Meta Science 98 (2014) 346–354

Contents lists available at ScienceDirect

Meat Science

journal homepage: www.elsevier.com/locate/meatsci

Sustainability of meat production beyond carbon footprint: a synthesis of case studies from grazing systems in Uruguay

Valentín D. Picasso [a,*], Pablo D. Modernel [a,c], Gonzalo Becoña [b], Lucía Salvo [a], Lucía Gutiérrez [a], Laura Astigarraga [a]

[a] Facultad de Agronomía, Universidad de la República, Av. E. Garzón 780, Montevideo, Uruguay
[b] Instituto Plan Agropecuario, Bv. Artigas 3802, Montevideo, Uruguay
[c] Wageningen University, The Netherlands

ARTICLE INFO

Article history:
Received 15 April 2014
Received in revised form 5 July 2014
Accepted 7 July 2014
Available online 15 July 2014

Keywords:
Greenhouse gases
Climate change
Environmental impact
Grazing
Pasture
Feedlot

ABSTRACT

Livestock production has been challenged as a large contributor to climate change, and carbon footprint has become a widely used measure of cattle environmental impact. This analysis of fifteen beef grazing systems in Uruguay quantifies the range of variation of carbon footprint, and the trade-offs with other relevant environmental variables, using a partial life cycle assessment (LCA) methodology. Using carbon footprint as the primary environmental indicator has several limitations: different metrics (GWP vs. GTP) may lead to different conclusions, carbon sequestration from soils may drastically affect the results, and systems with lower carbon footprint may have higher energy use, soil erosion, nutrient imbalance, pesticide ecotoxicity, and impact on biodiversity. A multidimensional assessment of sustainability of meat production is therefore needed to inform decision makers. There is great potential to improve grazing livestock systems productivity while reducing carbon footprint and other environmental impacts, and conserving biodiversity.

1. Introduction

Livestock production is growing worldwide because of the increased demand for animal proteins. Beef cattle production has increased in the last three decades almost 40% worldwide, being the Americas one of the regions that led this development (FAO, 2013). At the same time, the need to reduce the sector's greenhouse gas (GHG) emissions and its overall environmental footprint has become a top priority for industry and policy makers (Gerber et al., 2013). Carbon footprint has become the indicator to quantify the GHG emission intensity, usually expressed from the standpoint of the consumer as kg of CO_2 equivalent (De Vries & de Boer, 2010). A recent FAO report challenged the global livestock sector as a large contributor to climate change representing 14.5% of anthropogenic GHG emissions, and therefore, a sector with major opportunities for mitigation (Gerber et al., 2013). Due to their high cattle numbers the Latin America and the Caribbean have the largest challenges and opportunities.

In Uruguay cattle graze year-round on natural grasslands from the Campos biome (Royo Pallarés, Berretta, & Maraschin, 2005), improved pastures with legumes and P fertilizer added, and seeded pastures (i.e., mixtures of temperate grasses and legumes replacing the native vegetation, also known as ley). Cow–calf systems breed heifers at around 2.5 years of age, and calves are weaned at 6 months of age, and 130 to 150 kg of live weight (LW), with a national weaning rate between 63 and 70% (DIEA, 2013). Backgrounding of steers (from 150 to 350 kg LW) is usually done on native grasslands and seeded pastures. Finishing of steers (up to 500 to 550 kg LW) is also mostly done on pastures, and only 10% of the steers are finished in feedlots. The expansion of agriculture (driven by no tillage soybean production) has reduced the area of grasslands to 70% of the country area, and pushed livestock production to marginal lands, as well as providing opportunities for intensification of livestock systems based on higher inputs and grains. In this context, Uruguay has increased its beef production more than 45% since 1980 (DIEA, 2013), representing currently almost 75% of the GHG emissions of the whole country (MVOTMA, 2010). Therefore, climate change mitigation and adaptation, soil erosion control, grassland biodiversity conservation, and water quality are major environmental priorities for the Ministry of Livestock, Agriculture and Fisheries of Uruguay (MGAP, 2013).

The range of beef carbon footprint estimates among production systems in Uruguay is large (Becoña, Astigarraga, & Picasso, 2014; Modernel, Astigarraga, & Picasso, 2013), so that there is a high potential for reducing GHG emissions. In beef cow–calf grazing systems, using forage efficiently by optimizing forage allowance is a key mitigation option that can increase beef productivity and reduce carbon footprint

* Corresponding author. Tel.: +598 23543460.
E-mail address: vpicasso@fagro.edu.uy (V.D. Picasso).

http://dx.doi.org/10.1016/j.meatsci.2014.07.005
0309-1740/© 2014 Elsevier Ltd. All rights reserved.

per kg of beef and per ha (Becoña et al., 2014). In beef backgrounding–finishing systems, confinements (feedlots) have lower GHG emissions than grazing systems, and improving productivity of grazing systems can greatly mitigate GHG emissions (Modernel et al., 2013).

Carbon footprint studies in the beef sector worldwide have identified GHG mitigation opportunities at global and local scales for tackling climate change (some examples are shown in Table 1). Improving efficiency of beef production, through increasing animal intake quantity and quality, increasing reproductive efficiency, and daily weight gain, may result in significant reductions of GHG emissions from ruminants. Furthermore, in beef grazing systems, grazing efficiency could play a central role in GHG emission mitigation (Herrero et al., 2013).

Despite the major contribution of carbon footprint to the understanding and mitigation of GHG emissions, sustainability of food production is a much broader concept than carbon footprint. Furthermore, there is little recognition of the role livestock grazing systems play in storing carbon, protecting biodiversity and utilizing marginal land that cannot be used for crops. Therefore, the literature on sustainability and life cycle assessment (LCA) can significantly contribute to inform meat industry and policy makers. The objectives of this paper were to quantify carbon footprint using various metrics and several other environmental variables: fossil energy consumption, soil erosion, nutrient balance, pesticide ecotoxicity, and impact on biodiversity, among fifteen beef grazing systems in Uruguay. Our hypotheses were: i) that the carbon footprint of different beef grazing systems will change when using different metrics for assessment and ii) that significant tradeoffs exist among alternative environmental variables, especially between carbon footprint and impact on biodiversity.

2. Materials and methods

2.1. Description of the systems

Beef production cycle in Uruguay can be divided in cow–calf and finishing stages. The first one includes the reproduction process, producing calves of 150 kg live weight (LW) on average that would enter the meat production stage. Finishing includes an initial phase where the animal grows from 150 to 350 kg LW on average (backgrounding) and the fattening phase, going from 350 kg LW to slaughter weight (500 kg LW on average). Farms can specialize in breeding (cow–calf farms), finishing, or both (complete cycle farms). The most common management in farms based on natural grasslands include high stocking rates, with consequent overgrazing (Carvalho & Batello, 2009) and low forage allowance (i.e., kilograms of forage every 100 kg of animal LW, Sollenberger, Moore, Allen, & Pedreira, 2005) which limits livestock production. Other forage sources used by farmers are improved grasslands (natural grasslands oversown with legumes) and seeded pastures (i.e., ley) mostly comprised by exotic perennial species such as Fescue (*Festuca arundinacea* Schreb.), white clover (*Trifolium repens* L.) and birds foot trefoil (*Lotus corniculatus* L.). In both cases phosphorus and nitrogen fertilizer are applied. Seeded pastures achieve acceptable yields until 3 to 4 years, when a crop is sown as part of a crop–pasture rotation. Annual fodder crops in winter (ryegrass and oats) and summer (sorghum fodder) are used.

In order to represent a wide range of beef producing cycles in Uruguay, this study compared 15 beef production cycles, which were the combination of three cow–calf systems and five finishing systems. The boundary of our study is the primary production phase (i.e., farming systems), not the entire beef value chain.

The three cow–calf systems were averages of groups of farms previously identified through statistical clustering of seven production and environmental variables from 20 cow–calf farms, as described in Becoña et al. (2014). The three more contrasting clusters of cow–calf systems were included in the analysis for this paper. The "low performance" cow–calf farms (LP) had the lowest forage production allowance and high stocking rates and poor herd reproductive parameters. The "intermediate performance" cow–calf farms (IP) had an intermediate stocking rate and forage production allowance, resulting in better reproductive performance, but a low efficiency in heifer raising, and intermediate beef productivity. Finally, the "high performance" cow–calf farms (HP) had high beef productivity and excellent reproductive performance, sustained by high stocking rates on optimal forage production allowance, resulting in minimal carbon footprint.

The five finishing systems were identified based on previous published literature and expert opinion, as combinations of two typical backgrounding and three fattening systems (Modernel et al., 2013). Backgrounding systems were based on grazing, either native grasslands (G) or seeded pastures (P). Fattening was based on grasslands (G), seeded pastures (P), or feedlot (F). Five different combinations of these background–finishing stages were included in the analysis of this paper: G–G, G–P, P–P, G–F, and P–F.

A summary of each system's nutritional characteristics and productive performance is presented in Table 2. Nutritional requirements were used to calculate the relative area for each system needed to produce the required amount of feed, using national technical coefficients

Table 1
Comparison of grasslands and pasture based beef systems carbon footprint (kg $CO_{2e}\cdot$kg LW^{-1}) from various studies.
Modified from Becoña et al. (2014).

Beef system	Feed base	Mean	Country	Reference
Cow–calf	Native grasslands	28.7	Uruguay	Becoña et al. (2014)
	Native grasslands and improved pastures	20.8		
	Native grasslands and seeded pastures	16.0		
	Mixed hay and pasture	10.4	Canada	Beauchemin, Henry Janzen, Little, McAllister, & McGinn (2010)
	40% legume pasture, grass hay, and wheat	10.5	USA	Pelletier, Pirog, & Rasmussen (2010)
Finishing only	40% legume pasture, brome pasture, grass and alfalfa hay	8.7	USA	Pelletier et al. (2010)
Backgrounding–finishing	Native grasslands	16.7	Uruguay	Modernel et al. (2013)
	Native grasslands–seeded pastures	13.0		
	Seeded pastures	9.5		
	Native grasslands–feedlot	10.5		
	Seeded pastures–feedlot	6.9		
	Pastures and supplements	19.3	Australia	Peters et al. (2010)
	Native grasslands	42.6	Brazil	Ruviaro, de Léis, Lampert, Barcellos, & Dewes (2014)
	Improved natural grass	20.2		
	Native grass/ryegrass	29.6		
	Improved grasslands/sorghum	23.4		
	Cultivated ryegrass and sorghum	20.0		
	Native grass suppl. with protein mineralized salt	33.3		
	Native grass suppl. with protein energy mineralized salt	23.4		

Table 2
Diet characteristics and productive performance of three cow–calf, two backgrounding, and three fattening beef systems in Uruguay.
Adapted from Becoña et al. (2014) for cow calf systems; and Modernel et al. (2013) for backgrounding–fattening systems.

System	Cow–calf			Backgrounding		Fattening		
	Breeding and calf up to 150 kg LW			Steer 150 to 350 kg LW		Steer 350 to 500 kg LW		
	Low performance	Intermediate performance	High performance	Grassland	Seeded pasture	Grassland	Seeded pasture	Feedlot
Diet composition [a] (dry matter % for backgrounding–finishing)	Native pasture (95%) and improved pastures (5%)	Native pasture (93%) and improved pastures (7%)	Native pasture (66%) and improved pastures (34%)	100% native pasture	Seeded pasture (61%), native pasture (30%) and sorghum grain (9%)	100% native pasture	Seeded pasture (93%), sorghum grain (6.5%), and rice bran (0.5%)	Sorghum grain (60.5%), rice bran (12%), rice husk (14%), rice hay (8.5%), vitamins and minerals (5%)
Calf weaning rate (%) [a]	61	82	85	–	–	–	–	–
Average daily gain ($kg \cdot animal \cdot day^{-1}$) [a]	–	–	–	0.4	0.7	0.4	0.7	1.4
Dry matter intake ($kg \cdot animal \cdot day^{-1}$) [b]	9.6	10.5	11.3	9.9	7.9	12	8.4	13.2
Days to achieve final weight [a,c]	–	–	–	486	285	366	214	102
Area under grazing (%) [a]	100	100	100	100	96	100	95	0
Stocking rate $LU \cdot ha^{-1}$ [b]	0.4	0.6	0.6	0.8	1.2	0.8	1.8	3.0
Live weight productivity ($kg \cdot ha^{-1}$) [b]	73	89	125	162	240	124	264	443

LW: Live weight, LU: Livestock units.
[a] Information provided by farmers.
[b] Estimated using coefficients from Mieres (2004).
[c] Time to go from 150 to 350 kg in backgrounding and from 350 to 500 in finishing systems.

for crop and forage production (Table 3). These were the activity data inputs for calculating environmental impacts.

Beef LW productivity ($kg \cdot ha^{-1}$) was calculated for each of this fifteen combinations as follows. In cow–calf systems, total LW production (sum of LW kg sold minus bought, plus the difference in kg stocks, minus kg consumed on farm) was divided by the total farm area dedicated to raise cattle. In the finishing systems, the difference between final and initial LW was the LW gain (production), and then divided by the area needed to feed the animals. This total area was calculated from the nutritional requirements of the animals (NRC, 2000) to achieve the LW gain of each system and the average yield productivity for each feed source (Table 3).

2.2. Environmental impact assessment

2.2.1. Life cycle assessment (LCA) methodology

We used a partial LCA methodology to study some of the environmental impacts of the beef production systems, including farm activities and the production of farm inputs. All impacts were related to a functional unit of 1 kg of animal live weight (LW) and summarized into environmental impact categories (Brentrup, Küsters, Lammel, Barraclough, & Kuhlmann, 2004). Based on the description of each system, we conducted an inventory analysis compiling all resources needed for and all emissions released by each system and related them to the defined functional unit (ISO (International Organization for Standardization), 1998). Then, inventory data were multiplied by characterization factors (CF) to give indicators for the different environmental impact categories (e.g., climate change, resource depletion, ecotoxicity, eutrophication, biodiversity). A detailed description of the estimation of the impacts for each category follows.

In order to integrate the information of the different environmental categories, a summary of environmental impact index was calculated, which is an optional step in the LCA methodology. In order to summarize the information of variables measured in different units, a standardization of each variable was performed (i.e., value for each system minus the mean for the fifteen systems, divided by the standard deviation). We did not perform a normalization or weighing as recommended by LCA for lack of reference data for the region (Brentrup et al., 2004).

2.2.2. Greenhouse gases (climate change)

Methane (CH_4), nitrous oxide (N_2O), and carbon dioxide (CO_2) were accounted in calculating GHG emissions based on Intergovernmental Panel on Climate Change (IPCC) equations from Chapter 10 — Emissions from Livestock and Manure Management (IPCC, 2006). Detailed assumptions for calculations are described in previously published papers (Becoña et al., 2014; Modernel et al., 2013). The coefficients and emission factors used in equations for enteric fermentation, manure management (CH_4), and production and distribution of animal feed

Table 3
Yield, inputs and nutritional characteristics of fodder used on nutritional calculations of the 3 cow–calf and 5 finishing systems of Uruguay.
Adapted from Becoña et al. (2014) for cow calf systems and Modernel et al. (2013) for finishing systems.

	Yield	Dig. OM	CP	ME	N fertilizer ($kg\ N \cdot ha^{-1}$)	Diesel ($L \cdot ha^{-1}$)	Pesticides ($kg \cdot ha^{-1}$)
	$Mg\ DM \cdot ha^{-1}$	(%)	(%)	$Mcal \cdot kg^{-1}$ DM			
Native grasslands	5.5	55	9–10	2.1	0	0	0.0
Improved grasslands	5.0	60	13	2.2	0	12	0.4
Seeded pastures	7.5	63–67	15–18	2.4	9	12	5.0
Annual fodder crops (oats)	3.4	65	16	2.4	92	37	6.0
Sorghum grain	4.1	85	8.6	3.3	67	30	10.7
Rice bran	1.5	44	10	2.2	46	29	6.0
Rice husk	0.7	73	15	2.2			

ME: Metabolic energy concentration, Dig. OM: Organic matter digestibility, CP: crude protein.

(N_2O and CO_2) are presented in Table 4. Methane emissions from enteric fermentation were estimated based on the gross energy consumed per day, and the diet digestibility and crude protein. Feedlot systems use diesel combustion for feed distribution. A tier 2 IPCC (2006) approach and information from MIEM (2010) were used in calculating diesel and glyphosate emission factors. Methodology described by Spielmann, Dones, and Bauer (2007) and 9.0×10^{-3} kg CO_2-eq. Mg·km^{-1} from Ledgard (com. pers.) was used to calculate emission factors of extraction of raw materials, manufacture, and transport of fertilizers and pesticides. Means of transportation and amounts imported in the last 5 years were obtained from MGAP (2013).

Greenhouse gas emissions were expressed in kg of CO_2 equivalent per kg of live weight (LW) produced in the whole meat production process. Global warming potential was 1 for CO_2, 25 for CH_4 and 298 for N_2O for 100 years. Emissions were also weighted by the global temperature change potentials (GTP), following Reisinger and Ledgard (2013) with values of 1, 4, and 265 for CO_2, CH_4 and N_2O respectively for 100 years (Myhre et al., 2014). Carbon stock in soil was assumed to remain constant as recommended by IPCC (2006). However, sensitivity to the C sequestration by soils was explored as described in detail in the Discussion section.

2.2.3. Energy consumption and soil erosion (resource depletion)

Fossil fuel energy consumption was estimated using the coefficients provided by local databases and farm-specific data, using the model Agroenergia described by Llanos, Astigarraga, Jacques, and Picasso (2013). The energy use per kg of LW was estimated over one growing cycle and expressed in MJ·kg LW^{-1}. Energy consumption for inputs was 39, 280, and 33 MJ·kg^{-1} for diesel, pesticides, and fertilizers, respectively (Modernel et al., 2013).

Soil erosion rates for each system were estimated using EROSION 5.0 (Garcia Prechac, Clerici, Hill, & Brignoni, 2005), a model based on the Universal Soil Loss Equation and its later Revision (USLE/RUSLE) adapted and calibrated for Uruguay soils. Homogeneous soil types (Brunosol subeutrico típico — Typic Argiudolls), length (100 m) and inclination (3%) slopes where considered, in order to make the systems comparable. Erosion rates where expressed in kg soil·kg LW^{-1} (Modernel et al., 2013).

Table 4
Coefficients and emission factors to calculate GHG emissions of beef systems in Uruguay. Modernel et al. (2013).

	Coefficient/emission factor	Source
CH_4		
Ym (feedlot) (% GE)	3.0	IPCC (2006) Table 10.3
Ym (grazing) (% GE)	6.5	
Bo (m^3 CH_4.kg of VS^{-1})	0.1	IPCC (2006) Table 10A-5
MCF (feedlot)	32	IPCC (2006) Table 10A-5
MCF (grazing)	1.5	
N_2O		
EF_3 (kg N_2O-N)	0.02	IPCC (2006) Table 11.1
EF_4 (kg N_2O-N)	0.01	IPCC (2006) Table 11.3
EF_5 (kg N_2O-N)	0.0075	IPCC (2006) Tables 10.5 and 11.3
CO_2		
EFc (kg CO_2-eq. kg gas oil^{-1})	2.9	IPCC (2006) and MIEM (2010)
Input emission factors		
Herbicides (CO_2 eq. L^{-1}) [a]	18.3	Spielmann et al. (2007),
Insecticides (CO_2 eq. L^{-1}) [a]	14.8	Ledgard, S. (com. pers.),
Seeds (kg CO_2·kg^{-1}) [a]	0.2	MGAP (2013) and
Fertilizers (kg CO_2.kg^{-1})[a]	0.4	Carámbula (1981)

Y_m: Conversion methane factor (% of gross energy lost as methane); GE: Gross energy intake (MJ·day^{-1}); Bo: Maximum methane producing capacity for manure produced by livestock category (m^3·CH_4 kg of VS excreted^{-1}); VS: Excreted volatile solids (kg MS·animal·day^{-1}); MCF: Methane conversion factors for manure management system in the climate region; EF_3: Emission factor according to the manure management and region; EF_4: Emission factor according to manure management system; EF_5: Emission factor according to manure management system; EFc: Fuel Factor emission (gas–oil) (2.98 kg CO_2 eq. kg fuel^{-1}).
[a] Average value for each category.

2.2.4. Nutrient imbalances (eutrophication)

Nitrogen (N) and Phosphorus (P) nutrient imbalance ratio (NIR) was calculated for each system using the methodology proposed by Koelsch & Lesoing (1999) and adapted by Modernel et al. (2013). This indicator is a ratio between nutrient outputs and inputs used in the system. Values over one represent nutrient surpluses (higher surpluses represent higher risks of water eutrophication) and values less than one represent that nutrients are being exported at a higher rate than incorporated in the system. Nitrogen inputs were accounted for fertilizers and biological fixation from legumes while the animal LW gained in the period was considered as output. Nitrogen content in animals LW was estimated with an equation that considers empty LW and the nitrogen–protein ratio while phosphorus was estimated as 0.69% of LW (NRC, 2000).

2.2.5. Pesticide ecotoxicity

Pesticide ecotoxicity was calculated as the standard LCA method using USEtox (www.usetox.org), a method which calculates characterization factors for human toxicity and freshwater ecotoxicity (Rosenbaum et al., 2008). Characterization factors for each pesticide were obtained by multiplication of scale-specific fate factors, exposure factors and effect factors, added among different compartments (continental and global). The characterization factor for aquatic ecotoxicity (ecotoxicity potential) provided an estimate of the potentially affected fraction of species (PAF) integrated over time and volume per unit mass of each pesticide used (PAF m^3 day kg^{-1}). Because of lack of available data in USEtox, we were not able to calculate the human toxicity factors, and we report results only for ecotoxicity. Total use of each pesticide for the production of a kg of animal LW was multiplied by the corresponding ecotoxicity characterization factor and added to calculate the total pesticide ecotoxicity per kg of LW.

2.2.6. Land use impact on biodiversity

To evaluate the impact of beef production on biodiversity, we used a modified version of the grassland conservation index (Viglizzo, 2012), a recently developed metric to evaluate the condition of the natural grassland at farm level. For this study we only had data available of the different vegetation types at the production system (land use), so we only calculated the agrobiodiversity component (ABD) as the sum of the proportion of farm area on each land use multiplied by a relative biodiversity value of each land use (1 for natural grasslands, 0.9 for seeded pastures, 0.6 for annual fodder crops, and 0.5 for grain crops). These values are actually very similar to the ones reported in other LCA studies (e.g., species richness characterization factors for biodiversity from Souza et al., 2013, and naturalness degradation potential from Brentrup et al., 2004). We defined the impact on biodiversity index (IBI) as 1 — ABD, so that a larger value of the index means a larger land use impact on natural grassland biodiversity.

2.3. Statistical analysis

In order to identify differences in GHG emissions among beef production phases for the different metrics, an analysis of variance was performed with the three cow–calf systems and the five finishing systems. A simple linear regression model for various metrics of GHG emissions was fitted using live weight productivity as the independent variable, in order to explore the impact of increasing productivity on the various metrics of GHG emissions. A principal component multivariate analysis and a pairwise scatterplot and correlation matrix were constructed to describe the relationship among all environmental variables, with the main purpose of identifying trade-offs or linear associations among variables. All analyses were conducted in Infostat software (Di Rienzo et al., 2011) and figures were created in R (R Core Development Team, 2008).

3. Results

3.1. Greenhouse gas emissions per kg of live weight

Greenhouse gas emissions were on average 21.9 and 11.3 kg CO_{2e}·kg LW^{-1} for cow–calf and finishing systems respectively, using the GWP metrics. The GHG emissions per kg of LW in cow–calf phase were significantly larger than those of finishing phase ($P = 0.02$). The average GHG emissions were reduced to 7.3 and 4.9 kg CO_{2e}·kg LW^{-1} when using the GTP metrics, and there were no significant differences between the two phases ($P = 0.06$). Using GWP metrics, the average relative contribution of methane was 76% and nitrous oxide was 24% in cow–calf systems. These values switched to 36% for methane and 73% for nitrous oxide when using the GTP metrics. For the finishing systems, using GWP methane accounted for 61% and nitrous oxide for 34%, changing to 23% and 68% when using GTP (Fig. 1).

Greenhouse gas emissions ranged from 9.7 to 20.3 kg CO_{2e}·kg LW^{-1} in the fifteen combinations of cow–calf and backgrounding–finishing, using the GWP metrics. These values were reduced to 40% on average when using the GTP metrics (Fig. 2). The differences in greenhouse gas emissions among the systems were reduced too. Furthermore some of the rankings between the systems changed. For instance, the backgrounding–finishing system on seeded pasture emitted more GHG than the one on grasslands–feedlot, when using the GTP metrics.

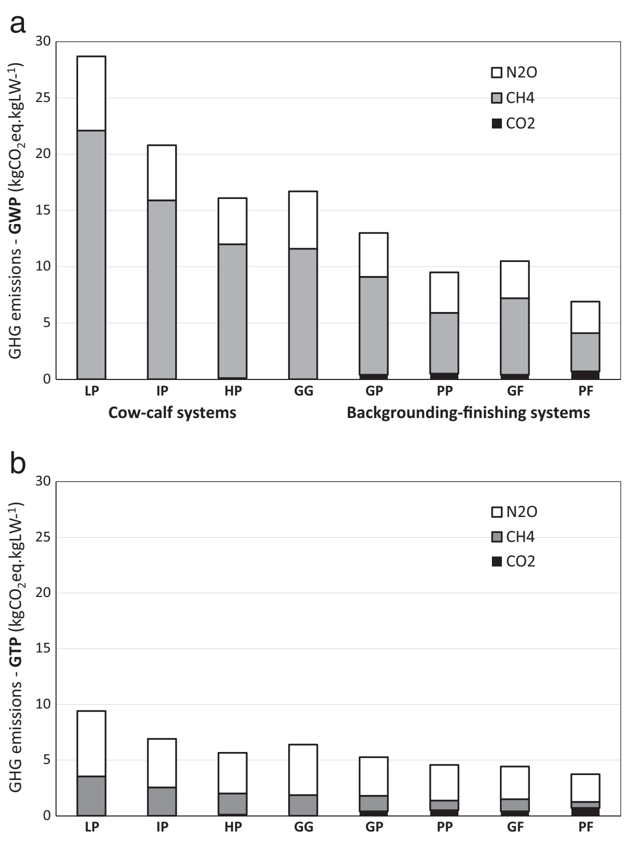

Fig. 1. Greenhouse gas emissions (CO_2, CH_4, N_2O) per kg of live weight produced in three cow–calf systems (LP = low performance, IP = intermediate performance, HP = high performance) and five finishing systems (GG = grasslands–grasslands, GP = grasslands–pasture, PP = pasture–pasture, GF = grasslands–feedlot, PF = pasture–feedlot) in Uruguay, using two alternative metrics: a) global warming potential (GWP) and b) global total potential (GTP).

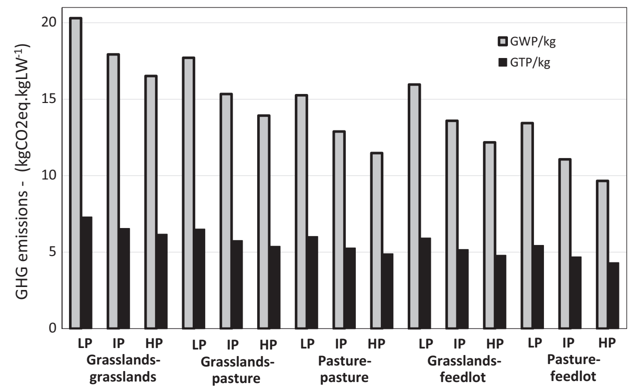

Fig. 2. Greenhouse gas emissions per kg of live weight produced in fifteen combinations of three cow–calf systems (LP = low performance, IP = intermediate performance, HP = high performance) and five finishing systems in Uruguay, using two alternative metrics: Global warming potential (GWP, gray bars) and global total potential (GTP, black bars).

3.2. Greenhouse gas emissions per hectare

Carbon footprint, i.e., the total amount of greenhouse gas emissions throughout the life cycle of a product is usually expressed from the standpoint of the consumer, as kg of CO_2 equivalent per kg of a product. However, it can also be expressed from the standpoint of the producer, as kg of CO_2 equivalent per unit of area (ha) of the production system (Reisinger & Ledgard, 2013). In this study we considered the variation both in cow–calf systems and backgrounding–finishing systems, and we found contradictory results depending on the metrics used. When using GWP, as LW productivity increased, both GHG emissions per kg and per ha were reduced. However, when using GTP, as LW productivity increased, GHG emissions per kg were reduced, but no significant trend was identified for GHG emissions per ha (Fig. 3).

3.3. Other environmental impacts and trade-offs

Trade-offs between environmental variables was explored by two methods. First, a principal component multivariate analysis was carried out with all fifteen systems and nine variables. The first two principal components explained 92.5% of the total variation. The first axis was positively associated with LW productivity, nutrient balances, energy consumption, and pesticide ecotoxicity, while negatively associated with

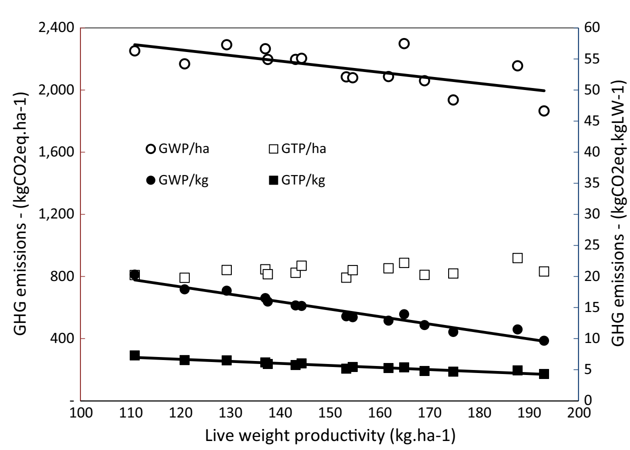

Fig. 3. Greenhouse gas emissions per kg of LW produced (closed markers) and per hectare of land (open markers) as a function of LW productivity (LWP, kg LW per ha) in fifteen combinations of three cow–calf systems and five finishing systems in Uruguay, using two alternative metrics: Global warming potential (GWP, circles) and b) global total potential (GTP, squares). Predicted values of significant linear regressions are represented with a line. GWP/kg = 32.81–0.12 LWP, adjusted $R^2 = 94\%$, $P < 0.0001$; GTP/kg = 11.03–0.04 LWP, adjusted $R^2 = 85\%$, $P < 0.0001$; GWP/ha = 2690.97–3.60 LWP, adjusted $R^2 = 42\%$, $P = 0.0056$; GTP/ha = 737.69 + 0.65 LWP, adjusted $R^2 = 13\%$, $P = 0.0994$.

GHG emissions per kg LW. The second axis was positively associated with soil erosion, energy use and pesticide ecotoxicity, while negatively associated with impact on biodiversity, P balance, and LW productivity.

Second, a scatterplot matrix representing pairwise scatterplots and correlation coefficients was calculated for all variables (Fig. 4). Live weight productivity was negatively associated with GHG emissions per kg, while positively associated with all other environmental impacts (except soil erosion). GHG emissions per kg were negatively associated with all other variables (except again soil erosion). It is interesting to highlight that GHG emissions per kg were negatively associated with fossil energy consumption, and with inputs use (N and P balances, and pesticide risk). Energy consumption, N and P balances, pesticide risk index, and soil erosion were all positively correlated among them. Impact on biodiversity was positively correlated to the latter variables, except soil erosion.

4. Discussion

4.1. Limitations of carbon footprint as environmental impact indicator

The carbon footprint is a useful indicator for evaluating the environmental impact of beef production on climate change. It provides guidance in order to identify systems, technologies, or processes that livestock production can improve in order to mitigate this impact. Estimations of GHG emissions have shown that improving production efficiency is highly relevant in order to reduce their emissions. Furthermore, carbon footprint has brought to the same table farmers, industry, consumers, policy makers, and researchers, to work together to address one of the most challenging problems humanity is facing.

However, the use of carbon footprint for evaluating environmental impact of beef production has some serious limitations: 1) different metrics (GWP vs. GTP) give different results and may lead to different recommendations, 2) carbon sequestration from soils is not always accounted for, and 3) significant trade-offs exist between carbon footprint and other relevant environmental variables.

First, there are different metrics (i.e., GWP, GTP) to account for the relative impact of the different GHG emissions that change significantly the absolute values of the GHG emissions, and in some cases, may change the relative ranking of alternative systems and management practices, giving contradictory results. The global warming potential (GWP) is an index of the total energy added to the climate system by a gas relative to that added by CO_2. However, the GWP does not lead to equivalence with temperature or other climate variables. The GWP for a time horizon of 100 years was adopted as a metric to implement the multi-gas approach embedded in the United Nations Framework Convention on Climate Change (UNFCCC) and was made operational in the 1997 Kyoto Protocol (Myhre et al., 2014). The global temperature change potential (GTP) is closer than GWP to the effects on temperature and is defined as the change in global mean surface temperature at a chosen point in time in response to an emission pulse, relative to that of CO_2 (Myhre et al., 2014). Both GWP and GTP are strongly affected by the choice of time horizon, but GTP has a larger uncertainty. The anthropogenic emissions of CH_4 rank second after the anthropogenic emissions of CO_2 in terms of potential impact on the global Earth radiative budget. They both produce an important greenhouse effect, but CH_4 has a much higher radiative efficiency and a much shorter life-time in the atmosphere than CO_2. As a consequence, the climatic impact of the relative emissions and emission reductions of these two gases is very

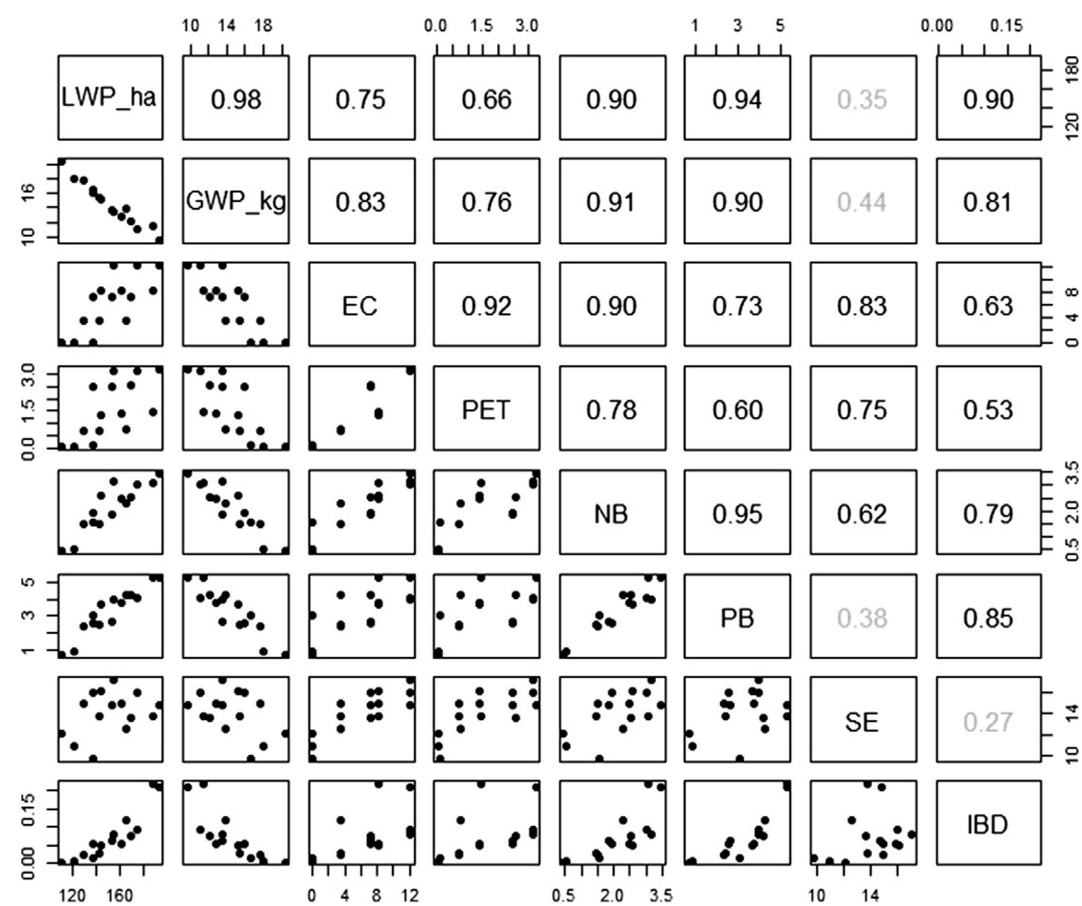

Fig. 4. Scatterplot matrix representing pairwise scatterplots (below diagonal) and pairwise Pearson correlation coefficients (r, above diagonal) for eight variables estimated for fifteen beef systems in Uruguay. Correlation coefficients in bold are significantly different from zero ($P < 0.05$). LWP/ha = live weight productivity ($kg \cdot ha^{-1}$); GWP/kg = GHG emissions using GWP metrics ($kg\ CO_{2e} \cdot kg\ LW^{-1}$); EC = Energy consumption ($MJ \cdot kg\ LW^{-1}$); PET = Pesticide Ecotoxicity; NB = Nitrogen Balance; PB = Phosphorus Balance; SE = Soil Erosion ($kg\ soil \cdot kg\ LW^{-1}$); IBD = Impact on Biodiversity.

different, depending on the period of time considered for its assessment (Myhre et al., 2014). Several studies highlight the importance of moving beyond GWP based emission equivalence (Lauder et al., 2013; Shine, Fuglestvedt, Hailemariam, & Stuber, 2005). The main complicating feature in considering trade-offs between methane and most other greenhouse gases is the short atmospheric lifetime, compared to the timescales that dominate the response of CO_2. The GWP and GTP are fundamentally different by construction and different numerical values can be expected. In particular, the GWPs for short-term life gases as methane, over the same time frames, are higher than GTPs due to the integrative nature of the metric. The choice of metric type and time horizon will for many components have a much larger effect than improved estimates of input parameters and can have strong effects on perceived impacts of emissions and abatement strategies (Myhre et al., 2014), and at regional and national scales, particularly for countries that may face binding economy-wide emission targets under future agreements, with large fractions of non-CO_2 emissions in their national inventories (Reisinger & Ledgard, 2013).

Second, it is still debated how much carbon under grasslands and pastures can be sequestered, and this also has a major impact on GHG balance (Soussana, Tallec, & Blanfort, 2010). Carbon sequestration in soils may have a large range of variation (Jones & Donnelly, 2004). According to Lal (2004), it is reasonable to expect 200 to 600 kg $C\cdot ha^{-1}\cdot yr^{-1}$ of C sequestration in soils under temperate humid climate by improving grazing management, and the same can be expected for crop rotations under no tillage. Salvo (2014) found for no tillage crop pasture rotations in Uruguay values of C sequestration of 391 to 445 kg $C\cdot ha^{-1}\cdot yr^{-1}$ after 8 years. For the fifteen beef systems considered in this study, a sensitivity analysis was conducted to the rate of C sequestration in soils, ranging from 100 to 600 $C\cdot ha^{-1}\cdot yr^{-1}$ (equivalent to 367 to 2200 $CO_2\cdot ha^{-1}\cdot yr^{-1}$). Reduction in GHG emissions assuming 100 $C\cdot ha^{-1}\cdot yr^{-1}$ of carbon sequestration was 17% and 44% using the GWP and the GTP metrics respectively in average for all systems. Furthermore, the reduction in GHG emissions assuming 600 $C\cdot ha^{-1}\cdot yr^{-1}$ of carbon sequestration was 103% and 263% using the GWP and the GTP metrics respectively. This means that in feasible scenarios where C sequestration in soils is high, the net C balance may be negative, and the livestock systems may become net sinks of GHG. The type of pasture, species composition, grazing intensity and management, and other factors affect this balance. Estimations of carbon footprint should account for the sensitivity to carbon sequestration in soils, given the high impact that this variable has on the GHG balance. Depending on the system, and the metric used, livestock grazing systems could be net sinks or net sources. This is a major point and should not be overlooked when assessing the contribution of livestock systems to climate change.

Finally, and most importantly, there are significant trade-offs between carbon footprint and other relevant environmental variables. This is the main reason why carbon footprint should not be used alone for environmental assessment. And this point has three major dimensions. One is the interesting trade-off between GHG emissions and fossil energy consumption in the livestock systems analyzed in this paper. The scientific consensus from IPCC is that the increased greenhouse effect is due to the burning of fossil fuels, which started after the industrial revolution and continues until today (Solomon, 2007). It is paradoxical that reducing carbon footprint in livestock systems could result in increasing fossil fuels use by these systems. A second dimension is the trade-off between global and local environmental issues (Modernel et al., 2013). Climate change is a global problem, while soil erosion, pesticide toxicity, and water eutrophication by nutrients are local problems, which must be assessed and addressed by local decision makers. The third dimension is the trade-off between carbon footprint and biodiversity. In the livestock systems analyzed in this paper, reducing carbon footprint is associated with increasing the impact on native grassland biodiversity. Climate change poses serious threats to humanity and biodiversity. Mitigation of climate change should not be associated with directly reducing biodiversity habitat.

4.2. Multidimensional environmental impact assessment

All the reasons listed above suggest that a broad set of indicators is needed to inform decision makers. Sustainability science literature has contributed in the last decades with conceptual frameworks and methods to design and assess multiple attributes and indicators (López-Ridaura, Keulen, Ittersum, & Leffelaar, 2005; Sarandón & Flores, 2009; Speelman, López-Ridaura, Colomer, Astier, & Masera, 2007). Some examples are discussed below.

An amoeba (or radar) plot is presented in Fig. 5, showing standardized values for the beef production cycles with the five different finishing systems averaged over the three cow–calf systems. The grasslands–grasslands system has maximum GHG emissions, and minimum values for all other impacts. On the opposite side, the pasture–feedlot system has minimum GHG emissions and maximum values for all other variables. The other systems are intermediate situations. This plot has the advantage that shows the information of all impacts, and allows for identifying tradeoffs among variables and which variables have more impact on each system. It is most informative. It does not give a summary metric for the overall environmental impact, though.

The environmental impact index calculated as the average of all standardized variables suggested that the systems have significantly different environmental impacts, and that grasslands systems have less impact than seeded pastures systems and all these grazing systems have less impact than the feedlots (Table 5). It suggested also that grazing systems have significantly less impact than those with feedlots. This is consistent with the amoeba plot previously shown. However, normalization and alternative weighing of variables may change these results.

The choice of method of assessment, variables, normalization and weighing may change the conclusions of the analysis. Stakeholder participation and interdisciplinary teams are two fundamental elements of the process in order to succeed. Furthermore, the definition of thresholds and reference values is essential. The LCA methodology has standardized methods and reference values for certain regions (e.g., Europe). It is important to develop regional databases to be able to apply this methodology in full.

In order to provide a detailed analysis of the empirical examples of the issues discussed above, in this paper the environmental assessment was not complete. Other relevant environmental variables and attributes were left behind, although there are studies from Uruguay documenting some of them. The water footprint, for instance, is a key variable that should be included in further studies (Ran, Deutsch, Lannerstad, & Heinke, 2013). The resilience of the livestock systems to extreme climatic events (e.g., drought) is an increasingly relevant attribute of sustainability and metrics for quantitatively assessing this are being developed (Picasso et al., 2013). The impacts of the animal nutrition on the quality and health attributes (fatty acids and mineral composition) of the beef produced have also been documented (Saadoun & Cabrera, 2012). Finally, a complete sustainability assessment must also integrate social and economic indicators to the environmental ones.

4.3. The potential of grazing systems in the region

The objectives of this paper were to describe some environmental variables associated with real beef production systems, though the upper bounds of productivity of grazing systems in the region are not included in the range of systems described here. In the region it has been proved that increasing the meat productivity and conserving natural resources are complementary objectives (Carvalho & Nabinger, 2009). Long term experiments on grazing management have shown that it is possible to triplicate meat productivity of current farming systems based on natural grasslands by moderate forage allowance, without using external inputs or replacing the natural grasslands. Carvalho and Nabinger (2009) reported animal productivity up to 230 kg $LW\cdot ha^{-1}$ in livestock grazing systems with controlled grazing intensity (forage allowance of 8% in spring and 12% the rest of the

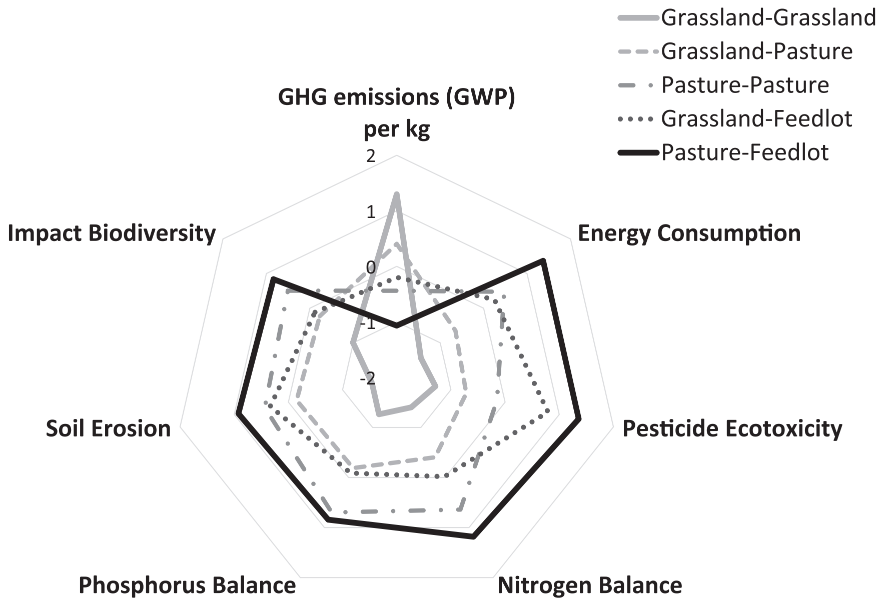

Fig. 5. Amoeba plot for standardized environmental impact variables for beef production cycles with five different finishing systems in Uruguay (each value is the average over three cow–calf systems): GG = grasslands–grasslands, GP = grasslands–pasture, PP = pasture–pasture, GF = grasslands–feedlot, PF = pasture–feedlot.

year) in southern Brazil. They also reported up to 1000 kg LW·ha^{-1} in livestock grazing systems with improved grazing management and the addition of fertilizer and temperate grasses.

In Uruguay, long term grazing experiments with cow–calf systems controlling the grazing intensity (10% forage allowance) also found significant increases in productivity up to 160 kg LW·ha^{-1} (Carriquiry et al., 2012; Soca, Espasandín, & Carriquiry, 2013). This strategy follows the new paradigm of ecological intensification by improving the performance of current faming systems on yield, income and ecosystem services (Bommarco, Kleijn, & Potts, 2012), achieving a triple win: adaptation of farming systems, enhancement of food security and mitigation of climate change. This framework is being followed in Uruguay and the region, with public policies oriented to preserve the natural grasslands and the adaptation of these systems to climate change, as well as NGO's studying and promoting incentives to preserve the natural grasslands in relation with livestock production systems (Bilenca & Miñarro, 2004; Parera, Paullier, & Bosso, 2012).

These examples suggest that there is great potential to improve the productivity of grazing livestock systems in the region, without the addition of inputs, but just improving grazing management. These improvements have the potential to simultaneously increase productivity, reduce GHG emissions, and the other environmental impacts, while conserving biodiversity (Carvalho, Nabinger, Lemaire, & Genro, 2009).

Table 5
Environmental impact index for beef production cycles with five different backgrounding–finishing systems in Uruguay.

	Environmental Impact Index	
Grasslands–grasslands	−0.95	a
Grasslands–pasture	−0.28	b
Pasture–pasture	0.31	bc
Grasslands–feedlot	0.14	c
Pasture–feedlot	0.78	d
P grassland/pasture vs. feedlot	0.0129	

Larger values indicate greater environmental impact. Values with the same letter are not significantly different (Tukey, P = 0.05). The P value for the contrast between systems with feedlot finishing vs. grazing finishing is shown.

5. Conclusion

The use of carbon footprint as an indicator for evaluating environmental impact of beef grazing systems has several serious limitations. The choice of GHG emission metric can have strong effects on perceived impacts of emissions and mitigation strategies. Beef systems with grazing finishing have greater GHG emissions than feedlot finishing when GWP is used as metric, but the difference becomes minimal when the GTP is used instead. Estimations of carbon footprint should account for the sensitivity to carbon sequestration in soils, because C sequestration may change the GHG balance from net sources to sinks. There are significant trade-offs between global and local environmental impacts: climate change is a global problem, while soil erosion, pesticide ecotoxicity, water eutrophication by nutrients, and grassland biodiversity loss are local problems, which must be assessed and addressed by local decision makers. Beef systems with grazing finishing have lower impact on all those variables than feedlot systems. Therefore multiple metrics to assess environmental impacts and benefits of livestock grazing systems are more suitable. The examples described here from grazing systems in Uruguay suggest that there is great potential to improve the productivity of grazing livestock systems, by improving grazing management, and at the same time reducing GHG emissions, and other environmental impacts, while conserving biodiversity.

Acknowledgments

The authors would like to acknowledge the funding from CSIC-UDELAR (grant # Calidad 2012 Sustentabilidad), and the helpful comments of two anonymous reviewers who improved significantly the quality of this paper.

References

Beauchemin, K. A., Henry Janzen, H., Little, S. M., McAllister, T. A., & McGinn, S. M. (2010). Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. *Agricultural Systems*, *103*(6), 371–379. http://dx.doi.org/10.1016/j.agsy.2010.03.008.

Becoña, G., Astigarraga, L., & Picasso, V. D. (2014). Greenhouse gas emissions of beef cow–calf grazing systems in Uruguay. *Sustainable Agriculture Research*, *3*(2), 89–105. http://dx.doi.org/10.5539/sar.v3n2p89.

Bilenca, D., & Miñarro, F. (2004). Identificación de áreas valiosas del pastizal en las Pampas y Campos de Argentina, Uruguay y sur de Brasil. In D. Bilenca, & F. Miñarro (Eds.), Buenos Aires: Fundación Vida Silvestre Argentina.

Bommarco, R., Kleijn, D., & Potts, S. G. (2012). Ecological intensification: Harnessing ecosystem services for food security. *Trends in Ecology & Evolution*, *28*(4), 230–238. http://dx.doi.org/10.1016/j.tree.2012.10.012.

Brentrup, F., Küsters, J., Lammel, J., Barraclough, P., & Kuhlmann, H. (2004). Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. *European Journal of Agronomy*, *20*(3), 265–279. http://dx.doi.org/10.1016/S1161-0301(03)00039-X.

Carámbula, M. (2009). Producción de semillas de plantas forrajeras. In Montevideo (Ed.), *Hemisferio Sur*.

Carriquiry, M., Espasandin, A., Astessiano, A., Casal, A., Claramunt, M., Do Carmo, M., & Soca, P. (2012). *La cría vacuna sobre campo nativo: Un enfoque de investigación jerárquico para mejorar su productividad y sostenibilidad, 48. (pp. 41–48). Montevideo: Veterinaria, 41–48*.

Carvalho, P. C. D. F., & Batello, C. (2009). Access to land, livestock production and ecosystem conservation in the Brazilian Campos biome: The natural grasslands dilema. *Livestock Science*, *120*(1–2), 158–162. http://dx.doi.org/10.1016/j.livsci.2008.04.012.

Carvalho, P., & Nabinger, C. (2009). Ecofisiología de Sistemas Pastoriles: Aplicaciones para su Sustentabilidad. *Agrociencia*, *XIII*(3), 18–27.

Carvalho, P. C. F., Nabinger, C., Lemaire, G., & Genro, T. C. M. C. M. (2009). Challenges and opportunities for livestock production in natural pastures: The case of Brazilian Pampa Biome. In R. Feldman, G. E. Oliva, & M. B. Sacido (Eds.), *Proceedings of the IX International rangeland congress: Diverse rangelands for a sustainable society* (pp. 9–15). Rosario: Universidad de Rosario.

De Vries, M., & de Boer, I. J. M. (2010). Comparing environmental impacts for livestock products: A review of life cycle assessments. *Livestock Science*, *128*(1–3), 1–11. http://dx.doi.org/10.1016/j.livsci.2009.11.007.

Di Rienzo, J. A., Casanoves, F., Balzarini, M. G., Gonzalez, L., Tablada, M., & Robledo, C. W. (2011). *InfoStat. Córdoba, Argentina: Universidad Nacional de Córdoba.* (Retrieved from http://www.infostat.com.ar/).

DIEA (2013). *Anuario estadístico agropecuario 2013.* , 270 Ministerio de Ganadería Agricultura y Pesca (MGAP); (Montevideo,Retrieved from http://www.mgap.gub.uy/portal/hgxpp001.aspx?7,5,754,O,S,0,MNU;E;2;16;10;9;MNU).

FAO (2013). *FAOSTAT.* (Retrieved from http://faostat3.fao.org/faostat-gateway/go/to/home/E on April 12 2014).

Garcia Prechac, F., Clerici, C., Hill, M., & Brignoni, A. (2005). EROSION Version 5.0, Programa de computacion Para el uso de la USLE/RUSLE en la Region Sur de la Cuenca del Plata. *Version operativa en Windows DINAMA-UNDP, Proyecto URU/03/G31 y CSIC-UDELAR*. Montevideo, Uruguay: Facultad de Agronomía, UDELAR (Retrieved from www.fagro.edu. uy/?manejo/).

Gerber, P. J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A., & Tempio, G. (2013). *Tackling climate change through livestock — A global assessment of emissions and mitigation opportunities*. Rome: FAO, 115 (Retrieved from http://www.fao.org/docrep/018/i3437e/i3437e.pdf).

Herrero, M., Havlík, P., Valin, H., Notenbaert, A., Rufino, M. C., Thornton, P. K., Blümmel, M., Weiss, F., Grace, D., & Obersteiner, M. (2013). Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proceedings of the National Academy of Sciences of the United States of America*, *110*. (pp. 20888–20893). 52. http://dx.doi.org/10.1073/pnas.1308149110.

IPCC (2006). IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. In H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe (Eds.), Japan: (Hayama: IGES).

ISO (International Organization for Standardization) (1998). Environmental management—Life cycle assessment—Goal and scope definition and life cycle inventory analysis. *International Standard ISO 14041*. Geneva: ISO.

Jones, M. B., & Donnelly, A. (2004). Carbon sequestration in temperate grassland ecosystems and the influence of management, climate and elevated CO_2. *New Phytologist*, *164*(3), 423–439. http://dx.doi.org/10.1111/j.1469-8137.2004.01201.x.

Koelsch, R., & Lesoing, G. (1999). Nutrient balance on Nebraska livestock confinement systems. *Journal of Animal Science*, *63–71*.

Lal, R. (2004). Soil carbon sequestration impacts on global climate change and food security. *Science*, *304*(5677), 1623–1627. http://dx.doi.org/10.1126/science.1097396.

Lauder, A.R., Enting, I. G., Carter, J. O., Clisby, N., Cowie, A. L., Henry, B. K., & Raupach, M. R. (2013). Offsetting methane emissions — An alternative to emission equivalence metrics. *International Journal of Greenhouse Gas Control*, *12*, 419–429.

Llanos, E., Astigarraga, L., Jacques, R., & Picasso, V. (2013). Eficiencia energética en sistemas lecheros del Uruguay. *Agrociencia Uruguay*, *17*(2), 99–109.

López-Ridaura, S., Keulen, H. V., Ittersum, M. K. V., & Leffelaar, P. A. (2005). Multiscale methodological framework to derive criteria and indicators for sustainability evaluation of peasant natural resource management systems. *Environment, Development and Sustainability*, *7*(1), 51–69. http://dx.doi.org/10.1007/s10668-003-6976-x.

MGAP (2013). *Ministerio de Ganadería, Agricultura y Pesca.* (Retrieved April 10, 2014, from http://www.mgap.gub.uy/).

MIEM (2010). *La matriz energética en Uruguay—2009.* (Retrieved April 10, 2014, from www.miem.gub.uy/web/energia).

Mieres, J. (2004). *Guía para alimentación de rumiantes. Serie Técnica*, *142*, La Estanzuela, Uruguay: INIA.

Modernel, P., Astigarraga, L., & Picasso, V. (2013). Global versus local environmental impacts of grazing and confined beef production systems. *Environmental Research Letters*, *8*(3), 035052. http://dx.doi.org/10.1088/1748-9326/8/3/035052.

Mvotma, D. (2010). *Inventario national de gases de efecto invernadero 2004.* Resumen ejecutivo. Cambridge University Press, 39 (Montevideo. Retrieved April 10, 2014, from http://www.mvotma.gub.uy/ciudadania/biblioteca/documentos-de-ambiente/item/10002742-inventario-nacional-de-emisiones-de-gases-de-efecto-invernadero-2004.html).

Myhre, G., Shindell, D., Bréon, F. -M., Collins, W., Fuglestvedt, J., Huang, J., & Zhang, H. (2014). Anthropogenic and natural radiative forcing. In T. F. Stocker, D. Qin, G. -K. Plattner, M. Tignor, S. K. Allen, J. Boschung, & P.M. Midgley (Eds.), *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 659–740). Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

NRC (2000). *Nutrient requirements of beef cattle: Seventh revised edition: Update 2000.* Washington, DC: National Academy Press, 249.

Parera, A., Paullier, I., & Bosso, A. (Eds.). (2012). *Incentivos para conservar los pastizales naturales del Cono Sur. Una oportunidad para gobiernos y productores rurales* (pp. 111). Montevideo: Aves Uruguay.

Pelletier, N., Pirog, R., & Rasmussen, R. (2010). Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. *Agricultural Systems*, *103*(6), 380–389. http://dx.doi.org/10.1016/j.agsy.2010.03.009.

Peters, G. M., Rowley, H. V., Wiedemann, S., Tucker, R., Short, M.D., & Schulz, M. (2010). Red meat production in Australia: Life cycle assessment and comparison with overseas studies. *Environmental Science & Technology*, *44*(4), 1327–1332. http://dx.doi.org/10.1021/es901131e.

Picasso, V., Lizarralde, C., Molina, C., Bartaburu, D., Morales, H., & Astigarraga, L. (2013). A quantitative assessment of robustness to drought in livestock grazing systems in Uruguay. *First Conference on Global Food Security*. The Netherlands: Noordwijkerhout.

R Core Development Team (2008). *R: A language and environment for statistical computing.* Vienna, Austria: R Foundation for Statistical Computing (Retrieved from http://www.r-project.org).

Ran, Y., Deutsch, L., Lannerstad, M., & Heinke, J. (2013). Rapidly intensified beef production in Uruguay: Impacts on water related ecosystem services. *Aquatic Procedia*, *1*, 77–87. http://dx.doi.org/10.1016/j.aqpro.2013.07.007.

Reisinger, A., & Ledgard, S. (2013). Impact of greenhouse gas metrics on the quantification of agricultural emissions and farm-scale mitigation strategies: A New Zealand case study. *Environmental Research Letters*, *8*(2), 25019. http://dx.doi.org/10.1088/1748-9326/8/2/025019.

Rosenbaum, R. K., Bachmann, T. M., Gold, L. S., Huijbregts, M.A. J., Jolliet, O., Juraske, R., Koehler, A., Larsen, H. F., MacLeod, M., Margni, M., McKone, T. E., Payet, J., Schuhmacher, M., Meent, D., & Hauschild, M. Z. (2008). USEtox—The UNEP-SETAC toxicity model: Recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. *The International Journal of Life Cycle Assessment*, *13*(7), 532–546. http://dx.doi.org/10.1007/s11367-008-0038-4.

Royo Pallarés, O., Berretta, E. J., & Maraschin, G. E. (2005). The South American Campos ecosystem. In J. M. Suttie, G. Reynolds, & C. Batello (Eds.), *Grasslands of the world* (pp. 171–219). Rome: FAO.

Ruviaro, C. F., de Léis, C. M., Lampert, V. D. N., Barcellos, J. O. J., & Dewes, H. (2014). Carbon footprint in different beef production systems on a southern Brazilian farm: A case study. *Journal of Cleaner Production.* http://dx.doi.org/10.1016/j.jclepro.2014.01.037.

Saadoun, A., & Cabrera, C. (2012). *Calidad nutricional de la carne bovina producida sobre pasturas en Uruguay.* , *1*. (pp. 97–101). Montevideo: Veterinaria, 97–101.

Salvo, L. (2014). *Fluxos e balanco de gases de efeito estufa em solo do Uruguai afetado por sistemas de manejo.* Universidade Federal Do Rio Grande Do Sul (Tese doutorado).

Sarandón, S. J., & Flores, C. C. (2009). Evaluación de la sustentabilidad en agroecosistemas: una propuesta metodológica. *Agroecologia*, *4*, 19–28.

Shine, K., Fuglestvedt, J., Hailemariam, K., & Stuber, N. (2005). Alternatives to the global warming potential for comparing climate impacts of emissions of greenhouse gases. *Climate Change*, *68*, 281–302.

Soca, P., Espasandín, A., & Carriquiry, M. (2013). *Efecto de la oferta de forraje y grupo genético de las vacas sobre la productividad y sostenibilidad de la cría vacuna en campo natural. Serie Tecn.* INIA.

Sollenberger, L. E., Moore, J. E., Allen, V. G., & Pedreira, C. G. S. (2005). Reporting forage allowance in grazing experiments. *Crop Sicence*, *45*(3), 896–900. http://dx.doi.org/10.2135/cropsci2004.0216.

Climate Change 2007—The Physical Science Basis. Solomon, S. (Ed.). (2007). *Working Group I Contribution to the Fourth Assessment Report of the IPCC*, *Vol. 4*. Cambridge University Press.

Soussana, J. F., Tallec, T., & Blanfort, V. (2010). Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. *Animal: An International Journal of Animal Bioscience*, *4*(3), 334–350. http://dx.doi.org/10.1017/S1751731109990784.

Souza, D.M., Flynn, D. F. B., DeClerck, F., Rosenbaum, R. K., Melo Lisboa, H., & Koellner, T. (2013). Land use impacts on biodiversity in LCA: Proposal of characterization factors based on functional diversity. *The International Journal of Life Cycle Assessment*, *18*(6), 1231–1242. http://dx.doi.org/10.1007/s11367-013-0578-0.

Speelman, E. N., López-Ridaura, S., Colomer, N. A., Astier, M., & Masera, O. R. (2007). Ten years of sustainability evaluation using the MESMIS framework: Lessons learned from its application in 28 Latin American case studies. *International Journal of Sustainable Development & World Ecology*, *14*(4), 345–361. http://dx.doi.org/10.1080/13504500709469735.

Spielmann, M., Dones, R., & Bauer, C. (2007). Life cycle inventories of transport services. *Final report ecoinvent data v2.0. 14. Dübendorf, Swiss Centre for LCI. Verified January 17, 2014* (From http://www.ecoinvent.org/support/old-doc/rep/reports-for-guests-users).

Viglizzo, E. F. (2012). Hacia un Indicador de Conservación de Pastizales (ICP). In A. Parera, I. Paullier, & A. Bosso (Eds.), *Incentivos para conservar los pastizales naturales del Cono Sur. Una oportunidad para gobiernos y productores rurales* (pp. 97–111). Montevideo: Aves Uruguay.