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Greenhouse gas balance and carbon footprint of pasture-based beef cattle production systems in the tropical region (Atlantic Forest biome)

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*The production of beef cattle in the Atlantic Forest biome mostly takes place in pastoral production systems. There are millions of hectares covered with pastures in this biome, including degraded pasture (**DP**), and only small area of the original Atlantic Forest has been preserved in tropics, implying that actions must be taken by the livestock sector to improve sustainability. Intensification makes it possible to produce the same amount, or more beef, in a smaller area; however, the environmental impacts must be assessed. Regarding climate change, the C dynamics is essential to define which beef cattle systems are sustainable. The objectives of this study were to investigate the C balance (t $CO_{2e.}$/ha per year), the intensity of C emission (kg $CO_{2e.}$/kg BW or carcass) and the C footprint (t $CO_{2e.}$/ha per year) of pasture-based beef cattle production systems, inside the farm gate and considering the inputs. The results were used to calculate the number of trees to be planted in beef cattle production systems to mitigate greenhouse gas (**GHG**) emissions. The GHG emission and C balance, for 2 years, were calculated based on the global warming potential (**GWP**) of AR4 and GWP of AR5. Forty-eight steers were allotted to four grazing systems: DP, irrigated high stocking rate pasture (IHS), rainfed high stocking rate pasture (RHS) and rainfed medium stocking rate pasture (RMS). The rainfed systems (RHS and RMS) presented the lowest C footprints (−1.22 and 0.45 t $CO_{2e.}$/ha per year, respectively), with C credits to RMS when using the GWP of AR4. The IHS system showed less favorable results for C footprint (−15.71 t $CO_{2e.}$/ha per year), but results were better when emissions were expressed in relation to the annual BW gain (−10.21 kg $CO_{2e.}$/kg BW) because of its higher yield. Although the DP system had an intermediate result for C footprint (−6.23 t $CO_{2e.}$/ha per year), the result was the worst (−30.21 $CO_{2e.}$/kg BW) when the index was expressed in relation to the annual BW gain, because in addition to GHG emissions from the animals in the system there were also losses in the annual rate of C sequestration. Notably, the intensification in pasture management had a land-saving effect (3.63 ha for IHS, 1.90 for RHS and 1.19 for RMS), contributing to the preservation of the tropical forest.*

Keywords: sustainability, soil carbon stock, emission intensity, land-saving effect, mitigation

Implications

Pasture degradation should be avoided because it causes low productivity and has a high environmental impact, especially related to the high carbon footprint of beef production in these pastures. It also results in a waste of land. Recovery and intensification of pasture-based beef cattle production systems simultaneously improve carbon sequestration and mitigate greenhouse gases emissions, in addition to having a land-saving effect. It also leads to reductions in carbon footprint per unit of product and the number of trees required for the abatement of greenhouse emissions. The medium intensification system presented the lowest carbon footprints, with possible carbon credits.

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Introduction

Beef cattle production is an important activity in Brazil, both in terms of herd size and land area. It currently has 214.7 million head in 162.2 million ha of pasture, with a stocking rate of 1.32 head/ha equivalent to 0.93 AU/ha (1 AU = 450 kg live BW) as per ABIEC (2019) and IBGE

(2018). Although the area with pastures in the Atlantic Forest biome has decreased, we still have 36.42 million ha, which is a much smaller area than the 47.05 million ha occupied in 1990. Some pasture areas were destined for other crops or forest regeneration, that is, from 30.5 million ha in 1990 to 32.8 million ha in 2018 (MapBiomas platform, https://plataforma.mapbiomas.org/).

In economic terms, beef cattle accounts for 8.7% of Brazil's GDP, with 20% of its production exported. The production mostly takes place in pastoral production systems, where only 12.6% of the herd is finished in feedlots. This makes the correct management of pastures a strategic point for the maintenance of the Brazilian beef cattle industry. Although 19.4 million ha of its pastures have been converted to other activities as a result of increased productivity (from 1.63 @/ha in 1990 to 4.5 @/ha per year in 2018; 1@ = 15 kg hot carcass), Brazil still has 49.1 million ha of pastures which are degraded and in some degree of regeneration (ABIEC, 2019).

The projections of Brazilian agribusiness from 2015/2016 to 2025/2026 predict an increase in the livestock sector (MAPA, 2016). The meat production is expected to continue its rapid growth (rates of 2.4% per year) in the next decade (OECD-FAO, 2015, quoted by MAPA, 2016). Just as important as economic performance and livestock growth, the way livestock food is produced has been a relevant focus of consumer questioning of greenhouse gases (**GHG**) emissions and the consequent global warming that causes climate change.

Regarding the effects of climate change, it should be remembered that livestock activity produces GHG in the form of methane (CH_4) from enteric fermentation, nitrous oxide (N_2O) from use of nitrogenous fertilizers and CH_4 and N_2O from manure management and deposition of animal manures on pastures. Some carbon dioxide (CO_2) is also produced on animal farms from fossil fuel and energy usage (O'Mara, 2012). In 2015, Brazilian agriculture emitted 429 000 gG carbon dioxide equivalent ($CO_{2e.}$) and accounted for 31.3% of total national net GHG emissions (1 274 000 gG $CO_{2e.}$). Regarding the $CO_{2e.}$ emitted by agriculture, about 60% comes from enteric fermentation (MCTIC, 2020). This reinforces the need to adopt mitigation measures by livestock management strategies. To meet these needs, Brazil has developed the Sectorial Climate Change Mitigation and Adaptation Plan for the Consolidation of a Low Carbon Economy in Agriculture (Plan ABC), MAPA (2012), which promotes the recovery of pasture areas and the adoption of integrated production systems, among other efforts.

Agricultural soils may act as sinks or sources of GHG, depending on how they are managed. Undesirable anthropogenic actions may cause reductions in soil C stocks and promote CO_2 emissions. This worsens the climate change problem. On the other hand, successful management techniques can improve soil C stocks, mitigate GHG emissions and contribute to minimizing global warming and climate change (Soussana *et al.*, 2010; O'Mara, 2012).

The intensification of grassland productivity by manipulation of both primary production and stocking density leads to complex environmental impacts. As intensification increases, positive impacts, such as C sequestration, are progressively impaired by negative impacts linked to excessive active N forms. Hence, in each unique environmental setting, a threshold level of grassland intensification can be determined, above which any additional animal production is associated with further unacceptable environmental risks (Soussana and Lemaire, 2014).

Adequate physiological management of the forage and maintenance of soil fertility by liming and optimum fertilization are essential agronomic practices needed to restore degraded pasture (**DP**) (Oliveira *et al.*, 2003). Nitrogen is one of the most important nutrients required to achieve this goal and to avoid degradation of pastures (Oliveira *et al.*, 2007). However, there are effects of C–N decoupling by domestic herbivores that reduce N availability and limit C uptake (Soussana and Lemaire, 2014). There is also isotopic evidence for oligotrophication of terrestrial ecosystems to N (due to elevated atmospheric CO_2 and longer growing season). These N declines will limit future terrestrial C uptake and increase nutritional stress for herbivores (Craine *et al.*, 2018).

This study aimed to investigate the C balance and the C footprint in pasture-based beef cattle production systems. Treatments differed according to different levels of pasture intensification (different levels of correctives, fertilizers – mainly N – and irrigation). The hypothesis was that the adequate physiological management of the forage and maintenance of soil fertility are essential agronomic practices needed to restore DP and, simultaneously, they may improve C sequestration and mitigate the GHG emission.

Material and methods

The study was carried out in São Carlos, SP, Brazil (22°1′ S and e 47°53′ W, 853 m above sea level). The prevailing climate is subtropical humid (Cwa (Köppen)), with two well-defined seasons (dry season, from April to September, and rainy season, from October to March) and a mean annual temperature of 20°C and an average cumulative annual rainfall of around 1360 mm.

Four representative Brazilian beef cattle finishing (Nelore steers) pasture-based production systems were studied. Pastures were supplemented with minerals in the rainy season and also with protein in the dry season (Supplementary Material Table S1). The study occurred during two periods: March 2012 to August 2013 (Period 1) and September 2013 to December 2014 (Period 2). The treatments consisted of different levels of pasture intensification, with two replicates per system (blocks) as described (Supplementary Material Table S2) were: intensively managed and irrigated *Megathyrsus* (*Panicum*) *maximum* Jacques (cv. Tanzânia) pasture with high stocking rate (IHS); intensively managed rainfed *Megathyrsus* (*Panicum*) *maximum* Jacques (cv. Tanzânia) pasture with high stocking rate (RHS); rainfed pasture with a mix of *Urochloa* (*Brachiaria*) *decumbens* Stapf (cv. Basilisk) and *Urochloa* (*Brachiaria*) *brizantha* (Hochst ex A. Rich) Stapf (cv. Marandu) grasses, with moderate stocking rate (RMS); degraded *Urochloa* (*Brachiaria*) *decumbens* pasture under extensive management (DP). The pastures (*Panicum sp*) in IHS were overseeded with oats (*Avena bysantina* Koch, cv. São Carlos – 60 kg/ha of viable seeds) and annual ryegrass (*Lolium multiflorum* Lam., cv. BRS Ponteio – 30 kg/ha of viable seeds) in the autumn. The intensively managed pastures (HIS and RHS) were divided into 12 paddocks (0.14 to 0.15 ha each), which were grazed for 3 days in a rotational grazing system with 36 days of rest. The RMS pastures were divided into six paddocks (0.55 ha each), which were grazed for 6 days, also in a rotational system with 30 days of rest. The DPs (two paddocks of 1.7 ha) were kept under continuous grazing. Pastures were managed under variable stocking rates ('put and take'). Stocking rates were adjusted accordingly to the forage availability in each paddock. The stocking rate increased as the intensification of the systems increased. In the intensified systems IHS the stocking rate was 6.6 AU/ha; in RHS it was 4.2 AU/ha; in RMS it was 3.3 AU/ha, while in the DP it was only 1.4 AU/ha (1 AU = 450 kg live BW), according to Oliveira *et al.* (2018).

All pastures (except DP) were limed and fertilized with superphosphate and potassium chloride to achieve 20 mg/dm^3 of P and 4% K in soil cation exchange capacity, according to Oliveira *et al.* (2008). Lime was applied once a year. Annual top-dressing N-fertilizer was applied at the rate of 600 kg N/ha in IHS, 400 kg N/ha in RHS and 200 kg N/ha in RMS. Doses were divided into 5 applications during the rainy season in RHS and RMS and 10 applications in IHS (5 during the rainy and 5 during the dry season). The DP was not fertilized or limed. The IHS and RHS systems were implemented in 2002, and the DP and RMS systems in 1996. In relation to the date that soil samples were collected (2011), the management time for IHS and RHS systems was 9 years and 15 years for DP and RMS systems. These periods were considered to calculate the annual C accumulation rates (**ACAR**). The native forest soil was sampled to represent the original soil conditions in the experimental areas.

Soil carbon stock

The complete methodology is described in Segnini *et al.* (2019). Soil samples were collected in the pastures and the original vegetation area (Atlantic Forest – 'seasonal semi-deciduous forest') in the autumn of 2011 in a transition zone with two Hapludox soil types: Red Latosol and Yellow Latosol (both Oxisol according to the FAO classification system). Samples were collected at the depths: 0 to 5, 5 to 10, 10 to 20, 20 to 30, 30 to 40, 40 to 60, 60 to 80 and 80 to 100 cm, with six replicates (three replicates per block). For each replicate, two sub-samples were collected on two sides of the trench for each depth interval by using an Al ring of known volume for the subsequent evaluation of dry soil weight (at 110°C) and soil bulk density (mean of two sub-samples).

Preparation of soil samples and carbon stocks determination

The complete methodology is described in Segnini *et al.* (2019). Soil samples were air-dried at approximately 25°C, then gently crushed using a mortar and pestle and next passed through a 9-mesh sieve (particle size smaller than 2 mm). Sub-samples were finely ground to pass through a 100-mesh sieve (particle size smaller than 0.150 mm), for all analyses. Total C analysis was performed in duplicate on approximately 10 mg of soil using a CHN/S 2400ii elemental analyzer (Perkin Elmer, Waltham, MA, USA), calibration in Supplementary Material M1. Soil C stocks were estimated using the soil bulk density at each depth interval and the corresponding C content (Veldkamp, 1994). Subsequently, C stock data were corrected regarding soil compaction, according to the equation proposed by Sisti *et al.* (2004). Data from the soil under the natural vegetation were used as reference. The calculation of the equivalent soil mass was carried out for 0 to 100 cm layers according to Ellert and Bettany (1995), also using the native forest soil as reference (Supplementary Material Table S3, equations (1) and (2)).

Annual C accumulation rates were estimated (0 to 100 cm layers) by dividing the difference between C stocks in the pasture soils and the native forest (reference) by the number of years since pasture implementation. The formula used was ACAR = ((Carbon stocks systems − Carbon stocks Forest)/age systems)), according to equation (3), Supplementary Material Table S3 (Fernandes *et al.*, 2014). The age of the systems was considered as the number of years passed since the implementation of the pasture systems until the soil sampling date (2011).

Animal performance

The complete methodology is described in Oliveira *et al.* (2018). A total of 48 Nelore steers (24 steers per period; 271 ± 2.2 kg of live BW; 15 months old) were allotted to the four grazing systems (IHS, RHS, RMS and DP) at the same time.

Three steers (testers) were used to evaluate animal performance in each pasture replicate, and additional animals were used to adjust the stocking rate using the 'put and take' technique. Animals were weighed monthly (without fasting) and at the beginning and the end of the two experimental years; this was after overnight (16 h) fasting in order to calculate the average daily gains and the stocking rates. The animals were fasted for 16 h and weighed before transportation to the slaughterhouse to obtain the shrunk BW. All animals were slaughtered with a minimum of 430 kg of live BW on the same day.

Ruminal methane

The complete methodology is described in Sakamoto (2018). The sulfur hexafluoride (SF_6) gas tracer technique (Johnson and Johnson, 1995, and refined by Berndt *et al.*, 2014) was used for methane collection. This technique is consolidated (Boadi and Wittenberg, 2002) and can be used with confidence in studies comparing treatment effects

(Pinares-Patiño and Clark, 2008), especially for grazing animals (McGinn *et al.*, 2006). This technique uses a calibrated permeation capsule placed in the rumen. The calibration in the first year was 190 SF_6 gas of 2.396 ± 0.06 mg/day, and in the second year, it was 190 SF_6 gas of 1.753 ± 0.19 mg/day. The gas expelled through the mouth and nostrils was aspirated by a capillary tube adapted to a halter and connected to a canister under vacuum (collector), which was fixed on the neck of the animal. The methane collections were performed for five consecutive days, with the evacuated sampling canisters being changed every 24 h.

After each collection period, the sampling canisters were sent for chromatographic analysis. Their contents were diluted with pure N to determine the quantities of SF_6 and CH_4 gases, using a 'Greenhouse' GC-2014 gas chromatograph (Shimadzu, Chiyoda-ku, Tokyo, Japan), with a flame ionization detector (FID) and an electron capture detector (ECD), respectively (Supplementary Material M2). The concentrations of CH_4 and SF_6 found in the 'blank readings' were discounted from the concentrations found in the evacuated sampling canisters. Sampling took place in spring (September 21 to December 20), summer (December 21 to March 20), autumn (March 21 to June 20) and winter (June 21 to September 20) (Supplementary Material Table S4). CH_4 emissions from enteric fermentation (t $CO_{2e.}$/ha per year) were calculated according to equations (6), (7) and (8), Supplementary Material Table S3.

Nitrous oxide and methane fluxes from pasture

Gas samples were collected on an event basis for 2 years. Sampling took place in spring (September 21 to December 20), summer (December 21 to March 20), autumn (March 21 to June 20), and winter (June 21 to September 20) seasons (Supplementary Material Table S4).

Samples were collected using Polyvinyl chloride (PVC) chambers installed in the experimental plots, according to the chamber technique, as described in the protocol proposed by Zanatta *et al.* (2014), based on Parkin and Venterea (2010).

Six chambers were used per treatment (three replicates per block) and were allocated randomly, not considering the possible presence of feces and urine, but considering that in pastures with high stocking rates, feces and urine are dispersed in the whole area, making it difficult to identify places not contaminated by excreta (Gusmão *et al.*, 2015). In every sampling cycle, samples were taken initially for five consecutive days and, afterward, at 2- to 3-day intervals until a total of 10-day samplings were completed in each season; this involved 22 days of sampling per cycle. The first samplings occurred 24 h after fertilizer application because there were three treatments with N fertilization. Sampling started between 0800 and 0100 h and was sampled three times after fixing the chamber lids at intervals of 0, 30 and 60 min. Overall, samples were collected in 30 chambers (4 treatments and the forest × 6 chambers) for a total of 7200 sampling events (30 chambers × 3 sampling times × 10 samplings × 4 seasons × 2 years). The analysis was carried out in a Thermo Scientific™ TRACE™ 1310 GC with an automatic injector. The concentrations of CH_4 and CO_2 were determined with a FID and the concentrations of N_2O ECD. External calibration was employed in order to quantify the analytes (Supplementary Material M3)

For the calculation of gas fluxes (F), the gas increment for times (t0, t30 and t60) was calculated first, considering the linear adjustment model and the molecular volume correction for the temperature inside the chamber (T) during sampling and using the formula described in the protocol proposed by Zanatta *et al.* (2014):

$$F = (\Delta C \Delta t^{-1}) \times (MVm^{-1}) \times (VA^{-1}),$$

where $\Delta C\ \Delta t^{-1}$ represents the rate of change of the gas inside the chamber per unit of time (ppb/h); M is the molecular weight (g); V and A are volume (l) and chamber area (m^2), respectively; Vm is the molecular volume of the gas (l), corrected as a function of the temperature inside the chamber during sampling (1 mole of gas occupies 22.4 l under normal temperature and pressure conditions – CNTP), by multiplying 22.4 by (273 + T)/273, with T being the average temperature inside the chamber in degree Celsius.

N_2O and CH_4 emissions from N fertilization and animal wastes (t $CO_{2e.}$/ha per year) were calculated according to equations (9), (10) and (11), Supplementary Material Table S3.

Carbon balance

Carbon balance (Table 1) was calculated as the difference between the ACAR of the pastures and the emissions of $CO_{2e.}$ originated from the beef cattle activity during 1 year (CH_4 emissions from enteric fermentation, N_2O and CH_4 emissions from N fertilization and animal wastes), using AR4 methodology (IPCC, 2007) (global warming potential (**GWP**) $CH_4 = 21$, $N_2O = 310$), AR5 methodology (IPCC, 2014) (GWP $CH_4 = 27.75$, $N_2O = 265$) and the conversion factor of C to $CO_{2e.}$ = 3.67 (Supplementary Material Table S3, equations (4) and (5)).

The intensities of GHG emissions – Ei (Tables 2 and 3), which considered only GHG emissions (Supplementary Material Table S3, equation (12)) or C balance (Supplementary Material Table S3, equation (13)) – were calculated as the division between the GHG emission or C balance and the product output: stocking rate (steers/ha), live BW (kg/ha per year), carcass (kg/ha per year) and carcass edible portion (**CEP**; kg/ha per year). The number of trees required to mitigate the emissions of GHG from beef cattle production systems was calculated (Table 4), using these results.

To achieve better results in terms of productivity and GHG mitigation, it was necessary to adjust the inputs to perform some agricultural operations as the distribution of limestone and fertilizer and to use electricity for center-pivot irrigation. Then the question arose as to how the use of these inputs could impact C balance and emission intensities. To solve this issue, emissions from the manufacture and distribution of these inputs, as well as the use of electricity (GHG emissions

Table 1 *Carbon (C) balance between greenhouse gases (GHG) emissions and removals in beef cattle grazing systems*

Item	Treatments				RMSE	*P*-value
	IHS	RHS	RMS	DP		
Annual C accumulation rates (t CO_{2e}/ha per year)	-0.81^{b}	1.92^{a}	1.80^{a}	-1.07^{b}	0.51	0.0043
GHG removals (t CO_{2e}/ha per year)	-2.97^{b}	7.07^{a}	6.59^{a}	-3.95^{b}	1.87	0.0043
GHG emissions (t CO_{2e}/ha per year)						
AR4	10.43^{a}	6.92^{b}	5.28^{c}	2.28^{c}	0.65	<0.0001
AR5	13.76^{a}	9.14^{b}	6.97^{b}	3.01^{c}	0.86	<0.0001
Carbon balance (t CO_{2e}/ha per year)						
AR4	-13.40^{c}	0.14^{a}	1.3^{a}	-6.23^{b}	0.86	0.0005
AR5	-16.74^{c}	-2.07^{b}	-0.38^{a}	-6.69^{ab}	2.05	0.0005

IHS = irrigated pasture with high stocking rate; RHS = rainfed pasture with high stocking rate; RMS = rainfed pasture with medium stocking rate; DP = degraded pasture; GHG = greenhouse gases; GWP = global warming potential.
Annual C accumulation rates = (C stocks in a treatment − C stocks in the forest)/management time.
Carbon balance = (GHG removals − GHG emissions).
AR4 methodology (IPCC, 2007) (GWP $CH_4 = 21$, $N_2O = 310$); AR5 methodology (IPCC, 2014) (GWP $CH_4 = 27.75$, $N_2O = 265$).
[a,b,c] Values within a row with different superscripts differ significantly at $P < 0.05$.

Table 2 *Emission intensity of greenhouse gases (GHG) as a function of animal performance in beef cattle production systems, considering only GHG emissions or carbon balance (CB)*

Item	Treatments				RMSE	*P*-value
	IHS	RHS	RMS	DP		
Productivity (kg BW/ha per year)	1386.15^{a}	866.71^{b}	656.05^{b}	220.54^{c}	187.27	<0.0001
GHG emission intensity (kg CO_{2e}/kg BW)						
AR4	7.88	8.34	8.51	10.37	1.46	0.1253
AR5	10.40	11.00	11.23	13.71	1.92	0.1218
CB emission intensity (kg CO_{2e}/kg BW)						
AR4	-10.29^{b}	0.61^{a}	1.90^{a}	-28.12^{c}	2.56	<0.0001
AR5	-12.81^{b}	-2.05^{a}	-0.81^{a}	-31.45^{c}	2.63	<0.0001

IHS = irrigated pasture with high stocking rate; RHS = rainfed pasture with high stocking rate; RMS = rainfed pasture with medium stocking rate; DP = degraded pasture; GHG = greenhouse gases; CB = carbon balance; GWP = global warming potential; BW = live body weight.
AR4 methodology (IPCC, 2007) (GWP $CH_4 = 21$, $N_2O = 310$); AR5 methodology (IPCC, 2014) (GWP $CH_4 = 27.75$, $N_2O = 265$).
[a,b,c] Values within a row with different superscripts differ significantly at $P < 0.05$.

of inputs) were added to GHG emissions, and the C balance was recalculated (C footprint).

Carbon footprint per ha (Table 5) was calculated (Supplementary Material Table S3, equation (14)) and also considered GHG emissions from the inputs (diesel used by machinery during liming and fertilization; electricity consumed during pasture irrigation; N, P and K production), according to Supplementary Material Table S3, equations (15), (16), (17), (18), (19), (20) and (21). The footprints for the beef cattle product output were calculated as the division between the C footprint per ha (Table 5) and the product outputs: stocking rate, live BW and carcass (Tables 2, 3 and 4). The number of trees required to mitigate the GHG emission was recalculated (Table 5) with consideration of C footprint per ha and the stocking rate (steers/ha) using the GWP_S of the AR4 methodology.

Annual carbon sequestration potential rate for Eucalyptus

Data from integrated systems with trees of Eucalyptus genus (333 trees/ha) in another experimental area situated near the area used in this study were collected in April 2016, where 40 trees (5 years old) were used to determine the wood volume and to obtain wood rings. Samples were subsequently used to determine biomass and C pools of tree trunks. These data were used to build the equations for estimating stem volume and tree biomass. The equations estimated trunk volume varying from 131.81 to 155.73 m^3 and trunk biomass from 56.92 to 67.47 Mg/ha in the livestock-forest system and crop-livestock-forest system, respectively. Diameter at the beginning and end of each segment and the segment mass were measured. Subsequently, a trunk sample (15 cm ring) was obtained from each segment to determine the moisture content after oven drying at 60°C until constant weight. For these samples, density (ratio of dry mass to volume) and C content (by elemental Analyzer Perkin Elmer model CHNS 2400ii) were also determined.

Annual C sequestration potential rate for eucalyptus (CO_{2e}/tree per year) considered that 333 trees/ha resulted in 65.2 t DM/ha (145 m^3 lumber, considering trunk density of 450 kg/m^3) with 0.45 t C/t DM; that provided 63.89 kg CO_{2e}/tree per year (((65.2 t de DM ha^{-1} × 0.45 t C/t DM)/(5 years × 333 trees)) × 3.67 × 1000).

Table 3 *Emission intensity of greenhouse gases (GHG) as a function of carcass traits and productivity of beef cattle production systems, considering only GHG emissions or carbon balance (CB)*

Item	Treatments				RMSE	*P*-value
	IHS	RHS	RMS	DP		
Carcass (kg carcass/ha per year)	767.34[a]	480.33[b]	365.83[b]	117.49[c]	94.17	<0.0001
GHG emission intensity (kg CO_{2e}/kg carcass)						
AR4	14.06[b]	14.92[b]	15.18[b]	19.49[a]	2.38	0.0526
AR5	18.56[b]	19.69[b]	20.03[b]	25.75[a]	3.13	0.0509
CB emission intensity (kg CO_{2e}/kg carcass)						
AR4	−18.34[b]	1.02[a]	3.4[a]	−52.94[c]	4.75	<0.0001
AR5	−22.84[b]	−3.75[a]	−1.44[a]	−59.21[c]	4.88	<0.0001
CEP (kg CEP/ha per year)	707.01[a]	445.09[b]	330.66[b]	104.84[c]	83.76	<0.0001
GHG emission intensity (kg CO_{2e}/kg CEP)						
AR4	15.22[b]	16.00[b]	16.32[b]	21.88[a]	2.44	0.0236
AR5	20.09[b]	21.13[b]	21.54[b]	28.91[a]	3.2	0.0228
CB emission intensity (kg CO_{2e}/kg CEP)						
AR4	−19.86[b]	1.03[a]	3.70[a]	−59.34[c]	5.17	<0.0001
AR5	−24.73[b]	−4.08[a]	−1.51[a]	−66.37[c]	5.30	<0.0001

IHS = irrigated pasture with high stocking rate; RHS = rainfed pasture with high stocking rate; RMS = rainfed pasture with medium stocking rate; DP = degraded pasture; GHG = greenhouse gases; CB = carbon balance; GWP = global warming potential; CEP = carcass edible portion of the sum of edible portions of the Brazilian primal cuts.
AR4 methodology (IPCC, 2007) (GWP CH_4 = 21, N_2O = 310); AR5 methodology (IPCC, 2014) (GWP CH_4 = 27.75, N_2O = 265).
[a,b,c] Values within a row with different superscripts differ significantly at $P < 0.05$.

Table 4 *Trees needed to mitigate greenhouse gases (GHG) emissions in beef cattle production systems, considering carbon balance (CB)*

Item	Treatments				RMSE	*P*-value
	IHS	RHS	RMS	DP		
Stocking rate (steers/ha per year)	7.60[a]	4.76[b]	3.6[b]	1.64[c]	0.44	<0.0001
CB mitigation trees (n. trees/ha)						
AR4	−218.75[c]	2.34[a]	21.27[a]	−101.71[b]	31.07	0.0005
AR5	−273.24[c]	−33.84[ab]	−6.34[a]	−113.67[b]	33.52	0.0005
CB mitigation trees (n. trees/steer)						
AR4	−29.11[b]	1.08[a]	6.27[a]	−63.89[c]	7.66	0.0001
AR5	−36.34[b]	−6.50[a]	−1.37[a]	−71.24[c]	7.93	0.0002
CB mitigation trees (n. trees/kg BW)						
AR4	−0.17[a]	0.01[a]	0.03[a]	−0.46[b]	0.05	0.0004
AR5	−0.21[a]	−0.03[a]	−0.01[a]	−0.51[b]	0.05	0.0003
CB mitigation trees (n. trees/kg carcass)						
AR4	−0.30[a]	0.02[a]	0.05[a]	−0.86[b]	0.09	0.0003
AR5	−0.37[a]	−0.06[a]	−0.02[a]	−0.97[b]	0.09	0.0003
CB mitigation trees (n. trees/kg CEP)						
AR4	−0.32[a]	0.02[a]	0.06[a]	−0.97[b]	0.09	0.0003
AR5	−0.40[a]	−0.07[a]	−0.02[a]	−1.08[b]	0.09	0.0002

IHS = irrigated pasture with high stocking rate; RHS = rainfed pasture with high stocking rate; RMS = rainfed pasture with medium stocking rate; DP = degraded pasture; n. trees = number of trees; GHG = greenhouse gases; CB = carbon balance; GWP = global warming potential; BW = live body weight; CEP = carcass edible portion of the sum of edible portions of the Brazilian primal cuts.
AR4 methodology (IPCC, 2007) (GWP CH_4 = 21, N_2O = 310); AR5 methodology (IPCC, 2014) (GWP CH_4 = 27.75, N_2O = 265).
[a,b,c] Values within a row with different superscripts differ significantly at $P < 0.05$.

This result was used to calculate the number of trees necessary to mitigate the GHG emissions of different production systems.

Statistical analyses

After verifying the residue normality by the Shapiro–Wilk test, data were analyzed by the MIXED procedure of SAS software (SAS Inst. Inc., Cary, NC, USA) with repeated measures (Supplementary Material M4). The model included the fixed effects of treatment (four pasture-based beef cattle production systems) and year (1 and 2) and their interactions (treatments × year) for average C balance, C footprint, Ei and number of trees simulation. The effect of block (area replicate) was considered as random factors. The matrix that

Table 5 *Greenhouse gases (GHG) emissions due to the use of inputs used on beef cattle production systems and carbon (C) footprint of beef cattle (with GHG off-setting potential), using AR4 methodology*

Item	Treatments				RMSE	*P*-value
	IHS	RHS	RMS	DP		
GHG emission of inputs (t $CO_{2e.}$/ha per year)	2.31[a]	1.36[b]	0.85[c]	0.00[d]	0.0085	<0.0001
C footprint per ha (t $CO_{2e.}$/ha per year)	−15.71[b]	−1.22[a]	0.45[a]	−6.23[ab]	2.27	0.0042
BW C footprint (kg $CO_{2e.}$/BW)	−10.21[a]	−1.02[a]	0.60[a]	−30.00[b]	0.0003	0.0005
Carcass C footprint (kg $CO_{2e.}$/kg carcass)	−20.15[a]	−2.00[a]	1.91[a]	−50.29[b]	0.005	0.0005
C footprint mitigation trees (n. trees/ha)	−256.48[b]	−19.89[a]	7.29[a]	−101.71[ab]	37.17	0.0042
C footprint mitigation trees (n. trees/steer)	−34.22[ab]	−3.60[a]	2.36[a]	−63.90[b]	9.24	0.0033

IHS = irrigated pasture with high stocking rate; RHS = rainfed pasture with high stocking rate; RMS = rainfed pasture with medium stocking rate; DP = degraded pasture; n. trees = number of trees; GWP = global warming potential; BW = live body weight.
C footprint per ha = (GHG removals − (GHG emissions + GHG emissions of inputs)).
AR4 methodology (IPCC, 2007) (GWP CH_4 = 21, N_2O = 310).
[a,b,c,d] Values within a row with different superscripts differ significantly at $P < 0.05$.

best fit the data was the autoregressive covariance structure. The effects were considered significant at $P \leq 0.05$. All means are presented as least-square means, and effects were separated by the PDIFF option of SAS and Tukey average test.

Results

The negative numbers shown in the tables mean C deficit in the systems, that is, the GHG emissions were higher than the removals, and the positive numbers represent C credits obtained in beef cattle production systems. In the case of the number of trees, negative values mean a deficit of trees in the production systems. In contrast, positive numbers mean C credits equivalent to the number of trees presented in Tables 1, 2, 3, 4 and 5.

The annual soil C sequestration rates were positive for the intensively managed rainfed pasture systems with high and moderate stocking rates (RHS and RMS, respectively). Negative annual soil C sequestration rates were obtained for the DP and intensively managed and irrigated pasture system (IHS). Turning these results into GHG removals in $CO_{2e.}$, the same result pattern was obtained (Table 1).

Greenhouse gases emissions were higher, and corresponded with higher stocking rate, in the irrigated treatment (IHS). Using the GWPs proposed in AR5 (IPCC, 2014), the emission was even higher (Table 1). Contributing to these results was the higher enteric methane emission from the higher stocking rate associated with the higher GWP attributed to methane in AR5.

The C balance generated C credits only in the rainfed systems – RHS and RMS (Table 1), when the AR4 methodology was applied. When applying the AR5, all systems presented a C balance deficit, which shows higher emission than GHG removal. The treatments with the highest C deficit were IHS and DP (Table 1). In IHS, both the higher stocking and the loss of soil C stocks contributed to the negative C balance, while in the DP, the main cause was the loss of soil C stocks.

When only GHG emissions are considered, there is no difference between treatments for emission intensity per live BW because the increase in emission is linked to the increase in productivity (Table 2). When considering the C balance for the calculation of C balance emission intensity (CB emission intensity) per BW, the worst result was for the DP (which integrates lower productivity with soil C loss). Degraded pasture was followed by IHS, which, despite the high productivity through the use of inputs and irrigation, was not able to sequester C in the soil when compared to the forest soil. Rainfed systems (RMS and RHS) presented the best results for CB emission intensity per BW, with C credits when using the AR4 methodology (Table 2).

The GHG emission intensity, as a function of carcass traits and productivity of different beef cattle production systems, and considering only GHG emissions or C balance, presented different results (Table 3). When considering only the intensity of emissions per kg of carcass, the DP system presented higher emission intensity than the other production systems (RMS, RHS and HIS). For the CB emission intensity per kg of carcass, which considers the C balance, the higher emission was obtained for DP, followed by IHS. The rainfed systems (RHS, RMS) may even generate C credits when considering the products of carcass traits and using the AR4 methodology (Table 3). The same response pattern to the treatments was observed for emission intensity per kg of CEP (Table 3).

With the results of CB and Ei, the number of trees needed to mitigate GHG emissions in the beef cattle production systems per area unit and per kg of product was calculated (Table 4). The IHS system required the highest number of trees to mitigate GHG emissions per ha and per steer, followed by the DP system (Table 4). The RMS and RHS required fewer trees in the AR5 methodology and presented a potential tree surplus in the AR4 methodology (Table 4). When expressed by kg of BW, kg of carcass and kg of CEP, the DP system was the one system that required more trees; the IHS system had higher productivity and equaled the

RHS and RMS systems. These systems (RHS and RMS) presented a potential tree surplus for the meat produced in the AR4 methodology (Table 4).

The highest GHG emission due to inputs was in IHS, followed by RHS and RMS. In the DP system, this value was considered null. Even considering the use of inputs, when calculating the C footprint per unit area (ha), the RMS system still generated C credits, followed by RHS, DP and IHS. The same response patterns were obtained for C footprint per BW and per carcass (Table 5).

The IHS system required the largest number of trees, followed by the DP system, which did not differ from RHS and RMS systems when using the C footprint (with inputs) to calculate the tree requirement per ha. Only RMS remained with a potential tree surplus by the AR4 methodology (Table 5). When using the C footprint per steer (with inputs) to calculate the tree requirement, the greatest tree requirement was for DP (as it has the lowest stocking), followed by IHS, RHS and RMS. Again, only RMS remained with a tree surplus potential for each animal slaughtered using the AR4 methodology (Table 5).

The intensification of pasture management resulted in higher forage production, stocking rate (Table 4) and production of meat. This resulted in a land-saving effect of 3.63, 1.90 and 1.19 ha in IHS, RHS and RMS, respectively, for each ha of intensified pasture, as calculated according to equation (22), Supplementary Material Table S3.

Discussion

Carbon balance

Beef cattle production in RMS and RHS systems, in which pastures were fertilized with 200 and 400 kg N/ha per year, respectively, presented the best results for the ACAR (with the forest as reference) and, consequently, demonstrated higher potential for GHG removal (Table 1). In contrast, the IHS and DP systems lost soil C and were unable to perform GHG removal.

Concerning C balance, as GHG emissions increased with increasing stocking rates, the IHS system presented the highest C deficit, followed by the DP system. In contrast, the RHS and RMS systems presented C credits when using the GWPs of the AR4. The complex relationships between grazing intensity, increased productivity and nutrient management can contribute to reduce GHG emissions and enhance GHG sinks in grazing lands (Soussana *et al.*, 2010 and O'Mara, 2012). Irrigation also affected the results.

According to Braz *et al.* (2013), well-managed tropical pastures can increase soil C stocks, while soils under poorly managed or DPs may lose C when compared to soils under the original vegetation (forest). Grass species with abundant root systems during pasture recovery and intensification processes must be the reason for this behavior (Oliveira *et al.*, 2007). Factors such as the increase in rhizodeposition and the litter layer contribute to increased soil C stocks, but this depends on the use of fertilizers, principally N (Oliveira *et al.*, 2007).

The dynamics of OM and the C : N ratio are fundamental to explain the C stock results because the C and N cycles are interconnected (Soussana and Lemaire, 2014).

In the case of RHS and RMS, the results indicate that there were favorable dynamics involving soil management processes and pasture physiology. Soil correction and fertilization increased forage and root production. The equilibrium in stocking rates and grazing intensity favored soil litter deposition and root deposition, which in turn influenced soil C accumulation rates and promoted increased C sequestration that consequently improved C balances.

In the DP system (without liming or fertilization), there was mineralization of soil organic matter (**OM**), low rhizodeposition and absence of litter layer due to the absence of soil nutrients and overgrazing in the dry season. These factors led to C depletion in the soil, including soil C stock existing before the pasture degradation process (Segnini *et al.*, 2019).

In the irrigated system, two factors may explain the results: the dynamics of OM in irrigated soils and the increase in stocking rate. According to De Bona *et al.* (2006), irrigation can increase soil C input; however, sometimes it is not enough to increase C stocks because soil management may modify the irrigation effect on soil stocks (Bayer *et al.*, 2006). According to Andrén *et al.* (1992), an increase in soil water content caused by irrigation provides favorable conditions for microbial activity, intensifying microbial OM decomposition and C mineralization. These factors may have contributed to the lower C stocks detected in IHS (Segnini *et al.*, 2019).

Certainly, higher stocking also contributed to increasing the C deficit in the HIS's C balance, both by C exported in animal products and by enteric methane emissions. Soussana *et al.* (2010) compiled results showing that C export by animal products (meat, milk), which is only a small fraction of the ingested grass C, is 0.6% of C intake in extensive meat production systems and may become much higher in intensive dairy production systems (e.g. 19% to 20% of C intake).

Carbon losses through methane emissions from the enteric fermentation also explain the results, as the annual methane emissions (animal stocking rate × individual enteric methane emission (Table 1)) were 469.2, 317.4, 229.8 and 110.7 kg CH_4/ha per year for the IHS, RHS, RMS and DP systems, respectively (Sakamoto, 2018).

Emission intensity by greenhouse gases emissions or carbon balance

Ei per LBW was the same for all treatments when considering only GHG emissions because the increase in emission was linked to the increase in productivity (Tables 2 and 3). Enteric methane emission was the major contributor to total GHG emissions, as reported by Sakamoto (2018), and the emission of N_2O and methane from the pastures was exceptionally low, as in Oliveira *et al.* (2016). However, when considering the C balance for the calculation of CB Ei per LBW, the undesirable result for the degraded system (which integrated lower productivity with soil C loss) was evident. The IHS system, which was although unable to sequester C in the soil when compared to the forest soil, had this effect mitigated by high live weight productivity (Table 2).

High and medium stocking rainfed systems presented the best results for CB Ei when considering the C balance generated using the AR4 GWP, C credits (Table 2). This response was possible due to the yield values presented by each system (Table 2) and the systems' ability to sequester C, as discussed by Segnini *et al.* (2019).

When the carcass yields are also integrated (productivity issues aside), the efficiency of the production systems starts to interfere even more in the GHG Ei, which consolidates the inadequate outcome for the degraded system (Table 3). This is due to the different values obtained for hot carcass weight (kg) and dressing (%), according to Oliveira *et al.* (2018).

When considering not only emissions but also C balance to calculate CB Ei and in addition to system production efficiency issues, C sequestration also interfered with results for both kg of carcass and CEP. The best results were obtained for the RMS and RHS, followed by the IHS and, again, the worst result for the degraded system (Table 3). These results are important for Brazil because they validate the investments made in the Low Carbon Agriculture Plan, whose main objective is the recovery of 15 million ha of DP areas (Plan ABC, MAPA, 2012).

Tree requirements to mitigate greenhouse gases emissions

Knowing that some pasture-based livestock production systems have C deficits as a result of the C balance, some researchers have studied the effect of inserting trees into production systems to mitigate GHG emissions (Nguyen *et al.*, 2012; Figueiredo *et al.*, 2017; Cunha *et al.*, 2016) (Table 4).

In this experiment, the interest was in calculating the number of trees required to mitigate GHG emission, considering the C balance within the farm gate. The trees must be developing while the animals are being raised, and values have been annualized. When animals are slaughtered, they can be replaced, and new animals can benefit from emissions abatement from the annual growth rate and accumulation of C in the trunks of the eucalyptus trees. The trees may be inserted in a separated tree area or integrated system (livestock and forest), in the latter case. The trees would bring in a new set of dynamics; they would have an effect on pasture production, herbage growth rates in response to N applied, soil moisture, animal performance and animal comfort, among other typical aspects of integrated production systems.

The results obtained per area show that the IHS system required the highest number of trees, followed by the DP, RHS and RMS systems. When expressed by kg BW, kg carcass and kg CEP, the DP pasture was the one that required most trees; the IHS system, as it had a higher productivity, equaled the RHS and RMS systems. The latter, using the AR4 methodology, presented a potential tree surplus for the meat produced (Table 4).

In the case of IHS, trees could be planted in adjacent areas, given the difficulty of irrigation management along with the trees. For DP systems, it seems more viable to recover pastures to reduce this need, including adopting integrated production systems containing forestry, as proposed by Nguyen *et al.* (2012), Cunha *et al.* (2016) and Figueiredo *et al.* (2017). The results of RHS and RMS systems are important because they can support the production of low emission meat without the need for tree planting.

Carbon footprint (with inputs and greenhouse gases off-setting potential)

The different responses obtained (increased production, C loss or sequestration, increase or decrease in GHG emissions, C balance with or without C deficit) as a function of the different pasture-based livestock production systems were due to the different soil and pasture managements adopted, which included the use of agricultural practices (liming, fertilization and irrigation) that consume inputs (diesel for agricultural operations, electricity and fertilizers) (Table 5).

Calculating C footprint and tree requirements considering these issues is important, because it is necessary to know if the advantages presented in C balance still are advantageous when considering C footprint, where emissions from input use are considered. Carbon footprint and CB were the same in DP because input emissions were not considered because it is an extractive system in which inputs were not used. For the other systems, the higher the intensification, the higher the GHG emission from the inputs and the higher the C deficit, with the IHS system having the highest C deficit (Table 5).

Soussana *et al.* (2010), who worked with biophysical modeling, also found different responses as a function of different beef cattle production systems. According to their prediction, the attributed GHG balance was positive for grazing sites (indicating a sink activity). However, it was negative for systems where the forage was cut and for mixed sites (indicating a source activity). Therefore, grazing management seems to be a better strategy for removing GHG from the atmosphere than cutting management. It had C credits of 3.2 t/ha per year, as occurred in the RMS system, which generated C credits of 0.45 t/ha per year. For Brazilian conditions, Figueiredo *et al.* (2017) modeled three production systems (DP, well-managed pasture and crop-livestock-forest system) and obtained C credits for the crop-livestock-forest system, C deficits of 0.8 for the DP and 6.84 t $CO_{2e.}$/ha for the well-managed pasture. Our results, which were obtained in field experiments and not modeled, were worse for the DP, with a deficit of 6.23 t $CO_{2e.}$/ha per year. Results were better for the well-managed pasture (RMS system), which did not present a C deficit as predicted by Figueiredo *et al.* (2017), but C credits of 0.455 t $CO_{2e.}$/ha per year.

When C footprint is expressed per kg of BW gain or kg of carcass in 1 ha, the DP system had the worst results because the productivity of this system is very low, unlike IHS, RHS and RMS, which presented similar results. Rainfed pasture with medium stocking rate presented C credits for each unit of livestock product produced. Again, the measured results differ from the predicted results by Figueiredo *et al.* (2017), who found a C footprint with a C deficit of 18.5 kg CO_{2e}/BW for the DP system, while a C deficit of 30 kg CO_{2e}/BW was measured (Table 5).

For the RMS system, a C footprint with a deficit of 7.6 kg CO_{2e}/BW was predicted (Figueiredo *et al.*, 2017), but in this research, a C footprint with C credits of 0.6 kg CO_{2e}/BW was measured. Carbon footprints per kg of carcass ranged from a C deficit of −50.3 kg CO_{2e}/kg carcass to a credit of 1.91 kg CO_{2e}/kg carcass (Table 5). They did not reach the value predicted by Stanley *et al.* (2018) in the USA of 6.65 kg CO_{2e}/kg carcass for a proposed adaptive multi-paddock grazing system. The difference is due to the high rate of C sequestration observed by Stanley *et al.* (2018). The C accumulation rates found in this experiment were lower, possibly because they were calculated using the forest as a reference and not the time zero of conversion of the production system (baseline) or land use before conversion, such as for the DP.

According to O'Brien *et al.* (2019), these differences between the predicted values in the models and those found in the case studies occur because of regional characteristics. There is a need to perform a harmonization of the models so that the predicted values are closer to the observed values, and the models can be applied in more regions.

The requirement for trees, considering the inputs included in the C footprint calculation, was higher for the IHS system, which required 256.5 trees/ha, followed by the DS and RHS with 101.7 and 19.9 trees/ha, respectively.

When evaluating the tree requirement per steer, the RMS system presented a surplus of 7.3 trees per steer (Table 5). Due to the high stocking rate of the IHS and the low rate of the DP, the order of treatment changes, where the DP of the system required more trees per steer (63.9), followed by IHS (34.2) and RHS (3.6). The RMS system generated a surplus of 2.36 trees per steer. Knowing these values becomes important for planning integrated production systems or even for those planning to produce C-neutral meat, with emissions mitigated by tree planting.

Other aspects related to livestock production are important. Farms with higher productivity maximize their output from the resources invested and the emissions linked to adult animals and consequently reduce their C footprint per kilogram of meat produced. Besides its primary function of producing meat, farming systems based on pastures usually keep the animals in their natural habitat, providing benefits to society, such as the sustainable management of renewable natural resources, conservation of biodiversity and the maintenance of socio-economic viability for many rural areas, especially in remote areas. In addition and in agreement with our results, the intensification of pasture management would result in a land-saving effect (3.63 ha to IHS, 1.90 to RHS and 1.19 to RMS) and allow the preservation of the Brazilian Atlantic Forest, which has great biodiversity with 146 species of woody trees in this case study. Relative to GHG emission, this fact is important because, when computing root and understory forest biomass, total plant biomass totaled 220.5 t/ha and biomass C stored 104.6 t/ha. These results demonstrate the importance of public policies and economic incentives for the improvement of pastures as means of environmental preservation.

The best pasture management strategies, therefore, can be an effective alternative for sustainable meat production because they help to mitigate GHG and also bring many other environmental, social and economic benefits.

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Declaration of interest

The authors declare that they have no conflicts of interest.

Ethics statement

Project No. 05/2016 is in accordance with the precepts of Law No. 11 794, of 8 October 2008, Decree No. 6 899, of 15 July 2009, and the rules issued by the National Council for the Control of Animal Experimentation and was approved by the ETHICS COMMITTEE ON THE USE OF ANIMALS of Embrapa.

Software and data repository resources

SAS and Microsoft Excel were used to store data in a secure repository. The data are restricted, depending on the user's access profile and password.

Supplementary material

To view supplementary material for this article, please visit https://doi.org/10.1017/S1751731120001822

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