characterises a second common practice in RS. Cattle ranching is a widespread tradition in this state, such that this crop-livestock (CL) rotation offers an excellent opportunity to use fallow land to increase the forage supplies for cattle. Intensification of this CL system is reached by reducing fallow periods and sowing temperate mixed pastures for cattle (Theisen, 2017). More recently, soybean has been introduced in the rotation with rice in lowland areas. The expansion of soybean into areas of rice increased by 205% over the last 10 years, reaching 341,188 ha in 2021 (IRGA, 2022). The diversification of rice-based cropping systems in RS is primarily explained by the lack of profitability of rice, increasing prices of soybean and incentives promoted by the ABC Plan for the adoption of more sustainable agricultural systems (Coltro et al., 2017; SOSBAI, 2018; Theisen et al., 2017).

Rice production is also associated with environmental externalities. The use of flooded paddy fields and the intensity at which rice is cultivated in southern Brazil distinguishes it from other crops. Paddy fields are anthropic land in which the natural landscape passes a high level of modification for the construction of paddies and water channels. Field irrigation during the growing season demands a large volume of water, turning it scarce in some regions during this period (Coltro et al., 2017; Theisen, 2017). Moreover, the production and transportation of farm inputs contribute to the emissions of harmful substances and the depletion of finite resources upstream of the supply chain. Similarly, machinery operations during rice cultivation burn fossil fuel, also emitting harmful substances into the air. Fertilisation of the fields leads to nutrient losses, which also have the potential of causing the contamination of groundwater, acidification of the environment and eutrophication of water bodies (Cai et al., 2018; Coltro et al., 2017; Nunes et al., 2017).

Similar to dairy farming, GHG emissions are also a concern in rice production. While GHG are emitted along with the production and transportation of farm inputs, the largest share of emissions occurs at the farm during rice cultivation. On the farm, GHG are emitted from fossil fuel burned in machinery, nutrient losses from fertilisers and soil management activities (Cai et al., 2018; Coltro et al., 2017; Nunes et al., 2017). Remarkedly, the flooded environment created during rice cultivation favours the anaerobic decay of organic matter, leading to the production of high quantities of methane (Nunes et al., 2016; Yodkhum et al., 2017; Zschornack et al., 2016). This feature makes flooded rice production a hotspot of GHG emissions in the national agricultural sector. In Brazil, rice is responsible for about 2% of the national emissions from the agricultural sector (~10 Mt CO_2eq.), with methane representing around 75% of this total (Coltro et al., 2017; Nunes et al., 2017; SEEG, 2020). Naturally, most of these emissions originate in southern Brazil due to the large areas occupied by paddy fields in this region.

Improving and developing new paddy field-based agricultural systems are globally suggested as strategies for mitigating environmental impacts from rice production and diversifying farm income (Arunrat et al., 2016; Cai et al., 2018; Coltro et al., 2017; Ramsden et al., 2017). Therefore, understanding the trade-offs between economic and environmental outcomes of emerging paddy field-based systems is crucial to guide farmers in adopting more sustainable farming systems. Several studies investigated the agronomic and environmental performance of paddy fields agricultural systems in RS, e.g., (Bayer et al., 2014; Coltro et al., 2017; Nunes et al., 2017, 2016; Theisen et al., 2017). However, studies applying a set of economic and environmental indicators to evaluate and compare long-term field experiments in this region are still scarce in the literature.

1.2 Research objectives and outline

The main objective of this dissertation is to investigate the economic, environmental and joint performance of dairy and paddy fields agricultural systems in Brazil. The goals are to contribute to the interdisciplinary literature concerning the sustainability of agricultural systems in the country and propose improvements. The dissertation is presented in five chapters. The first one comprises of the general introduction, which is followed by three research chapters and closes with a general conclusion.

Objectives of Chapter 2:

- To investigate the heterogeneity and structure among dairy farms in Paraná, BR.
- To identify strategies that dairy farmers can adopt to reduce the carbon footprint of their farms.

Objectives of Chapter 3:

- To estimate the environmental efficiency and its determinants for pasture-based dairy farms in Minas Gerais, BR.
- To calculate the shadow price of methane emitted in dairy farms.

Objectives of Chapter 4:

- To assess the economic and environmental performance of experimental paddy field-based agricultural systems in the Rio Grande do Sul, BR.

Chapter 2

2 Carbon footprint and mitigation strategies among heterogeneous dairy farms in Paraná, Brazil[15]

Abstract

Dairy production is a vital part of the Brazilian agri-food system, providing food security, employment, and income in rural areas. Nevertheless, rearing dairy cattle leads to greenhouse gases (GHG) emissions which may contribute to global warming and consequently climate change. The remarkable heterogeneity among dairy farms as well as the lack of representative research pose constraints to the development of effective actions to reduce GHG emissions in emerging countries. In this study, we explore a large farm survey to group farms and derive their carbon footprint (CF). Cluster analysis and life cycle assessment are applied to a sample of 911 farms. The results of the analysis categorized the farms into four groups. Statistical comparisons indicated a significant difference in the CF between groups for producing one kg of fat and protein corrected milk (FPCM). The mean CF results ranged from 1.75 kg CO_2eq. (kg FPCM)$^{-1}$ in Group 1 (G1) to 3.27 kg CO_2eq. (kg FPCM)$^{-1}$ in Group 4 (G4). While G1 was composed of larger farms, on average having more access to technologies and technical support, G4 was composed of less specialized producers, owning dual-purpose herds. We also identified and discussed key strategies and management practices that can be adopted by farmers for reducing the CF of dairy farming. Research and policy should strive to accelerate farmers' adoption of intensification technologies and practices, though following sustainable intensification practices that also account for regional socioeconomic development.

Keywords: life cycle analysis, farm typology, carbon footprint, environmental management, GWP star

[15] This chapter was published as: Everton Vogel, Caetano Luiz Beber. Carbon footprint and mitigation strategies among heterogeneous dairy farms in Paraná, Brazil. **Journal of Cleaner Production** 349 (2022) 131404. https://doi.org/10.1016/j.jclepro.2022.131404. CRediT authorship contribution statement: Everton Vogel: Project administration, Conceptualization, Methodology, Data Curation, Formal analysis, Writing – original draft, Writing – review & editing. Caetano Luiz Beber: Investigation, Conceptualization, Writing – review & editing. I would like to thank Dr. Ayobami Adetoyinbo for his comments in the early version of the manuscript during the Doctoral seminar.

2.1 Introduction

Brazilian dairy production has grown significantly in the last decades. It increased from 15.8Mt to 34.5 Mt between 1990 and 2019, placing Brazil among the top five milk producers in the world (Embrapa, 2020; IBGE, 2018). Milk is produced in more than 1.17 million farms across the country (IBGE, 2018) and it is a fundamental part of the Brazilian economy, supporting national food security, providing employment and safeguarding livelihoods in rural areas. In addition to the socioeconomic importance, there is a growing awareness regarding the role of dairy systems for the environmental sustainability. Well-managed dairy systems are recognized for protecting grassland ecosystems, preserving soils, and contribute to nutrient cycling (Kok et al., 2020; Vázquez-González et al., 2021). Conversely, dairy systems may be responsible for depleting scarce resources and producing undesirable outputs that can result in environmental impacts, for example, by using natural resources, such as land, water, and fossil energy; and the excretion of N and P compounds (Bartl et al., 2011; Mu et al., 2017). Moreover, dairy systems are also responsible for greenhouse gas (GHG) emissions, e.g., methane, nitrous oxide, and carbon dioxide. GHG emissions may cause global warming and consequently contribute to climate change, in turn triggering major damage to natural and anthropogenic systems (IPCC, 2019a). Worldwide the dairy cattle herd is responsible for about 30% of all GHG emissions from the livestock sector (~ 2.1 Gt CO_2eq. yr^{-1}) (Gerber et al., 2013; Herrero et al., 2016). As the global demand for dairy products is expected to increase in the coming decades, improving the environmental efficiency of the dairy supply chains became paramount to tackle climate change (IPCC, 2019a; Roe et al., 2019; Willett et al., 2019).

Sustainable intensification of farming systems in low and middle income countries (LMIC) has been suggested as a strategy to increase food supplies while improving the environmental performance in the agri-food sector (IPCC, 2019a; Roe et al., 2019; Willett et al., 2019). Intensification has also been recommended for less specialized dairy farms, which normally display low productivity and high GHG emissions per litre of milk produced (Gerber et al., 2011; Herrero et al., 2016). Dairy systems and management practices among dairy farmers in LMIC are highly heterogeneous. Farmers operating highly specialized farms with housed herds coexist with medium and small-holders, sometimes owning only a few dual-purpose cows reared on pastures (de Léis et al., 2015; Gerber et al., 2013; González-Quintero et al., 2021). Accounting for this heterogeneity during the design of agricultural development strategies is crucial for reaching GHG mitigation targets and avoiding harmful socioeconomic prejudices (Gerber et al., 2013; Ortiz-Gonzalo et al., 2017).

A primary step to improve the environmental efficiency of the dairy sector is to understand the origins and fate of the GHG along the production chain. The Life Cycle Assessment (LCA) approach has been suggested as the standard technique to inventory GHG and calculate the Carbon Footprint (CF) of dairy products (FAO, 2017, 2016; IDF, 2010). LCA has been applied to quantify and understand the sources of GHG from dairy systems in several countries (Seó et al., 2017; Velarde-Guillén et al., 2022). For instance, it was adopted to compare conventional and organic milk production in Europe (Cederberg and Mattsson, 2000; Kristensen et al., 2011; Thomassen et al., 2008); and for comparing the performance of milk produced among small-holders in Peru (Bartl et al., 2011). LCA was also applied to investigate the differences between confined and pasture-based systems in Ireland (O'Brien et al., 2014) and Canada (Arsenault et al., 2009). Case studies conducted in Brazil evaluated: pasture-based, semi-confined, and confined production systems (de Léis et al., 2015); the substitution of total mix ration for pasture in the cows' diet in Southern Brazil (Ribeiro-Filho et al., 2020); and semi-intensive system in Bahia state North-eastern Brazil (Carvalho et al., 2021). Except for Carvalho et al. (2021), only high-yield production modes have been evaluated for Brazilian conditions, thus representing only a small share of milk producers in the country (de Léis et al., 2015).

LCA studies are very data demanding, requiring a significant amount of financial resources and labour to be completed (Thoma et al., 2013). This feature leads researchers to take two different paths to compare the environmental impact of dairy systems. The first relies on in-depth analysis of small samples or scenarios from selected farming systems, e.g., (Bartl et al., 2011; Cederberg and Mattsson, 2000; de Léis et al., 2015; Ribeiro-Filho et al., 2020). A second approach relies on national statistics or surveys, in most cases not specifically designed for conducting LCA studies (Gerber et al., 2011; González-Quintero et al., 2021). The first approach is fundamental for understanding the environmental impacts of specific modes of production while the second allows insights at national or regional level, enabling a better understanding of the heterogeneity among production systems operating in a region.

In this study, we follow the second approach and analyse a large sample of milk producers from Paraná, Southern Brazil. The aim is to gain insights into the structure of the dairy farms operating in Paraná and calculate their carbon footprint. We employ clustering techniques and life cycle assessment in a joint approach. In particular, we intend to categorize groups of farms that represent the wide range of producers operating in the region by characteristics such as their structure, practices, socioeconomic aspects and CF. We then discuss how sustainable intensification and other GHG mitigation strategies can be applied to these different groups. Our study therefore contributes to the literature by broadening the analysis of CF to less specialized dairy farms, where to our knowledge very few studies have been conducted. We expect that this approach can support researchers and extension workers. More specifically, those engaged in implementing climate protection actions in developing countries face a lack of resources to conduct proper field surveys to understand the heterogeneity among farming systems in their region.

In the next section, we briefly describe the region of study, the dataset, and the methods applied. In section 3, we present the results of the farms' clustering and LCA linked to a discussion on the last changes that occurred in the region and their implication for our results. Following this, we discuss the results of the CF calculations and a range of GHG mitigation strategies related to sustainable intensification of dairy farming, as well as strategies for producers that have been operating intensive systems. The paper closes with the research and policy recommendations and some concluding remarks.

2.2 Materials and methods

This study relates to milk producers from Paraná state, Southern Brazil. The state is located at [25°25′S 49°15′W] and has an area of 199.298,9 km², Figure 1 (IBGE, 2021). The state is located in a subtropical region covered by two climate classifications Cfa[16] and Cfb, with annual rainfall ranging from 1300 to 2500 mm (Alvares et al., 2013).

[16] Cfa: humid subtropical oceanic climate, without dry season, with hot summers; Cfb: humid subtropical oceanic climate, without dry season, with temperate summer (Alvares et al., 2013).

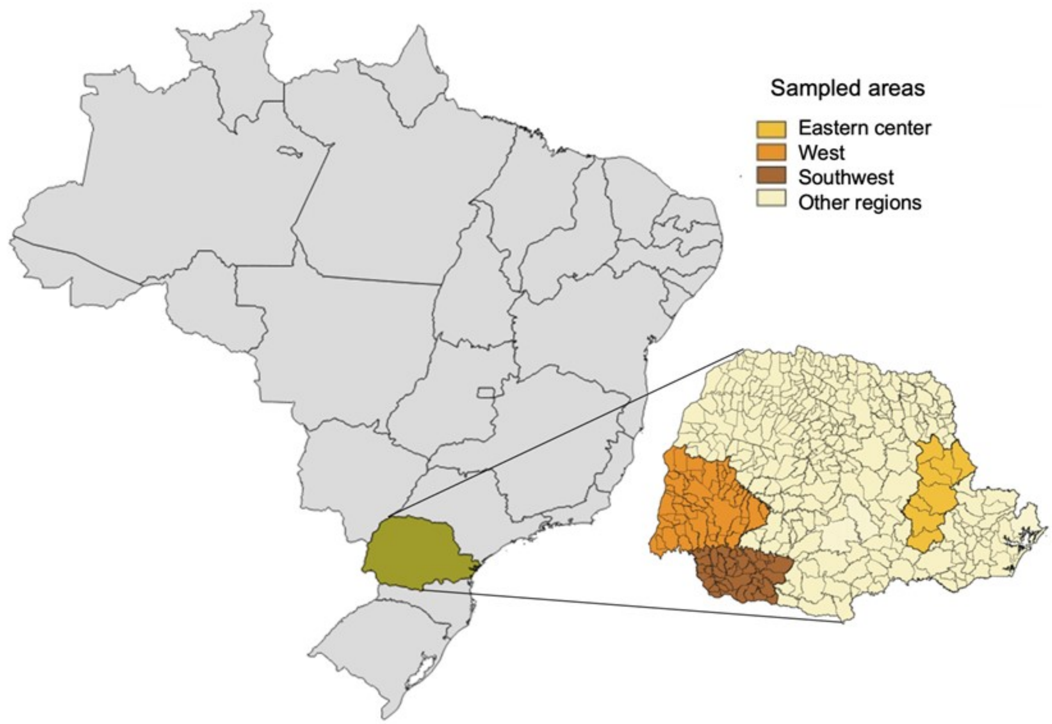

Figure 2-1. Location of Paraná in Brazil and Paraná with the sampled regions highlighted. Adapted from (IPARDES, 2009).

Paraná is the third-largest milk producer in the country, with 87,063 dairy farms and an annual production of ~ 3.2 Mt in 2017 (IBGE, 2018). Table 1 provides an overview of the dairy farms' structure in Paraná according to the number of dairy cows in the herd.

Data were collected in a general farm survey conducted by the Institute of Economic and Social Development of Paraná (IPARDES) in 2007. Four areas were considered for sampling purposes. The first three represented the state's subregions of Eastern centre, West, and Southwest. These are considered the main dairy basins in the state, accounting for 62% of the total milk production in Paraná (IBGE, 2006). The fourth area, termed Other regions, grouped all remaining regions of Paraná due their smaller share in the state's production, Fig. 1.

Table 2-1. Structure of the dairy farms in Paraná according to the number of cows in the herd.

Location	Cow in the herd (N)	Farm (N)	Milked cow (N)	Milked cow (%)	Milk (t yr^{-1})	Milk (%)	Yield (t $cow^{-1}yr^{-1}$)
Paraná	1-10	534,409	65,171	8	137,373	4	2.11
	11-20	20,038	121,810	14	324,545	10	2.66
	21-50	25,908	316,828	36	1,075,779	33	3.40
	51-100	7,873	181,695	21	776,197	24	4.27
	101-500	4,042	151,198	17	732,851	23	4.86
	>500	473	34,848	4	206,290	6	5.92
Paraná	Total	86,793	871,550		3,253,035		3.97
Brazil	Total	1,170,860	11,478,463		30,103,106		2.67

Source: Adapted from the Brazilian National Agricultural Census, reference year 01.10.2016 to 30.09.2017 (IBGE, 2018).

2.2.1 Multivariate analysis

Data were first inspected for outliers and filtered to remove farmers that did not commercialize any milk or milk products in the period. The selection of variables followed the premise that different farm and herd characteristics generate groups with contrasting carbon footprints. The variables considered were: area of the farm, area of pasture, number of lactating cows, total number of animals in the herd, herd composition, availability of specialized breeds, milk production per cow, annual milk production of the farm, CF, and estimated capital (i.e., machinery, equipment, and buildings used in the dairy enterprise). To conduct the multivariate analysis we followed the steps described by Alvarez et al. (2014) and Hair et al. (2017). The selected variables were submitted to the Kaiser-Maier-Olkin (KMO) measure of sampling adequacy and Bartlett test of sphericity (Hair et al., 2017). Only variables that presented an individual KMO ≥ 0.5 were maintained for analysis (Hair et al., 2017). Finally, we reached an overall KMO value score of 0.74. The suitability of the analysis was confirmed by applying the Bartlett test, which presented $p = 0.000$, confirming that the dataset was suitable to PCA. We maintained three principal components which explained 73% of the variability in the data (Hair et al., 2017). Factor loadings from the PCA were used to conduct the clustering. The optimal number of clusters was obtained based on scree plot ($k = 4$). With the predefined number of clusters, we applied the K-means algorithm to perform the clustering. Final validation of the results has been conducted by comparing the farm groups, considering the variables' characteristics, as described by Subirana et al. (2014). We conducted the analysis with R Core Team version 3.6.3, (2020), packages ‘ade4’ v.1.7-16 (Dray and Dufour, 2007) and ‘factoextra’ (Kassambara and Mundt, 2020).

2.2.2 Life Cycle Assessment

Life cycle assessment is an environmental evaluation technique applied to calculate the environmental burdens of products and services for their entire life cycle (Guinee et al. 2002). A full LCA study follows the International Organization for Standardization (ISO) 14040 and 14044 guidelines, which suggest a four-phase approach, namely goal and scope definition, Life Cycle Inventory (LCI), Life Cycle Impact Assessment (LCIA), and Interpretation (ISO, 2006a, 2006b).

The goal of the LCA in this study was to inventory and calculate the carbon footprint (CF) of dairy farms from Paraná, BR. To accomplish this goal, we adopted a time scope of one-year, accounting for the production of milk and liveweight gain at the farm. This cradle-to-farm gate approach is common when the focus of the study is on improving farm-level environmental management practices (Bartl et al., 2011; De Boer, 2003; de Léis et al., 2015; Flysjö et al., 2012). Dairy farms are multifunctional production units, and allocation of the burdens among the outputs is normally required (e.g., production of milk, calves, cull animals). We adopted a biophysical approach, based on energy consumption for milk and animal biomass production (Nemecek and Thoma, 2020; Zumwald et al., 2018). This allocation procedure was applied to allocate burdens between Fat and Protein Corrected Milk (FPCM) and liveweight gain at each farm. The system boundary included the production and transportation of feed and minerals, electricity, and the farming stage, including all animal categories declared by the producers. Due to the lack of information on the fate of the manure produced in the farms, and to be conservative in our calculations, we allocated emissions from manure spread in agricultural soils to the dairy farms. Similarly, not enough information regarding pasture management practices was available to estimate yearly pasture carbon storage or emissions, and thus it was considered in balance. Emissions associated with the production of medicines and construction of buildings were not accounted for due to their expected low impact.

The life cycle inventory analysis is the stage in which data regarding input and outputs for the product system under analysis is collected (Klöpffer and Grahl, 2014). Thus, we conducted an inventory for the farming stage, as well as for the production and transport of feed and farm inputs for the production of milk and animal liveweight (see FAO 2016). On-farm life cycle inventory of GHG emissions followed the refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2019b), considering a wet season of 180 days (Pereira et al., 2008). The inventory regarding feed consumption included pasture, silage, hay, salt, and concentrates. Farmers declared which animal categories received concentrate and feed supplements. The quantity of each feed ingredient in the diet was calculated based on animal category and animal productivity according to Embrapa (2005) and support from an animal scientist working since 2006 as a researcher on dairy production systems in Paraná; see Supporting Material (SM) Table SM1. Feed characteristics required for performing the calculations were retrieved from Feedipedia (2020) and Valadares Filho et al. (2020), and are presented in Table SM2. For the manure management calculations, we assumed that 20% of the manure from animals that were handled daily was collected and stored, the remaining 80% was assumed to be deposited directly onto pastures. Three manure storage systems were considered according to the farms' characteristics, namely liquid, solid, and dry lot. Emissions factors applied in the model are displayed in the SM, Table SM3. Furthermore, the LCI for the production and transport processes of fertilizers, seeds, feed, mineral salt, and electricity were retrieved from the ecoinvent® v.3.06 database (Ecoinvent, 2019). We adjusted the mentioned processes to the Brazilian electricity mix. The processes for the transportation of inputs were adjusted to represent the EURO 3 technology.

The life cycle impact assessment phase is dedicated to scaling the results of the LCI into the chosen impact categories (Klöpffer and Grahl, 2014). Our study follows the single issue LCA where only the carbon footprint is evaluated. To this end, we calculated the standard impact category Global Warming Potential (100-year time horizon) (GWP_{100} kg CO_2 equivalent (eq.)) (Guinée, 2002). The applied characterization factors for the farming stage were 27.2 and 29.8 for biogenic methane (CH_4) and fossil methane, respectively, and 273 for nitrous oxide (N_2O) (Forster et al., 2021).

The interpretation of the LCA results was conducted by comparing the GWP_{100} from the different farm groups by conducting a Kruskal-Wallis rank sum test followed by Dunn's Kruskal-Wallis multiple comparisons. In order to investigate the relationship between productivity and the CF of milk we also calculated the Spearman's rank correlation coefficient. Moreover, we compare the GWP_{100} results with the GWP* (GWP Star) method, which was recently proposed in order to improve the representation of the climate effects from short-lived climate pollutants (SLCPs), such as methane (Allen et al., 2018; Masson-Delmotte et al., 2021). The comparison and discussion between methods is presented towards the end of section 5.

2.3 Results

2.3.1 Farm Groups

The results revealed four farm groups that were identified by applying Principal Component Analysis (PCA) and K-means clustering. Table 2 presents the mean and standard deviation of the farm area, herd structure, and production characteristics of the sample and farm groups. In total, a sample of 911 farms were analysed and clustered. The number of animals owned by the farmers in the sample was 56,700 head, with 48.6% being cows, and an annual production of ~122,412 tonne of FPCM. Most farmers in the sample had five years or more of experience as managers. Women represented less than 10% of the managers in all groups (Table 3). Further characteristics regarding production and feed management of the sample and farm groups are presented in the SM, Table SM4.

Group 1 (G1) is formed of 128 farms, representing 14% of the sample. G1 was mostly represented by high productive farms with large herds (185 heads). The average number of cows per farm in the group was 102 with an average daily production per cow of 21.4 kg FPCM. About 77% of the herds in this group were composed of highly productive cows of the breed Holstein. The price received per litre of milk by farmers in G1 was on average R$ 0.08 higher than the other groups. The number of years of formal education of managers in G1 (8.5 years) was also higher than the other groups. In terms of labour, G1 farmers employed on average a total of 5.1 workers, with more than half being hired labour (3.2 workers). Furthermore, G1 also had a higher percentage of farmers accessing external financial support, affiliated to cooperatives and Unions, as well as receiving technical support.

Group 2 (G2) clustered 317 farms, representing 35% of the sample. The G2 is mainly represented by the medium size herds (~33 head) with an average of 16.8 dairy cows. Holstein cows also predominate as the main breed in G2 (46.9%), however, the use the Jersey breed is also representative in this group (18%). Average daily production per cow in G2 (13.6 kg FPCM) was lower than G1 but still higher than the sample average. Out of the four groups, G2 was the group that had the highest number of family workers engaged in the dairy activity (2.6 workers).

Table 2-2. Farm area, herd structure, and production characteristics for the sample and four farm groups.

Variable	Unit	Sample N=911	G1 N=128	G2 N=317	G3 N=326	G4 N=140	p.over all
Farm area	ha	61.1 (148)	142 (308)	34.0 (52.7)	18.0 (26.2)	149 (171)	<0.001
Pasture area	ha	33.2 (69.5)	59.7 (75.8)	17.5 (40.8)	8.36 (17.6)	102 (120)	<0.001
Herd	head	62.2 (84.4)	185 (114)	32.9 (14.4)	16.1 (8.91)	124 (97.2)	<0.001
Lactating cows	head	22.3 (34.0)	83.6 (54.4)	12.8 (5.76)	4.97 (2.81)	28.3 (19.5)	<0.001
Lactating + dry cows	head	30.3 (42.8)	102 (66.7)	16.8 (7.41)	7.24 (4.27)	49.2 (32.6)	<0.001
Holstein	% of the herd	33.2 (40.5)	77.5 (33.8)	46.9 (40.2)	12.6 (27.4)	9.73 (21.5)	<0.001
Jersey	% of the herd	13.1 (25.1)	9.71 (22.0)	18.0 (26.6)	13.2 (27.1)	5.01 (15.0)	<0.001
Girolando[a]	% of the herd	7.77 (23.2)	3.14 (14.2)	5.53 (18.0)	5.88 (20.9)	21.5 (36.9)	<0.001
Swiss brown	% of the herd	0.88 (6.30)	0.90 (5.98)	1.02 (5.34)	0.16 (1.64)	2.20 (12.4)	0.014
Other breeds	% of the herd	31.7 (42.4)	8.73 (23.5)	19.7 (34.6)	40.9 (45.5)	58.7 (44.3)	<0.001
Yield per cow per day	kg $FPCM^{-1}$	11.6 (6.42)	21.4 (6.05)	13.6 (4.76)	7.96 (3.29)	6.83 (2.72)	<0.001
Farm production	(t FPCM) yr^{-1}	134 (306)	682 (551)	65.1 (41.8)	13.6 (8.07)	71.5 (62.8)	<0.001
Price received	BRL (kg $FPCM)^{-1}$	0.55 (0.12)	0.62 (0.08)	0.54 (0.11)	0.54 (0.14)	0.54 (0.09)	<0.001
Farm income from milk	%	60.6 (32.0)	77.3 (25.0)	63.7 (29.4)	54.5 (34.0)	52.7 (31.9)	<0.001
Farm income from animals	%	7.39 (19.0)	3.27 (11.4)	2.74 (9.91)	3.49 (13.0)	30.8 (31.3)	<0.001

[a] Dairy cow breed developed in Brazil by cross breeding Holstein x Gir (*Bos indicus*).

Group 3 (G3) clustered the larger number of farms in our analysis (N= 326), representing 36% of the sample. G3 is represented by smallholder farms (~ 18 ha) owning herds averaging 16 animals with around 7 cows. Daily milk production per cow was on average 7.9 kg FPCM, which approximates the national average in 2017. Farmers belonging to this group relied more on non-specialized breeds (40.9%), which are generally less productive. G3 was the group that relied most on family-based work force, 2.16 from a total of 2.21 workers. This group concentrated 80 out 98 farmers from the sample that declared commercializing products derived from milk, thus indicating the importance this activity has for small producers (Table SM4).

Group 4 (G4) is represented by farmers with the largest farm area (149 ha) and the second largest herds (124 animals) after G1 (185 animals). However, the low number of lactating cows in the herds of G4 (28 heads) and low productivity per cow (6.83 kg FPCM) show a significant contrast between G1 and G4. The use of large pasture areas and high share of non-specialized breeds (58.7%) in G4 suggest these farmers use dual-purpose herds, reared in extensive management systems. Moreover, 50% of farms in G4 conduct only one milking daily (Table SM4). Despite the poor indicators, G4 farms have more than 50% of their income generated from milk.

Table 2-3. Management, labour, technical and financial support characteristics for the sample and four farm groups.

Variable	Unit	Sample N=911	G1 N=128	G2 N=317	G3 N=326	G4 N=140	p.overall
Age of farm manager	Year	51.1 (12.2)	50.7 (10.4)	48.9 (11.1)	51.3 (13.2)	56.3 (12.4)	<0.001
Sex of manager	Female	61 (6.76%)	8 (6.35%)	20 (6.33%)	28 (8.62%)	5 (3.68%)	0.266
	Male	842 (93.2%)	118 (93.7%)	296 (93.7%)	297 (91.4%)	131 (96.3%)	
Level of education of the manager	Year	5.44 (3.56)	8.55 (4.27)	5.15 (3.11)	4.38 (2.72)	5.81 (3.93)	<0.001
Family workers	Units	2.42 (1.17)	2.51 (1.67)	2.68 (0.99)	2.16 (0.85)	2.32 (1.48)	<0.001
Hired workers	Units	1.95 (1.95)	3.19 (2.34)	0.80 (0.68)	0.76 (0.78)	1.44 (0.96)	<0.001
Received technical support	No	367 (40.3%)	10 (7.81%)	101 (31.9%)	188 (57.7%)	68 (48.6%)	<0.001
	Yes	544 (59.7%)	118 (92.2%)	216 (68.1%)	138 (42.3%)	72 (51.4%)	
Associated at cooperative or association	No	201 (22.1%)	12 (9.38%)	55 (17.4%)	98 (30.1%)	36 (25.7%)	<0.001
	Yes	710 (77.9%)	116 (90.6%)	262 (82.6%)	228 (69.9%)	104 (74.3%)	
Acquired external financing	No	599 (65.8%)	45 (35.2%)	187 (59.0%)	262 (80.4%)	105 (75.0%)	<0.001
	Yes	312 (34.2%)	83 (64.8%)	130 (41.0%)	64 (19.6%)	35 (25.0%)	

2.3.2 Carbon footprint

Results of the Global Warming Potential, calculated based on the Life Cycle Assessment, displayed a significant difference between the four groups. All comparisons were significant with $p<0.01$, except for the comparison between G3 and G4 which had $p=0.027$. Our results, therefore provide evidence that clustering dairy farms also enables the identification of farm groups producing milk with distinct carbon footprints. Farm G1 presented a mean GWP_{100} of 1.75 kg CO_2eq. (kg FPCM)$^{-1}$, followed by G2, G3, and G4 with mean GWP_{100} of 2.20, 3.02, and 3.27 kg CO_2eq. (kg FPCM)$^{-1}$, respectively (Fig. 2a). Furthermore, the overall correlation between annual milk production per cow and the GWP_{100} was rho = -0.77 (Fig. 2b), which supports previous evidence that the CF of milk decreases substantially as yield increases.

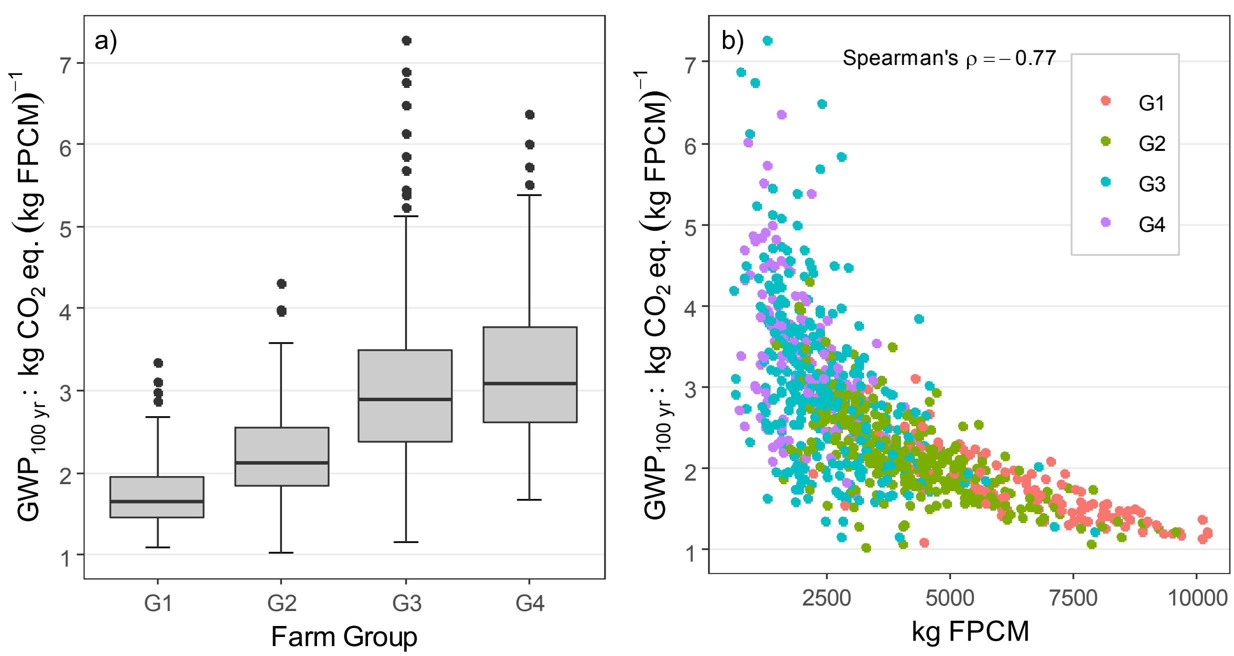

Figure 2-2. 2a) Global Warming potential 100-year time horizon (GWP100) for the production of one kg of Fat and Protein Corrected Milk (FPCM) according to farm group. 2b) Relationship between annual milk production per cow in kg FPCM and GWP100 (assuming 305 days lactation).

The relative contribution displayed in Figure 3, shows that enteric methane and feed related emissions are the main contributors to the CF of milk. The contribution of these sources ranged from 86% to 92% of the total GWP_{100}. However, it was possible to identify a reduction in the contribution of enteric methane from 64% to 55% when moving from the less specialized farms

(G4) towards more specialized ones (G1). Conversely, the contribution of feed related emissions increased from 28% in G4 to 31% in G1. This shift is commonly observed in LCAs of livestock and may be linked to several factors such as share of roughage in the animal's diet, feed quality, and animal genetics (De Boer, 2003; de Léis et al., 2015; Gerber et al., 2011). The representativeness of methane generated from manure management was the highest in G1. This is explained by the number of farmers using waste storage systems and handling manure in liquid form in this group and by our selection of methane emissions factor, Table SM3. Only 125 out of 911 farms in our sample had a waste store system, and 52% of them have been clustered in G1. Manure stored as slurry produces an anaerobic environment, facilitating methane production by bacteria (IPCC, 2019c).

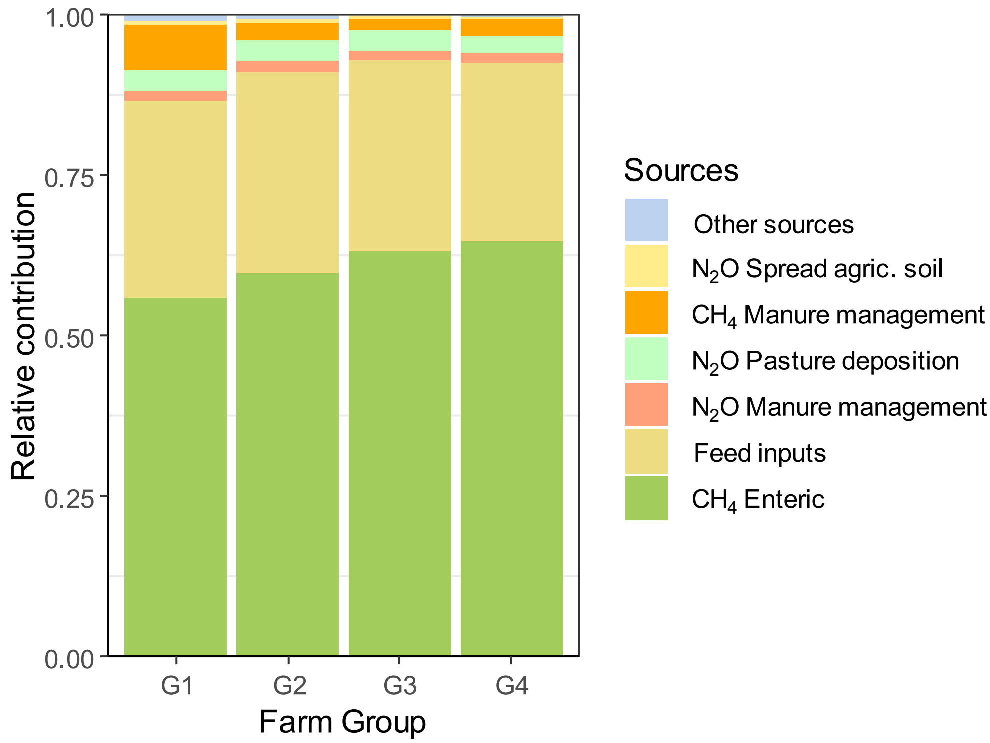

Figure 2-3. Relative contribution to Global Warming potential 100-year time horizon according to the origin of emissions.

2.4 Discussion

2.4.1 Recent developments in the dairy sector in Brazil and Paraná

New Federal regulations that came into force after the survey was conducted triggered some changes in the dairy sector in Brazil and consequently in Paraná. More specifically the Normative Instructions (NI) 51 from 2002, the NI 62 from 2011 and the NI 76, and NI 77 from 2018 defined more stringent SCC and TBC[17] parameters at national level. These normatives also required that farmers adjusted themselves by adopting new production practices and improved farm infrastructure. Consequently, due to the lack of financing and managerial skills, many farmers could not comply with the requirements and deadlines, exiting the activity. According to IBGE (2018) between 2006 and 2016 the number of farmers in the dairy sector in Paraná reduced by 27%. Farms with fewer than 10 dairy cows decreased by 27%, while farms with 21 to 50 dairy cows decreased by 22% and farms with 101 to 200 dairy cows decreased by 20%. Moreover, other factors also affected farmers' decisions to abandon the activity, such as market constraints (Beber et al., 2018) and lack of a successor (Bánkuti et al., 2018).

Farmers that informally processed their milk into dairy products (e.g., cheese, butter, and *requeijão*) also had to adapt. The informal market in Paraná corresponded to 16% of the volumes produced in 2016, while in 2006 it represented 23% of the volumes produced (IBGE, 2018). Intensification of inspections on milk manufacturing increased in recent years, putting pressure on farmers operating without a formal licence, which had to formalize their business to remain operative. In addition to pay for inspection services, many farmers had to improve their manufacturing facilities to comply with quality and sanitary standards. Such improvements required financing and technical supervision, which in many cases were not available. Consequently, farmers in this situation either abandoned the activity or remained working clandestinely with the risk of being penalized (Beber et al., 2019).

Regulations regarding manure management came into force in 2018 in Paraná. Farmers operating intensive systems with herds larger than 80 animals, and semi-intensive with herds larger than 180 animals, must apply for an environmental licence in order to operate (IAP, 2018). To be eligible to receive this licence, the farm must comply with practices and parameters for proper

[17] Since 2018 these figures are 300.000 CFU/ml for total bacterial count (TBC) and 500.000 SC/ml for somatic cell count (SCC).

waste storage and destination (e.g., use coated storage systems if manure is handled in liquid form). The regulation covers issues related to manure leaching, runoff, and discharge in water bodies; but it makes no mention of strategies to reduce GHG emissions during waste management. Moreover, by setting these numbers of animals as the threshold for demanding an environmental licence, the regulation does not cover the great majority of dairy farms in Paraná. Even small dairy farms can accumulate a significant amount of waste over time (e.g., washing-water, feed waste, bedding material, and manure). Besides the non-obligation to comply with this regulation, such farmers may be unaware of the problems manure can generate; and generally, do not receive any training on managing farm waste. This licencing obligation for large farms might have had the effect of reducing the CF of manure management from farms categorized in the G1, consequently increasing its CF gap in respect to the other groups from the time the data was collected to nowadays.

Despite the above-mentioned changes, on average, farm and herd structures across Brazilian dairy farms have not suffered major changes compared to our sample (Table 4). Farms with high productivity and low CF, as those characterizing G1, are still the exception in the Brazilian dairy sector (de Léis et al., 2015; IBGE, 2018). Most dairy farms in Brazil, such as those characterized by farms belonging to G2, G3 and G4, still operate with a low or medium technological level and display low productivity compared to the potential that can be reached in most regions of the country (Bánkuti et al., 2018; IBGE, 2018). For example, the average annual milk production per cow in Paraná was around 3.7 t in 2017. This value was below the average of the G2 (4.1 t) [18], and nearly half of the amount produced by G1 (6.5 t) in our sample from 2007, indicating the persistence of low productivities in this state.

Furthermore, given the annual milk production per cow at national level in 2017 (2.6 t), we can deduce that the other states of the country have a similar or even worse situation than Paraná. With this, despite the year of the data collection, our analysis provides relevant insights into the structure and CF of dairy farms in Paraná and Brazil. In addition, the heterogeneity of farms from our sample also indicates the existence of farms in different stages of structural and managerial development. Currently, in Paraná and in Brazil, it is still possible to find dairy farms from the most rudimentary to the most high-tech. Therefore, the farm groups resulted from the clustering and their attributed CF, brings valid and insightful reflections for the dairy farming of several other

[18] Annual production per cow was calculated assuming a lactation period of 305 days.

countries where farms and farmers are found in similar development stages to the ones in our sample.

Table 2-4. Structure of the dairy farms in Brazil according to the number of cows in the herd.

Cow in the herd (N)	Farm (N)	Milked cow (N)	Milked cow (%)	Milk (t yr^{-1})	Milk (%)	Yield (t $cow^{-1}yr^{-1}$)
1-10	383,171	853,241	7	1,446,286	5	1.70
11-20	238,020	1,252,053	11	2,674,775	9	2.14
21-50	306,134	3,259,614	28	8,412,359	28	2.58
51-100	131,109	2,493,013	22	6,960,521	23	2.79
101-500	97,166	2,999,498	26	8,603,615	29	2.87
>500	15,260	621,044	5	2,005,550	7	3.23

Source: Adapted from the Brazilian National Agricultural Census, reference year 01.10.2016 to 30.09.2017 (IBGE, 2018).

2.4.2 Carbon footprint and mitigation strategies

The LCA results unveiled the CF of three groups of farms that have been underexplored under the Brazilian conditions, namely G2, G3, and G4, which are the most representative groups for dairy producers in Paraná and Brazil (Bánkuti et al., 2018; IBGE, 2018). By comparing[19] our results with other studies, we identified that at the sample mean (2.6 kg CO_2eq. (kg FPCM)$^{-1}$)) our findings are close to the global average of 2.7 kg CO_2eq. (kg Milk)$^{-1}$ reported by Gerber et al. (2013). The results for G3 (3.02 kg CO_2eq. (kg FPCM)$^{-1}$) and G4 (3.27 kg CO_2eq. (kg FPCM)$^{-1}$) were above the global average, however, below the values for milk production in Asia (5.5 kg CO_2eq. (kg Milk)$^{-1}$) (Gerber et al., 2013) and the Peruvian highlands (5.42 kg CO_2eq. (kg ECM[20])$^{-1}$) (Bartl et al., 2011). Groups G1 and G2 presented mean values below the global average, 1.75 and 2.20 kg CO_2eq. (kg FPCM)$^{-1}$), respectively. Farms clustered in G1 presented an average GWP_{100} within the range of several OECD countries (Gerber et al., 2013; O'Brien et al., 2016; Thomassen et al., 2008).

Our results for G2, G3, and G4 also corroborate with the range found by González-Quintero et al. (2021) for Colombian milk production from dual-purpose herds. The authors found a mean

[19] We conduct only a general comparison, disregarding methodological choices and assumptions across studies (e.g. Functional unit, allocation rules, temporal and physical boundaries, impact category). These differences should be considered when detailed comparisons are within the goal of the study once significant differences may arise from them (de Vries et al., 2015; Schüler and Paulsen, 2019; Vogel et al., 2021).

[20] kg of Energy Corrected Milk (ECM) is another common functional unit applied in LCAs for milk production.

GWP_{100} value of 3.3 kg CO_2eq. (kg FPCM)$^{-1}$, with results ranging from 1 to 10.2 kg CO_2eq. (kg FPCM)$^{-1}$. Only farms belonging to G1 had GWP_{100} similar to previous values reported for dairy systems in Brazil. For example, de Léis et al. (2015) reported mean values of 0.776, 1.065, and 1.013 kg CO_2eq. (kg ECM)$^{-1}$ when comparing milk production in confined, semi-confined and pasture-based systems respectively. Likewise, Ribeiro-Filho et al. (2020) reported average values of around 1.06 kg CO_2eq. (kg ECM)$^{-1}$ for high productive systems in Southern Brazil. Carvalho et al. (2021) reported a value of 1.42 kg CO_2eq. (kg FPCM)$^{-1}$ for crossbred cows produced in semi-intensive production in North-eastern Brazil.

Therefore, we identified that the heterogeneity among farmers operating in a single state in Brazil is similar to the range described around the world (Gerber et al., 2013), clearly indicating the necessity of adopting different approaches to mitigate GHG emissions at regional scale. The fact that farms operating with high productivity display lower CF indicates a potential for mitigating emissions by increasing the productivity of low-yield animals. However, increasing the productivity while reducing the GHG intensity of dairy farms depends on a series of strategies that should be applied under a sustainable intensification framework (Gerber et al., 2013; Herrero et al., 2016; IPCC, 2019a). Options to improve the CF of smaller and less specialized farms operating with higher CF intensity are wide. In contrast, the range of mitigation options available for specialized farms already operating with low GHG intensity is evidently more limited. However, due to the size of their operations, small improvements can result in large absolute reductions. More specifically, GHG reduction strategies focus on actions to increase animal and herd performance, improve feed production and feeding management, optimize waste management, and increase energy efficiency (Gerber et al., 2011; Herrero et al., 2016). An extensive list of available and prospective strategies to mitigate GHG from cattle production has been discussed in the literature, e.g., (Gerber et al., 2013; González-Quintero et al., 2022; Grossi et al., 2019; Herrero et al., 2016; Llonch et al., 2017; Resende et al., 2020; Wattiaux et al., 2019). Based on these, we next present key strategies for the Brazilian conditions and discuss their practical implementation.

2.4.2.1 Feed and pasture management

Feed management is a cornerstone of dairy farms and has a large potential to reduce GHG emissions. Feeding animals all year-round with balanced diets based on high digestibility ingredients is important to explore the maximum production potential of dairy cows and reduce methane emissions from enteric fermentation (de Souza Filho et al., 2019; Herrero et al., 2016). However, feeding a dairy herd with a balanced diet is not an easy task and is among the challenges faced by dairy producers in Brazil. It depends on several factors such as knowledge of nutrition, financial resources, production system, and the farm's feed production capacity in terms of area, soil, topography, and weather conditions.

Most dairy farms in Brazil rely partially or entirely on grazing to provide animals with the necessary roughage. Thus, improving pasture quality and management is among the first actions to be implemented by farms with low performance (Gerber et al., 2013; IPCC, 2019a; Novo et al., 2013). Sowing season-adapted pastures with good nutritional quality grasses and the adoption of rotational grazing are among the simplest practices that can be implemented by less specialized producers (de Souza Filho et al., 2019; Novo et al., 2013; Pinheiro Machado Filho et al., 2021). Furthermore, farmers owning natural pastures can significantly improve their productivity and quality by over-seeding leguminous forages and seasonal grazes (Dick et al., 2015b; Novo et al., 2015; Ruviaro et al., 2015). Adoption of an intensified pasture-based system can reduce the expenses with feed and improve the profitability of dairy farms while promoting a reduction in GHG intensity of milk (O'Brien et al., 2015; Ruviaro et al., 2020). More recently, the modernization of crop-livestock-forestry integration systems in Brazil has also provided an excellent alternative to improve pasture and consequently the productivity and income in beef and dairy farms (Bungenstab et al., 2019; da Silva Cardoso et al., 2020). In those regions where dry season occurs, maintaining good pastures year-round is difficult without the support of irrigation and fertilization. Thus, the production of conserved forages such as silage and hay is an important strategy to complement the diet of the animals during the dry season. The southern region has an advantage in this regard. The region is located in the subtropical climatic zone, with favourable conditions to grow annual and perennial temperate pastures with good nutritional quality (Alvares et al., 2013; Ribeiro-Filho et al., 2020).

The inclusion of concentrated and feed supplements in the ration in less specialized farms is indispensable to fulfil the nutritional requirements of milking cows, improving milk productivity and reducing methane emissions from enteric fermentation (Gerber et al., 2013; Wattiaux et al., 2019). Nevertheless, this strategy depends on land availability for on-farm feed production or financial resources for purchasing it, which might make difficult the adoption by small-farmers. In addition, the use of co-products in the ration of dairy cattle has advantages to farmers and the environment. It has favourable cost-effectiveness for farmers and normally presents low CF (de Léis et al., 2015; Ruviaro et al., 2020). The use of co-products also reduces the demand for land to produce feed, and contributes to the circular economy (Van Zanten et al., 2019). When using co-products, however, farmers must adjust the diet of the animals accordingly. Besides, if the co-product has low-digestibility (e.g., straw) on-farm emissions from enteric fermentation are likely to increase. Avoiding wastage of feedstuffs is another strategy to reduce emissions from dairy systems. Wastage should be avoided across all feed supply chain, e.g., production, transportation, storage, and feeding. Depending on the level of management and feed storage systems, forage losses can reach up to 30% (FAO, 2017).

Farmers operating intensive systems tend to have adequate feed management strategies. In these cases, animals stay housed year-round and all feed must be provided in-house in the form of roughages, concentrates, protein supplements, minerals, and vitamins. Feeding can then be optimized for each animal according to its physiological and productive stage (e.g., calves, heifers, lactating cows, and dry cows). Problems often associated with zero-grazing systems are the high GHG intensity for the production and transportation of feedstuffs (O'Brien et al., 2015; Ruviaro et al., 2020), need for increased biosecurity measures, animals' welfare, and the high capital costs of facilities and machinery. The integration of grazing in the diet of high productive animals could be a feasible option to reduce this problem in southern Brazil, a region that can produce high quality pasture all year-round (Ribeiro-Filho et al., 2020). Further feed management strategies that specialized farmers can apply include the development of precision-feeding and the use of feed additives to reduce enteric methane production (Gerber et al., 2013; Tullo et al., 2019).

2.4.2.2 Herd management

The adoption of appropriate dairy breeds and herd management is an important step to increase the productivity of the herd and reduce GHG emissions. However, despite producing less milk, less specialized animals may bring advantages to farmers, e.g., farmers belonging to G3 and G4. Many farmers opt for dual-purpose herds to produce milk and beef in order to diversify their sources of income and reduce the risks of running a specialized dairy farm (González-Quintero et al., 2021). Even in this structure, strategies to speed up backgrounding and fattening of beef animals can reduce the overall CF of the farm (Gerber et al., 2013). Other advantages of owning dual-purpose animals are the resistance against diseases, better response to low-quality pastures and co-products feedstuffs, and lower immobilization of capital in animals. Furthermore, owning non-specialized animals also demands less operation management capacity from farmers.

In addition to a suitable breed, optimizing herd structure and reproductive performance of the dairy cows and heifers is a key strategy to improve the efficiency of dairy farms and consequently reducing GHG. The most important actions in this field are the reduction of calving interval, enhancement of pregnancy rate, and expanding the productive life of the cows (Gerber et al., 2013). In this regard, the use of Artificial Insemination (AI), and further extension of embryo transfer practices, are effective strategies to improve these parameters and reduce GHG intensity of dairy farms (Gerber et al., 2013; Herrero et al., 2016; Llonch et al., 2017). Herds with a small number of cows normally have underutilized bulls that require feed, space, and health care; increasing overall farms GHG emissions. However, the adoption of natural mating among less specialized farms in Brazil is still a common practice, in our sample for example, 55% of the farmers declared using this practice. By owning a bull, farmers do not need to detect the best moment to inseminate the cows (i.e., no need for oestrus detection), which requires specific training. Farmers also do not need to rely on off-farm professionals to perform AI (Beber et al., 2019). Large distances and the scarcity of dairy specialized technicians may make it difficult to schedule their visits in the best moment to apply the AI. In some cases, the AI straw and application may also have a charge that can be perceived as costly by farmers. Furthermore, proper biosecurity measures and animal health management is indispensable to increase animal health and welfare, reduce mortality rates, unexpected culling, and consequently GHG intensity of dairy herds (Gerber et al., 2013; Donal O'Brien et al., 2014).

2.4.2.3 Waste management

Manure and farm waste are significant sources of nutrients that are normally used for pasture or crop fertilization. Nevertheless, significant losses of N compounds and emissions of methane may arise along the waste management chain, possibly contributing to the farm CF and other environmental impacts (Wattiaux et al., 2019). Regardless of herd size or production system, selecting an appropriate waste storage system can reduce substantially absolute on-farm emissions (IPCC, 2019c). Pasture-based farms have an expressive share of the manure deposited directly on pastures. And except for small-scale farms, where dung is collected manually, any attempt to handle dung deposited on pastures is unfeasible. But, even pasture-based systems can accumulate waste on the collecting yard and milking area generating GHG emissions, and if not properly handled it can also leach and run off into water bodies during the rainy season (Palhares et al., 2012). In our sample, only 14% of the farmers declared owning storage systems, showing a critical point for improvement (Table SM4). The new regulations implemented have improved the situation in the last decade, however most farms still lack proper waste management systems.

Farmers that already own waste storage systems can adopt a series of practices to reduce emissions, for example, by decreasing storage time, covering piles in solid storage systems, or applying aeration in liquid storage systems (Gerber et al., 2013; Herrero et al., 2016). The adoption of anaerobic digestion systems is an effective technology to treat manure and recover energy in dairy farms. This is a compelling technology to curb methane emissions from farms handling methane in liquid form, as we have identified for farmers belonging to G1. The implementation of this technology has been supported by the Low Carbon Agriculture Plan (ABC)[21], but adoption is still low among dairy farms in Paraná. The State registered only 66 digesters in 2019, most of them installed in pig farms (CiBiogas, 2021).

2.4.2.4 Energy efficiency

The adoption of energy-saving technologies is quoted as a prominent strategy to reduce the CF and costs of dairy farms. Milking and milk cooling systems account for the higher usage of standard electricity across farms. Farms that have animals housed however, have extra expenditures and GHG emissions, e.g., with feed processing, cleaning, lighting, and ventilation of barns. Thus, while any farm can adopt management practices and technologies for energy-saving, this strategy offers larger impact to highly productive systems. Moreover, energy contributes to a small share of the CF from milk, having an expected lower cost-effectiveness compared to the strategies previously described (Gerber et al., 2011).

2.5 Policy implications and future research

Future development and expansion of the dairy sector in Brazil will inevitably pass along the path of production intensification. Thus, developing a framework based on sustainable

[21] In Brazil, GHG reduction strategies in the agricultural sector have been guided by the Low Carbon Agriculture Plan (ABC) (Brasil, 2012; Bungenstab et al., 2019). The ABC plan was designed by the Ministry of Agriculture, Livestock and Supply to support research and the adoption of improved agricultural practices to decarbonize the agricultural sector in the country.

intensification of dairy farms in the country will be essential for achieving mitigation targets in the livestock sector. Nevertheless, intensification based only on actions to reduce GHG intensity by increasing yields is often criticized for being narrowly framed (Clay et al., 2020; Garrett et al., 2018; IPCC, 2019a). This approach may lead to animal welfare issues and distinct environmental impacts (Clay et al., 2020; IPCC, 2019a; Novo et al., 2015).

The Low Carbon Agriculture Plan (ABC) from the Brazilian Government has yielded some remarkable results in the development of improved and integrated production systems, such as crop-livestock, crop-livestock-forestry integration, pasture recovery, and agroforestry (Bungenstab et al., 2019; Costa et al., 2018; de Moraes et al., 2014; Martinelli et al., 2019). Most actions sponsored by the ABC plan enable farmers to earn environmental and economic benefits (Costa et al., 2018; Florindo et al., 2018; Pashaei Kamali et al., 2016). However, the relative low rate of adoption among farmers indicates that the next generations of Low Carbon Agriculture in Brazil should go beyond credit subsidies and actions at the plot level (Cortner et al., 2019). Given the heterogeneity of farming systems in Brazil and the concentration of small-holder farmers in some sectors, climate-smart policies in the country should consider a transition within a broader process of socio-environmental change in rural areas (Clay et al., 2020; IPCC, 2019a).

The low professionalization of human resources and the lack of proper farm management is mentioned as main factors affecting the adoption of suitable technologies and good practices by farmers (Beber et al., 2019). Therefore, the provision of adequate extension services and training to all farmers is an initial fundamental step to implement sustainable intensification strategies (Gerber et al., 2013). Extension services can facilitate the diffusion of knowledge and technologies at the farm level. Thus, qualifying extensionists in sustainable production practices is a key action to promote the development of the dairy sector in Paraná and Brazil. Extensionists should be trained and provided with tools to support and train farmers that are willing to implement changes. For instance, extensionists should be able to support farmers in selecting sustainable production systems that are in accordance with farmer's beliefs and intention (e.g., traditional, integrated, multifunctional, or agroecological systems). Systems which are the best adapted to farmers' socioeconomic circumstances and the farms' pedoclimatic conditions (Novo et al., 2015). Although decisions must be taken by farmers themselves, extensionists must also be able to guide farmers through the production process, for example in adjusting the scale of their operations, in selecting appropriate breeds, in drawing the feeding management strategy and in the development of sanitary, health and reproductive controls. Embrapa[22] has developed a promising programme (Full Bucket programme) which focuses on technology transfer and adaptative learning through the creation of demonstration dairy farms and in training extensionists and technicians across the country (Novo et al., 2015, 2013). The core strategies of the Full Bucket programme are aligned with most practices reviewed in our study (see, Novo et al., 2013). Therefore, expanding this program and making it accessible to a larger number of extensionists and farmers is crucial for sustainably developing dairy production in Brazil.

It is important to notice that production technologies and technical knowledge available in Brazil already allow the ambitious and financially capable producer to build intensive dairy operations similar to those in the developed countries, with high productivity and low carbon footprint (de Léis et al., 2015; Ribeiro-Filho et al., 2020). Nevertheless, research in Brazil should strive to determine which systems and levels of intensification are more suitable for the country. Such studies should take a broader view of the farming systems, accounting for issues such as animal welfare, farm ecosystem services as well as economic development and climate change adaptation strategies. Improving production based on the circularity concept is another emerging field in the livestock sector that also desires more research in the Brazilian context (Van Zanten et al., 2019, 2018). Integrated crop-livestock, crop-livestock-forestry systems and agroecological systems could offer improvements in this direction. Optimizing semi-intensive systems might offer a noteworthy strategy to reduce the CF of milk while avoiding problems from the super-intensive systems (e.g., animal health and welfare issues, reliance on off-farm sources of feed, waste management and consumer acceptance) (Hennessy et al., 2020; Ruviaro et al., 2020).

There is strong evidence that the intensification of degraded pastures can reverse their status of net carbon emitters into carbon sinks, increasing soil carbon pool significantly over time (Oliveira et al., 2021; Segnini et al., 2019). Nevertheless, carbon sequestration and storage are difficult to quantify or estimate for large samples, and thus, generally considered in a steady-state in LCA studies. (Dick et al., 2015b; Stanley et al., 2018). More research on pasture carbon dynamics, considering regional climate, soil, and pasture management is required under Brazilian conditions. For example, Oliveira et al. (2021) and Segnini et al. (2019) also show that intensive systems with a high stocking rate per ha are less effective in storing carbon when compared to

[22] Brazilian Agricultural Research Corporation. https://www.embrapa.br/en/international

moderate stocking systems. Thus, further experiments are required to determine ideal animal density and fertilisation levels to optimise carbon sequestration in pastures. Moreover, determination of storage and stabilization prediction curves, as well as emissions by specific soil management practices need to be determined in order to better account for soil carbon in dairy LCAs (Corbeels et al., 2016; de Sant-Anna et al., 2017).

The development of certified pasture-based milk production has plenty of opportunities in Brazil. Research institutes and processing companies should foster such projects as a market strategy. For example, the Brazilian dairy industry should strive to reach similar developments as those achieved by the Carbon Neutral Brazilian Beef project (Alves et al., 2017; Resende et al., 2020) and receive a premium price to produce milk within a sustainable framework. Besides, the dairy industry could benefit by adding value to their products and promoting fair social and green marketing.

Regardless of the size of the operation in commercial dairy farms, profitability is one of the most relevant determinants for the adoption of new technologies and management practices. Most intensification technologies can also promote higher returns in the short to mid-term. Thus, expanding financial credit to a wider range of dairy farmers, and promoting a business environment in which farmers are keen on taking the initial risks is also necessary to expand the adoption of new technologies (Gerber et al., 2013). The provision of credit is one of the main strategies of the ABC plan (Brasil, 2012b), for which the low rate of adoption is one of the factors responsible for the moderate success of the program. Increasing awareness of farmers to the options available is necessary; especially farmers that do not receive any technical assistance are less likely to be aware of governmental programs and credit possibilities.

In contrast, some farmers may have technical and financial conditions to improve the profitability and environmental performance of their operations but have no intention to do so (Rossi Borges and Oude Lansink, 2015; Senger et al., 2017). Research to explore the factors affecting farmers intention to intensify and adopt sustainable practices, need to be further explored in the dairy sector in Brazil. These can help in the development of effective nudging strategies for farmers' compliance with sanitary and environmental regulations, as well as to promote their pro-environmental behaviour (e.g., Duflo et al., 2011; Peth et al., 2018). The adoption of technologies that can mitigate GHG emissions, but bring little or no direct economic benefits might indeed meet resistance among farmers. In these cases, the standard credit strategies applied so far in Brazil are likely to fail, indicating the need for new economic incentives to cover this gap (e.g., taxes abatement subsidies, payment for ecosystem services, milk pricing bonus, or linking credit access to adoption of sustainable practices). Ultimately, more severe regulation and penalization of negligent producers is needed. The license for dairy farming required in Paraná serve this purpose, compelling the adoption of certain sustainable practices among larger producers. Within small-farmers such requirement could be costly and lead to exit of the activity, thus having a negative socioeconomic spillover. Therefore, training and advice must compensate in the support for the adoption of sustainable practices among these farmers. Such trade-offs between environment and economic efficiencies and benefits must be further studied and measured.

The milk processing sector is in constant development in Brazil and Paraná but it is generally not considered competitive if compared to other agricultural supply chains in the country (e.g., beef and grains) (Beber et al., 2019). Private processing companies and cooperatives operating in Southern Brazil are facing several challenges to maintain their business operations. While new international companies started operations in the region, others less competitive were merged or dissolved (Beber et al., 2021, 2018). This rather unstable market environment may hinder farmers' decisions to invest in new technologies and thus spoil any attempt to promote climate friendly practices and policies. Consequently, this is also delaying the socioeconomic development in the region, where more than 87,000 farms produce milk, corresponding to approximately 28.5% of the farms in Paraná. Developing regional environmental governance frameworks may be required to develop the supply chain in Paraná and grant access to international markets and global value chains. Government, business, and civil society need to provide a clear direction (and the necessary resources) to farmers on which pathways to follow in order to develop a sustainable agri-food system (Béné et al., 2019). In this regard it is fundamental to recognize the heterogeneity among farms and farmers and the maintenance of an adequate socioeconomic environment, where each dairy farmer could be capable of thriving. This can be achieved through stable market conditions and fair prices, as well as resources to farmers willing to process and market their own milk. Public research and extension/advisory services should be expanded in the region, since technicians supported by the processing sector tend to focus only on issues related to productivity and milk quality. Lastly, the country should be prepared to provide meaningful alternatives to farmers and businesses that will not cope with challenges ahead and, eventually, leave the production chain.

The IPCC Global Warming 100-year time horizon is by far the most used climate change metric applied to compare agricultural systems, however it is recognized for misrepresenting the climate effects from short-lived climate pollutants (SLCPs), such as methane (Allen et al., 2018; Masson-Delmotte et al., 2021). This may lead to major implications to policy design in the agri-food sector (Ridoutt, 2021). The GWP* is presented in terms of CO_2w.e. (warming-equivalent) and allows a better representation of future warming of SLCPs when compared to CO_2eq. by accounting for their change rate over a certain timeframe (Smith et al., 2021). We therefore applied the GWP* to our data and compare the results with GWP_{100}. We assumed that farms had a stable dairy herd and applied the equation suggested by Smith et al. (2021) for the calculations. A stable herd is a conservative approach since the national dairy herd has decreased in the last two decades (IBGE, 2018).

The GWP* results, presented in Figure 4, show contrasting lower values to those of the GWP_{100} presented in Figures 2 and 3. The GWP* for G1 to G4 was 0.27, 0.35, 0.53, and 0.61 kg CO_2w.e. kg $(FPCM)^{-1}$, respectively. The ranking of groups was not altered. Moreover, when comparing the groups based on GWP*, we observed a significant difference between all of them, ($p <0.001$) (Fig 4a). In terms of contribution analysis, it is clear that methane reduced its relative importance to the total impact (Fig 4b). These results confirm the importance of improving the production of less specialized farms as discussed earlier. Due to the higher contribution of feed-related emissions, the results also reinforce the importance of developing sustainable feed production systems for the Brazilian conditions, considering the trade-offs between biogenic methane and fossil emissions for feed production. Moreover, more research has to be conducted applying the GWP* metric to evaluate more accurately the possible contributions of the Brazilian agri-food sector to climate change.

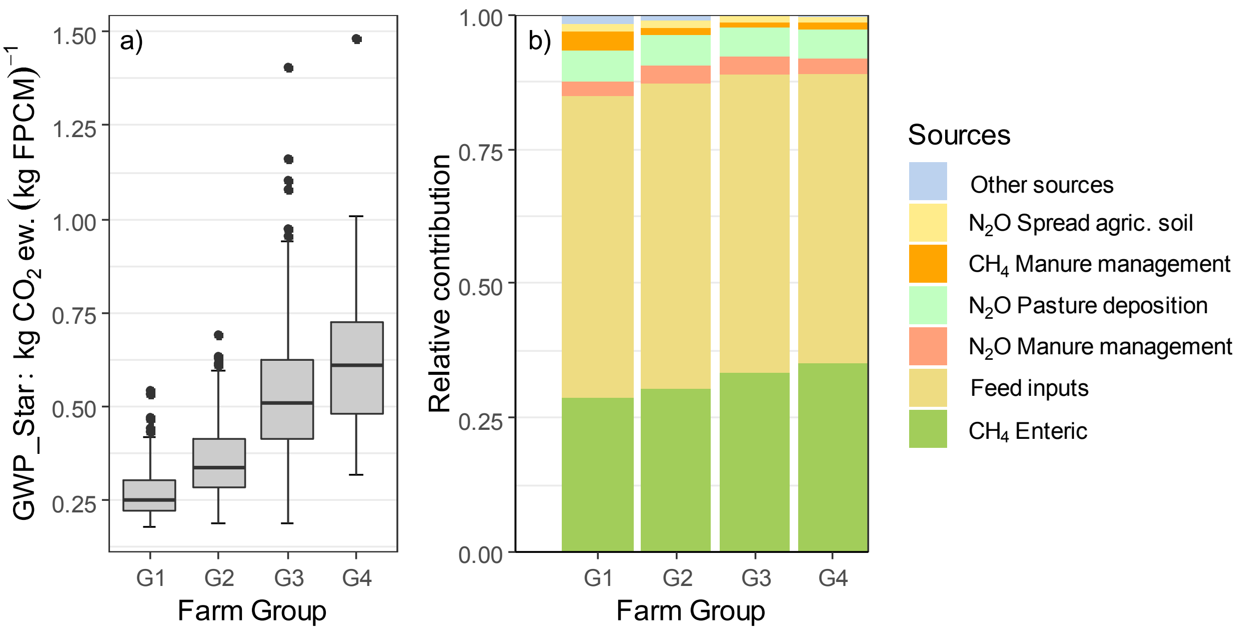

Figure 2-4. 4a) Global Warming Potential Star (GWP*) for the production of one kg of Fat and Protein Corrected Milk (FPCM) according to each farm group. 4b) Relative contribution to GWP* according to the origin of emissions.

2.6 Conclusions

Tackling climate change is a matter of priority in the livestock sector. Nevertheless, guiding best management strategies in practice is sometimes blurred by the lack of information regarding regional characteristics of heterogeneous farming systems. In the present study, we integrate multivariate statistical analysis and expert knowledge to cluster dairy farms and explore their carbon footprint. Our approach identified four groups of farms showing distinct characteristics and significant differences in the Global Warming Potential comparison. Detailed analysis of the characteristics of the clusters indicated that farms operating with higher CF represented the majority of the dairy farms in Paraná. On average, they were likely to have less specialized herds with lower productivity, receive less technical support, and have less farm machinery and infrastructure when compared to farms that displayed lower CF. Research needs to advance in some areas to identify better paths towards the sustainability of the Brazilian dairy sector. However, the body of knowledge and technologies available in the country already allows significant improvements if adopted by farmers, which suggests that the high CF of dairy farms is only one of the problems faced by farmers operating in Brazil. Reducing the CF in the dairy sector therefore will largely depend on moving farmers from the status quo towards sustainable intensification. To reach this goal, actions should be integrated into a broader framework of regional environmental governance and socioeconomic development.

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Chapter 3

3 Environmental efficiency and methane abatement costs of dairy farms from Minas Gerais, Brazil[23]

Abstract

Increasing dairy farm productivity while simultaneously mitigating greenhouse gases emissions is a common policy goal in many countries. In this paper, we assess trade-offs and synergies between these goals for pasture-based dairy farms in Brazil. We apply stochastic frontier analysis within a translog hyperbolic distance function specification, including methane emissions as undesirable output and accounting for annual climatic types. Our results indicate that, on average, farmers can improve their production by 9.4% while simultaneously decreasing methane emissions by 8.7%. The adoption of more productive cows and improved pastures have a positive effect on the environmental efficiency of the farms. Farmers operating in warmer and dryer climate types tend to have lower environmental efficiency. Calculation of shadow prices for methane emitted at farms indicates that the mean abatement costs of methane is US $2,254 per tonne. Overall, by reducing inefficiency, dairy farmers can significantly increase farm production while simultaneously contribute to national commitments to mitigate methane emissions.

Keywords: shadow price, technical efficiency, eco-efficiency, GHG mitigation, Balde cheio, Köppen classification

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3.1 Introduction

Dairy farming is fundamental to the economy of many countries– markedly low- and middle-income countries (LMIC), where it also plays a pivotal role in providing food security, employment, and livelihoods in rural areas (FAO, 2010; OECD-FAO, 2021). At the same time, dairy farming contributes to greenhouse gas (GHG) emissions, which are major drivers of global warming. Globally, the dairy herd is responsible for emitting around 2.1 Gt of CO_2eq. (Carbon dioxide equivalent)[24], representing ~ 30% of all emissions in the livestock sector (Gerber et al., 2013; Herrero et al., 2016). These emissions are comprised of carbon dioxide, nitrous oxide and remarkably methane, which represents more than 50% of all emissions. GHG emissions from dairy farming vary considerably across countries and production systems, however, a strong negative correlation between the carbon footprint of milk and animal productivity has been identified, suggesting that improving the productivity of dairy cows is an effective strategy to improve the environmental sustainability of dairy farms (Gerber et al., 2011; Vogel and Beber, 2022). Consequently, dairy farmers become key players for achieving the goals set in the Paris Agreement. Especially by mitigating methane emissions, which is the most concerning GHG emitted at dairy farms and is considered one of the most cost-effective strategies to slow down global warming (Reisinger et al., 2021; UN-CCAC, 2021).

Globally, policymakers face the challenge of designing strategies to mitigate GHG emissions to comply with international climate commitments and national laws while maintaining and improving socioeconomic and ecosystem services provided by dairy farms (Brazil, 2021b; Clay et al., 2020; Gerber et al., 2013; Ravichandran et al., 2020). However, the implementation of such strategies at farms is complex and context-specific, generating outcomes that are likely to produce synergies as much as trade-offs (Campbell et al., 2018; Clay et al., 2020; Novo et al., 2015).

In this study, we assess economic and environmental synergies and trade-offs of pasture-based dairy farms managed under the influence of sustainable development strategies. We analyse a sample of Brazilian dairy farmers participating in Embrapa's [25] *Balde Cheio* (Full Bucket-FB) programme in the State of Minas Gerais and investigate their ability to maximize desirable outputs

[24]Carbon dioxide equivalent (CO_2eq.) based on the Global Warming Potential 100 years-time horizon (GWP_{100}).
[25] Brazilian Agricultural Research Corporation, https://www.embrapa.br/en/international.

while minimizing methane emissions. We estimate a stochastic translog hyperbolic distance function, allowing for asymmetric treatment of desirable and undesirable outputs in the multi-output production frontier (Cuesta et al., 2009; Le et al., 2020; Mamardashvili et al., 2016; Skevas et al., 2018). Moreover, this approach allows to identify drivers of environmental inefficiency and the calculation of shadow prices for methane.

The Brazilian dairy farming is evolving rapidly and has become one of the main components of the national agri-food sector. According to the last agricultural census, in the period 2006-2017, the number of dairy farms in the country decreased from 1.35 M to 1.17 M farms (13%), while the number of milked animals reduced by 9%, from 12.7 M to 11.5 M cows; conversely, milk production increased by 70% in the same period. In 2020, The national milk production reached 36.5 Mt, generating around US $12 billion in value for farmers and placing Brazil as the third-largest dairy milk producer in the world (Embrapa, 2021; Rocha et al., 2020). Moreover, the national dairy production contributes to local food security in rural areas. For instance, more than one-quarter of the milk produced in the country does not enter the dairy processing industry (IBGE, 2018), indicating that it is either consumed directly by the household or commercialised locally through short supply chains. On the environmental side, Brazilian dairy farms play an important role in the conservation of grassland and key biodiversity areas in the form of Legal Reserve and Permanent Preservation Areas which are spared at farms (Embrapa Territorial, 2020). Nevertheless, by hosting one of the largest dairy herds in the world, the country contributes substantially to GHG emissions. In 2019, dairy farming in Brazil was responsible for emitting 53.8 Mt CO_2eq., representing 2.5% of the national and 9.3% of the agri-food sector CO_2eq. emissions (SEEG, 2020). Overall, the national dairy herd presents low productivity and high GHG intensity with methane accounting for almost three quarters of all emissions (SEEG, 2020).

A number of studies analysed the environmental efficiency of dairy farms. Early approaches treated externalities as inputs in the production function, focusing on farmers' ability to minimise the surplus of N (Nitrogen) and P (Phosphorus) compounds in Dutch dairy farming (Reinhard et al., 2002, 2000, 1999). Mamardashvili et al. (2016) investigated the environmental efficiency and abatement costs of N surplus in Swiss dairy farms located in mountainous areas. The authors applied hyperbolic and enhanced hyperbolic distance functions to investigate the farmers' ability to expand the production of desirable outputs while reducing Nitrogen pollution. Applying a similar approach, Skevas et al. (2018) revisited the Dutch case to investigate the effects of agri-environmental policies and production intensification on the environmental efficiency of dairy farms. Adenuga et al. (2019) compared the environmental efficiency and abatement costs of N surplus for dairy farms on the island of Ireland. In terms of P surplus, March et al. (2016) applied the non-parametric Data Envelopment Analysis (DEA) to assess the environmental efficiency of dairy farms in Scotland while Adenuga et al. (2020) compared farmers from Northern Ireland by applying the stochastic hyperbolic distance function. Studies evaluating the environmental efficiency of dairy farmers in terms of GHG emissions also have gained attention in the dairy sector. A pioneering study considering GHGs in the environmental efficiency of dairy farms was proposed by Njuki and Bravo-Ureta (2015). The authors employed a quadratic directional distance function with a CO_2eq.-pollution index to investigate the impacts of GHG regulations in the USA dairy sector. The same approach was applied by Njuki et al. (2016) to study the effects of dairy enterprise size on environmental efficiency and abatment costs of CO_2eq. of dairy farms in the northeastern of the USA . Wettemann and Latacz-Lohmann (2017) applied DEA techniques to derive ranges of efficiencies and abatement costs for specialized dairy farms in northern Germany. Le et al. (2020) employed the stochastic hyperbolic distance function to compare technical and environmental efficiency and calculate CO_2eq.-abatement costs for dairy production in Alberta, Canada.

We expand the literature on environmental efficiency of dairy farms in multiple directions. First, most studies thus far have evaluated intensive high-productive systems in developed countries, e.g., (Adenuga et al., 2019; Le et al., 2020; Njuki et al., 2016; Reinhard et al., 1999; Skevas et al., 2018; Wettemann and Latacz-Lohmann, 2017). In contrast, we analyse pasture-based dairy production in Brazil, where dairy farms present, on average, low yields, operate with limited access to technology and face different policy incentives. Second, instead of evaluating a CO_2eq.-index, we focus exclusively on methane emissions as undesirable output. Thus, we provide a better understating of the environmental efficiency of dairy farms in terms of the most important GHG emitted in the dairy sector. In this approach, we also calculate methane-specific shadow prices, providing an indication of the abatement costs of this GHG for dairy farms in Brazil. This might be of interest for national policy design, particularly given the recent commitments the Brazilian government assumed to cut methane emissions as a signing party of the Global Methane Pledge[26].

[26] Signatory countries committed to cutting global methane emissions by 30% from 2020 levels by 2030 (EU, 2021).

Lastly, we include the Annual Climate Type concept in our production function to evaluate the effects that climatic regions have on farms' environmental efficiency. This approach is based on the Köppen-Geiger climate classification and might be relevant since there is increasing evidence of the impact of climatic elements on technical (Gori Maia et al., 2021; Perez-Mendez et al., 2019) and environmental efficiency (Le et al., 2020; Njuki et al., 2016; Njuki and Bravo-Ureta, 2015) of dairy farms.

3.2 Methods

3.2.1 Theoretical framework

The theoretical foundations for investigating production in a dynamic environment, where a bundle of inputs is employed to produce multiple outputs, were introduced by the seminal works of Debreu (1951) and Shephard (1953, 1970). Ever since, distance functions (DF) proved very useful in the empirical measurement of efficiency, notably by Farrell (1957) (Kumbhakar and Lovell, 2003). Under this framework, an input distance function seeks the maximum radial contraction of the input vector at constant output. Conversely, the output distance function seeks the maximum radially expansion of output vectors, at given inputs (Kumbhakar and Lovell, 2003). Despite being extensively applied to evaluate the production processes of marketable goods, the idea of radially expanding outputs altogether is limited when undesirable by-products are part of the decision-making unit outputs.

These limitations gave rise to further developments of the DF taking the form of Directional Distance Functions (DDF) (Chambers et al., 1996). One of the advantages of this approach is the possibility of applying the output DDF to evaluate the environmental efficiency of decision-making units by seeking a maximum increment in desirable outputs while simultaneously reducing undesirable outputs (Chambers et al., 1998; Chung et al., 1997). This mechanism is enabled by introducing a directional vector into the function in an additive form to scale desirable and undesirable outputs in opposite directions (Färe et al., 2005; Färe and Grosskopf, 2000). Several empirical studies evaluating environmental efficiency follow from these developments, e.g., (Njuki et al., 2016; Njuki and Bravo-Ureta, 2015; Picazo-Tadeo et al., 2005; Riera and Brümmer, 2022). Limitations associated with the DDF are that the results are subjective to the selection of the directional vectors, which are normally arbitrarily chosen (Atkinson and Tsionas, 2016; Holtkamp and Brümmer, 2018). Besides, it does not satisfy the property of commensurability, i.e., the results are sensitive to measurement units (Peyrache and Coelli, 2009; Skevas et al., 2018).

Another approach to estimate the environmental efficiency is the Hyperbolic Distance Function (HDF) proposed by Färe et al. (1989), based on the work of Färe et al. (1985). Instead of projecting a straight line towards the frontier, the graph representation follows a hyperbolic path allowing inputs and outputs to be treated asymmetrically (Färe et al., 1985). Färe et al. (1989) developed their framework applying the non-parametric DEA approach. The parametric stochastic framework considering the HDF was proposed by (Cuesta and Zofío, 2005), while proper adjustments to accommodate undesirable outputs were amended by (Cuesta et al., 2009). The HDF satisfies the commensurability property (Skevas et al., 2018) and overcomes the arbitrariness of selecting a directional vector. Moreover, the HDF also allows for the calculation of shadow prices for non-marketable by-products, therefore providing a suitable framework for our study on dairy farms[27].

To characterise the technology set with undesirable by-products, an additional vector representing undesirable outputs is appended to the traditional representation. It is then represented by a feasible combination of vectors of inputs $x = (x_1, x_2, \dots, x_n)$, desirable outputs $y = (y_1, y_2, \dots, y_n)$ and undesirable by-products $b = (b_1, b_2, \dots, b_n)$. Following Cuesta et al. (2009), the technology can be represented by the graph set

$$T = \{(x, y, b): x \in R_+^K, \mathrm{y} \in R_+^M, \mathrm{b} \in R_+^R,\ x\ can\ produce\ (y, b)\}. \tag{1}$$

The corresponding hyperbolic distance function can be defined as in eq. (2). Where $D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b})$ represents the hyperbolic distance function and θ is a scalar. Given the available amount of inputs, the HDF represents a maximum expansion of the desirable output vector and equiproportionate contraction of the undesirable output vector, placing producers at the boundary of the production technology T.

$$D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b}) = min\left\{\theta > 0\text{: } \left(\boldsymbol{x}, \frac{\mathrm{y}}{\theta}, \boldsymbol{b}\theta\right) \in T\right\} \tag{2}$$

[27] The modelling of undesirable outputs in productive efficiency is a rapidly expanding field of research, for a critical review of recent approaches refer to Dakpo et al. 2016.

$D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b})$ ranges between 0 and 1. If a farm presents $D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b}) = 1$ it is located at the boundary of the production possibility set and is considered environmentally-adjusted technical efficient (Dalheimer, 2020). If the technology satisfies the traditional axioms, then our hyperbolic distance function satisfies the properties P1 to P4 below (Cuesta et al., 2009; Cuesta and Zofío, 2005; Färe et al., 1985).

P1. Almost homogeneity: $D_H(\boldsymbol{x}, \mu\boldsymbol{y}, \mu^{-1}\boldsymbol{b}) = \mu D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b}); for\ \mu > 0$

P2. Non-decreasing in desirable outputs: $D_H(\boldsymbol{x}, \lambda\boldsymbol{y}, \boldsymbol{b}) \leq D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b}); \lambda \in [0,1]$

P3. Non-increasing in undesirable outputs: $D_H(\boldsymbol{x}, \boldsymbol{y}, \lambda\boldsymbol{b}) \leq D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b}); \lambda \geq 1$

P4. Non increasing in inputs: $D_H(\lambda\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b}) \leq D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b}); \lambda \geq 1$

Following the almost homogeneity condition and selecting a normalizing output variable M, we can set $\theta = \frac{1}{y_M}$, and express $D_H(\boldsymbol{x}, \boldsymbol{y}, \boldsymbol{b})$ as

$$D_H\left(\boldsymbol{x}_i, \frac{\mathbf{y}_i}{y_M}, \boldsymbol{b}_i y_M\right) = \frac{1}{y_M} D_H(\boldsymbol{x}_i, \mathbf{y}_i, \boldsymbol{b}_i). \tag{3}$$

By taking logs of both sides of eq. (3) we reach

$$lnD_H(\boldsymbol{x}_i, \mathbf{y}_i, \boldsymbol{b}_i) = lnD_H\left(\boldsymbol{x}_i, \frac{\mathbf{y}_i}{y_M}, \boldsymbol{b}_i y_M\right) + lny_{Mi}. \tag{4}$$

The hyperbolic efficiency is defined as $HE_i = D_H(\boldsymbol{x}_i, \mathbf{y}_i, \boldsymbol{b}_i)$. We substitute and rearrange the equation solving for lny_M; and finally, append an error term v_i to capture statistical noise:

$$-\ln y_{Mi} = lnD_H\left(\boldsymbol{x}_i, \frac{\mathbf{y}_i}{y_M}, \boldsymbol{b}_i y_M\right) - lnHE_i + v_i. \tag{5}$$

3.2.1.1 Shadow price

The shadow price can be interpreted as the production of desirable output that must be foregone to reduce one unit of the undesirable output under analysis (Färe et al., 2005; Zhou et al., 2014). Shadow prices are particularly relevant for studying production systems where by-products are not marketable. An ingenious approach to calculating shadow prices is based on the duality between the hyperbolic distance function and the profitability (Return to the dollar) function (Färe et al., 2002; Färe and Grosskopf, 1998).

Assuming that a producer seeks to maximise profit, she faces the problem described in eq.(6) (Cuesta et al., 2009; Färe et al., 2002).

$$\prod(x,\ p_y, p_b) = \begin{matrix} max \\ x, y \end{matrix} \left\{ \frac{p_y y}{p_b b} :\ D_H(x, y, b)\ \leq 1 \right\} \tag{6}$$

where p_y is the price of desirable output and p_b is the unknown price of the undesirable output. The first-order conditions to the problem in eq. (6) are equal to eq. (7) and eq. (8), respectively.

$$\frac{p_y y}{p_b b} = \lambda \frac{\partial D_H}{\partial y} y = \lambda \left(\frac{\partial ln D_H}{\partial ln y} \right) D_H \tag{7}$$

$$\frac{p_y y}{p_b b} = -\lambda \frac{\partial D_H}{\partial y} b = -\lambda \left(\frac{\partial ln D_H}{\partial ln b} \right) D_H. \tag{8}$$

By taking the ratio between eq. (8) and eq. (7) and applying the implicit function theorem to the distance function yields eq. (9), which allows the calculation of the shadow price of the undesirable output b in terms of the main desirable output y_m.

$$-\ p_y \frac{\frac{\partial D_H}{\partial b}}{\frac{\partial D_H}{\partial y_M}} = \ p_y \frac{d y_M}{db}\ ]_{D_H=1} \tag{9}$$

It is noteworthy that the shadow price refers to the estimation at the frontier, assuming the farmer is fully efficient, i.e., $D_H\ = 1$.

3.2.2 Methane emissions

Direct measurement of methane emissions is complex and expensive, thus we estimate the emissions following the Guidelines for National Greenhouse Gas Inventories (IPCC, 2019b). Methane originated from enteric fermentation and manure management are the two sources considered in the guidelines. Enteric fermentation emissions are derived based on daily feed intake of the herd. We calculate the daily gross energy (GE) intake and apply the simplified tier 2 method to calculate the daily dry matter intake (DMI) for each animal category declared by the farmers (i.e., cows, calves, heifers, bulls) (IPCC, 2019c). Finally, we apply the equations for predicting enteric methane based on DMI described by Ribeiro et al. (2020).

Methane originated from manure is derived from information on manure volatile solids (VS) content and manure management system. The VS excretion is calculated based on daily gross energy intake of the animals and feed quality (IPCC, 2019b). Based on expert information, we assume that 80% of the manure from animals handled on a daily basis was deposited on pastures, the remaining 20% was deposited onto barns, milking parlour or handling areas, and thus entered the storage system. Liquid and solid manure storage systems are considered based on the manure management system declared by farmers. The default value of 0.19 m^3 CH_4 (kg VS)$^{-1}$ is adopted as the maximum methane producing capacity of VS excreted (IPCC, 2019b).

3.2.3 Study area and Data

We analyse a sample of 208 dairy farms distributed across the state of Minas Gerais (MG), south-eastern Brazil, Figure 1. MG has an area of ~586,522 km^2 and is covered by three out of six Brazilian biomes (IBGE, 2021). The State has a long tradition in milk production and is the largest milk producer in Brazil (IBGE, 2018). In 2021, MG produced a total of 9.4 Mt milk, representing 27% of the national production (Embrapa, 2021).

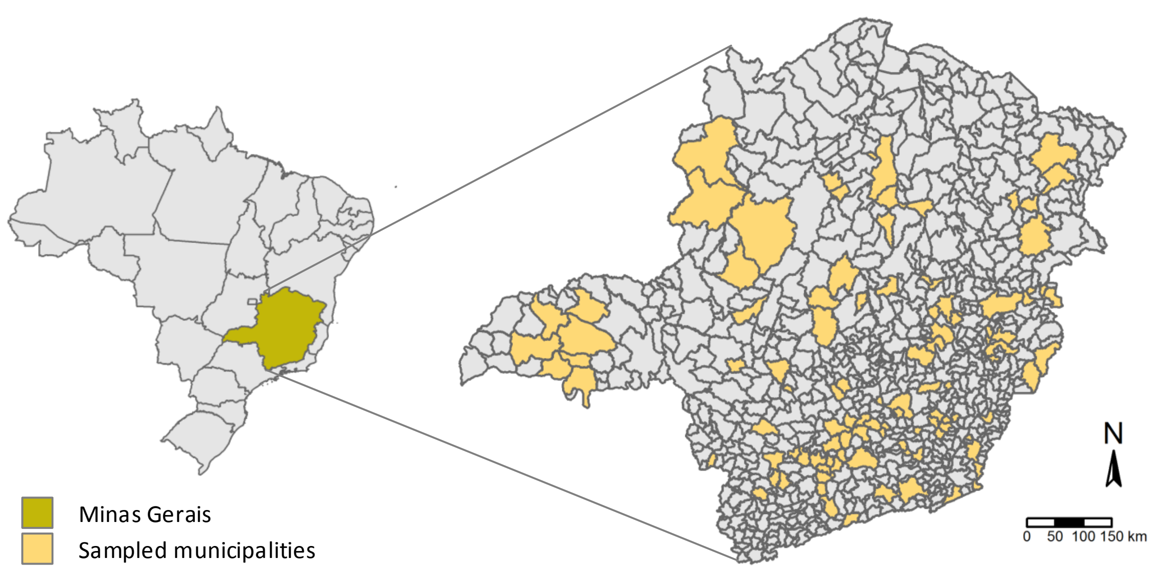

Figure 3-1. Location of the state of Minas Gerais and sampled municipalities.

The cross-sectional dataset was collected in 2017 as part of Embrapa's *Balde Cheio* (Full Bucket-FB) programme.[28] The FB programme was created by the Embrapa's South-eastern Livestock Research Centre in 1999 and aims at sustainable intensification of dairy farms in Brazil through technology transfer and participatory learning. The database includes a complete socioeconomic characterisation of the household and technical and economic information related to the dairy enterprise. The sample includes exclusively pasture-based producers, which is the most common dairy production system in Brazil. The descriptive statistics of selected farm variables are presented in Table 1.

Table 3-1. Variables overview and summary statistics

Variable (N=208)	Mean	Std.Dev	Min	Max
Capital (1,000 US$[a])	2.53	2.3	0.21	12.24
Purchased feed (1,000 US$)	15.45	13.95	0.99	78.11
Other expenses (1,000 US$)	11.76	10.84	1.08	51.65
Land (ha)	40.9	35.41	1	217
Labour (Working units)	1.73	0.77	1	4
Lactating cows (N)	23.74	14.57	5	82
Herd Size (N)	62.1	38.4	9	213
Milk sold (t FPCM[b])	108.74	83.72	15.37	440.59
Animals sold (1,000 US$)	4.66	5.11	0	29.9
Methane CH_4 (t)	4.95	3.28	0.88	20.87
Buyer (N)	4.62	2.34	1	12
Daily milk yield (kg cow^{-1})	12.45	3.55	4.12	23.12
Experience (years)	20.73	13.62	2	60
Improved pasture (% of pastures)	0.15	0.18	0	1
Milk price (US$)	0.36	0.04	0.28	0.56
Cows in the herd (%)	0.75	0.09	0.41	0.91
Technical visits (N)	13.67	4.65	0	35
Bull in the herd (yes: 1; no: 0)	0.71			
Hired labour (yes: 1; no: 0)	0.82			
Rent land (yes: 1; no: 0)	0.27			

[a]USD-BRL: 3.192 (BACEN, 2022).[b] Fat and Protein Corrected Milk.

Variables selection for the environmental production function is based on recent studies exploring the technical and environmental efficiency of dairy farms, e.g. (Adenuga et al., 2020; Le

[28] For a complete description of the programme and its modus operandi see Novo et al. (2015), https://doi.org/10.1080/14735903.2014.945320 .

et al., 2020; Mamardashvili et al., 2016; Njuki et al., 2016; Skevas et al., 2018). The variable (Capital) represents the opportunity cost of capital invested in buildings and machinery, plus depreciation costs. (Purchased feed) is the sum of all feedstuffs purchased in the year including roughage, concentrates, calve feed, and mineral supplements. (Other expenses) include operating expenses with fertilizers, veterinary services, medicines, artificial insemination costs, and overheads. (Land) is the area available for feed production, i.e., forage and grain. (Labour) is measured in terms of working units per year. (Lactating cows) represents the number of lactating cows in the herd. (Methane) is annual amount of methane emitted on the farm from enteric fermentation and manure sources (see section 2.2. for details). All monetary values have been converted to 2017 US dollars by applying the USD-BRL exchange rate of 3.192 (BACEN, 2022).

Furthermore, to investigate the influence of year-specific climate elements on environmental efficiency, we include the Annual Climate Type (ACT) in our model (Dubreuil et al., 2019). The ACT relies on the Köppen-Geiger climate classification algorithm, which accounts for seasonal temperature and precipitation variations for grouping climatic types and regions (Trewartha and Horn, 1980). Climatology data for each municipality have been retrieved from the *National Aeronautics and Space Administration (NASA) Langley Research Center (LaRC) Prediction of Worldwide Energy Resource (POWER)* project[29]. And the 'ClimClass' R package (Eccel et al., 2016) was employed to derive two levels Köppen Annual Climate Types, see (Table 2).

Table 3-2. Annual Climate Types (ACT), number of farms by ACT, and summary of climate elements for the year 2017.

Köppen ACT	Farms	P_total*	P_winter	P_summer	T_avg	T_w.m	T_c.m
Aw[a]	87	938.6	322.4	616.2	23.6	26.6	19.2
Cw[b]	100	967.6	211.3	756.2	20.9	23.3	16.4
Cs[c]	11	931.6	264.6	667.0	20.9	23.5	16.3
BS[d]	10	550.9	239.2	311.8	23.3	25.8	18.5

[a]Aw (Tropical with dry winter); [b]Cw (Humid Subtropical With dry winter); [c]Cs (Humid Subtropical With dry summer); [d]BS (Dry Semi-arid); *P_total: total precipitation depth (mm); P_winter: precipitation depth in the 6 coldest months (mm); P_summer: precipitation depth in the 6 warmest months (mm) T_avg: average temperature (°C); T_w.m: average temperature of the warmest month (°C); T_c.m: average temperature of the coldest month (°C).

[29] https://power.larc.nasa.gov/data-access-viewer/

3.2.4 Empirical model

We estimate the stochastic version of the translog hyperbolic distance function (Cuesta et al., 2009). The Stochastic Frontier Analysis was proposed independently by Meeusen and van Den Broeck (1977) and Aigner et al. (1977) and enables separating technical inefficiency from random disturbances which are out of the control of the producers (Kumbhakar and Lovell, 2003).

Our model considers three outputs, including one undesirable, and six inputs. Letting $i = 1, 2 \ldots N$ represent the number of dairy farms, the main desirable output is represented by annual milk production (y_M), and the secondary desirable output is the income of animals sold (y_s). The undesirable output is methane emissions (b). The six inputs are capital (x_1), lactating cows (x_2), labour (x_3), land (x_4), feed (x_5), and other expenses (x_6). The annual climate type (c) is a four-levels controlling variable intended to gain insights into the annual climate types effect on environmental efficiency. We set the ACT (Aw) as the reference, since it presents the highest mean temperature throughout the year. The final specification for the hyperbolic distance function to be estimated is presented in eq. (10). We scaled the variables by their geometric mean before taking logarithms.

$$\begin{aligned}
-lny_{Mi} &= \alpha_0 + \sum_{k=1}^{6} \alpha_k ln(x_{ki}) + \frac{1}{2}\sum_{k=1}^{6}\sum_{l=1}^{6} \alpha_{kl} ln(x_{ki}) ln(x_{li}) \\
&\quad + \beta_0 ln(b_i^*) + \frac{1}{2}\beta_{00} ln(\, b_i^*)^2 + \sum_{k=1}^{6} \chi_{k0} ln(x_{ki}) ln(\, b_i^*) \\
&\quad + \delta_2 ln(\, y_{si}^*) + \frac{1}{2}\delta_{22} ln(\, y_{si}^*)^2 + \sum_{k=1}^{6} \gamma_{k2} ln(x_{ki}) ln(\, y_{si}^*) \\
&\quad + \rho_{20} ln(\, y_{si}^*) ln(b_i^*) + \omega_0 c_i + v_i + u_i
\end{aligned} \tag{10}$$

Where $b_i^* = b_i \times y_{Mi}; y_{si}^* = y_{si}/y_{Mi}$. The composite error term is $\varepsilon_i = v_i + u_i$, where v_i is the error term which captures random noise and has a normal distribution $v_i\ N\sim(0, \sigma_{vi}^2)$, and $u_i = -lnHE_i$ is the hyperbolic inefficiency term following a half-normal distribution $u_i\ N^+\sim(u_i, \sigma_{ui}^2)$. Additionally, we considered heteroskedasticity in both v_i (eq.(11)) and u_i, (eq.(12)) (Caudill et al., 1995; Wang, 2002).

$$\sigma_{ui}^2 = e^{z_i'\zeta} \quad (11)$$

$$\sigma_{vi}^2 = e^{w_i'\tau} \quad (12)$$

Where z_i is a farm-specific vector of variables that affect the variance of the inefficiency term while w_i is a farm-specific vector of variables that affect the variance of the noise term, and ζ and τ are parameters to be estimated. A positive sign of σ_{ui}^2 indicates the variable z_i under consideration has a positive effect on inefficiency. Similarly, if σ_{vi}^2 displays a positive sign, it suggests the variable w_i under consideration increases production uncertainty (risk) (Mamardashvili et al., 2016; Wang, 2002).

We follow the recent literature and availability of data variables to select *z* and *w* variables. Table 3 presents the *z* and *w* variables considered in the model and respective expected signs.

Table 3-3. Variables and expected signs for evaluating heteroskedasticity.

Variable	σ_{ui}^2	sign	σ_{vi}^2	sign
(Buyer)	z_1	+	w_1	-
(Milk yield)	z_2	-	w_2	+/-
(Time farming)	z_3	+		
(Improved pasture)	z_4	+/-		
(Cows in the herd)	z_5	-		
(Tech. support)	z_6	-	w_3	-
(Bull in the herd)	z_7	+	w_4	+/-
(Hired labour)	z_8	+	w_5	-
(Rent land)	z_9	+	w_6	+/-

Following Battese and Coelli (1988) farm-specific point estimate hyperbolic efficiency (HE_i) scores are calculated according to the conditional distribution of u given ε:

$$HE_i = \mathrm{E}\,[e^{-u_i} |\, \varepsilon_i]. \quad (10)$$

The estimation of the distance function parameters is conducted by maximum-likelihood using the R software (R Core Team, 2019) and the package ‘npsf’ (Badunenko et al., 2020).

3.3 Results and discussion

3.3.1 Production technology

The first-order maximum-likelihood estimates for the production technology, determinants of environmental inefficiency and associated standard errors are presented in Table 4. The complete list of coefficients is presented in appendix A. All first-order coefficients presented the expected signs, with exception of labour which was not statistically significant. Moreover, the coefficient of undesirable output has a negative sign, confirming the existence of trade-offs between desirable and undesirable outputs.

The first-order coefficients in the translog hyperbolic distance function may directly be interpreted as elasticities (Cuesta et al., 2009). Thus, we observe that the number of lactating cows has the largest distance elasticity, followed by feed and other expenses. Land and capital exhibit very low elasticities when compared with the other inputs. This is in line with most recent studies evaluating environmental efficiency in dairy farming, e.g., (Adenuga et al., 2020, 2019; Mamardashvili et al., 2016; Skevas et al., 2018). The results also indicate that, on average, the dairy farmers in our sample operate with decreasing returns to scale – suggesting that if farmers double inputs, outputs will not increase proportionally.

In terms of outputs, we observe that the desirable by-product income from livestock sold has a small contribution to the production function, which is expected in dairy enterprises, e.g, (Le et al., 2020). In addition, the undesirable output presents a large elasticity and the expected negative sign, indicating that increases in methane emissions will shift farms away from the production frontier, consequently reducing their environmental efficiency (Skevas et al., 2018).

Table 3-4. First order parameters and heteroskedasticity model estimates

Technology	D_H		SE
α_0 (Intercept)	-0.218	***	0.040
α_1 (Capital)	-0.043	***	0.012
α_2 (Lactating cows)	-0.207	***	0.051
α_3 (Labour)	0.012		0.023
α_4 (Land)	-0.019	*	0.009
α_5 (Feed)	-0.154	***	0.028
α_6 (Other expenses)	-0.111	***	0.024
β_1 (Methane)	-0.257	***	0.029
δ_2(Animals sold)	0.005	**	0.002
ω_1 (Cw)	-0.042	**	0.013
ω_2 (Cs)	-0.034	*	0.015
ω_3 (BS)	-0.031		0.024
Heteroskedasticity in σ_u^2			
ζ_0 (Intercept)	3.881	**	1.425
ζ_1 (Buyer)	0.092		0.059
ζ_2 (Milk yield)	-0.481	***	0.074
ζ_3 (Time farming)	-0.015		0.010
ζ_4 (Improved pasture)	-1.773	*	0.880
ζ_5 (Cows in the herd)	-3.807	*	1.631
ζ_6 (Tech. support)	-0.055		0.036
ζ_7 (Bull in the herd)	0.239		0.312
ζ_8 (Hired labour)	0.695	*	0.370
ζ_9 (Rent land)	-0.107		0.342
Heteroskedasticity in σ_v^2			
τ_0 (Intercept)	-16.849	***	2.457
τ_1 (Buyer)	0.335	*	0.137
τ_2 (Milk yield)	0.683	***	0.123
τ_3(Tech. support)	0.014		0.065
τ_4 (Bull in the herd)	-1.905	**	0.065
τ_5 (Hired labour)	-0.721		0.629
τ_6 (Rent land)	1.110	*	0.642
Log_Likelihood	236.15		
Mean EE	0.9141		
Std.Dev	0.0873		

$^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$

The Aw (Tropical with dry winter) annual climate type variable was selected as the baseline for comparing the influence of the climate variables on the production function. This climate classification covers nearly 35% of the area of MG and characterises the climate of the Brazilian Cerrado (savanna) biome (Alvares et al., 2013). We find differences between Aw and the Cw

(Humid Subtropical With dry winter) and the Cs (Humid Subtropical With dry summer). These results suggest that dairy farming in subtropical climates is environmentally more suitable than under tropical conditions. This is reasonable since temperatures exceeding the thermal comfort of cows is one of the main factors influencing the productivity of dairy farms, and we expect this to be more frequent under tropical conditions (Gori Maia et al., 2021; Mukherjee et al., 2013). Moreover, dry winters with warmer temperatures, as found in Aw, may have a stronger influence on pasture growth and quality, consequently exerting a negative effect on the environmental efficiency of farmers operating in this climatic region.

Despite the contrasting characteristics of Aw and BS (Dry Semi-arid) in terms of precipitation, we find no differences between the two climate types. The mean annual rainfall in the municipalities classified as BS was 58% of the volume of rain received by farmers in Aw, (Table 2). In terms of temperature, however, the two climate types are similar, presenting a difference of 0.3°C in the annual average temperature.

Also noteworthy is the fact that we identified BS ACT in Minas Gerais. Previous studies that use older Climate Normals data have found no semi-arid climate types in the State, e.g., (Alvares et al., 2013). In our updated Köppen-Geiger model, however, we determine municipalities that presented dry semi-arid conditions. These results are consistent with more recent climatology studies, which also identify BS climate types in MG, e.g., (Dubreuil et al., 2019; Martins et al., 2018). The presence of BS climate types in MG can be seen as evidence of climate change unfolding in the northern region of the State (Dubreuil et al., 2019). This trajectory is likely to continue for the coming years and further pressure milk productivity and environmental efficiency in the region.

3.3.2 Technical-environmental performance and determinants

The mean environmental efficiency of the sample is depicted in Figure 2 and was 0.91, ranging from 0.61 to 0.99, indicating that most farmers in the sample exhibit high environmental efficiency. These results suggest that, on average, farmers can increase outputs by 9.4% (1/0.91) while simultaneously reducing methane emissions by 8.7% (1-0.91). By reducing inefficiency, farmers could meaningfully contribute to national commitments for reducing methane emissions and still benefit from it by increasing farm output at the same time. For instance, if the farmers in our sample completely eliminate inefficiency, it would represent an annual reduction in 86 tonnes in methane emissions. Moreover, since the farmers in our sample are already engaged in a programme intended to improve farm productivity, we expect that improving the performance of the average smallholder milk producer in Minas Gerais can achieve higher contributions to mitigating methane emissions.

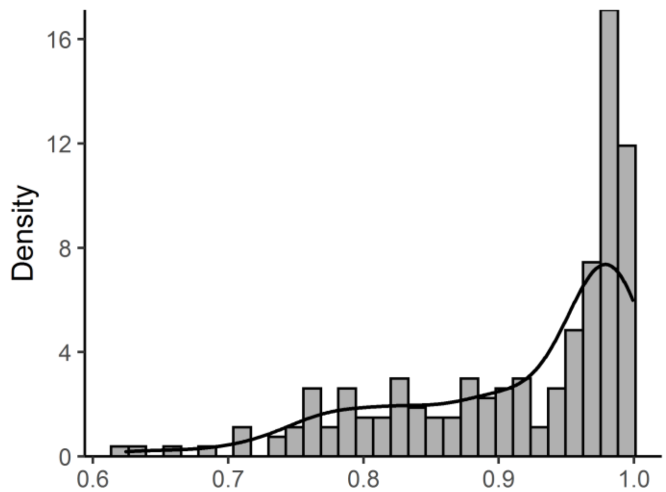

Figure 3-2. Environmental efficiency scores of dairy farms from Minas Gerais.

To put in perspective the effect of the exogenous variables on environmental inefficiency we present their marginal effects in Table 5.

Table 3-5. Marginal effects of determinants of inefficiency.

Variable	Mean	Std.Dev	Min	Max
Buyer	0.005	0.004	0.000	0.024
Milk yield[a]	-0.024	0.022	-0.127	-0.001
Time farming	-0.001	0.001	-0.004	0.000
Improved pasture	-0.088	0.083	-0.468	-0.004
Cows in the herd	-0.188	0.178	-1.006	-0.009
Technical support	-0.003	0.003	-0.014	0.000
Bull in the herd	0.012	0.011	0.001	0.063
Hire labour	0.034	0.032	0.002	0.184
Rented area	-0.005	0.005	-0.028	0.000

[a] Variables in bold presented significance in the heteroskedasticity model, $p < 0.1$.

Milk yield, presents a negative significant influence on environmental inefficiency. This is expected and in line with previous literature (Le et al., 2020; Mamardashvili et al., 2016; Reinhard et al., 2002; Shortall and Barnes, 2013), and can be associated to some extent with the genetic quality of the herd (Le et al., 2020). Therefore, our results confirm the evidence that increasing milk yield per cow is crucial for both economic and environmental efficiency of dairy farms. Low-yield dairy cows in low and middle income countries is one of the most pressing issues of sustainability of dairy farms (González-Quintero et al., 2022; Novo et al., 2013; Vogel and Beber, 2022). Nevertheless, the improvement of dairy farms in practice warrants a systems-thinking approach. For instance, the successful adoption of high productive breeds depends on several factors, such as suitable feed supply, climate and rearing conditions that attend the requirements of the selected breed, and farmers with know-how to manage high-yielding animals (Novo et al., 2015).

The share of improved pastures (Improved pasture) has a negative influence on environmental inefficiency. This is expected since improved pastures produce more forage per unit of land, reducing land use. Additionally, improved pastures tend to have higher digestibility and lower natural detergent fibre; which in turn contributes to lower feed conversion rate (FCR) and methane production from enteric fermentation. It is not surprising that pasture improvement ranks first in the list of activities that farmers shall focus on to improve farms' sustainability in the FB programme (Novo et al., 2015). Our results are supported by a considerable body of literature providing evidence that sustainable intensification of degraded and low-quality pastures positively contributes to land sparing, soil carbon storage, and reduction of GHG intensity of beef and dairy cattle (IPCC, 2019a; O'Brien et al., 2016; Oliveira et al., 2021; Ruviaro et al., 2015).

The share of lactating cows among cows in the herd (Cows in the herd) has a negative effect on inefficiency. This result provides evidence that adjusting herd structure to reach the best productive performance possible also improves the environmental efficiency of dairy farms. Fundamentally, this is a key indicator in dairy farms and should ideally be around 84% (Bachman and Schairer, 2003; Kuhn et al., 2006). Nonetheless, most dairy farms in Brazil are short of reaching this level.

We find that contracting labour (Hired labour) has a significant positive effect on farms' inefficiency. This somewhat confirms the entrepreneurial view that farms exclusively run by the family receive better care, leading to higher efficiency. Family labour is also less expensive as it normally is informal and does not include social security expenses. The traditional efficiency literature reports no pattern regarding the influence of the share of family labour on efficiency (Zhu and Lansink, 2010).

Remarkably, we observe the existence of trade-offs between production efficiency and risk for some variables. Milk yield presented a significant negative sign in the z-model and a significant positive sign in the v-model, suggesting that adopting more productive cows increases efficiency but also production risk. There are many factors that can contribute to these results. For example, that animals with higher production are more susceptible to diseases and metabolic disorders, inflicting abrupt and unexpected drops in production and increasing expenses with treatments (Brito et al., 2021; Knaus, 2009). They are also more demanding in terms of diet, requiring a higher level of managerial skills to provide a balanced diet year-round, according to animals' categories and productive cycle (Brito et al., 2021; Hoischen-Taubner et al., 2021). Moreover, the capital invested in more productive animals is higher which also increases losses in case of unexpected culling (Hoischen-Taubner et al., 2021). The same pattern was found for renting land; it significantly increases production risk but is beneficial to production efficiency. While renting land is associated with contractual expenses, we expect that farmers use rented land to produce high quality pasture or silage, such that it improves farm environmental efficiency. Conversely, the presence of breeding bulls in the herd significantly decreases risk, but at the same time has a negative effect on environmental efficiency.

3.3.3 Shadow price of methane emissions

The farm-specific shadow price for methane emissions is calculated with respect to the desirable output milk by using the sample mean of milk price. Since input and output variables have been normalised to estimate the production frontier, we adjust the shadow price by multiplying the result of eq. (9) by the ratio of the desirable output by the undesirable output (Mamardashvili et al., 2016). The resulting mean shadow price value is US $2,254, suggesting the opportunity cost of reducing an extra tonne of methane emitted in terms of foregone milk production would be equivalent to 6,261 kg FPCM. Therefore, under the present technology, improving farming efficiency is the most cost-effective path to mitigate emissions of the dairy farms. Notwithstanding the shadow price calculation assumes that farms are operating on the production boundary, thus shadow price values for inefficient farms may be overestimated (Adenuga et al., 2019).

To the best of our knowledge, this is the first study that has applied hyperbolic distance function to derive the shadow price of methane from dairy farms, making cross studies comparison very limited. Scaling our results to CO_2eq., by applying the conversion factor of 27.2 (Masson-Delmotte et al., 2021), we reach a value of US $83 per one tonne of CO_2eq.. Results from studies that evaluated whole farm CO_2eq. emissions varied considerably. For instance, Njuki and Bravo-Ureta (2015) reported values ranging from US $43 to US $950 per tonne of CO_2eq. for the US dairy production. The mean value reported for milk production in Germany was 165 € (US $186)[30] per tonne of CO_2eq. (Wettemann and Latacz-Lohmann, 2017), while Le et al. (2020) reported a value of Can $308.29 (US $230) [31] per tonne of CO_2eq. in Canada. Naturally, direct comparisons are not only limited by differing environmental efficiency models, but also by regional milk prices and assumptions in modelling GHG emissions, which differ considerably among studies.

3.4 Concluding remarks

Dairy farming has a crucial function in generating farm income, providing food security and employment, as well as safeguarding livelihoods in rural areas in many low and middle-income counties. Nevertheless, dairy farming is also an important contributor to greenhouse gas emissions, an externality of global concerns. Low productive cows in adverse climate settings as much as inadequate management practices compromise farm productivity and are also likely to affect their environmental performance. However, research on the environmental performance of dairy farming is limited to developed countries and high-productive systems. In this paper, we addressed this gap and analyse the environmental performance of pasture-based dairy production in Minas Gerais state in Brazil. The stochastic translog hyperbolic distance function was applied considering methane emissions as undesirable output. This approach allowed us to derive farms' specific environmental efficiency scores, identify key variables that affect efficiency and risk in milk production, and also to derive the economic/environmental trade-off in the form of shadow price for methane.

More specifically, our contribution is fourfold. First. we demonstrated that farmers can improve farms' environmental performance by increasing milk and animal liveweight outputs while simultaneously reducing methane emissions. On average, dairy farms participating in the

[30] https://www.exchangerates.org.uk/EUR-USD-spot-exchange-rates-history-2017.html
[31] https://www.exchangerates.org.uk/CAD-USD-spot-exchange-rates-history-2020.html

Full Bucket programme can increase production by 9.4% while reducing methane emissions by 8.7%. These results unveil the potential dairy farms can contribute to the Brazilian commitments for reducing methane emissions just by becoming more efficient at present input use.

Second, increasing milk yield of the cows improves environmental efficiency. This confirms previous evidence from the traditional production efficiency literature as well as life cycle assessment studies. However, policymakers should be aware that the adoption of high productive breeds does not solve the problem per se, improvement of small holder dairy farming must follow a planned sequence of steps based on a systems-thinking approach. The share of improved pastures also displayed a positive effect on the environmental efficiency of the dairy farms. This is an indication that improving forage quality goes beyond increasing milk yield but also contributes to the environmental performance of farms. This is a crucial issue in the Brazilian dairy sector which relies predominantly on pastures. In addition, we identified that increasing the share of lactating cows in the herd contributes positively to environmental efficiency. Nevertheless, adjusting herd indices to reach best performance is still a challenge for farmers in Brazil, even to those farmers taking part of a programme intended to improve farm performance. Despite having high-qualified technical support, improving the performance of dairy farms is a slow process which normally take several years to reach desirable outcomes. From this, we can deduce that long term policy and technical support is very desirable to improve the sustainability of dairy farming. Moreover, creating mechanisms to accelerate this process is urgently required, given the current urgency in reducing GHG emissions to curb global warming.

Third, our study introduced the Annual Climate Type classification in the estimation of the production frontier. The results provided evidence that dairy farmers operating in tropical and semi-arid climates are at a disadvantage when compared to farmers from areas with a humid subtropical climate. These results reinforce the necessity of considering regional climate types for designing agri-environmental policies.

Fourth, to the best of our knowledge, this is the first study that has addressed the shadow price of methane emissions from dairy farms in Brazil. We found a mean value of US $2,254 per tonne of methane and thus provide a benchmark regarding the costs of reducing methane emissions from dairy farms in Brazil. This may be useful for policy making, considering that only eliminating farm inefficiency may not be sufficient to reach the ambitious targets set by the country in international environmental agreements. However, the high abatement costs per farm suggests policymakers might face challenges in implementing mechanisms that require farmers to internalize abatement costs without any external support.

Finally, we discuss some limitations of our study. Our sample was composed exclusively of farmers taking part in a voluntary opt-in programme designed to improve farm efficiency, thus extrapolating our results for the whole population of dairy farmers in Brazil warrants caution due to selection bias. Given the actions promoted by the FB programme we expect that, on average, smallholder farmers will display lower performance than farmers in our sample. The cross-sectional characteristic of our database did not allow us to explore the dynamics in climate and annual extreme weather conditions faced by farmers in Minas Gerais. Moreover, due to the limited number of observations we derived a two levels ACT which includes a main climate group and the seasonal precipitation characteristics. Further studies considering three-level ACT classification are expected to provide further insights into the climate influence on the efficiency of dairy farms. Due to the lack of feasible measurement techniques, methane emissions needed to be calculated indirectly. This certainly added some uncertainty to our results. Lastly, this study focused exclusively on methane, which is currently the most concerning externality in the Brazilian dairy sector, nevertheless trade-offs between methane and other undesirable outputs need to be further explored in future studies in the Brazilian conditions.

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Chapter 4

4 Environmental and economic performance of paddy field-based crop-livestock systems in Southern Brazil[32]

Abstract

CONTEXT: The adoption of improved crop-livestock (CL) systems is an important strategy to enhance the sustainability of agricultural systems. However, before new CL systems technologies are brought into commercial practice their environmental and economic performance should be scrutinized. *OBJECTIVE:* We apply life cycle assessment and profit analysis to compare two technically improved paddy-based crop-livestock systems with a baseline scenario.
METHODS: Attributional life cycle assessment and profit analysis is applied to compare the performance of the three CL systems. We selected six impact categories to investigate the influence of production-related and area-related functional units. Each CL system has been evaluated by employing the crop-by-crop approach as well as the whole system analysis. The baseline scenario represents a traditional CL system in Southern Brazil, where flooded rice is grown as the spring-summer crop followed by beef cattle being released into paddy fields to graze rice straw and regrowth after the cereal is harvested. The experimental systems were managed under no-tillage, sowing of ryegrass in winter for cattle, fertilisation based on soil analysis and crop requirements, agrochemical application based on best management practices; additionally, in one of the improved systems, soybean have been introduced in the rotation with rice. *RESULTS AND CONCLUSIONS:* Results indicate that the improved experimental systems presented higher productivity and profitability per ha than the baseline system. The environmental performance was substantially affected by the functional units selected for the evaluation. Production-related functional units benefited the improved systems; conversely, the area-related functional unit benefited the less intensive baseline system. Generally, the CL system where soybean was included in the rotation presented considerably better performance; this was particularly evident with regard to the global warming potential, agricultural land occupation, and water depletion impact categories. Increasing farm inputs in the CL system without including soybean in the rotation increased profit and production per area but brought little advantages to the environmental performance.
SIGNIFICANCE: We expand the understanding of the environmental and economic performance of paddy field-based crop-livestock systems in Southern Brazil, and provide valuable information to researchers, as well as the broader audience interested in the sustainability of agricultural systems. **Keywords**: Farm management, sustainable intensification, carbon footprint, costing, life cycle analysis

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4.1 Introduction

Land-based agricultural systems are indispensable to assure global food security and livelihoods in rural areas in the coming decades (Alexandratos and Bruinsma, 2012; Willett et al., 2019). Despite its importance, agricultural expansion and a number of core agricultural practices have been linked to negative effects on humans, natural resources, and the environment (Foley et al., 2011; Tilman et al., 2011; Willett et al., 2019). According to the International Panel on Climate Change (IPCC) (2019a), the clearing of native vegetation for agricultural purposes is associated with habitat and biodiversity loss, and the release of greenhouse gases (GHG). GHG are released from biomass burned during both the clearing and soil carbon degradation processes during cultivation. Another notable example is the excessive use of fertilisers, which may lead to acidification of soils, natural resource depletion, eutrophication of water bodies, and global warming (Foley et al., 2011; IPCC, 2019a). Correspondingly, fossil fuels used in agricultural machinery release several harmful pollutants into the atmosphere, e.g., GHG and acidifying substances (IPCC, 2019a). In addition, livestock husbandry generates a substantial amount of nutrient rich manure, and emissions of methane (IPCC, 2019a). Finding solutions to these problem is a matter of urgency and will require strong actions of local and national governments in tandem with engagement from the stakeholders linked to the agri-food supply chains (Campbell et al., 2018; Charles et al., 2014; IPCC, 2019a; Roe et al., 2019).

Brazil ranks as a key agricultural producer in the world and consequently, the country's agriculture also contributes to many of the abovementioned problems (OECD/FAO, 2018). Thus, one of the main challenges for the agricultural sector in Brazil is to continue to meet global demands while simultaneously improving its environmental sustainability (Martinelli et al., 2010; OECD/FAO, 2015). One strategy that may be of great assistance in reaching this goal is the sustainable intensification of traditional crop-livestock (CL) systems; improved CL systems can promote land sparing, a reduction of environmental impacts, and ultimately increase the productivity of the land (Charles et al., 2014; De Oliveira Silva et al., 2018; Garrett et al., 2018; IPCC, 2019a). This strategy is one of the key actions of the Brazilian Low Carbon Agriculture (ABC) Program, which was launched to support research and the adoption of improved agricultural practices in the country (Brasil, 2012b). The program focuses on the restoration of degraded pastures, development and adoption of integrated systems, no-tillage cropping systems, agroforestry, and systems capable of achieving nitrogen fixation biologically (Brasil, 2012b). The

ABC's efforts have contributed to an increase in the establishment of agronomical experiments and the adoption of CL systems across the country (Bungenstab et al., 2019; Salton et al., 2014; Zotarelli et al., 2012). This is not dissimilar to old agricultural frontiers, such as in the lowlands of the Pampa biome in Rio Grande do Sul (RS), where the development and the adoption of integrated systems has increased (e.g., de Moraes et al., 2014; Theisen et al., 2017). Relevant agricultural activities in the Pampa lowlands include the production of beef cattle in natural pastures, and cultivation of flooded paddy rice. These activities are sometimes rotated as a CL system (i.e., cattle graze in paddy fields after the rice is harvested, or during rice-fallow seasons) (Martins et al., 2017; Reis, 1998). Improving this traditional CL system, by rotation of rice and soybean as the summer crop, and the sowing of winter pastures, is a promising technology towards sustainable intensification in the Pampa (de Moraes et al., 2014).

There exists a significant body of research evidencing the precedence of improved and integrated systems in comparison to traditional agricultural systems. For example, no-tillage and the use of cover crops can significantly increase soil organic carbon (Salton et al., 2014; Zotarelli et al., 2012). Furthermore, the use of rotation of leguminous crops and catch crops can reduce the needs of nitrogen fertilisers, and consequently reduce GHG emissions (Cai et al., 2018; Zotarelli et al., 2012). In economic terms, improved and integrated systems also generally present similar or indeed better returns when compared to traditional systems (Costa et al., 2018; Pashaei Kamali et al., 2016; Poffenbarger et al., 2017; Ramsden et al., 2017). However, despite the expected superiority of improved systems, the associated benefits and trade-offs should be carefully evaluated before promoting their adoption by farmers. First, it must be acknowledged that the performance of the cropping systems may vary significantly according to their location, rotation arrangements, and the species of crop and livestock adopted in the systems (Goglio et al., 2018; Nemecek et al., 2011a). And second, because the outputs of improved systems are also subject to market prices as their counterparts.

Life cycle assessment (LCA) is a quantitative environmental assessment technique that has been widely applied to evaluate and compare the environmental performance of cropping and livestock systems (Henriksson et al., 2012; Roy et al., 2009; Ruviaro et al., 2012). LCA allows researchers to assess a range of environmental impacts under a unique functional unit, and to account for upstream emissions to produce and deliver farm inputs. Generally, agricultural LCAs compare baseline, or business as usual practices with modelled scenarios or experimental data. For example, to compare integrated and organic systems (Jeswani et al., 2018; Nemecek et al., 2011a; Nunes et al., 2017), or extensive and intensive systems (Cardoso et al., 2016; Dick et al., 2015b; Nemecek et al., 2011b; Ruviaro et al., 2015). LCA was further applied to various integrated rotational systems of soybeans (Maciel et al., 2015; Matsuura et al., 2017; Prudêncio da Silva et al., 2010; Zortea et al., 2018), melon (Santos et al., 2018), and integrated crop-livestock-forestry systems (Costa et al., 2018; Figueiredo et al., 2015).

The use of LCA to assess the environmental impacts of paddy rice has increased in recent years. This can largely be attributed to the importance of this crop as a staple food, and the potential impacts associated with its production (e.g., the release of methane, and water use) (Abdul Rahman et al., 2019; Cai et al., 2018; Coltro et al., 2017; Habibi et al., 2019; Nunes et al., 2017, 2016; Ramsden et al., 2017; Yodkhum et al., 2017). Moreover, some authors have applied LCA in a joint approach by integrating economic indicators into their analysis (e.g., Costa et al., 2018; Kamali et al., 2016; Ramsden et al., 2017; Ruviaro et al., 2016; Zortea et al., 2018). With the exception of a number of studies that focus on carbon footprint, the majority of the studies have applied a range of indicators to identify hot spots and trade-offs among the agricultural systems.

Despite the substantial number of studies investigating the sustainability of agriculture in Brazil, there remains a limited number of works which focus on the application of a set of environmental and economic indicators to evaluate paddy field-based crop-livestock systems. Thus, in this study, we first apply attributional LCA to assess and compare the environmental performance of three CL systems based on paddy fields, and second, we evaluate the profitability of these systems by calculating and comparing their returns to land and management. We conduct our discussion based on a dual approach, first analysing each crop/livestock individually in a crop-by-crop approach followed by a system analysis, where we investigate each CL system as a whole (Goglio et al. 2018). Hence, this study seeks to expand the understanding of the environmental and economic performance of paddy field-based crop-livestock systems, and strives to provide invaluable information to researchers, as well as the broader audience interested in the sustainability of agricultural systems.

4.2 Materials and methods

4.2.1 Location and description of the systems

Data for this study relates to cropping livestock systems located in the Pampa Biome, State of Rio Grande do Sul (RS), Brazil. The experimental area has 18 ha and was established in Autumn-Winter 2013 on a commercial farm (4,600 ha) located in the municipality of Cristal; coordinates [30°59′59″ S, 52°02′54″ W]. The climate classification in the region is the humid subtropical oceanic climate, without dry season, with hot summers (Cfa), the mean annual temperature is 20.2 °C, and the mean annual rainfall is 1,385 mm (Alvares et al., 2013). The soil in the area was classified as haplic Gleysol (US Soil Taxonomy Entisol). The area was three years fallow and, prior to any farm operation occurring, the soil was sampled (0-10 cm). Soil samples presented the following results: pHH2O: 5.5, clay: 240 g kg^{-1}, silt: 230 g kg^{-1}, sand: 530 g kg^{-1}, organic matter: 18.0 g kg^{-1}, Cation exchange capacity (pH 7): 10.6 cmolc dm^{-3}, Base saturation: 56%, Aluminium saturation: 3%, available P ($Mehlich^{-1}$): 10 mg dm^{-3}, and exchangeable K: 76 mg dm^{-3}.

To conduct the study, we analyse data from the first four years of two treatments of the experiment. Each treatment was composed of two production seasons with livestock being produced in autumn-winter and crops in the spring-summer seasons. Both treatments were designed with three replicates with areas varying from 1 to 1.5 ha. To conduct the analysis in this study the mean values among treatments normalised to one ha were used.

The first treatment includes the production of beef cattle as the autumn-winter enterprise, followed by flooded paddy rice as the spring-summer enterprise thereafter (BR system) (see Table 1). The BR system was conducted under no-tillage management with the exception of the first season. During this period, when the area was ploughed and harrowed, lime was applied and incorporated into the soil, and the paddies were prepared (i.e. levees construction). The annual cycle of operations was as follows: commencing in April/May, spontaneous plants and crop regrowth were suppressed with herbicides; subsequently, Italian ryegrass (*Lolium multiflorum* L.) was sown and fertilised with N-P-K fertiliser. During the ryegrass growing season, topdressing fertilisation and plant protection agents were applied according to the requirements of the grass. When ryegrass reached grazing conditions, castrated bullocks (total of 695 kg liveweight (LW) ha^{-1}) were introduced in the area for a period averaging 70 days. In Spring (October/November), after removing the cattle from the area, spontaneous plants and ryegrass leftovers were suppressed with herbicides. Then, rice (*Oryza sativa* L.) was dry-seeded with localized placement of N-P-K fertiliser. During the crop growing period, the fields were continuously flooded with support of electric pumps. Topdressing fertilisation, plant protection agents, and liquid fertilizers were applied according to the culture requirements with support of tractors. Finally, when rice reached the stage of maturation, fields were let to dry out, allowing combine harvesting. Average yields in the BR CL system were 212 kgLW gain per hectare for the autumn-winter enterprise (BR_cattle), and 11,189 kg ha^{-1} for the spring-summer enterprise (BR_rice).

The second treatment was composed of beef cattle as the autumn-winter enterprise, followed by the rotation of soybean (*Glycine max* L.) and rice as the spring-summer enterprise thereafter (BSR system) (Table 1). Generally, farm operations followed the same sequence as in the BR system. In the BSR's autumn-winter enterprise, however, cattle occupation density entering the system was lower (443 kgLW ha^{-1}) with longer grazing season (109 days). In the spring-summer enterprise, soybean and rice were rotated. While the farm operations followed the same steps as in the BR system, in the BSR system the soybean did not receive irrigation nor nitrogen fertilisation. The average yields in the BSR CL system were 245 kgLW gain per ha for cattle (BSR_cattle), 3,712 kg ha^{-1} for soybeans (BSR_soybean), and 12,018 kg ha^{-1} for rice (BSR_rice).

Crop cycles were, on average, 146 and 147 days for rice and soybean, respectively. Fertilisation, spontaneous plants, and pest management followed regional guidelines and best agronomical practices seeking high yields. Inputs and outputs of both systems were documented and are presented in more detail in section 2.2.2 (Life Cycle Inventory).

The Baseline System (BL) was modelled to represent the production of rice as the spring-summer crop with beef cattle grazing rice straw and regrowth (Table 1). Technical and agronomical data for rice production was retrieved from peer-reviewed LCAs and agronomic studies developed in the region. In addition, we used data from *Rice Production Budgeting,* published regularly by the Rio Grande do Sul Rice Institute (IRGA). Data for cattle grazing rice leftovers was modelled according to Balbino et al. (2012). Soil characteristics in the BL system were assumed to be the same as in the experimental systems. The rice in the BL system was assumed to be cultivated under minimal tillage, which is the technique applied in about 60% of the rice area grown in RS (SOSBAI, 2018). The main features of the minimal tillage are that tillage and levees construction is conducted well before the sowing season, allowing the growth of spontaneous plants in the fields (SOSBAI, 2018). Rice is then dry-seeded after the suppression of the spontaneous plants that serve as soil cover (SOSBAI, 2018). Remaining management activities follow common operations as described in the experimental systems (i.e., irrigation, application of plant protection agents, topdressing ferritization, and harvest). Due to soil management for the rice crop, in the BL system, we assume that young cattle (437 kgLW ha^{-1}) entered the area soon after the rice was harvested. They were subsequently grazed for 53 days. Following this period, the soil was tilled and set to rest until October when sowing starts, (see Table 1). Average yields for the BL CL system were derived from the literature; the values applied were 56 kg LW gain per ha for cattle (BL_cattle) (Balbino et al., 2012), and 7,450 kg ha^{-1} for rice (BL_rice) (IRGA, 2022).

Table 4-1. Experimental and baseline crop-livestock systems presented by system and season.

Crop season	Crop-livestock system		
	BR[a]	BSR[b]	BL[c]
Autumn-Winter 2013	Ryegrass + Cattle	Ryegrass + Cattle	Cattle
Spring-Summer 2013-14	Rice	Soybean	Rice
Autumn-Winter 2014	Ryegrass + Cattle	Ryegrass + Cattle	Cattle
Spring-Summer 2014-15	Rice	Rice	Rice
Autumn-Winter 2015	Ryegrass + Cattle	Ryegrass + Cattle	Cattle
Spring-Summer 2015-16	Rice	Soybean	Rice
Autumn-Winter 2016	Ryegrass + Cattle	Ryegrass + Cattle	Cattle
Spring-Summer 2016-17	Rice	Rice	Rice

[a] BR: Beef cattle Rice crop-livestock system; [b]BSR: Beef cattle Soybean Rice crop-livestock system; [c]BL: Baseline crop-livestock system

4.2.2 Life cycle assessment

According to the International Organization for Standardization (ISO) 14040 and 14044 an LCA study contains four iterative phases, namely, goal and scope definition, Life Cycle Inventory (LCI), Life Cycle Impact Assessment (LCIA), and Interpretation (ISO, 2006a, 2006b).

4.2.2.1 LCA goal and scope

The main goal of the LCA in this study was to compare the environmental performance of the three systems selected. As agricultural systems are multi-functional, a dual approach is recommended when evaluating their performance (e.g., system LCA and product LCA), (see Goglio et al. (2018)). Nemecek et al. (2011a) described three functions of interest when evaluating cropping systems, namely, land management function, financial function and the productive function. Therefore, we first evaluate the productive function by conducting a crop-by-crop assessment using one tonne of grain adjusted to 13% moisture and one tonne of cattle liveweight as Functional Unit (FU). Second, we investigate the CL systems as a whole, considering land management and a productive function selecting one hectare and year (ha·a), and one tonne of crude protein (tCP) as FUs.

The experimental and baseline systems deliver rice, soybean, and cattle for fattening. Therefore, the boundaries for this study followed the cradle-to-experiment gate approach, (Fig.1). Infrastructure was not included in the LCI. Furthermore, land transformation was not included in this study because most paddy field areas in the RS have been integrated into agricultural land for longer than 20 years. The temporal boundary for the whole CL systems was from autumn-winter 2013 to summer 2016-2017 (i.e., two cycles of the farm stage in figure 1); while for the crop-by-crop approach, we assessed the eight cropping seasons individually averaging the results of each crop/livestock system. The farming stage for the three systems and the production of young stock were investigated as foreground processes. The remaining stages were considered background processes and were drawn from an LCI database. The systems produce only one output per season; hence no allocation was needed at the farm gate. Allocation in the production of young stock was solved using economic allocation. We follow the same rule to allocate lime to the individual crops in the crop-by-crop analysis.

To compare the three systems, we selected six environmental impact categories, namely, Global Warming Potential (GWP), terrestrial Acidification Potential (AP), freshwater Eutrophication Potential (EP), Fossil Depletion (FD), Agricultural Land Occupation (ALO), and Water Depletion (WD). The selection of the impact categories followed data availability and the relevance of the impact to the agricultural sector in Brazil (Matsuura et al., 2017; Ugaya et al., 2019). Some limitations of our study are discussed in section (4.4).

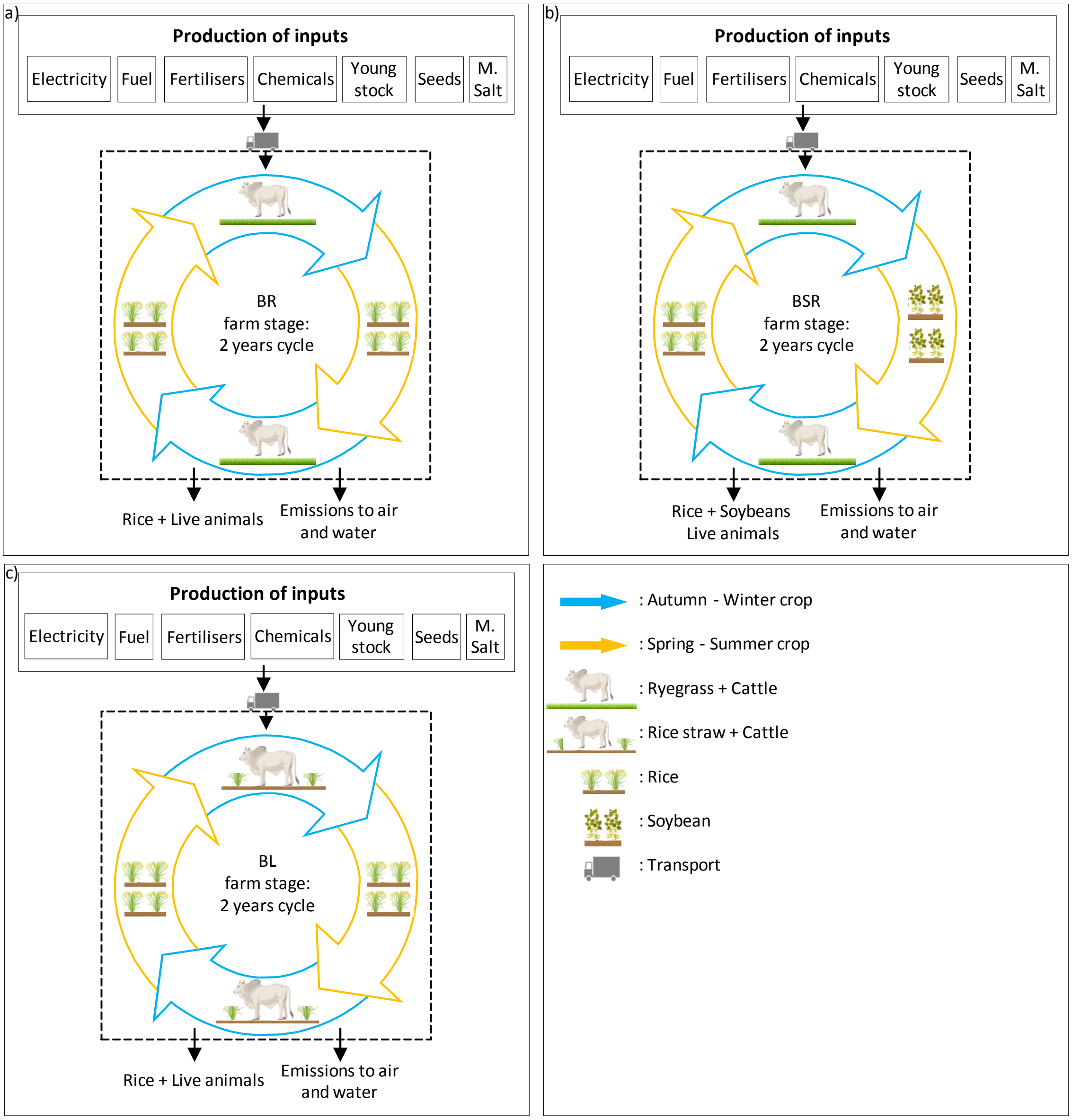

Figure 4-1. System boundaries representing one cycle of two years of the three crop-livestock systems evaluated a) CL beef cattle rice (BR), b) CL beef cattle soybean rice (BSR), c) CL baseline (BL).

4.2.2.2 Life cycle inventory

The first step in the LCI was to collect data related to the inflows and outflows of capital goods for the three CL systems. For the experimental BR and BSR systems, data was acquired from the agronomic protocols recorded during the experiment. Data for inputs and outputs for the BL system were screened from literature. The production of young stock was not part of the experiments; however, due to the relevance of the cow-calf stage to overall impacts in beef production (Dick et al., 2015a; Ruviaro et al., 2015) we also modelled this process based on information derived from the literature. The second step was to complete the LCI by calculating the emissions to air and water using models. Gross energy and protein content in soybean and rice were estimated based on the whole grain (Feedipedia, 2020; Nunes et al., 2017), See Supporting Information (SI), Table S1. Gross energy in cattle biomass was estimated based on the fat and protein content, and their calorific values (Valadares Filho et al., 2016), Table S1. The protein content of cattle was based on liveweight (FAO, 2017), Table S1. Uncertainty was propagated across the model using the equations provided by IPCC (2019d). The LCI for the farming stage per crop and ha·a are presented for the BR system in Table 2, BSR system in Table. 3, and the BL system in Table 4.

To model the suckler cow herd, we assumed that the herd was reared in natural pastures in the Pampa Biome. The herd was divided into animal categories, i.e., suckler cows, dry cows, bulls, calves, and reposition growing animals. The system's outputs accounted for were young stock (~190 kg LW per head) and cull animals to slaughter. The main characteristics regarding the herd are presented in the Tables S2 and S3. The results of the LCI for the production of calves, considering mass and economic allocation is presented in SI Table S4.

To complete the LCI we calculated direct and induced (indirect) emission to air and water for the three CL systems. Direct emissions of dinitrogen monoxide (N_2O) from crop and animal production were estimated following the IPCC guidelines (IPCC, 2019e), accounting for Nitrogen (N) content in mineral fertilizer, cattle manure, N mineralized due to soil erosion, and crop residues, Equation S1 in SI (Eq. S1). Similarly, induced emissions of N_2O from atmospheric deposition leaching and runoff of N compounds were estimated according to IPCC (2019e), (Eq. S2) and (Eq. S3), respectively. Carbon dioxide (CO_2) emissions from liming was considered only for the BR and BSR systems. Limestone was assumed to be dolomite and therefore the Emission Factor (EF) applied was 0.13, (Eq. S4) (IPCC, 2006). CO_2 from urea fertilisation was calculated based on the EF of 0.20 (Eq. S5) (IPCC, 2006). Ammonia (NH_3) volatilisation was calculated based on the total N applied to agricultural soil and its fraction volatilised as NH_3-N; we applied the EF of 0.142 for urea, 0.03 for Ammonium Nitrate (AN), and 0.197 for animal manure and crop residues (IPCC 2019c p.11.46-47). Similar to NH_3, nitrogen oxides (NO_x) emissions were calculated based on total N applied to agricultural soil and its fraction volatilised; EF applied were of 0.011 for urea, 0.029 for Ammonium Nitrate (AN), and 0.015 for animal manure and crop residues (IPCC 2019c p.11.46-47).

Furthermore, we calculated methane (CH_4) emissions to air from paddy fields and cattle production. Daily methane emissions from flooded paddies were derived from recent field measurements from rice production in RS. We applied the daily EF of 4.06 kg ha^{-1} for the no-till systems (BR, BSR), and 4.50 for minimal tillage (BL), Table S1. Flooding period for BR and BSR systems varied from 106 to 121days, based on the minimum maturity cycle of the rice variety as declared by SOSBAI (2018). For the BL system, we assumed 100 days as in Nunes (2016) and IRGA (2018). Methane emissions from cattle enteric fermentation and manure management were calculated per animal category using growth characteristics, daily feed intake, feed characteristics, daily volatile solids excreted, and regional emission factors (FAO, 2016; IPCC, 2019c). The daily enteric fermentation EF was derived from digestible energy from feed and animals' gross energy requirement applying Eq. S6 through S14 (IPCC, 2019c). Methane from manure deposited on pastures was calculated based on volatile solids (VS) excretion and the maximum methane producing capacity (EF 0.19) (Eq. S15, S16) (IPCC, 2019c). The amount of N and Phosphorous (P) excreted by animals was calculated using the mass balance approach (Eq. S17, S18, S19) (IPCC, 2019c).

Emissions to water accounted for were nitrate (NO_3) and phosphate (PO_4) from leached nutrients, phosphate from runoff P compounds, and P from soil erosion. The EF applied to NO_3 leaching was 0.24 of the total N (IPCC 2019c p.11.25). Emissions of P compounds were estimated according to Nemecek and Schnetzer (2012), (Eq. S20, S21, S22). As no uncertainty is provided to EF related to P compounds emissions, the highest adjusted uncertainty value found in our model was adopted as a proxy.

Water consumption for rice irrigation in the BR and BSR systems was estimated based on pump capacity and electricity consumption; for the BL system, the value was based on Coltro et al. (2017). Water consumption by cattle was estimated following Hicks et al. (1988). Emissions associated with fuel burned in agricultural machinery were calculated according to EFs provided by Nemecek and Kagi (2007).

Information regarding background production processes of fertilisers, agrochemicals, mineral salt, seeds, and electricity were retrieved from the ecoinvent® v.3.01 database, adjusting to the Brazilian electricity mix (Ecoinvent, 2019). The transportation of the inputs was also considered selecting EURO 3 technology. The selected processes are presented in Table S5.

Table 4-2. Life cycle inventory for inputs and outputs, and field emissions from experimental crop-livestock system Beef cattle and Rice (BR), presented per crop and hectare-and-year.

Description	Unit	Crop-livestock system BR					
		ha·a	CV[a]	Cattle	CV	Rice	CV
Inputs from nature							
Land occupation	ha	1.00E+00		1.10E+00	(0.004)	8.96E-02	(0.054)
Water	m^3	1.10E+04		8.11E+00	(0.004)	9.85E+02	(0.054)
Inputs from technosphere (economic)							
Seed	kg	1.34E+02		3.31E+01	(0.004)	9.30E+00	(0.140)
Lime	t	1.13E+00		4.56E-01	(0.004)	6.37E-02	(0.001)
Young animals liveweight	kg	6.94E+02		7.66E+02	(0.001)		
N, from Ammonium Nitrate	kg	3.00E+01		1.10E+01	(0.004)	1.79E+00	(0.054)
N, from Urea	kg	2.48E+02		1.27E+02	(0.085)	1.19E+01	(0.050)
P2O5, from triple superphospahte	kg	1.88E+02		1.30E+02	(0.105)	6.27E+00	(0.054)
K2O, from Postasium Chloride	kg	2.39E+02		1.35E+02	(0.078)	1.04E+01	(0.072)
Electricity	kWh	5.68E+02				5.09E+01	(0.073)
Diesel agricultural machinery	kg	9.89E+01		1.31E+01	(0.115)	7.79E+00	(0.065)
Mineral salt	kg	1.08E+01		1.20E+01	(0.074)		
Plant protection agents	L	1.85E+01		3.86E+00	(0.279)	1.35E+00	(0.240)
Liquid fertilizer	L	2.11E+00				1.91E-01	(0.468)
Transport, lorry 16-32 tons	tkm	1.32E+02		4.29E+01	(0.092)	8.36E+00	(0.020)
Transport, lorry 7.5-16 tons	tkm	3.63E+01		3.91E+01	(0.003)	7.70E-02	(0.295)
Outputs							
Production yield[b]	t			1.00E+00		1.00E+00	
Production of energy equivalent	GJ	1.79E+02		8.27E+00		1.53E+01	
Production of protein equivalent	kg	1.17E+03		1.69E+02		9.14E+01	
Emissions to air							
Methane	kg	5.21E+02	(0.145)	5.01E+01	(0.096)	4.27E+01	(0.316)
Direct dinitrogen monoxide	kg	4.96E+00	(0.134)	4.23E+00	(0.227)	1.01E-01	(0.891)
Induced dinitrogen monoxide	kg	2.90E+00	(0.257)	1.58E+00	(0.661)	1.31E-01	(0.782)
Carbon dioxide	kg	9.31E+02	(0.080)	2.13E+02	(0.237)	6.59E+01	(0.192)
Ammonia	kg	5.47E+01	(0.356)	3.43E+01	(0.824)	2.11E+00	(1.242)
Nitrogen oxides	kg	1.41E+01	(0.360)	8.12E+00	(0.973)	5.99E-01	(1.070)
Emissions to water							
Nitrate	kg	1.38E+02	(0.447)	9.20E+01	(1.095)	4.89E+00	(1.520)
Phosphate	kg	2.36E+00	(0.657)	1.68E+00	(1.788)	7.49E-02	(1.757)
Phosphorus	kg	2.14E-01	(0.868)	2.70E-02	(1.945)	1.70E-02	(1.945)

[a] Coefficient of Variation; [b] grain adjusted to 13% moisture, and cattle as liveweight

Table 4-3. Life cycle inventory for inputs and outputs, and field emissions from experimental crop-livestock system Beef cattle Soybean and Rice (BSR), presented per crop and hectare-and-year.

Description	Unit	Crop-livestock system BSR							
		ha·a	CV[a]	Cattle	CV	Soybean	CV	Rice	CV
Inputs from nature									
Land occupation	ha	1.00E+00		1.45E+00	(0.012)	2.71E-01	(0.100)	8.34E-02	(0.066)
Water	m^3	5.51E+03		1.13E+01	(0.007)			9.17E+02	(0.066)
Inputs from technosphere (economic)									
Seed	kg	9.54E+01		4.34E+01	(0.012)	1.02E+01	(0.193)	7.76E+00	(0.015)
Lime	t	1.13E+00		5.97E-01	(0.010)	1.18E-01	(0.001)	8.21E-02	
Young animals liveweight	kg	4.46E+02		6.46E+02	(0.006)				
N, from Ammonium Nitrate	kg	2.00E+01		1.45E+01	(0.012)			1.67E+00	(0.066)
N, from Urea	kg	1.83E+02		1.66E+02	(0.082)			1.12E+01	(0.013)
P2O5, from triple superphospahte	kg	2.08E+02		1.77E+02	(0.117)	2.71E+01	(0.100)	5.84E+00	(0.066)
K2O, from Postasium Chloride	kg	2.29E+02		1.81E+02	(0.075)	2.48E+01	(0.408)	9.81E+00	(0.096)
Electricity	kWh	2.78E+02						4.64E+01	(0.074)
Diesel agricultural machinery	kg	8.34E+01		1.72E+01	(0.121)	1.52E+01		7.22E+00	(0.126)
Mineral salt	kg	1.24E+01		1.79E+01	(0.034)				
Plant protection agents	L	1.52E+01		4.51E+00	(0.533)	3.06E+00	(0.083)	1.09E+00	(0.376)
Liquid fertilizer	L	9.49E-01				9.33E-02	(0.096)	1.33E-01	(0.820)
Transport, lorry 16-32 tons	tkm	1.27E+02		5.86E+01	(0.086)	1.38E+01	(0.114)	1.01E+01	(0.014)
Transport, lorry 7.5-16 tons	tkm	2.37E+01		3.34E+01	(0.007)	1.58E-01	(0.104)	6.12E-02	(0.448)
Outputs									
Production yield[b]	t			1.00E+00		1.00E+00		1.00E+00	
Production of energy equivalent	GJ	1.36E+02		8.27E+00		2.05E+01		1.53E+01	
Production of protein equivalent	kg	1.30E+03		1.69E+02		3.45E+02		9.14E+01	
Emissions to air									
Methane	kg	2.80E+02	(0.185)	7.20E+01	(0.096)			3.85E+01	(0.316)
Direct dinitrogen monoxide	kg	4.61E+00	(0.129)	5.57E+00	(0.244)	9.51E-02	(0.292)	9.65E-02	(0.888)
Induced dinitrogen monoxide	kg	2.29E+00	(0.277)	2.12E+00	(0.653)	4.18E-02		1.25E-01	(0.784)
Carbon dioxide	kg	8.27E+02	(0.096)	2.89E+02	(0.231)	6.61E+01	(0.250)	8.40E+01	(0.204)
Ammonia	kg	4.41E+01	(0.378)	4.64E+01	(0.800)			2.00E+00	(1.243)
Nitrogen oxides	kg	1.10E+01	(0.412)	1.09E+01	(0.980)			5.65E-01	(1.071)
Emissions to water									
Nitrate	kg	1.20E+02	(0.478)	1.26E+02	(1.103)	2.13E+00	(1.551)	4.69E+00	(1.520)
Phosphate	kg	2.53E+00	(0.654)	2.28E+00	(1.793)	2.89E-01	(1.795)	6.98E-02	(1.757)
Phosphorus	kg	1.31E-01	(0.991)	3.44E-02	(1.945)	8.46E-03	(1.945)	1.52E-02	(1.945)

[a] Coefficient of Variation; [b] grain adjusted to 13% moisture, and cattle as liveweight

Table 4-4. Life cycle inventory for inputs and outputs, and field emissions from crop-livestock Baseline (BL), presented per crop and hectare-and-year.

Description	Unit	Crop-livestock system BL					
		ha·a	CV[a]	Cattle	CV	Rice	CV
Inputs from nature							
Land occupation	ha	1.00E+00		2.03E+00	(0.012)	1.34E-01	(0.014)
Water	m³	1.10E+04		7.32E+00	(0.007)	1.48E+03	(0.014)
Inputs from technosphere (economic)							
Seed	kg	1.20E+02				1.61E+01	(0.014)
Lime	t						
Young animals liveweight	kg	4.37E+02		8.88E+02	(0.006)		
N, from Ammonium Nitrate	kg	1.50E+01				2.01E+00	(0.014)
N, from Urea	kg	9.20E+01				1.24E+01	(0.014)
P2O5, from triple superphospahte	kg	6.00E+01				8.05E+00	(0.014)
K2O, from Postasium Chloride	kg	9.00E+01				1.21E+01	(0.014)
Electricity	kWh	7.46E+02				1.00E+02	(0.014)
Diesel agricultural machinery	kg	1.33E+02				1.78E+01	(0.019)
Mineral salt	kg	5.20E+00		1.06E+01	(0.074)		
Plant protection agents	L	8.07E+00				1.08E+00	(0.187)
Liquid fertilizer	L	2.78E-01				3.72E-02	(0.087)
Transport, lorry 16-32 tons	tkm	3.10E+01				4.16E+00	(0.011)
Transport, lorry 7.5-16 tons	tkm	2.25E+01		4.49E+01	(0.006)	5.60E-02	(0.189)
Outputs							
Production yield[b]	t			1.00E+00		1.00E+00	
Production of energy equivalent	GJ	1.18E+02		8.27E+00		1.53E+01	
Production of protein equivalent	kg	7.64E+02		1.69E+02		9.13E+01	
Emissions to air							
Methane	kg	4.75E+02	(0.150)	5.04E+01	(0.096)	6.04E+01	(0.316)
Direct dinitrogen monoxide	kg	9.57E-01	(0.391)	3.39E-01	(1.621)	1.06E-01	(0.884)
Induced dinitrogen monoxide	kg	1.18E+00	(0.337)	3.14E-01	(0.623)	1.37E-01	(0.771)
Carbon dioxide	kg	1.47E+02	(0.125)			1.97E+01	(0.250)
Ammonia	kg	1.98E+01	(0.513)	6.96E+00	(0.299)	2.20E+00	(1.239)
Nitrogen oxides	kg	5.46E+00	(0.482)	1.43E+00	(1.952)	6.38E-01	(1.069)
Emissions to water							
Nitrate	kg	5.54E+01	(0.569)	3.42E+01	(1.384)	5.17E+00	(1.519)
Phosphate	kg	9.19E-01	(0.680)	5.15E-01	(1.561)	8.93E-02	(1.782)
Phosphorus	kg	2.10E-01	(0.868)	4.90E-02	(1.945)	2.50E-02	(1.945)

[a] Coefficient of Variation; [b] grain adjusted to 13% moisture, and cattle as liveweight

4.2.2.3 Life Cycle Impact Assessment and interpretation

Two LCIA methods were applied to convert the LCI results into the six impact categories selected. First, the (IPCC) Global Warming Potential (GWP kg CO_2 equivalent (eq.)) selecting the 100-year time horizon (Myhre et al., 2013). Second, we selected the Recipe midpoint (H) World Recipe H, considering the following impact categories and respective units: terrestrial Acidification Potential (AP: kg SO_2 eq.), freshwater Eutrophication Potential (EP: kg P eq.), Fossil Depletion (FD: kg oil eq.), Agricultural Land Occupation (ALO: $m^2 \cdot a$), and Water Depletion (WD: m^3) (Goedkoop et al., 2008). Both methods were implemented using the SimaPro® v. 8.2.0 software, which was used to assemble the life cycle for each system and conduct the calculations (PRé Sustainability, 2019).

The results were generated by running 1,000 Monte Carlo (MC) iterations with a 95% confidence interval using SimaPro. Infrastructure was excluded from the analysis. To compare the LCIA results, we first used the traditional deterministic approach by comparing point values. In addition, we used a discernibility analysis, which is a pairwise method used to contrast two or more alternatives (Mendoza Beltran et al., 2018). Discernibility analysis compares the alternatives by identifying the number of MC runs in which one system presented lower impact than the other (Heijungs and Kleijn, 2001; Mendoza Beltran et al., 2018). To narrow our interpretation of the discernibility analysis, we emphasised the results where one system presented a comparatively lower impact than the other system in at least 95% of the number of MC runs (Heijungs and Kleijn, 2001).

4.2.3 Economic performance

Profitability was calculated for each enterprise of the CL systems as the return to land and management (i.e., gross revenue less total costs, not including costs of management and land) (Kay et al., 2016; Poffenbarger et al., 2017). The averages for the four years cycle are presented per crop/livestock season and CL system.

Budgets for each CL system and cropping season were constructed following the guidelines from the Brazilian National Food Supply Company (Conab, 2010). To complete the budgets, we used the data described in the preview section and the *Rice Production Budgeting* (IRGA, 2022). The following costs were included in the analysis: cost of seeds, electricity, transportation, machinery operations, insurance, hired labour, licences for operating, and drying expenses (IRGA, 2022); fertilisers and depreciation (Conab, 2020; IRGA, 2022); agrochemicals (Conab, 2020), completed with field survey at regional retailers; purchased cattle (Emater-RS, 2020); and interests following the Brazilian benchmark interest rate (i.e., Selic rate). We consulted output prices for soybeans which were recorded by Agrianual (2019), for rice as recorded by IRGA (2020), and for beef cattle according to Emater-RS (2020).

In all three systems, farm machinery operations and labour were accounted for by adopting the operations time and equipment efficiency for each activity according to IRGA (2020). Machinery used in the field operations for the soybean crop was assumed to be the same, and assumed to have the same efficiency as for the rice crop in the experimental systems (i.e., sowing, spraying, and harvesting). Young stock was assumed to be purchased when ryegrass was suitable for grazing, or after rice harvesting in the BL system, and sold at the end of the period in the system. Thus, animals left the system before being ready to slaughter. For this reason, when calculating the return from the cattle enterprises, we computed only the live weight gain in the period in which the animals remained in the systems, assuming ready-to-slaughter prices. Moreover, no freight or transaction costs were added to the calculations of cattle enterprises. The results were adjusted in response to inflation and are presented in US dollars to the year 2017.

4.3 Results

4.3.1 Crop-by-crop analysis

In the crop-by-crop analysis, each production season was assessed individually. The results for the production of one tonne of cattle indicate that the two experimental systems (BR_cattle; BSR_cattle) presented lower impacts than the baseline system (BL_cattle) in four out of the six impact categories evaluated, namely GWP, AP, EP, and ALO (Fig. 2a to 2f). Generally, young stock production was the process with greater burdens, and together with the farming stage it was responsible for 65% or more of the impacts in five of the categories analysed GWP, AP, EP, ALO, and WD. For the production of rice, the experimental BR_rice and BSR_rice systems presented higher performance than the BL_rice system in all six impact categories (Fig. 2g to 2l). Moreover, the farming stage dominated five out of six impact categories for rice production (GWP, AP, EP, ALO, and WD). The contribution of the farming stage to each impact category was similar for all three rice systems, responsible for approximately 89%, 87%, 58%, 83%, and 99% of the impacts for GWP, AP, EP, ALO, and WD, respectively. The soybean crop presented very low GWP, AP, and WD; conversely, the EP and ALO were higher than for the rice crop (Fig. 2i and 2j).

The BSR_cattle system presented the lowest GWP among the three systems (22,826 kg CO_2 eq. tLW^{-1}) and was followed by the BR_cattle and BL_cattle system with a GWP of 24,503 and 25,470 kg CO_2 eq. tLW^{-1}, respectively (Fig. 2a). The production of young stock in low productive suckler herds, as modelled in this study, was very resource-demanding (e.g., land, and water) and produced a significant amount of GHG. This, in turn, contributed significantly to the impact categories associated with these issues. Methane emissions were the most important contributor to GWP with absolute values of 19,088, 16,852, and 21,952 kg CO_2 eq. tLW^{-1} for BR_cattle, BSR_cattle, and BL_cattle, respectively. Most of the methane was emitted during the production of the young stock that entered the systems (i.e., 93% BR_cattle; 89% BSR_cattle; 94% BL_cattle). For the production of rice, the BL_rice system presented a GWP 34% and 47% higher than the BR_rice and BSR_rice systems, respectively. As expected, the GWP for rice production was also dominated by methane emissions, totalling 80%, 78%, and 83% for BR_rice, BSR_rice, and BL_rice, respectively. The BSR_soybean system presented the lowest GWP per FU (262 kg CO_2 eq. t^{-1}) (Fig. 2g). The farming stage contributed only 39% of the total GWP for soybean production; these impacts primarily originated from CO_2 emissions from lime application, followed by emissions from the use of fuel and fertilisers.

For the terrestrial acidification potential, the BL_cattle system presented the highest value per FU, followed by the experimental BR_cattle and BSR_cattle systems with an AP of 729 and 651 kg SO_2 eq. tLW^{-1}, respectively (Fig. 2b). The main contributor to the AP in the experimental systems was NH_3 emissions from manure deposited onto pastures and fertilisers applied during ryegrass production. The AP for the production of one tonne of rice was lower in the experimental systems than the BL_rice system (7.4% BR_rice; 12% BSR_rice). For rice production, NH_3 from N fertilisation was the main contributor to the AP. The BSR_soybean, once again, presented the lowest impact with an AP of 1.0 kg SO_2 eq. t^{-1} (Fig. 2h). This lower value was expected, as no N fertilisation was applied for soybean production.

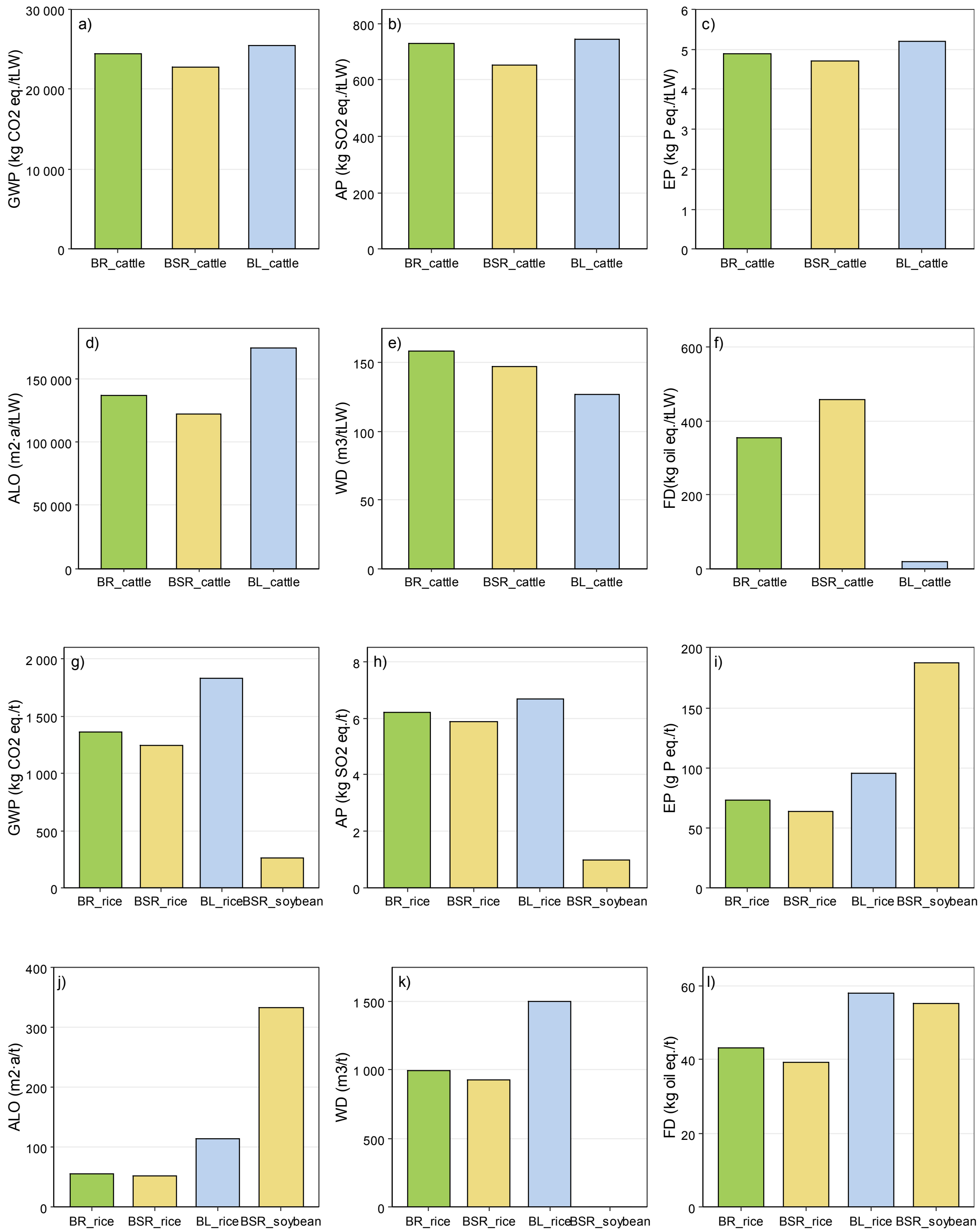

Figure 4-2. Crop-by-crop life cycle impact assessment for the production of one tonne of cattle liveweight (LW) and one tonne of crop. (GWP) Global Warming Potential, (AP) terrestrial Acidification Potential, (EP) freshwater Eutrophication Potential, (FD) Fossil Depletion, (ALO) Agricultural Land Occupation) and (WD) Water Depletion.

Freshwater eutrophication potential was very similar in the three systems for the production of cattle (Fig. 2c). Regardless of the system, EP for cattle production was dominated by the loss of P-compounds during the production of young stock, 74% and 82% for the experimental systems BR_cattle and BSR_cattle, respectively, and 96% for the BL_cattle system. Also, due to the P-fertilisation for the production of ryegrass, the contribution of the experiment stage was higher for the BR_cattle (11%) and BSR_cattle (16%) systems than for the BL_cattle (4%) system. EP for rice production was 23% and 34% lower in the experimental systems BR_rice and BSR_rice, respectively, when compared to the BL_rice system. Farm emissions contributed 58% of the EP in all three systems; this was followed by the production of P-fertilisers with approximately 19%. The BSR_soybean crop presented an EP higher than rice crops (0.187 kg P eq. t^{-1}) (Fig. 2i). This result was firstly driven by the loss of P-compounds at the farm (56%), and secondly by the production of P-fertilisers (31%).

The impacts related to fossil depletion for cattle production were significantly higher in the two experimental systems. The FD for the BR_catlle and BSR_cattle was 18 and 24 folds higher than the BL_cattle system (Fig. 2f). These results were dominated by the production of fertilisers, which contributed to 82% and 84% of FD for the BR_cattle and BSR_cattle system, respectively. For the production of rice, the FD for the BL_system (58 kg oil eq. t^{-1}) was 34% and 48% higher than the BR_rice and BSR_rice, respectively. The soil management in the BL_rice system (minimal tillage) was more machinery demanding than the no-tillage practice applied in the experimental systems, which increased the FD substantially in the BL_rice system. FD contributions for crop production were primarily from production of fertilisers (BL_rice 45%, BR_rice 55%, BSR_rice 57%, BSR_soybean 44%) and from fuel (BL_rice 35%, BR_rice 21%, BSR_rice 21%, BSR_soybean 32%). FD for the production of agrochemicals ranged from 7% for the BL_rice, to 21% for the BSR_soybean system.

Results of the ALO for the production of one FU of cattle in the BL_cattle system (174,912 $m^2 \cdot a\ tLW^{-1}$) indicate a significantly higher impact than the experimental systems BR_cattle (136,890 $m^2 \cdot a\ tLW^{-1}$) and BSR_cattle (121,900 $m^2 \cdot a\ tLW^{-1}$) (Fig. 2d). The area occupied for the production of young stock contributed 95%, 90% and 86% of the total ALO for BR_cattle, BSR_cattle, BL_cattle, respectively. For the production of rice, the ALO result for the BL_rice system (114 $m^2 \cdot a\ t^{-1}$) was more than two folds higher than the experimental systems (Fig. 2j). In addition, the production of soybean in the BSR_soybean system presented the highest ALO among the crop systems (332 $m^2 \cdot a\ t^{-1}$).

The Water Depletion was mainly driven by drinking water for the cattle production and irrigation water for rice production. The experimental BR_cattle and BSR_cattle systems presented higher WD than the BL_cattle system; this was mainly driven by the inputs to produce ryegrass in the experimental systems. Conversely, in the production of rice, the baseline BL_rice system presented the highest WD (1,503 $m^3\ t^{-1}$), which was 51% and 62% higher than the BR_rice and BSR_rice system, respectively. No irrigation was used for soybean production and therefore, the BSR_soybean WD was very low compared to the other systems (3.5 $m^3\ t^{-1}$) (Fig. 2k).

4.3.2 Environmental performance of the whole crop-livestock systems

The whole system results indicate that the experimental BSR system generally performed better than the BR and BL systems (Table 5). However, this tendency was only partially confirmed when applying discernibility analysis with the 95% threshold selected by the authors (Fig. 3).

Table 4-5. Life cycle impact assessment results for three crop-livestock systems and two functional units.

Impact Category	Unit	$ha \cdot a^a$			tCP^b		
		BR	BSR	BL	BR	BSR	BL
IPCC GWP 100a	kg CO_2eq	37,405	23,699	25,940	31,897	18,293	34,101
Terrestrial acidification	kg SO_2eq	721	489	414	615	376	535
Freshwater eutrophication	kg Peq	5.4	4.1	3.1	4.6	3.0	4.3
Fossil depletion	kg oileq	800	654	444	680	501	582
Agricultural land occupation	m^2a	127,693	85,785	84,173	108,676	65,739	110,221
Water depletion	m^3	11,131	5,681	11,162	9,609	4,329	14,692

[a]hectare and year; [b]Tonne of crude protein

In the area-related evaluation, we analysed the impacts of producing one hectare of each CL system disregarding the productivity of the systems. Following this approach, the less intensive BL system performed better than the two experimental systems. It performed considerably better than the BR system in five out of six impact categories and better than the BSR system in three out of the six impact categories assessed. Although the BSR system performed better than the BL in GWP and WD, only WD was significantly lower. The GWP in the BSR system was 23,699 kg CO_2

eq. (ha·a)$^{-1}$ against 25,940 kg CO_2 eq. (ha·a)$^{-1}$ in the BL system with 84% of the MC runs favouring the BSR system. When comparing the BSR and BR systems, the BSR system displayed a considerably better performance, with 100% of the MC runs having a lower impact in five out of six impact categories (Fig. 3). Moreover, point value for EP in the BSR system was 4.1 kg P eq. (ha·a)$^{-1}$ and in the BR system 5.4 kg P eq. (ha·a)$^{-1}$, which yielded 90% of the MC runs in favour of the BSR system (Fig. 3).

Unit of reference →		ha·a				tCP		
		0%	50%	100%		0%	50%	100%
Impact category								
Global Warming Potential	$j\downarrow k\rightarrow$	BR	BSR	BL	$j\downarrow k\rightarrow$	BR	BSR	BL
	BR		0%	0%	BR		0%	79%
	BSR	100%		84%	BSR	100%		100%
	BL	100%	16%		BL	21%	0%	
Terrestrial Acidification Potential	$j\downarrow k\rightarrow$	BR	BSR	BL	$j\downarrow k\rightarrow$	BR	BSR	BL
	BR		0%	0%	BR		0%	8%
	BSR	100%		4%	BSR	100%		100%
	BL	100%	96%		BL	92%	0%	
Freshwater Eutrophication Potential	$j\downarrow k\rightarrow$	BR	BSR	BL	$j\downarrow k\rightarrow$	BR	BSR	BL
	BR		10%	1%	BR		5%	18%
	BSR	90%		3%	BSR	95%		87%
	BL	99%	97%		BL	83%	14%	
Fossil Depletion	$j\downarrow k\rightarrow$	BR	BSR	BL	$j\downarrow k\rightarrow$	BR	BSR	BL
	BR		0%	0%	BR		0%	0%
	BSR	100%		0%	BSR	100%		100%
	BL	100%	100%		BL	100%	0%	
Agricultural Land Occupation	$j\downarrow k\rightarrow$	BR	BSR	BL	$j\downarrow k\rightarrow$	BR	BSR	BL
	BR		0%	0%	BR		0%	56%
	BSR	100%		46%	BSR	100%		100%
	BL	100%	55%		BL	44%	0%	
Water Depletion	$j\downarrow k\rightarrow$	BR	BSR	BL	$j\downarrow k\rightarrow$	BR	BSR	BL
	BR		0%	43%	BR		0%	100%
	BSR	100%		100%	BSR	100%		100%
	BL	57%	0%		BL	0%	0%	

Figure 4-3.Discernibility analysis with the percentage of the Monte Carlo runs in which crop-livestock system ***j*** had lower environmental impact than crop-livestock system ***k***.

Adapted from (Mendoza Beltran et al., 2018)

When considering the production-related functional unit, we observed a trend in favour of the experimental systems. Point value results of producing one tonne of protein in the BSR system were lower than the BR and BL systems for all impact categories (see Table 5). This superior performance was confirmed in the discernibility analysis where the BSR system performed considerably better than the BR and BL systems in all impact categories but EP against the BL system (see Fig. 3). Differences in the comparison between BR and BL systems were not as prominent in the discernibility analysis where the BR system presented only the ALO impact category with considerably lower impacts. The GWP in the BSR system was lower than the BR and BL systems, by 42% and 46%, respectively. In the ALO impact category, the differences between the BL system presented a value of 44,482 m^2a higher than the BSR system. In terms of water depletion, the difference between the BSR and BL systems totalled 10,363 m^3 per tonne of protein produced.

4.3.3 Economic analysis

Figure 4 shows the cumulative distribution function of the return to land and management per hectare on an annual basis. The experimental BR and BSR systems presented a higher return to land and management than the BL system. The return to land and management was, on average, highest in the BR system (\$997.75 $(ha{\cdot}a)^{-1}$), followed by the BSR system (\$457.40 $(ha{\cdot}a)^{-1}$) and BL system (\$255.70 $(ha{\cdot}a)^{-1}$) (Figure 4, and SI Results Table S6). Despite the overall better performance of the experimental CL systems, the cattle enterprises generated negative return to land and management. These negative results were mainly driven by the costs of inputs required to cultivate the ryegrass pasture (Table S6).

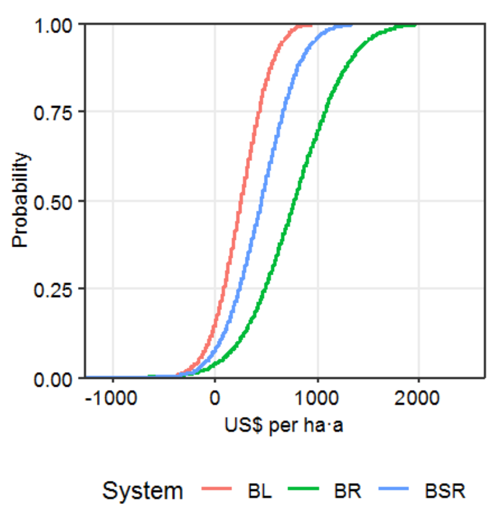

Figure 4-4. Cumulative Distribution Function of the Profit (return to land and management) for the three crop-livestock systems. Beef cattle rice (BR), beef cattle soybean rice (BSR), baseline (BL).

4.4 Discussions

4.4.1 Crop-livestock systems and sustainable intensification of lowlands

Based on the results of our analysis, it is possible to suggest that the BSR CL system is generally environmentally advantageous over the traditional BL and improved BR system. However, the degree to which we can support this assertion depends on the FU adopted to compare the systems, to confirm the complexity involved in analysing crop-livestock systems and the necessity of multiple-approach evaluations (Goglio et al., 2018; Nemecek et al., 2011a). Three main characteristics of the BSR system can be associated with its better performance: the lower density of young stock entering the system, soybean crop, and higher yields.

The lower animal stocking density per area in the BSR system allowed a higher daily weight gain and a longer grazing season. With this strategy, the BSR system benefited twice; first, from the lower environmental load carried into the system by young stock and second, from increased liveweight gain in high-quality pastures. Moreover, a longer grazing season in ryegrass assures feed supply in a period where natural pastures experience less vegetative growth in the region (de Moraes et al., 2014). Further environmental optimisation of this system may seek to find an optimal initial stocking density that expands the grazing season, considering pasture support capacity and pasture production cycle.

The soybean crop was the most important element responsible for the considerably lower GWP and WD in the BSR system. Two advantages of using soybean in the BSR system are identified. First, soybean does not require irrigation; and second, it does not require N-fertilisation. In contrast to rice, soybean cultivation requires drained soils, and thus, paddy fields need to be dried out for planting soybean. With this practice, the common emissions of methane that occur during the production of rice are avoided for a whole season. Moreover, soybean is a leguminous crop, and its cultivation in South Brazil is conducted, as stated, without the application of N-fertilisers. Resultantly, cultivation of this crop does not lead to emissions associated with the production and use of N-fertilisers, a considerable contributor to overall GWP in agricultural systems (Nemecek et al., 2011a). Despite the fact that, in our study, the soybean crop reached higher yields and was successful, in practice, some areas of lowlands with paddy fields are difficult to drain properly hindering the wide adoption of soybean in these areas. A possible solution for this problem may be the construction of ridges in the paddy fields. However, this practice characterises the adoption of a more distinct CL system than the ones evaluated in our study (Theisen et al., 2017).

The higher inputs of synthetic fertilisers applied to the experimental systems were the main driver of the high yields reached in these systems. The yields of rice in the BR and BSR systems were about 50% and 61% higher than in the BL_rice system which represented regional averages, and the BSR_soybean system yield was, on average, 37% higher than the regional average (~ 2.7 t ha^{-1}). Furthermore, the production of quality ryegrass to support livestock was only made possible by fertilised production. High-yield systems benefit when certain impacting factors show no increase, or minimal increase, with the intensification of the production (e.g., land occupation, water for irrigation, methane release from paddy fields) (Nunes et al., 2016). Therefore, the experimental systems, in particular the BSR, displayed a better performance in the GWP, ALO, and WD impact categories. Regardless of having similar management, the BSR_rice yields were higher than the BR_rice, which may be an effect of the leguminous crop in the BSR system. Although the LCA approach adopted in this study could not confirm this hypothesis, there is substantial evidence that the adoption of leguminous crops can reduce the requirements of mineral fertilisers and improve cropping systems yields (Arunrat et al., 2016; Cai et al., 2018; Nemecek et al., 2011b; Theisen et al., 2017).

Despite the higher yields that can be reached by increasing the application of fertilisers, this agricultural practice may contribute to nutrient loss and GHG emissions at the farming stage. The high demand for fertilisers was also the main driver of the poor performance of the two experimental systems in the FD, AP, and EP impact categories. The intensity in which this affects the CL systems is more evident when applying the area-related analysis. Nemecek at al. (2011b) confirmed that the reduction of fertilisers can significantly decrease environmental impacts in cropping systems, however, trade-offs are very likely to occur (e.g., reduction of yields). Moreover, these authors also stress the importance of maintaining some level intensity to ensure a balance between environmental impacts and economic returns of the farming systems. Thus, finding a balance between the use of fertilisers and the volume of production may be an important target to ensuring the sustainability of CL systems in lowland paddy fields.

The adoption of no-till management in the experimental systems led to the reduction of fossil fuel usage and related emissions. Additionally, no-till management may also reduce soil carbon mineralisation and improve soil quality (Nemecek et al., 2011b). In practice, however, some lowland areas are not well-drained and the use of mechanisation may leave tracks on the fields limiting no-till management in the next crop. In these areas, partial tillage may be an option to level the fields, thus allowing proper mechanisation (Theisen et al., 2017).

Finding an optimal production strategy in CL systems becomes even more complex when including the economic pillar in one's analysis. This issue was clear in our study when we found that the system with better environmental performance (BSR) was not as efficient in economic terms. These results were not surprising, as conflicts between environmental and economic goals are often found in ecoefficiency evaluations of agricultural systems (Cai et al., 2018; Nemecek et al., 2011b; Pashaei Kamali et al., 2016; Poffenbarger et al., 2017). In fact, achieving desirable scores across all indicators in the same systems may be impossible (Nemecek et al., 2011b). Strategies aiming for a more sustainable CL system should strive to maximise synergy among the most relevant indicators without losing track of trade-offs among those deemed of a lesser priority (Devkota et al., 2019). Responsibility for developing such strategies must go beyond the farm gate and include key stakeholders invested in improving the sustainability of the agricultural supply chains (Béné et al., 2019).

4.4.2 Level of investigation and multifunctionality of the crop-livestock systems

Evaluating the CL systems using production-related and area-related functional units allows a broader understanding of the trade-offs among systems. Nevertheless, when integrating cattle as part of the system rotation, the use of dry matter as the production-related FU seemed not appropriated, and thus, the production of protein has been considered in this study. This approach allowed us to associate the results to a generic unit of reference that was common among all elements of the CL systems evaluated. Applying this method led to slightly different results. For example, the BSR system presented with its best performance in the protein-related FU, which is an effect of the high protein content of the soybean crop produced in this system. Further, the results confirmed that area-related environmental performance tends to benefit less intensified systems, such as the BL in this study; conversely, in economic terms the BL system presented the worse performance. Still, defining the most appropriate FU for multifunctional CL systems remains a challenging task (Van Der Werf et al., 2014).

Furthermore, the crop-by-crop approach permits the identification of hotspots within each cropping season. In the CL systems in this study, with the exception of liming, all other farm operations were crop-specific and therefore burdens were specifically allocated to the season in which the farm operation occurred. This may be a rather simplistic approach, as the effects of one crop season over the next cannot be tracked. However, a high degree of agronomical knowledge and information of the CL system dynamics would be necessary to successfully apply more complex methods (Goglio et al., 2018).

4.4.3 Comparison to other studies

Due to methodological choices and limitations among LCA studies, direct comparison of results across LCA studies would be far from best practice. In fact, comparisons can be even misleading (de Vries et al., 2015; Vázquez-Rowe et al., 2017). Another constraint when comparing the results of our study with others conducted in RS, is the presence of a degree of correlation among them, as some of the parameters applied in our model were drawn from regional studies. This is particularly true with regard to the modelling of the BL system and the production of young stock. Thus, we conduct our comparison based on the results of crop-by-crop analysis, from the cattle systems to the crop systems, emphasising some of the main issues encountered during the process.

4.4.3.1 Beef cattle comparison

Difficulties to compare studies may appear when the system boundaries of the studies are not identical. For example, most cattle LCA studies conducted in Brazil reported their findings in terms of ready for slaughter weight (~ 420 to 440 kgLW per animal) (e.g., (Cardoso et al., 2016; Dick et al., 2015b; Ruviaro et al., 2016, 2015)). Conversely, the CL systems in this study delivered animals with mean liveweights of 249, 294, and 214 kg for the BR_cattle, BSR_cattle, and BL_cattle systems, respectively (i.e., backgrounding animals). Disregarding the boundaries of the systems, the production of one tonne of LW in this study was still within the ranges reported for the region. In terms of GWP, Pashaei Kamali et al. (2016) reported 26,800 kg CO_2 eq. tLW^{-1} for a crop-livestock system in Southern Brazil with cattle grazing soybean residues; this value is similar to the results of the BL_cattle system in this study. Additionally, the authors reported a value of 18,700 kg CO_2 eq. tLW^{-1} for improved pastures, which was lower than the experimental systems in this study. This can be associated with the shorter time animals remained in the high-quality pasture in our study. Pasture quality is one of the main drivers of GHG emissions in pasture-based beef production (Bilotto et al., 2019; Cardoso et al., 2016; Dick et al., 2015b; Florindo et al., 2018;

Ruviaro et al., 2015). Moreover, our results were also between the ranges found for cow-calf and finishing stages of animals produced in natural and improved pastures in other regions of South America. On the Uruguayan beef production, Becona et al. (2014) reported values of 34, 600 kg CO_2 eq. t weaned calf^{-1} and 20,800 kg CO_2 eq. tLW^{-1}, while Picasso et al. (2014) found mean values around 21,900 kg CO_2 eq. tLW^{-1} for cow-calf systems and 11,300 kg CO_2 eq. tLW^{-1} for finishing systems. In Argentina, Nieto et al. (2018) reported values of 23,600 kg CO_2 eq. tLW^{-1} for cow-calf systems and 19,600 kg CO_2 eq. tLW^{-1} at the farm gate.

Results presented with mismatched functional units may also generate uncertainties when comparing LCA studies. This issue occurs quite often in LCAs from livestock production; for example, when system boundaries are set to the farm gate, but the results are reported using carcass weight or boneless meat as the functional unit (Wiedemann and Yan, 2014). We encountered this issue when comparing AP and EP results with the finding from Lupo et al. (2013) for the Northern Great Plains of the USA. Rescaling of the results was needed to reach an approximation of the results reported by the authors (~181 kg SO_2 eq. tLW^{-1} and ~1.37 kg P eq. tLW^{-1} at the farm gate) which are 3 to 4 times lower than the results of our study. Similarly, Nguyen et al. (2010) evaluated EU beef production and reported their values in carcass weight; the authors found AP of 210 kg SO_2 eq. t carcass weight^{-1}, for animals coming originally from suckler herds. AP and EP impact categories are underexplored for pasture-based cattle production, thus limiting the capacity to draw comparisons to regional studies.

The fossil depletion impact category that was not evaluated by most livestock studies, or was assessed as part of a different impact category. For example, Pashaei Kamali et al. (2016) assessed the impact category *fossil energy use* and reported that improved pasture systems have a significantly higher impact compared to low input natural pasture systems (i.e., ~27 folds). This tendency is consistent with results pertaining to cattle production, as shown in this study, where the experimental systems presented a 23 folds higher FD than the BL_cattle system. The value of 4 kg oil eq. tLW^{-1} reported by Dick et al. (2015b) for animals reared in natural pasture was lower than the BL_cattle in this study; conversely, the authors reported higher emissions for improved systems, namely 455 and 577 kg oil eq. tLW^{-1}. The main driver of FD in our study was the production of farm inputs, especially fertilisers and fuel.

ALO for the BL_cattle was 26% and 17% lower than the findings from Dick et al. (2015b) and Pashaei Kamali et al. (2016), respectively, for natural pastures in the RS. BL_cattle result was sounder with the crop residue scenario assessed by Pashaei Kamali et al. (2016) (i.e., 184,300 ($m^2 \cdot a$ tLW^{-1}). Conversely, the experimental BR_cattle and BSR_cattle systems presented an ALO higher than the improved pastures systems from both, Dick et al. (2015b) (25,100 $m^2 \cdot a$ tLW^{-1}) and Pashaei Kamali et al. (2016) (37,000 $m^2 \cdot a$ tLW^{-1}). Besides, the ALO finds of our study are generally higher than values reported for most production system in Europe and North America (de Vries et al., 2015). The higher ALO for beef production in Brazil may be associated with the rather extensive management practices adopted in the country, where animals graze on pastures all through the year.

Water-related impacts from beef cattle production remain underexplored in the context of Brazil. Dick et al. (2015b) assess the daily water consumption per head of cattle in their inventory (50 L). However, no further analysis was carried out to associate this value with their final FU. On the other hand, water-related impacts are well studied in the Australian agricultural sector. For example, the WD results for cattle in our study were similar to those values for *consumptive water* found by Ridoutt et al. (2011) for grass-fattened and feedlot-finished animals in the Australian beef cattle production (139 to 160 m^3 tLW^{-1}). This study also presented similar results to the values reported by Wiedemann et al. (2016) for domestic market beef production in New South Wales (117.9 and 196 (m^3 tLW^{-1})). Despite the similarities in these results, methodological choices among the studies may have significatively impacted the results. The inventory for the farming stages in this study was limited to the water withdrawn for cattle consumption and rice irrigation, which limited the application of more sophisticated water-related impact categories and consequently, comparisons to other studies.

4.4.3.2 Crop comparisons

Methodological choices in our study have significantly influenced the results of the GWP for rice production. GWP for rice in our study was slightly higher than those results reported in other studies in the region (BR_rice 1,350 kg CO_2 eq. t^{-1}; BSR_rice 1,248 kg CO_2 eq. t^{-1}; and BL_rice 1,821 kg CO_2 eq. t^{-1}). For example, Coltro et al. (2017) reported 690 kg CO_2 eq. t^{-1}, while Nunes et al. (2016) reported values of 979 and 1,015 kg CO_2 eq. t^{-1} for minimal tillage. But, Coltro et al. (2017) and Nunes (2016) applied methane daily EF of 1.51 and 1.74 kg day^{-1}, respectively; these values were derived from the IPCC (2006) guidelines. In our study, however, EFs were derived from field measurements in the regions and were higher than those derived in other studies. In the screening phase of this study, the equation S23 was applied to derive EFs according to the

IPCC (2019f) guidelines, which generated EFs of 1.13 for the no-tillage system and 1.79 kg day^{-1} for minimal tillage systems. If applied in this study, the IPCC derived EFs would have produced significantly lower GWP results. Furthermore, Abdul Rahman et al. (2019) reported GWP values (1,389 kg CO_2 eq. t^{-1}), when applying IPCC derived EFs for conventional rice farming in Malaysia, which was similar to those described in this study. Yet, despite taking special care in defining their functional unit, Abdul Rahman et al. (2019) did not provide clear information about the moisture content of the grains in their FU, which turns comparisons with other studies difficult. The same issue was found when trying to compare our results with those from Coltro et al. (2017).

Due to choices of impact categories, the WD for rice in our study could not be directly compared with the *Water* impact category assessed by Nunes et al. (2017) (approximately 14 m^3 per t of milled parboiled rice). However, the findings of our study are similar to those detailed in the *Blue water footprint* impact category for Brazilian rice (670 m^3 t^{-1}), reported by Chapagain and Hoekstra (2011).

Some of the above-mentioned challenges were also found when comparing the production of soybean in the BSR_soyben system to other published studies (e.g., contrasting LCIA methods, and grain moisture content not declared). Generally, the BSR_soyben system presented similar environmental performance than the average values reported for the region. In terms of GWP the value was slightly lower than the values of 352, 338, and 287 kg CO_2 eq. t^{-1} reported by Maciel et al. (2016), Prudêncio da Silva et al. (2010), and Zortea et al. (2018), respectively. The difference in yields may have driven the majority of the GWP differences among the studies. AP for soybean in this study was higher than the value found by Zortea et al. (2018) 0.533 kg SO_2 eq. t^{-1}, but, lower than the 2.5 kg SO_2 eq. t^{-1} found by Prudêncio da Silva et al. (2010). The ALO in our study was considerably lower than the value of 2,017 $m^2 \cdot a$ t^{-1} found by Prudêncio da Silva et al. (2010), which is a consequence of the high yields in the BSR experimental system.

4.4.4 Strengths and limitations

LCAs evaluating CL systems using long-term experiments are still scarce in Brazil and thus, assessing the four-years experimental CL systems was one of the advantages of this study. On the other hand, some variables that could have extended our results were not available, e.g., soil carbon changes due to no-tillage management, and biological nitrogen fixation. Therefore, by accounting only for the carbon losses due to erosion using models, we may have overestimated the

GWP of the experimental systems. In addition, due to the lack of measurements of N fluxes, the influence of the soybean crop over the CL system could not be evaluated. This remains an area warranting further exploration.

In this study, in addition to multiple functional units, a two-step approach was applied to generate the results and support their interpretation, i.e., uncertainty propagation during the LCI preparation and Monte Carlo analysis during the LCIA assessment. Nevertheless, the interpretation phase was limited to point values and discernibility analysis, and therefore no hypothesis testing was conducted to confirm if differences among the three systems were statistically significant.

Furthermore, to compare the environmental performance of the CL systems we applied two recognised LCIA impact methods (i.e., IPCC and ReCiPe) that provided insights into six important impact categories studied in the agricultural sector. However, we recognize that this selection could have been broader, for example, relevant impact categories not accounted for in this study are those linked to toxicity and biodiversity, see (Matsuura et al., 2017) and (Nemecek et al., 2011a). Moreover, when selecting the ReCiPe LCIA method, we chose to evaluate the water depletion impact category, and although we believe it did not affect the comparison among the systems that we evaluated, there are other water-related impact categories that could have provided more thorough results regarding water usage and consumption, e.g., (Boulay et al., 2018). Similarly, we selected only return to land and management as the measures of the profitability of the CL systems; this suited the goals of our study. However, there are many other profitability indicators at the farm level that could have been applied.

4.5 Conclusion

The present study compared and discussed the life cycle assessment and return to land and management of two experimental and a baseline paddy field-based crop-livestock systems located in the Rio Grande do Sul, Brazil. The experimental CL systems presented higher productivity and profitability per ha than the baseline system. In terms of environmental performance, however, the two experimental systems presented distinct results when compared to the baseline system. In the protein-related analysis, the environmental performance of the intensified BSR CL system, which rotations consisted of beef cattle, soybean, and rice, was considerably better in all but one impact category (i.e., EP). The BR CL system consisted of beef cattle and rice; and regardless of using protein-related or area-related functional units, it did not perform considerably better than the baseline or the BSR system in most impact categories. This indicate that increasing inputs to improved productivity, without the adoption of soybean, did not improve the environmental performance of the system. Moreover, when assessing the area-related functional unit, the environmental performance of the baseline system was considerably better than both experimental systems. In the BL system, rice was produced applying business as usual level of inputs, and cattle were fed only on rice residues left in the field; these features were the main factors responsible for the low impacts per ha in the BL system. Further, the economic performance of the baseline system was positive but lower than that of the experimental systems.

Further strategies contributing towards sustainable paddy fields-based crop-livestock systems need to go beyond the paddy fields. One of the most important points of note is the improvement of the performance of suckler herds. It must be noted that in most cases, this process occurs on a different farm, thus hindering any action at the backgrounding and finishing stages. Additionally, finding a reasonable balance between inputs use, productivity and environmental impacts should be further explored for the rice crop. The integration of soybean crops in the rotation appears to be the most advantageous strategy to reduce impacts in rice-based crop-livestock systems in South Brazil. The expansion of soybean over rice fields should, however, be well coordinated to avoid the collapse of rice supplies.

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Chapter 5

5 General Conclusions

The Brazilian agricultural sector plays an essential role in providing food security, employment and safeguarding livelihoods in the country. In addition, Brazil is a leading exporter of several agricultural products, supplying the increasing demand overseas. Consequently, the agricultural sector constitutes an essential part of the national economy, comprising of more than one-quarter of the national GDP. The country has favourable climate and richness of natural resources necessary for farming and husbandry, such as land and water; hence Brazil is considered a key player to suppling agri-food products to the growing global population in coming decades. Nevertheless, to advance and assume this position and to reach the status of a developed country, Brazil needs to overcome some critical issues prescribed by the international agenda for sustainable development as well as comply with important international environmental commitments. Therefore, fostering economic growth and social progress in rural areas while improving the environmental sustainability of agricultural systems are essential goals the country must strive for in the coming years. Sustainable intensification and integration of agricultural systems emerged in the national policy agenda as an option in this direction. However, the effective implementation of more sustainable agricultural systems in the country is constraint by several issues. The heterogeneity among Brazilian regions, agricultural systems and farm endowments hinders the development of targeted policies and actions. Moreover, the lack of financial capacity and management skills from farmers, as well as the lack of profitability of farming activities are also limiting issues described in the literature.

In this dissertation we investigated some of the abovementioned issues for two of the most important agricultural systems in Brazil. The first two research chapters (2-3) were dedicated to dairy farming, and the last one (chapter 4) was dedicated to paddy field-based crop-livestock systems. In the remainder of this chapter, we summarize the main findings of the research and discuss the implications of the results for policymakers and other stakeholders; finally, we close the chapter with a discussion about the limitations and future research possibilities.

5.1 Main Findings

One of the first steps to designing and implementing sustainable production systems in a region or country is to understand the structure of the farms operating in the area and the status of the indicators of interest. Chapter 2 was dedicated to understanding the structure and carbon footprint of dairy farms and discussing strategies to mitigate GHG emissions of dairy farming in Paraná and Brazil. The results of the chapter show that farms selling milk in Paraná are well represented by four groups. The carbon footprint (GWP_{100}) per kg of FPCM of the groups was as follows G1:1.75 kg CO_2eq.; G2: 2.20 kg CO_2eq.; G3: 3.02 kg CO_2eq.; G4: 3.27 kg CO_2eq.. The range of values is similar to those found in dairy systems around the globe, confirming the high heterogeneity of dairy farms in Paraná also in terms of CF. Farms operating with high CF represent the majority of the dairy farms in Paraná. On average, they are likely to have less specialized herds with lower productivity, receive less technical support and have less farm machinery and infrastructure compared to farms that displayed lower CF. Moreover, we confirm the strong negative correlation between carbon footprint per kg FPCM and animal productivity. Four main areas of action to mitigate GHG were identified, namely, feeding, herd, waste and energy management. Given the low productivity and high CF of some of the groups analysed and the general status of the dairy farming in the country, improving feeding and animal productivity seems the most important pillars to fortify in the country. Waste management and energy savings practices are also relevant – especially for farms with large herds and automated systems. Small improvement in farms hosting large herds may lead to high absolute GHG reduction due to gains of scale. The sensitivity analysis between the two methods of characterizing carbon footprint did not influence the rank of the groups. The GWP*, however, indicates that the Global Warming Potential calculated with this metric is significantly lower than the GWP_{100}. Moreover, methane still remains an important source of potential warming, but not as strong as in the GWP_{100} method.

In chapter 2, we provided evidence from Brazil to the growing body of literature associating low animal productivity with high GHG intensity in dairy farming. Nonetheless, our discussion in chapter 2 was based only on partial productivity and thus somehow limited, since we did not consider all farm endowments simultaneously in the analysis. In chapter 3, we analysed the environmental efficiency of dairy farms to overcome this restriction. We evaluated the joint performance of dairy farms with the stochastic translog hyperbolic distance function, accounting for methane emissions as undesirable output. The results show that farmers could improve their environmental performance by increasing milk and animal liveweight outputs while reducing methane emissions without increasing farm inputs. On average, dairy farmers in our sample can increase farm production by 9.4% while reducing methane emissions by 8.7%. When exploring variables that influence the environmental efficiency, we found that increasing the milk yield of cows improves environmental efficiency of the farms – this is in line with preceding evidence from the traditional production efficiency literature and partially with our findings from chapter 2. Similarly, the share of improved pastures also displayed a positive effect on the environmental efficiency of the dairy farms, strengthening the importance of feed management to the environmental performance in dairy farming. In addition, we identified that adjusting the share of lactating cows in the herd contributes positively to environmental efficiency. Conversely, hiring labour was associated with low environmental performance. The introduction of the Annual Climate Type classification in the estimation of the production frontier provides an indication that dairy farmers operating in tropical and semi-arid climates are somehow at a disadvantage when compared to farmers from areas with a humid subtropical climate. Furthermore, the possibility of promoting the reduction of GHG emissions through taxes has become current in policy settings. Nonetheless, in chapter 3, we show that such approach might be very costly for dairy farmers in Brazil. On average, the abatement cost of methane emitted by dairy farms was US $2,254 per tonne, representing about one-quarter of the sampled farms' revenue.

In chapter 4, we evaluated and compared two experimental crop-livestock (CL) systems implemented in paddy fields in RS, southern Brazil, and compared them with a modelled Baseline system (BL). The experimental CL systems were identified as (BR) for Beef cattle-Rice CL integration and (BSR) for the Beef cattle-Soybean-Rice CL integration. The crop-by-crop analysis indicate that the environmental performance of the crops varies considerably according to the system and impact category under analysis, with crops from BSR presenting a slight tendency of better performance. In the evaluation of the whole systems, the experimental CL systems present higher productivity and profitability per ha than the baseline system. In terms of environmental performance, however, the experimental systems present contrasting results. In the protein-related functional unit analysis, the environmental performance of the intensified BSR CL system, was considerably better in all but one impact category (i.e., Eutrophication Potential). Clearly trade-offs between economic and environmental performance emerged in the experimental systems. For instance, farmers would have a better economic return if adopting the BR system, however, regardless of using protein-related or area-related functional units, it did not perform considerably better than the BL or the BSR systems in most impact categories. Moreover, when assessing the area-related functional unit, the environmental performance of the baseline system was considerably better than both experimental systems, remarkedly against BR. In the BL system, rice was produced by applying business as usual level of inputs, and cattle were fed only on rice residues left in the field; these features were the main factors responsible for the low impacts per ha in the BL system.

5.2 Implications of the results

5.2.1 Dairy farming

Based on the results presented in chapter 2 and 3, we deduce that developing long-term sustainable intensification for dairy farming in Brazil seems a reasonable path to improve the overall sustainability of the dairy sector in the country. However, caution should be taken since a share of dairy producers in the country already operate intensive systems; these can also be adjusted to become more sustainable, yet through other strategies. Based on the results and discussion of the two chapters, we present some points to notice for designing and implementing policy and actions to improve the sustainability of dairy farming in Brazil.

- *Recognize the heterogeneity among dairy farms*: we show that there is high heterogeneity among dairy farms in Paraná and that this is very likely to reproduce across Brazil (Chapter 2). Acknowledging this heterogeneity during the design and implementation of national policies and strategies for the dairy sector might be relevant to increasing the success of developing more sustainable dairy farming in the country. The differences in farm structure, socioeconomic indicators, and the carbon footprint of milk suggest that farmers may require approaches to improve farm sustainability and thrive. In this regard, policymakers and extension services need to recognize that a share of farms (specialized and high productive) tend to display low CF intensity but have high overall emissions due to the size of their operations. Conversely, farms that are not specialized or farms with dual-purpose herds display higher GHG intensity. In these farms lies a great potential to apply sustainable intensification actions. Additionally, regional climatic characteristics also influence the environmental sustainability of dairy farms (Chapter 3) and increase heterogeneity among regions. Taking climatic zones into account in long-term planning of dairy systems in the country may be desirable, especially in the development of adaptative strategies against heat and drought in tropical and semi-arid regions and extreme rain falls in subtropical regions.

- *Improving feeding management*: Providing dairy cows with a balanced diet is crucial to improve animal productivity and reduce GHG intensity (Chapter 2). This was supported in (Chapter 3) in the case of methane emissions, provided that farms with a higher share of improved pastures presented higher environmental efficiency. Improving pastures is a central strategy toward sustainable intensification of livestock production, and thus prioritised in the ABC+ Plan and Full Bucket programmes. Policymakers and extension services should facilitate the adoption of improved pastures by farmers. And more important, support farmers in maintaining good quality pastures over the years to avoid grassland degradation. In addition to being essential to increase animal productivity, improved grassland also has a great capacity to store carbon, reduce soil erosion, and increase farm biodiversity. Furthermore, the adoption of concentrate and mineral supplements in the dairy cow's ration is also important to reach a balanced diet and increase milk production. Increasing the use of such feedstuffs will significantly increase the productivity and reduce GHG emissions for less specialized farms (Chapter 2). On the other hand, the intensive use of concentrate feed might be undesirable. Despite being crucial for maximizing milk yield in dairy farms, it can lead to feed-food competition, higher production costs and digestive disorders in dairy cows. Therefore, the rational use of concentrate feedstuffs should be promoted, aiming to find an optimum between the two extremes.

- *Herd management*: another crucial factor associated with the success of a dairy enterprise is herd productivity. Farms with a higher number of specialized dairy cows have higher productivity and lower CF intensity (Chapter 2). This is also true for the environmental efficiency of dairy farms in terms of methane (Chapter 3).

Therefore, besides improving feed quality, increasing the genetic quality of the national dairy herd may be one of the most important goals to improve the sustainability of dairy farming in the country. However, policymakers and other stakeholders should consider some fundamental points. First, promote the development of adapted breeds according to the regional characteristics. For example, European dairy breeds have great productive potential, however, they are not well adapted to the warm climate types of most of the country. Farmers producing in these regions using grass-based systems should rear more adapted breeds, e.g, Girolando. Second, adopting high productive breeds does not solve the problem per se, feeding and farm management must be adjusted accordingly. In Chapter 3, we show that maintaining the appropriate number of cows in the herd improves the environmental efficiency of dairy farms; this is directly related to farm management actions. Three, contrary to feeding which can be improved in one season, improving the genetic quality of the national herd will need more time, requiring long-term planning and regionally targeted policies and actions.

- *Manure management*: proper management of the manure accumulated in dairy farms reduces the potential of causing environmental impacts and offers an opportunity to recover energy and nutrients, consequently, reducing the reliance of the farms on external inputs. In Brazil, a significant number of farmers miss this opportunity and do not have a manure storage system or use manure as fertiliser (Chapter 2). Developing and adapting manure storage systems according to the dairy systems in the country is important strategy in the path to sustainable intensification. However, it is noteworthy that environmental and operational trade-offs emerge from the different manure storage systems (chapter 2). For example, handling manure in solid form avoids emissions of methane, but may increase emissions of N compounds. Besides, solid manure is somehow more difficult to spread. Conversely, manure handled in liquid form is much easier to spread with tractors, but it has a high methane emission capacity during storage if not treated. As farmers intensify and increase the size of their operations there is a tendency to adopt liquid storage systems. In these cases, adopting anaerobic digestors for recovering energy from methane can significantly reduce the emissions of this gas, generating a win-win situation. Despite being promoted by the ABC Plan in the last decade, the adoption of such systems by dairy farmers appears still not feasible. Therefore, increasing incentives for farmers to adopt proper manure storage systems and making small-scale digestors feasible might be necessary in the country.

- *Build capability*: Eliminating inefficiency is one of the most cost-effective ways to improve farm environmental sustainability (Chapter 3). It allows farmers to reduce GHG emissions and concurrently produce more milk and livestock without adding farm inputs. Moreover, improving feeding and herd management are crucial to increase farm productivity and environmental sustainability (Chapter 2 and 3). Technology and technical knowledge in Brazil already allows for the development of highly productive and efficient dairy enterprises. However, the low productivity of dairy farms in the country and the number of farmers leaving the activity in recent years indicates some critical issues in the dairy sector. Improving farmers' capability to sustainably intensify their systems is crucial to increasing the success of national policies for the sector. As identified in Chapter 2, the provision of adequate extension services and training to farmers and the young generation of farmers is essential. Dairy farms are complex enterprises that require a systems-thinking management approach. The Full Bucket programme applies such an approach; thus increasing the enrolment of farmers in this and other similar programmes is crucial to the development of the national dairy sector. Moreover, creating mechanisms to accelerate this process is urgently required given the current urgency in reducing GHG emissions to curb global warming.

- *Finding a leverage point*: The significant reduction in GHG intensity reached by increasing animal productivity is appealing for suggesting the adoption of high-productive breeds and striving for maximum productivity of cows (Chapter 2 and 3). However, other important factors should be considered during sustainable intensification of dairy farms. First, other relevant environmental impacts associated with dairy farming will likely emerge with intensification, e.g., those linked to N and P surplus. Second, Holstein cows are the most productive and in theory the ones that would produce milk with the lowest CF intensity. However, as already mentioned, this and other European breeds are not suitable for some climatic zones in Brazil. Moreover, the use of highly productive breeds needs to be associated with high-quality feed and management, which is still lacking among many farmers in the country. Stakeholders in the supply chain therefore need to understand that regional characteristics and different production systems will produce milk with contrasting CF. Policymakers should adopt a holistic view during policy design, and instead of relying only upon the minimum CF intensity target, consider other indicators, for example, animal welfare, other environmental impacts, and ecosystem services produced at the farms. Nonetheless, the increasing focus on climate-friendly labels may put constraints in this regard.

5.2.2 Paddy field-based agricultural systems

The results of chapter 4 show that rice is the most productive but also the most demanding crop for paddy fields in southern Brazil. Despite the limitations of being in an experimental set, in general, we deduce that improved crop-livestock integration are good options to increase land productivity and reduce the environmental impact intensity of paddy field-based agricultural systems production. Moreover, diversification of production by rotating rice and soybean as summer crops is a reasonable path to reduce many environmental impacts of the system, markedly GWP. Based on Chapter 4, stakeholders interested in paddy field agricultural systems in Rio Grande do Sul could consider the following:

- *Finding a leverage point*: The results of Chapter 4 indicate that if farmers make decisions based on profit maximization and given that they could implement the experimental systems analysed, they would select the improved Beef cattle-Rice (BR) system. This system, however, did not perform best in many of environmental impacts assessed, showing trade-offs between economic and environmental indicators. Policymakers and stakeholders should be aware that rotation of rice with soybeans and livestock is better regarding environmental sustainability, but not as good in economic returns. The promotion of more sustainable agricultural systems in this region, therefore, is likely to require incentives for farmers.

- *System-based and crop-by-crop environmental assessment*: The main advantage of applying attributional LCA is the possibility of comparing single crops as well as whole productive systems. Policymakers and stakeholders should be aware that the results of these approaches may lead to contrasting results, influenced by functional units and allocation rules applied in the study, as unveiled in chapter 4. The advantage of system-based comparisons lies in the possibility of conducting a holistic comparison between agricultural systems. However, system-based analysis does not allow the identification of environmental impacts of the single elements that constitutes the agricultural system – this is reached by the crop-by-crop analysis. Crop-by-crop analyses are important for the identification of hotspots of emissions along the production chain and later for communicating the environmental impacts of the individual agri-food products to consumers. The crop-by-crop approach, however, requires the allocation of environmental burdens among outputs of the system, which sometimes is very complex. Despite of general guidance in the literature, there is no single rule for allocating environmental impacts of agri-food products. Common allocation rules in the literature are based on economic value, mass, protein content and energy content of the outputs. An emerging approach is based on the nutritional value of food products, however, the application of such approach to cradle-to-farm gate evaluations is still limited. Thus, policymakers and stakeholder should consider carefully these features when developing policies or interpreting environmental impact of products.

- *Improve up stream environmental impact of suckler herds*: The production of youngstock occurs in higher areas of the Pampa biome in extensive pastures. A significant share of the suckler herds in these regions have low productivity, consequently, producing youngstock with high embedded CF. Therefore, improving the upstream performance of suckler herds will positively influence the overall environmental performance of backgrounding and finishing stages of beef leaving paddy field systems, remarkably in the case of CF. Naturally, this also applies to any environmental impact and farm input.

- *Landscape-based governance*: promotion of any of the improved systems analysed will increase farm inputs and eliminate fallow. The impacts of scaling such systems must be further investigated in RS. Landscape or watershed impact assessment needs to be conducted; optimally by applying regionalized impact characterization factors. Moreover, landscape-based governance could be promoted in the region to optimize the rotation of different crops and livestock in order to minimize impacts at the landscape level, e.g., water usage.

- *The risk of permanent land use change*: the integration of soybean crops into the rotation with rice and beef is an advantageous strategy to reduce impacts in rice-based crop-livestock systems in RS. The expansion of soybean over rice fields should, however, be well coordinated to avoid the collapse of rice supplies in the country. The rapid advance of research in the adaptation of soybean varieties to lowland areas and the high market price of this crop in recent years suggests that this could be true in the future.

5.3 Limitations and future research

In chapter 2, we apply Life Cycle Assessment (LCA) and Cluster Analysis to evaluate the structure and carbon footprint of dairy farms. The limitation to conducting fieldwork and the lack of recent databased from dairy farming in Brazil led us to analyse a database from 2007 in this chapter. The dairy sector passed for several changes in the past years, thus conducting a new survey with the farmers of the sample analysed would provide a rich source for further socioeconomic and environmental research. The questionnaire applied, however, needs to be improved to accommodate questions that could improve the environmental assessment results. Especially the lack of information on the quantity of concentrate feed consumed by the animals limited the calculation of more precise CF for each farm. The lack of information also limited us to evaluate only a single impact category, the inclusion of other impacts should be considered in future work, e.g., eutrophication, and acidification potential. A better representation of the production systems adopted by the farmers is also desirable. Increasing the implementation of experiments and direct measurement of GHG for the diverse dairy systems would also help to produce better estimates of the impacts of milk production in the country. Furthermore, the contrasting results between global warming methods (GWP_{100} and GWP*) suggest the need for more research in this area. Considering the historical dairy herd size to derive more accurate contribution of dairy farming to the global warming is necessary. The consequences of using the GWP* method at the country and farm level, as well as in product labelling warrant more research since they may produce contrasting results. Furthermore, a point not explored extensively in this research is the fact that many farmers own dual-purpose herds. Owning dual-purpose herds is not an ideal in terms of GHG emissions and milk production, however, more studies need to be conducted to evaluate economic returns and other environmental impacts from this class of farmers.

In chapter 3, we employed the stochastic hyperbolic distance function to estimate the environmental efficiency of dairy farmers from Minas Gerais. Estimation results allowed us to calculate the shadow price for methane emissions, based on the duality between the hyperbolic distance function and the profit function. A limitation in our approach is the fact that the undesirable output, methane, was calculated and not observed. Nonetheless, we follow this approach because the measurement of GHG emissions at the production level is still not feasible. Our HDF was developed based on the assumptions of weak disposability of undesirable outputs and null-jointness between desirables and undesirables, thus not following the mass balance principle, characterized by methane production. This feature may also have influenced our results, e.g., efficiency scores and shadow prices. Although the theoretical framework suggests the possibility of estimating the hyperbolic distance function with multiple undesirable outputs, in our empirical application this was not practicable. Consequently, we selected the most concerning GHG emitted in the Brazilian dairy farms to evaluate as externality. Thus, further research to evaluate other GHG and environmental impacts in a joint production framework is required. An important theoretical requirement for estimation shadow prices in monotonicity. This requirement was attended for significant parameter estimates at the sample mean, however not completely for the single observations. Imposing monotonicity in the model is suggested as an approach to overcome this limitation, but this is not straightforward to apply and still remains a gap to be fulfilled in the case of the hyperbolic distance function. Moreover, Bayesian estimation methods also provide a robust approach to overcome this limitation. In terms of sample in chapter 3, we analysed data from the year 2017 from farmers enrolled in the Full Bucket programme. This indicates that sample selection issues are very likely to have influenced our results, limiting the interpretation of our results to the whole Brazil. The lack of representative sampling is a major issue in empirical agricultural studies in Brazil. Surveying farmers not taking part in the Full Bucket programme is crucial to evaluate the long-term socioeconomic and environmental impacts of this programme. Moreover, the use of panel data and larger samples could have improved the robustness of our conclusions and should be considered in future studies.

Overall, dairy farmers in Brazil operate in a market economy thus the low productivity of most farms across the country, in theory, could partially be associated with the lack of economic incentives. Therefore, more research on the influence of the market environment and milk prices in farmer's intention to intensify and adopt technologies might unveil important paths to improve the environmental sustainability of this supply chain in country.

In chapter 4, Life Cycle Assessment and farm profitability analysis was applied to compare two experimental crop-livestock systems with a baseline system. The results of our study show important trade-offs between economic and environmental impacts and also among environmental impacts for the different systems studied. Future work could advance in supporting the selection of production systems to be promoted, for example, by applying participatory multicriteria decision analysis. Regarding the methodology, we observe that LCA methodology is currently considered one of the most robust environmental assessment tools, e.g., this is the approach selected to conduct Environmental Product Declarations and environmental labelling. Nevertheless, LCA still presents some limitations for evaluating agricultural systems. As mentioned in Chapters 2 and 3, the lack of feasible tools to conduct field measurements of GHG emissions led us to rely on calculations. This also applies to other impact categories assessed in Chapter 4. Despite being constantly updated to improve accuracy, these models are still associated with some level of uncertainty. Proper representation and allocation of lagged benefits and impacts between crops/livestock in the agricultural systems is also still limited and remains a significant point for improving the method. Moreover, there are several procedures to characterise emissions into impact categories; these are sometimes designed to represent continent or country level and therefore offer only an approximation of the actual Potential Impact at the plot level. Our sample of experimental systems did not include a controlling system. This led us to conceptualize a baseline system based on the literature and expert opinion, which is far from ideal. Proper randomization and creation of business as usual treatments appear to be lacking in the design of large-scale long-term experiments in some areas in Brazil. Despite the difficulties in designing such experiments, researchers should be more rigorous in the establishment phase since this can be helpful in the future to improve the accuracy and validation of the results. As already mentioned, the experimental systems evaluated increase land productivity and reduces environmental impacts intensity, however, the effects of scaling such systems across RS must be further evaluated since they increase overall farm inputs and environmental impacts, e.g., at landscape level or watershed level. Besides, many situations that may occur in practice were not well represented in the experimental systems, e.g., flood event, fields with poor drainage, requirement of eventual tillage and logistics to bring youngstock into the systems.

Lastly, this dissertation and several field experiments across Brazil have provided evidence that intensification and integration of agricultural systems have a great potential to produce synergic outcomes concerning farm profitability, environmental and animal welfare issues. And such approaches have been extensively promoted by the ABC+ Plan and several other programmes across the country. The implementation and maintenance of integrated systems at farms, however, require a high level of agronomic and animal husbandry knowledge, as well as farm management. Evidence shows that at the actual stage of development in rural areas of Brazil, a significant share of farmers is having trouble to manage farms producing one crop or animal species. Therefore, the effective implementation of intensification and integrated agricultural systems must be assisted by qualified long-term technical support to farmers. Otherwise the same mistakes committed in the past will be repeated, increasing even further the socioeconomic inequality in rural areas of the country. Monitoring the economic and social impacts of environmental policies in the country may emerge as a relevant area of research to guide the sustainable development of rural areas.

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7 Appendix

7.1 Appendix Chapter 2 – (Supporting Material)

Supporting material to: Carbon footprint and mitigation strategies among heterogeneous dairy farms in Paraná, Brazil

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Table SM1 Concentrate consumption according to animal category.

Category	Unit	Value
Bulls	kg day^{-1}	1.5
Lactating cows	kg (kg Milk produced) $^{-1}$	1/3
Dry cows	kg day^{-1}	1.5
Cull cows	kg day^{-1}	1.5
Calves	kg day^{-1}	0.8
Heifers	kg day^{-1}	1.5
Backgrounding animals	kg day^{-1}	1.5
Fattening animals	kg day^{-1}	3

Source: (Embrapa, 2005) and expert judgment.

Table SM2 Forage and concentrate characteristics.

Name	Fed as	DM* %	NDF	TDN	CP	P
Avena sativa L.	Fresh forage	16.8	54.0	64.0	13.0	0.4
Brachiaria.spp	Fresh forage	27.5	70.2	53.8	8.6	0.2
Cynodon spp.	Fresh forage	31.0	69.4	59.5	9.9	0.2
Cynodon spp.	Hay	88.7	77.3	54.8	9.2	0.2
Lolium multiflorum	Fresh forage	19.4	54.0	65.0	18.6	0.3
Lotus corniculatus	Fresh forage	23.1	38.3	68.8	21.1	0.3
Natural pasture	Fresh forage	45.6	72.8	53.4	7.0	0.1
Panicum spp.	Fresh forage	28.7	72.4	58.3	9.1	0.2
Pennisetum spp.	Fresh forage	19.8	62.6	65.6	12.2	0.3
Saccharum officinarum L.	Silage	25.7	66.4	54.2	3.5	0.1
Saccharum officinarum L.	Fresh forage	28.9	53.5	62.8	2.8	0.1
Trifolium repens L	Fresh forage	16.8	45.7	66.0	20.9	0.4
Zea mays	Silage	31.1	54.0	63.2	7.2	0.2
Zea mays	Fresh forage	33.1	55.3	63.6	7.1	0.1
Maize meal	Concentrate	88.7	19.7	83.2	9.2	0.2
Soybean meal	Concentrate	88.6	14.8	81.2	48.8	0.6
Soybean Hulls	Concentrate	90.1	66.5	68.9	12.6	0.2
Rice meal	Concentrate	89.0	23.2	80.3	13.4	1.7
Dicalcium phosphate	Concentrate	98.4	-	-	-	18.5
Salt	Concentrate	99.0	-	-	-	0.3
Urea	Concentrate	97.9	-	-	281.9	-

Source: https://cqbal.com.br/; https://www.feedipedia.org. *DM: Dry Matter %; NDF: Neutral Detergent Fibre %DM; TDN: Total Digestible Nutrients %DM; CP: Crude Protein %DM; P: Phosphorus %DM.

Table SM3 Emissions factors applied for the inventory calculations.

Variable	Observation	Value	Unit	Source
Enteric CH_4	Range	19-23	g $(kgDMI)^{-1}$	(IPCC, 2019a) Table 10.12, pg.10.48
Methane Conversion Factor (MCF) Solid storage		5%		(IPCC, 2019c) Table 10.17, pg.10.74
MCF Dry lot		2%		(IPCC, 2019c) Table 10.17, pg.10.74
MCF Liquid		73%		(IPCC, 2019c) Table 10.17, pg.10.74
MCF Pasture		0.47%		(IPCC, 2019c) Table 10.17, pg.10.74
Bo: Maximum Methane producing capacity		0.19	m^3 CH_4 (kg $VS)^{-1}$	(IPCC, 2019c) Table 10.16, pg.10.72
Frac Gas_MS Nitrogen loss fraction due to volatilisation of NH_3 and NO_X from manure management	Solid storage	0.30	(kg NH_3–N + NO_x–N) (kg $N)^{-1}$	(IPCC, 2019c) Table 10.22, pg. 10.106
Frac Gas_MS	Dry lot	0.30	(kg NH_3–N + NO_x–N) (kg $N)^{-1}$	(IPCC, 2019c) Table 10.22, pg. 10.106
Frac Gas_MS	Liquid	0.30	(kg NH_3–N + NO_x–N) (kg $N)^{-1}$	(IPCC, 2019c) Table 10.22, pg. 10.106
EF4: Emission factor for N volatilisation and re-deposition	Wet season	0.014	kg N_2O–N (kg NH_3–N + NO_X–N volatilised$)^{-1}$	(IPCC, 2019e) Table 11.3, pg. 11.25
EF4	Dry season	0.005	kg N_2O–N (kg NH_3–N + NO_X–N volatilised$)^{-1}$	(IPCC, 2019e) Table 11.3, pg. 11.25
EF3: Emission Factor for direct N_2O-N emissions from manure management	Solid storage	0.01	kg N_2O-N (kg N excreted$)^{-1}$	(IPCC, 2019c) Table 10.21, pg. 10.100

Table SM3 Emissions factors applied for the inventory calculations. Continued.

Variable	Observation	Value	Unit	Source
EF3	Dry lot	0.02	kg N_2O-N (kg Nitrogen excreted)$^{-1}$	(IPCC, 2019c) Table 10.21, pg. 10.100
EF3	Liquid	0.005	kg N_2O-N (kg N excreted)$^{-1}$	(IPCC, 2019c) Table 10.21, pg. 10.100
Frac Leach_MS: Nitrogen loss fraction due to leaching from manure management	Solid storage	0.02		(IPCC, 2019c) Table 10.22, pg. 10.106
Frac Leach_MS	Dry lot	0.035		(IPCC, 2019c) Table 10.22, pg. 10.106
EF5 leaching runoff		0.011	kg N_2O–N (kg N leaching/runoff)$^{-1}$	(IPCC, 2019e) Table 11.3, pg. 11.25
Frac N_2 MS: Fraction of N_2-N emissions from manure management		3*EF3	kg N_2-N (kg N_2O-N)$^{-1}$	(IPCC, 2019c) Table 10.23, pg. 10.109
Frac GasM: Volatilisation from all organic N fertilisers applied, and dung and urine deposited by grazing animals	Urine Wet season	0.1062	(kg NH_3–N + NO_x–N) (kg N applied or deposited)$^{-1}$	(Bretas et al., 2020; Cardoso et al., 2019; Lessa et al., 2014; Simon et al., 2018)
Frac GasM	Urine Dry season	0.1346	(kg NH_3–N + NO_x–N) (kg N applied or deposited)$^{-1}$	(Bretas et al., 2020; Cardoso et al., 2019; Lessa et al., 2014; Simon et al., 2018)
Frac GasM	Dung Wet season	0.0318	(kg NH_3–N + NO_x–N) (kg N applied or deposited)$^{-1}$	(Bretas et al., 2020; Cardoso et al., 2019; Lessa et al., 2014; Simon et al., 2018)
Frac GasM	Dung Dry season	0.0493	(kg NH_3–N + NO_x–N) (kg N applied or deposited)$^{-1}$	(Bretas et al., 2020; Cardoso et al., 2019; Lessa et al., 2014; Simon et al., 2018)

Table SM3 Emissions factors applied for the inventory calculations. Continued.

Variable	Observation	Value	Unit	Source
EF3PRP: Emission Factor for direct N_2O-N emissions from manure deposited on pastures	Urine Wet season	0.0085	kg N_2O-N (kg N excreted)$^{-1}$	(Bretas et al., 2020; Cardoso et al., 2019; Lessa et al., 2014; Simon et al., 2018)
EF3PRP	Urine Dry season	0.0012	kg N_2O-N (kg N excreted)$^{-1}$	(Bretas et al., 2020; Cardoso et al., 2019; Lessa et al., 2014; Simon et al., 2018)
EF3PRP	Dung Wet season	0.0014	kg N_2O-N (kg N excreted)$^{-1}$	(Bretas et al., 2020; Cardoso et al., 2019; Lessa et al., 2014; Simon et al., 2018)
EF3PRP	Dung Dry season	0.0013	kg N_2O-N (kg N excreted)$^{-1}$	(Bretas et al., 2020; Cardoso et al., 2019; Lessa et al., 2014; Simon et al., 2018)
EF1: Emission factor for N additions from organic amendments to agricultural soil	Wet season	0.006	kg N_2O–N (kg N)$^{-1}$	(IPCC, 2019e) Table 11.1, pg. 11.11
EF1	Dry season	0.005	kg N_2O–N (kg N)$^{-1}$	(IPCC, 2019e) Table 11.1, pg. 11.11
Frac Leach (H): N losses by leaching/runoff in wet climates	Wet season	0.24	kg N (kg N additions or deposition by grazing animals)$^{-1}$	(IPCC, 2019e) Table 11.3, pg. 11.25
Nitrous Oxide	Global Warming Potential 100 year	273		(Forster et al., 2021) Table 7.SM.7
Methane Fossil	Global Warming Potential 100 year	29.8		(Forster et al., 2021) Table 7.SM.7
Methane Biogenic	Global Warming Potential 100 year	27.2		(Forster et al., 2021) Table 7.SM.7

Table SM4 Sample and cluster variables from dairy farms operating in Paraná in 2007.

Variable	Unit	Sample N=911	G1 N=128	G2 N=317	G3 N=326	G4 N=140	p.overall
Daily milking	1	180 (19.8%)	4 (3.12%)	21 (6.62%)	85 (26.1%)	70 (50.0%)	<0.001
	2	716 (78.6%)	110 (85.9%)	296 (93.4%)	240 (73.6%)	70 (50.0%)	
	3	15 (1.65%)	14 (10.9%)	0 (0.00%)	1 (0.31%)	0 (0.00%)	
Milking system	Manually	381 (41.8%)	1 (0.78%)	41 (12.9%)	268 (82.2%)	71 (50.7%)	<0.001
	Bucket milking machine	410 (45.0%)	30 (23.4%)	258 (81.4%)	58 (17.8%)	64 (45.7%)	
	Pipeline milking machine	120 (13.2%)	97 (75.8%)	18 (5.68%)	0 (0.00%)	5 (3.57%)	
Milk storage system	Milk can without cooling	57 (6.26%)	1 (0.78%)	3 (0.95%)	37 (11.3%)	16 (11.4%)	<0.001
	Fridge	55 (6.04%)	0 (0.00%)	4 (1.26%)	49 (15.0%)	2 (1.43%)	
	Freezer	212 (23.3%)	1 (0.78%)	39 (12.3%)	155 (47.5%)	17 (12.1%)	
	Milk can cooler	218 (23.9%)	6 (4.69%)	111 (35.0%)	52 (16.0%)	49 (35.0%)	
	Bulk milk cooler	369 (40.5%)	120 (93.8%)	160 (50.5%)	33 (10.1%)	56 (40.0%)	
Manure storage system	No	786 (86.3%)	63 (49.2%)	279 (88.0%)	315 (96.6%)	129 (92.1%)	<0.001
	Yes	125 (13.7%)	65 (50.8%)	38 (12.0%)	11 (3.37%)	11 (7.86%)	
Used manure	No	167 (18.3%)	6 (4.69%)	53 (16.7%)	78 (23.9%)	30 (21.4%)	<0.001
	Yes	744 (81.7%)	122 (95.3%)	264 (83.3%)	248 (76.1%)	110 (78.6%)	
Owned Tractor	No	529 (58.2%)	11 (8.59%)	166 (52.4%)	283 (87.3%)	69 (49.3%)	<0.001
	Yes	380 (41.8%)	117 (91.4%)	151 (47.6%)	41 (12.7%)	71 (50.7%)	
Pasture management	No	283 (31.1%)	20 (15.6%)	59 (18.6%)	171 (52.5%)	33 (23.6%)	<0.001
	Yes	628 (68.9%)	108 (84.4%)	258 (81.4%)	155 (47.5%)	107 (76.4%)	
Conducted pasture rotation	No	366 (40.2%)	44 (34.4%)	94 (29.7%)	183 (56.1%)	45 (32.1%)	<0.001
	Yes	545 (59.8%)	84 (65.6%)	223 (70.3%)	143 (43.9%)	95 (67.9%)	
Pasture renovation	No	483 (53.0%)	62 (48.4%)	140 (44.2%)	234 (71.8%)	47 (33.6%)	<0.001
	Yes	428 (47.0%)	66 (51.6%)	177 (55.8%)	92 (28.2%)	93 (66.4%)	
Produced feed supplement	No	195 (21.4%)	6 (4.69%)	48 (15.1%)	101 (31.0%)	40 (28.6%)	<0.001
	Yes	716 (78.6%)	122 (95.3%)	269 (84.9%)	225 (69.0%)	100 (71.4%)	
Purchased feed supplement	No	214 (23.5%)	7 (5.47%)	38 (12.0%)	123 (37.7%)	46 (32.9%)	<0.001
	Yes	697 (76.5%)	121 (94.5%)	279 (88.0%)	203 (62.3%)	94 (67.1%)	
Produced silage	No	448 (49.2%)	10 (7.81%)	98 (30.9%)	256 (78.5%)	84 (60.0%)	<0.001
	Yes	463 (50.8%)	118 (92.2%)	219 (69.1%)	70 (21.5%)	56 (40.0%)	
Produced hay	No	826 (90.7%)	90 (70.3%)	287 (90.5%)	315 (96.6%)	134 (95.7%)	<0.001
	Yes	85 (9.33%)	38 (29.7%)	30 (9.46%)	11 (3.37%)	6 (4.29%)	
Sell coproducts	No	813 (89.2%)	128 (100%)	306 (96.5%)	246 (75.5%)	133 (95.0%)	<0.001
	Yes	98 (10.8%)	0 (0.00%)	11 (3.47%)	80 (24.5%)	7 (5.00%)	
Breeding strategy	Artificial Insemination	391 (42.9%)	118 (92.2%)	173 (54.6%)	74 (22.7%)	26 (18.6%)	<0.001
	Controlled breeding	138 (15.1%)	3 (2.34%)	55 (17.4%)	66 (20.2%)	14 (10.0%)	
	Natural breeding	382 (41.9%)	7 (5.47%)	89 (28.1%)	186 (57.1%)	100 (71.4%)	

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7.2 Appendix Chapter 3

Annex A.
Parameter estimates of the hyperbolic distance function

Technology	D_H CH_4		SE
α0 (Intercept)	-0.218	***	0.040
α1 (Capital)	-0.043	***	0.012
α2 (Lactating cows)	-0.207	***	0.051
α3 (Labour)	0.012		0.023
α4 (Land)	-0.019	*	0.009
α5 (Feed)	-0.154	***	0.028
α6 (Other expenses)	-0.111	***	0.024
β1 (Methane)	-0.257	***	0.029
β00	0.239		0.236
α11	0.054	***	0.016
α22	2.518	***	0.581
α33	-0.023		0.117
α44	-0.004		0.011
α55	0.001		0.084
α66	0.060		0.053
α12	-0.103	*	0.059
α13	0.005		0.029
α14	0.023	*	0.011
α15	0.010		0.031
α16	0.129	***	0.033
α23	-0.265		0.173
α24	-0.026		0.064
α25	-0.863	***	0.178
α26	0.051		0.144
α34	0.025		0.023
α35	-0.323	***	0.063
α36	0.105	*	0.047
α45	0.051	*	0.021
α46	-0.057	**	0.019
α56	0.102	*	0.052
δ2(Animals sold)	0.005	**	0.002
δ22	0.001	**	0.001
χ10	-0.043		0.029
χ20	-0.682	*	0.348
χ30	0.230	*	0.094
χ40	0.005		0.037
χ50	0.317	**	0.114
χ60	-0.152		0.093
γ12	0.002	*	0.001
γ22	0.016	***	0.005

Annex A.
Parameter estimates of the hyperbolic distance function. Continued

Technology	$D_H CH_4$	SE	Technology
γ32	-0.011	***	0.002
γ42	0.003	***	0.001
γ52	0.001		0.003
γ62	0.005	**	0.002
ρ20	-0.008	***	0.003
ω2	-0.042	**	0.013
ω3	-0.034	*	0.015
ω4	-0.031		0.024
Heteroskedasticity in σ_u^2			
ζ0 (Intercept)	3.881	**	1.425
ζ1 (Buyers)	0.092		0.059
ζ2 (Milk yield)	-0.481	***	0.074
ζ3 (Time farming)	-0.015		0.010
ζ4 (Intensive pasture)	-1.773	*	0.880
ζ5 (Cows in the herd)	-3.807	*	1.631
ζ6 (Tech. support)	-0.055		0.036
ζ7 (Bull in the herd)	0.239		0.312
ζ8 (Hire labour)	0.695	*	0.370
ζ9 (Rent land)	-0.107		0.342
Heteroskedasticity in σ_v^2			
τ0 (Intercept)	-16.849	***	2.457
τ1 (Buyers)	0.335	*	0.137
τ2 (Milk yield)	0.683	***	0.123
τ3 (Bull in the herd)	0.014		0.065
τ4 (Hire labour)	-1.905	**	0.629
τ5 (Rent land)	-0.721		0.642
Log_Likelihood	236.15		
Mean EE	0.9141		
Std.Dev	0.0873		

7.3 Appendix Chapter 4 – (Supporting Information)

Environmental and economic performance of paddy field-based crop-livestock systems in Southern Brazil

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7.4 Table S1

Parameters applied in the calculations.

Description	Unit	Value	Source
Daily weight gain (BR)	kg LW	0.827	Experiment
Daily weight gain (BSR)	kg LW	0.956	Experiment
Daily weight gain (BL)	kg LW	0.461	Adapted from (Balbino et al., 2012)
Methane emission (CH_4)*	kg $d^{-1}ha^{-1}$	4.06 non-till; 4.50 min-tillage	(Bayer et al., 2014; Zschornack et al., 2018)
Crude protein in rice residue	% DM	4.2	https://www.feedipedia.org/node/557
N content of residue dry Matter	g kg^{-1}	7 ryegrass and rice; 8 soybeans	Table 11.1A (IPCC, 2019e)
Protein content of ryegrass	% DM	18.59	http://www.cqbal.com.br/
Protein content of natural pasture	% DM	6.96	http://www.cqbal.com.br/
Conversion from protein to N	kg $(kg\ N)^{-1}$	6.38 milk; 6.25 otherwise	FAO (2017)
Phosphorus (P) content of pasture	% DM	0.22	https://www.feedipedia.org
P content in milk	%	0.093	FAO (2017 p. 109)
P content in cattle live-weight	%	0.76 calves; 0.73 other	FAO (2017 p. 107-8)
N content in cattle live-weight	%	3.3 calves; 2.7 other	FAO (2017 p. 107-8)
Daily salt intake beef cow herd	kg $(kg\ LW)^{-1}$	0.050	(Dick et al., 2015a)
Empty body weight (EBW)	% of LW	85	FAO (2017)
Energy content in cattle EBW	MJ kg^{-1}	9.73	(Valadares Filho et al., 2016)
Energy content in soybeans dry matter	MJ kg^{-1}	23.6	https://www.feedipedia.org/node/42
Energy content in rice dry matter	MJ kg^{-1}	17.6	https://www.feedipedia.org/node/226
Protein cattle LW	kg kg^{-1}	0.169	FAO (2017 p. 107-8)
Protein soybeans dry matter	kg kg^{-1}	0.396	https://www.feedipedia.org/node/42
Protein rice dry matter	kg kg^{-1}	0.105	(Nunes et al., 2016)

*Assuming a cultivation cycle of 115 days

7.5 Table S2

Structure of the suckler cow herd modelled to account for young stock production.

Animal category	Head	Body-weight	
		Initial	Final
Cow, mature	221	380	380
Cow (Fail pregnancy)	140	380	380
Bull	18	600	600
Male, weaners	137	32	170
Female, weaners	137	32	150
Bull 2-year-old, replacement	3	170	328
Heifer 2-year-old, replacement	66	150	308
Bull 3-year-old, replacement	3	328	600
Heifer 3-year-old, first mating	64	308	380
Outputs			
Male weaners sold	134	190	
Female weaners sold	70	180	
Cull cow for slaughter	64	380	
Cull bull for slaughter	3	600	
Mortality animals < 1 year old	11	160[a]	
Mortality animals => 1 year old	9	378[a]	

[a] Weighted average from each animal category.

7.6 Table S3

Main characteristics of the suckler cow herd modelled to account for young stock production.

	Unit	Value	Source
Cow	head	425	(Malafaia et al., 2014)
Bull cow rate	%	4	(Malafaia et al., 2014)
Weaning rate	%	67	(Malafaia et al., 2014)
Yearly replacement rate of cows	%	15	(Malafaia et al., 2014)
Yearly replacement rate of bull	%	15	(Malafaia et al., 2014)
Yearly mortality < 1 year	%	4	(Malafaia et al., 2014)
Yearly mortality > 1 year	%	2	(Malafaia et al., 2014)
First calving	month	36	(Malafaia et al., 2014; Ruviaro et al., 2015)
Age of weaning	month	7	(Malafaia et al., 2014; Ruviaro et al., 2015)
Age at slaughter	month	36	(Malafaia et al., 2014; Ruviaro et al., 2015)
Calf weight	kg	32	(Ruviaro et al., 2015)
Weaned male weight	kg	170	(Dick et al., 2015a; Ruviaro et al., 2015)
Weaned female weight	kg	150	(Dick et al., 2015a)
Carcass yield male	%	50	(Malafaia et al., 2014)
Carcass yield female	%	48	(Malafaia et al., 2014)
Daily weight gain (Natural pasture)	kg	0.31	(Malafaia et al., 2014; Ruviaro et al., 2015)
Daily milk production	kg	1.1	(Dick et al., 2015a)
Land use kg of LW per hectare	kg LW	315	(Malafaia et al., 2014), based on winter season.

7.7 Table S4

Life cycle inventory for the production of one kg-liveweight weaned calves, leaving the farm averaging 190 kg LW, produced in natural grassland, presented for mass and economic allocation.

	Unit	Mass	CV	Economic [a]	CV
Inputs					
Drinking water	m^3 (kg LW)	0.121	0.060	0.139	0.060
Land occupation	ha (kg LW)	0.015	-	0.017	-
Salt consumption	kg (kg LW)	0.209	0.100	0.239	0.100
Transport	tkm (kg LW)	0.010		0.012	
Outputs to air					
Methane enteric	kg (kg LW)	0.723	0.045	0.828	0.045
Methane manure	kg (kg LW)	0.082	0.145	0.094	0.145
Direct dinitrogen monoxide	kg (kg LW)	0.007	1.946	0.007	1.946
Indirect dinitrogen monoxide, Atd	kg (kg LW)	0.003	0.295	0.004	0.295
Indirect dinitrogen monoxide, runoff/leach	kg (kg LW)	0.003	1.571	0.003	1.571
Ammonia	kg (kg LW)	0.290	0.260	0.332	0.260
Nitrogen oxides	kg (kg LW)	0.016	1.946	0.019	1.946
Outputs to water					
Nitrate	kg (kg LW)	0.353	1.517	0.404	1.517
Phosphate, leaching	kg (kg LW)	0.003	1.945	0.003	1.945
Phosphate, runoff	kg (kg LW)	0.010	1.945	0.011	1.945
Phosphorus, erosion	kg (kg LW)	0.0007	1.945	0.001	1.945

[a] economic allocation was calculated as 65% to calves and 35% to cull animals, based on averaged prices for the years under analysis (2013-2016).

7.8 Table S5

Background processes used to build the life cycle assessment (retrieved from EcoInvent 3).

Process	Adjustments
Electricity, medium voltage {BR}| market for | Alloc Def, U	-
Electricity, low voltage {BR}| market for | Alloc Def, U	-
Grass seed, Swiss integrated production, for sowing {CH}| production | Alloc Def, U	Adjusted to Brazilian electricity mix
Rice seed, for sowing {RoW}| production | Alloc Def, U	Adjusted to Brazilian electricity mix, and inclusion of 120 km transportation
Transport, freight, lorry 16-32 metric ton, EURO3 {RER}| transport, freight, lorry 16-32 metric ton, EURO3 | Alloc Def, U	-
Transport, freight, lorry 7.5-16 metric ton, EURO3 {RER}| transport, freight, lorry 7.5-16 metric ton, EURO3 | Alloc Def, U	-
Soybean seed, for sowing {CH}| production | Alloc Def, U	Adjusted to Brazilian electricity mix, soybean production in Brazil, and inclusion of 120 km transportation
Pesticide, unspecified {GLO}| market for | Alloc Def, U	-
Sodium chloride, powder {RER}| production | Alloc Def, U	Adjusted to Brazilian electricity mix, and inclusion of 120 km transportation
Ammonium nitrate, as N {RER}| ammonium nitrate production | Alloc Def, U	-
Urea, as N {RoW}| production | Alloc Def, U	-
Phosphate fertiliser, as P2O5 {RoW}| triple superphosphate production | Alloc Def, U	-
Potassium chloride, as K2O {RoW}| potassium chloride production | Alloc Def, U	-
Diesel {RoW}| market for | Alloc Def, U	-
Lime {CH}| production, milled, loose | Alloc Def, U	Adjusted to Brazilian electricity mix

7.9 Supporting Information equations

7.9.1 Equation S1

Direct Nitrous oxide (N_2O) from managed soils (adapted from equation 11.2 (IPCC, 2019e)).

$$N_2O = \left\{\left[(N_{sn} * EF_{1min}) + \left[(N_{cr} + N_{Som}) * EF_{1org}\right] + \left(N_{prp} * EF_3\right)\right] * \frac{44}{28}\right\}$$

Description	Unit	Source
N_2O: emissions of N_2O per hectare	kg ha^{-1}	
N_{sn}: amount of synthetic fertiliser N applied to soils	kg ha^{-1}	Inventory
N_{cr}: amount of N in crop residues (above-ground and below-ground)	kg ha^{-1}	(IPCC, 2019e) Tables 11.1A, 11.2
N_{Som}: amount of N in mineral soil that is mineralised	kg ha^{-1}	(IPCC, 2019e) p.11.19
N_{prp}: amount of urine and dung N deposited by grazing animals	kg ha^{-1}	Inventory
$EF_{1min/org}$: emission factor for N_2O-N emissions from fertiliser N inputs	kg kg^{-1}	(IPCC, 2019e) p.11.11, Table 11.1
EF_{1org}: emission factor for N_2O-N emissions from organic N inputs		(IPCC, 2019e) p.11.11, Table 11.1
EF_3: emission factor for N_2O-N emissions from urine and dung N deposited on pasture	kg kg^{-1}	(IPCC, 2019e) p.11.11, Table 11.1
44/28: conversion factor N_2O-N to N_2O	-	

7.9.2 Equation S2

Induced Nitrous oxide (N_2O) from atmospheric deposition (adapted from equation 11.11 (IPCC, 2019e)).

$$N_2O = \left\{\left[\left(N_{sn} * Frac_{gasf}\right) + \left(N_{prp} * Frac_{gasm}\right)\right] * EF_4 * \frac{44}{28}\right\}$$

Description	Unit	Source
N_2O: Induced emissions of N_2O	kg ha^{-1}	
N_{sn}: amount of synthetic fertiliser N applied to soils	kg ha^{-1}	Inventory
N_{cr}: amount of N in crop residues (above-ground and below-ground)	kg ha^{-1}	(IPCC, 2019e) Tables 11.1A, 11.2
N_{prp}: amount of urine and dung N deposited by grazing animals on pasture	kg ha^{-1}	Inventory
$Frac_{gasf}$: fraction of synthetic fertiliser N that volatilises as NH_3 and NOx	-	(IPCC, 2019e) p.11.46, Table A7-3
$Frac_{gasm}$: fraction of urine and dung N deposited by grazing animals (F_{PRP}) that volatilises as NH3 and NOx	-	(IPCC, 2019e) p.11.47, Table A8-1
EF_4: emission factor for N2O-N emissions from atmospheric deposition of N volatilised (NH_3–N + NO_x–N) on soils and water surfaces	kg kg^{-1}	(IPCC, 2019e) p.11.25, Table 11.3
44/28: conversion factor N_2O-N to N_2O	-	

7.9.3 Equation S3

Induced nitrous oxide (N_2O) from leaching/runoff from managed soils (adapted from equation 11.10 (IPCC, 2019e)).

$$N_2O = \left[\left(N_{sn} + N_{prp} + N_{cr} + N_{Som}\right) * Frac_{leach} * EF_5 * \frac{44}{28}\right]$$

Description	Unit	Source
N_2O: emissions of N_2O	kg ha^{-1}	
N_{sn}: amount of synthetic fertiliser N applied to soils	kg ha^{-1}	Inventory
N_{cr}: amount of N in crop residues (above-ground and below-ground)	kg ha^{-1}	(IPCC, 2019e) Tables 11.1A, 11.2
N_{prp}: amount of urine and dung N deposited by grazing animals on pasture	kg ha^{-1}	Inventory
N_{Som}: amount of N in mineral soil that is mineralised	kg ha^{-1}	(IPCC, 2019e) p.11.19
$Frac_{leach}$: fraction of all N added to/mineralised in managed soils in regions where leaching/runoff occurs that is lost through leaching and runoff	kg kg^{-1}	(IPCC, 2019e) Table 11.3
EF_5: emission factor for N_2O emissions from N leaching and runoff	kg kg^{-1}	(IPCC, 2019e) Table 11.3
44/28: conversion factor N_2O-N to N_2O	-	

7.9.4 Equation S4

CO_2 emissions from lime application (based on equation 11.12 (IPCC, 2006)).

$$CO_{2l} = \left[(\mathrm{L} * \mathrm{EF}_l) * \frac{44}{12}\right]$$

Description	Unit	Source
CO_{2l}: CO_2 emissions from lime application	Kg ha^{-1}	
L: amount of limestone applied	kg	
EF_l	-	(IPCC, 2006) p.11.27
44/12: conversion factor CO_2-C to CO_2		

7.9.5 Equation S5

CO_2 emissions from urea fertilisation (based on equation 11.13 (IPCC, 2006)).

$$\mathrm{CO_{2u}} = \left[(\mathrm{U} * \mathrm{EF}_u) * \frac{44}{12}\right]$$

Description	Unit	Source
CO_{2u}: CO_2 emissions from urea application	kg ha^{-1}	
U: amount of urea applied	kg	Table 1
EFu	-	(IPCC, 2006) p.11.32
44/12: conversion factor CO_2-C to CO_2		

7.9.6 Equation S6

Net energy for maintenance (based on equation 10.3 (IPCC, 2019c)).

$$\mathrm{NE_m} = [\mathrm{CF_i} * (\mathrm{Weight})^{0.75}]$$

Description	Unit	Source
NE_m: net energy required by the animal for maintenance	MJ d^{-1}	
CF_i: a coefficient which varies for each animal category	MJ d^{-1} kg^{-1}	(IPCC, 2019c) p. 10.25, Table 10.4
Weight: live-weight of animal	kg	Table S1

7.9.7 Equation S7

Net energy for activity (based on equation 10.4 (IPCC, 2019c)).

$$\mathrm{NE} = (\mathrm{C_a} * \mathrm{NE_m})$$

Description	Unit	Source
NE_a: net energy for animal activity	MJ d^{-1}	
C_a: coefficient corresponding to animal's feeding situation	MJ $d^{-1}kg^{-1}$	(IPCC, 2019c) p. 10.66, Table 10.5
NE_m: net energy required by the animal for maintenance	MJ d^{-1}	Equation S6

7.9.8 Equation S8

Net energy for growth (based on equation 10.6 (IPCC, 2019c)).

$$\mathrm{NE_g} = 22.02 * \left(\frac{\mathrm{BW}}{\mathrm{C} * \mathrm{MW}}\right)^{0.75} * \mathrm{WG}^{1.097}$$

Description	Unit	Source
NE_g: net energy needed for growth	MJ day^{-1}	
BW: the average live body weight (BW) of the animals in the population (phase)	kg	Table S1
C: coefficient	-	(IPCC, 2019c) p. 10.26
MW: Target weight at growth phase	kg	Table S1
WG: average daily weight gain of the animals in the phase	kg day^{-1}	Table S2, S4

7.9.9 Equation S9

Net energy for lactation (based on equation 10.8 (IPCC, 2019c)).

$$\mathrm{NE_l} = [\mathrm{Milk} * (1.47 + 0.40 * \mathrm{fat})]$$

Description	Unit	Source
NE_l: net energy for lactation	MJ d^{-1}	
Milk: amount of milk produced	kg d^{-1}	Table S2
Fat: fat content of milk by weight	%	(Cardoso et al., 2016)

7.9.10 Equation S10

Net energy for pregnancy (based on equation 10.13 (IPCC, 2019c)).

$$\mathrm{NE_p} = \left(\mathrm{C_{preg}} * \mathrm{NE_m}\right)$$

Description	Unit	Source
NE_p: net energy required for pregnancy	MJ d^{-1}	
C_{preg}: pregnancy coefficient	-	(IPCC, 2019c) p. 10.30, Table 10.7
NE_m: net energy required by the animal for maintenance	MJ d^{-1}	Equation S6

7.9.11 Equation S11

Ratio of net energy available in a diet for maintenance to digestible energy consumed (based on equation 10.14 (IPCC, 2019c)).

$$\text{REM} = \left\{1.123 - (4.092 * 10^{-3} * \text{DE\%}) + [1.126 * 10^{-5} * (\text{DE\%})^2] - \left(\frac{25.4}{\text{DE\%}}\right)\right\}$$

Description	Unit	Source
REM: ratio of net energy available in a diet for maintenance to digestible energy consumed	-	
DE%: digestible energy expressed as a percentage of gross energy	%	Brazilian tables of feed composition for cattle (cqbal.com.br)

7.9.12 Equation S12

Ratio of net energy available for growth in a diet to digestible energy consumed (based on equation 10.15 (IPCC, 2019c)).

$$\text{REG} = \left\{1.164 - (5.160 * 10^{-3} * \text{DE\%}) + [1.308 * 10^{-5} * (\text{DE\%})^2] - \left(\frac{37.4}{\text{DE\%}}\right)\right\}$$

Description	Unit	Source
REG: ratio of net energy available for growth in a diet to digestible energy consumed	-	
DE%: digestible energy expressed as a percentage of gross energy	%	Brazilian tables of feed composition for cattle (cqbal.com.br)

7.9.13 Equation S13

Gross energy for cattle (based on equation 10.16 (IPCC, 2019c)).

$$GE = \left[\frac{\left(\frac{NE_m + NE_a + NE_l + NE_p}{REM}\right) + \left(\frac{NE_g}{REG}\right)}{\frac{DE\%}{100}}\right]$$

Description	Unit	Source
GE: gross energy	MJ d^{-1}	
NE_m: net energy required by the animal for maintenance	MJ d^{-1}	Equation S6
NE_a: net energy for animal activity	MJ d^{-1}	Equation S7
NE_l: net energy for lactation	MJ d^{-1}	Equation S9
NE_p: net energy required for pregnancy	MJ d^{-1}	Equation S10
NE_g: net energy needed for growth	MJ d^{-1}	Equation S8
REM: ratio of net energy available in a diet for maintenance to digestible energy consumed	-	Equation S11
REG: ratio of net energy available for growth in a diet to digestible energy consumed	-	Equation S12
DE%: digestible energy expressed as a percentage of gross energy	%	Brazilian tables of feed composition for cattle (cqbal.com.br)

7.9.14 Equation S14

Enteric fermentation (based on equation 10.21 (IPCC, 2019c)).

$$EF = \left[\frac{GE * \left(\frac{Ym}{100}\right) * d}{55.65}\right]$$

Description	Unit	Source
EF: CH_4 emissions	kg	
GE: daily gross energy intake per head	MJ d^{-1}	
Y_m: methane conversion factor (GE converted to CH_4)	%	(IPCC, 2019c) p. 10.48, Table10.12
d: days the animal stays in the phase	d	
55.65: energy content of methane	MJ kg^{-1}	

7.9.15 Equation S15

Volatile solid excretion rates (based on equation 10.24 (IPCC, 2019c)).

$$VS = \left\{\left[GE * \left(1 - \frac{DE\%}{100}\right) + (UE * GE)\right] * \left[\left(\frac{1 - Ash}{18.45}\right)\right]\right\}$$

Description	Unit	Source
VS: volatile solid excretion per day on a dry-organic matter basis	kg d^{-1}	
GE: gross energy intake	MJ d^{-1}	
DE%: digestibility of the feed in percent	%	Brazilian tables of feed composition for cattle (cqbal.com.br)
UE: urinary energy excretion expressed as fraction of GE	%	(IPCC, 2019c) p. 10.70
Ash: the ash content of manure calculated as a fraction of the dry matter feed intake	%	(IPCC, 2019c) p. 10.70
18.45: conversion factor for dietary GE per kg of dry matter	MJ kg^{-1}	(IPCC, 2019c) p. 10.70

7.9.16 Equation S16

Emission factor from manure management (based on equation 10.23 (IPCC, 2019c)).

$$EF = \left\{(VS * d) * \left[B_o * 0.67\ kg/m^3 * \sum_{S,k} \frac{MCF_{S,k}}{100} * MS_{(T,S,k)}\right]\right\}$$

Description	Unit	Source
EF: CH_4 emission factor for cattle in the phase	kg	
VS: daily volatile solid excreted (dry matter basis)	kg d^{-1}	Equation S15
d: basis for calculating VS production	d a^{-1}	
B_O: maximum methane producing capacity for volatile solid excreted produced by cattle	m^3 kg^{-1}	(IPCC, 2019c) p. 10.71
0.67: conversion factor of m^3 CH_4 to kilograms CH_4	-	
$MCF_{(S, k)}$: methane conversion factors for each manure management system S in climate region k	%	(IPCC, 2019c) Annex 10B.6
$MS_{(T,S,k)}$: fraction of livestock category T`s manure handled using manure management system S in climate region k	%	Cattle is reared only on open pastures

7.9.17 Equation S17

Estimation of dry matter intake (IPCC, 2019c) p. 10.31.

$$\mathrm{DMI} = \left(\frac{\mathrm{GE}}{\mathrm{ED_{feed}}}\right)$$

Description	Unit	Source
DMI: dry matter intake	kg d^{-1}	
GE: daily gross energy intake per head	MJ d^{-1}	Equation S13
ED_{feed}: Energy density of feed	MJ kg^{-1}	18.45

7.9.18 Equation S18

Nitrogen excretion rate (based on equation 10.31A (IPCC, 2019c)).

$$N_{ex}=(N_{intake}-N_{retention})$$

Description	Unit	Source
N_{ex}: nitrogen excretion rate per animal	kg d^{-1}	
N_{intake}: daily N intake	kg d^{-1}	% of protein in daily DM intake
$N_{retention}$: daily N retention	kg d^{-1}	N in animal tissue, including new born calves; and milk produced. (see table S4)

7.9.19 Equation S19

Phosphorus excretion rate (based on equation 10.31A (IPCC, 2019c)).

$$P_{ex}=(P_{intake}-P_{retention})$$

Description	Unit	Source
P_{ex}: phosphorus excretion rate per animal	kg d^{-1}	
N_{intake}: daily P intake	kg d^{-1}	P in daily DM intake, and mineral salt P (45g kg^{-1})
$N_{retention}$: daily P retention	kg d^{-1}	P in animal tissue, including new born calves; and milk produced. (see table S4)

7.9.20 Equation S20

Phosphate leaching to ground water (adapted from Nemecek and Schnetzer, (2012 p.16))

$$PO_4^{3-} - P_{leach} = \left[P_{gwl} * D\right]$$

Description	Unit	Source
PO_4^{3-}-P_{leach}: quantity of Phosphorus leached to ground water per season	kg ha^{-1}	
P_{gwl}: daily average quantity of P leached to ground water for a land use category	kg ha^{-1}	(Nemecek and Schnetzer, 2012)
D: days of the crop rotation		

7.9.21 Equation S21

Phosphate run-off to surface water (adapted from Nemecek and Schnetzer, (2012 p.17))

$$PO_4^{3-} - P_{runoff} = \left(P_{rol} * D * 1 + \frac{0.2}{80} * P_2O_{5min} + \frac{0.4}{80} * P_2O_{5man}\right)$$

Description	Unit	Source
PO_4^{3-}-P_{runoff}: quantity of P lost through run-off to rivers season	kg ha^{-1}	
P_{rol}: daily average quantity of P lost through run-off	kg ha^{-1}	(Nemecek and Schnetzer, 2012)
P_2O_{5min}: quantity of P_2O_5 contained in synthetic fertilizer and	kg ha^{-1}	Inventory
D: days of the crop rotation		Inventory
P_2O_{5man}: quantity of P_2O_5 contained in solid manure	kg ha^{-1}	Inventory

7.9.22 Equation S22

Phosphorus emissions through water erosion to surface water (adapted from Nemecek and Schnetzer, (2012 p.17))

$$P_{er} = (S_{er} * P_{cs} * F_r * F_{erw} * D)$$

Description	Unit	Source
P_{er}: quantity of P emitted through erosion to rivers in the season	kg ha^{-1}	
S_{er}: quantity of soil eroded	kg $(ha\ d)^{-1}$	Inventory
P_{cs}: P content in the top soil	kg kg^{-1}	(Bayer et al., 2014; Zschornack et al., 2018)
Fr: enrichment factor for P (-)	-	
F_{erw}: Fraction of eroded soil that reaches the river	-	0.2 ryegrass, soybean; 1 rice
D: days of the crop rotation		Inventory

7.9.23 Equation S23

Daily methane emissions factor from rice cultivation (based on equation 5.2 (IPCC, 2019f)).

$$EF_{CH4} = \left(EF_c + SF_w + SF_p + SF_o\right)$$

Description	Unit	Source
EF_{CH4}: adjusted daily methane emission factor	kg ha^{-1} d^{-1}	
EF_c: baseline emission factor for continuously flooded fields without organic amendments		(IPCC, 2019f) p.5.59 Table 5.11 = (1.27)
SF_w: scaling factor to account for the differences in water regime during the cultivation period		(IPCC, 2019f) p.5.60, Table 5.12 = (1)
SF_p: scaling factor to account for the differences in water regime in the pre-season before the cultivation period		(IPCC, 2019f) p.5.61,Table 5.13 = (0.89)
SF_o: scaling factor should vary for both type and amount of organic amendment applied		(IPCC, 2019f) p.5.61-62 Equation 5.3 and Table 5.14 = (1: BR, BSR; 1.6: BL)

7.10 Supporting information results

7.10.1 Table S6

Table S6
Mean gross revenue, costs, and profit (return to land and management) for the crop-livestock systems.

		Crop-livestock system BR			Crop-livestock system BSR				Crop-livestock system BL		
	Unit	Ryegrass + Cattle	Rice	ha·a	Ryegrass + Cattle	Soybean	Rice	ha·a	Cattle	Rice	ha·a
Revenue											
Gross revenue	US$	1692.7	2912.3	4605.2	1256.3	1332.7	3002.2	3423.0	945.4	1941.5	2886.2
Operating expenses											
Purchased cattle	US$	1364.8	0.0	1365.3	878.3	0.0	0.0	878.9	859.4	0.0	859.4
Seed	US$	19.5	63.3	82.8	19.5	34.8	60.8	67.3	0.0	73.4	73.4
Fertiliser, lime	US$	304.3	322.5	626.3	313.6	207.5	333.3	584.1	0.0	210.6	210.6
Chemicals	US$	53.9	254.7	308.2	52.3	226.5	230.4	281.0	3.5	220.7	224.2
Electricity	US$	1.2	61.1	62.3	1.8	1.3	61.8	33.3	0.9	80.1	80.8
Freight inputs	US$	15.3	14.7	30.0	15.7	10.3	13.8	27.7	0.0	11.9	11.9
Machinery	US$	49.6	319.5	369.2	61.0	183.4	322.4	313.9	11.1	361.4	372.5
Insurance	US$	2.5	6.7	9.2	3.9	6.6	6.8	10.6	1.9	6.7	8.6
Labour	US$	15.0	159.6	174.6	20.2	81.2	164.7	143.2	8.1	138.5	146.6
Freight, drying	US$	0.0	359.4	359.3	0.0	96.2	391.3	243.8	0.0	239.3	239.3
Env. licence	US$	0.9	1.6	2.5	0.9	1.2	1.9	2.5	0.8	1.7	2.5
Interest on operating expenses	US$	36.2	61.6	97.8	37.9	38.9	62.2	88.5	14.4	57.4	71.8
Ownership expenses											
Machinery and facilities	US$	47.9	290.1	338.0	59.3	174.1	288.9	290.8	7.8	321.2	329.0
Profit (Return to land and management)	US$	-218.4	997.6	779.8	-208.2	270.8	1063.9	457.4	37.6	218.7	255.7

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7.4 Declaration

1. I, hereby, declare that this Ph.D. dissertation has no not been presented to any other examining body either in its present or a similar form. Furthermore, I also affirm that I have not applied for a Ph.D. at any other higher school of education.

Göttingen, 30/06/2022

Everton Vogel

2. I, hereby, solemnly declare that this dissertation was undertaken independently and without any unauthorised aid.

Göttingen, 30/06/2022

Everton Vogel