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From animals to machines. The impact of mechanization on the carbon footprint of traction in Spanish agriculture: 1900–2014

Eduardo Aguilera*, Gloria I. Guzmán, Manuel González de Molina, David Soto, Juan Infante-Amate

Universidad Pablo de Olavide, Ctra Utrera km 1, 41009, Sevilla, Spain

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A B S T R A C T

Mechanization of agriculture drastically increases labour productivity in crop production, playing a major role in industrialization by freeing up workforce for industry and services. These historical processes are well studied, but there is much less knowledge on their environmental implications, particularly the carbon footprint.

In this work, we aimed to reconstruct the complete historical process of mechanization in Spanish agriculture at the national scale, estimating the carbon footprint of traction through a life cycle assessment approach. The assessment includes greenhouse gas emissions from working animals and feed production, and accounts for the historical changes in the energy efficiency of the industrial production of machinery and fuels.

The results reveal an increase in the carbon footprint of traction from 3.1 Tg CO_2e yr^{-1} in 1900 to 11–12 Tg CO_2e yr^{-1} in the 1970s and 1980s, decreasing to 7–8 Tg CO_2e yr^{-1} in 2010–2014. Area-based emissions ranged 185–242 kg CO_2e ha^{-1} yr^{-1} in 1900–1933, when the practical totality of traction was animal, and 503–540 kg CO_2e ha^{-1} yr^{-1} in the 21st century, when animal traction had almost completely disappeared. Product-based emissions were similar at the beginning and at the end of the study period, as the productivity growth offset the area-based emissions growth. The results show a large peak in emissions during the main decades of the mechanization process. Thus, the large savings observed in the last three decades start from a very high emission level. The carbon footprint of traction could be reduced by mimicking the logic of traditional organic systems but still benefiting from modern technological efficiency, through the self-production of the fuel. Our analysis, however, shows that a simple shift to biofuels may actually increase greenhouse gas emissions and consume a large share of the current agricultural output. Therefore, its combination with significant reductions in fuel and feed demand would be necessary to achieve its mitigation potential.

1. Introduction

The alteration of natural ecosystems through the application of motive power constitutes the basis of agriculture, as this power is employed in all basic tasks, such as preparing the soil, distributing fertilizers, controlling competing wild plants, or harvesting. Indeed, “agriculture by definition involves the use of various implements (many of which must be mobile or transportable) and the application of directed force” (Spoor et al., 1987). Working animals supply most of the motive power in traditional organic agriculture, and also in early mechanized agriculture in which machines such as threshers were powered by animals. In these systems, traction animals are autonomously maintained within the agroecosystems, and they do not only provide all the motive power they require, but they also supply fertilizers, food and draft power to the society. Mechanization drastically cuts labor demand in agriculture (Ibarrola-Rivas et al., 2016), at the cost of the import of external energy and materials, based on fossil fuels (Gingrich et al., 2018). It progressed from inefficient and unreliable steam locomobiles in the 19th century to the development and refinement of internal combustion engine tractors during the first half of the 20th century (White, 2008). The process of mechanization has been highly uneven around the world, shaped by social and

* Corresponding author. Current affiliation: CEIGRAM-ETSIAAB, Universidad Politécnica de Madrid, Senda del Rey 13, 28040, Madrid, Spain.
E-mail address: eduardo.aguilera@upm.es (E. Aguilera).

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environmental constraints (Hayami and Ruttan, 1971). Some countries such as UK or US were highly mechanized by mid-20th century (Binswanger, 1986) while mechanization is still very low in some countries (Pingali, 2007). Draft animals were estimated to be still used in about 50% of the global land area in year 2000 (Wilson, 2003).

Greenhouse gas emissions (GHGe) from working animals mainly include enteric CH_4 and manure management N_2O and CH_4, as well as emissions from feed production and animal breeding. In mechanized systems, the combustion of fossil fuels is the main source of GHGe, while the production of machinery and fuel are also important contributors to the carbon (C) footprint. Comparisons of mechanical and animal traction have shown that the relative environmental performance of the two systems varies widely depending on the specific context, such as animal species, and on methodological choices such as systems boundaries and allocation criteria, ranging from significantly lower GHGe from animal traction using equines (e.g. Cerutti et al., 2014, Engel et al., 2012), to similar or significantly higher GHGe from animal traction than with mechanical traction using bovines (Gathorne-Hardy, 2016; Spugnoli and Dainelli, 2013).

The industrial production of most inputs used for agriculture experienced large efficiency improvements during the 20th century (e.g. Smil, 2000). Moreover, machine engines became more efficient and lighter. As a result, the total energy requirements of mechanical tasks such as tilling or harvesting decreased by more than 50% during the 20th century (Aguilera et al., 2015). Other processes, however, contributed to counteract this trend. The progressive depletion of high-grade ores means increasing energy consumption to extract and refine the materials (Gutowski et al., 2013). As a consequence, the energy efficiency of the production of raw materials and energy carriers may ultimately decline, showing an "inverted U″ shape, as has been observed for oil and gas production in the US and the world overall (Hall et al., 2014). On the other hand, many materials have been replaced by more energy intensive ones, such as materials able to bear the higher temperatures of high-efficiency engines (Stout and McKiernan, 1992), or electronics to fine-control machine functioning.

In spite of the in-depth changes experienced by traction during industrialization, the knowledge on the environmental implications of this transition is very scarce, as most long-term environmental studies of agriculture, including energy (e.g. Gingrich et al., 2018) or greenhouse gas emissions (GHGe) (e.g. Parton et al., 2015) assessments do not specifically address the traction subsystem. Therefore, to our knowledge, this work represents the first assessment of this transition, by analyzing the carbon footprint of working animals and mechanical traction in Spanish agriculture from 1900 to 2014. The studied period thus includes the practical totality of the industrialization process, as well as a multi-decadal initial period of animal-based traction and a final period of mechanical-based traction. This enables comparing the performance of animal and mechanical traction and to study the impacts of the historical changes in each technology and their combinations during the transitioning period.

The specific research questions are the following:

(1) What is the evolution of the carbon footprint of traction in Spanish agriculture?
(2) What is the evolution of the emission hotspots associated to traction?
(3) What is the evolution of the contribution of traction to the carbon footprint of Spanish food products?
(4) What are the GHG mitigation potentials of animal and mechanical (internal combustion) traction?"

2. Methods

2.1. Scope definition

The system studied is traction in Spanish agriculture, defined as all the processes implied in providing the motive power used in cropland cultivation, excluding the energy employed in irrigation. We chose both area-based and product-based functional units in order to reveal different aspects of the contribution of traction to the C footprint of crop production and to ease comparisons with studies from other areas and scales of analysis. As this study covers the totality of Spanish crop production, which is very heterogeneous, we chose carbon (C) and nitrogen (N) production as standardized mass functional units. The net output of cropland was considered, including used residues such as harvested or grazed straw, but excluding the fraction of the output that is fed to working animals, which is considered an internal loop of the system. This study employs a cradle to farm gate perspective focused on agricultural inputs used for traction. The studied system *Traction*, refers to all traction activities in Spanish agriculture, and it is subdivided into: (i) *Working animals*, which include draft animals used in agriculture (Section 2.2.1), and (ii) *Mechanical traction*, which includes energy (fuels production and use) and materials (machinery production and maintenance) employed for performing mechanical work in farm tasks, excluding water pumping for irrigation (Section 2.2.2). Data on mechanical traction is presented on an annual basis. However, this temporal resolution was not available for the data on working animals and functional units. In those cases, we employed decadal time steps matching those employed by Soto et al. (2016) and Guzmán et al. (2018), calculating the intermediate values through interpolation.

2.2. Inventory analysis

2.2.1. Working animals

Animals working in agriculture include oxen and other bovine, donkeys, mules and horses. The estimation of the number of working animals (Table A.1) was based on data from the Spanish Agricultural Yearbook (MAPA, 2018), which excludes animals used for transport and industry. Despite it is not specified in the Yearbook, we hypothesize that, aside from agricultural tasks, these animals may perform some transport of products and people at the local level, in a similar way as tractors do in mechanized systems. Animals exclusively dedicated to transport, however, are excluded from this analysis. We also excluded animals used for irrigation (Appendix A). The Yearbook animal census for the pre-1940 period was very unreliable, according to many historical sources (Soto Fernández et al., 2016). Therefore, we re-estimated it by calculating animal work requirements, verifying that the feed requirements of the re-calculated herd matched feed availability. The share represented by animals working in agriculture over the total number of animals of each species is shown in Table A2. Default power capacity factors (Table A.3) were adjusted by the live weight of animals (Table A.4) to estimate power capacity factors of working animals in Spain (Table A.5). The full procedure is described in Appendix A.

2.2.2. Mechanical traction

2.2.2.1. Machinery. The following parameters were estimated yearly for each type of farm machinery: the census, including number and rated power of the machines, the number of annual registrations in the census and removals from it, the average lifespan, and the weight of new, removed and average machines of the census (Appendix A). See Section 2.3.2 for a description of the way these data are used to estimate machinery emissions. The machine types studied are all of those reported by the statistical sources, including locomobiles, threshers, tractors, harvesters, other motors (static), tillage machinery and other farm implements. The average lifespan of machinery (Fig. A1) was estimated based on annual registrations (Fig. A2) and removals, assuming that the machines removed in a given year were the oldest ones in the census.

2.2.2.2. Fuels. We harmonized the fuel consumption data from primary sources (Appendix A, Fig. A4) expressing them as petajoule (PJ) gross energy, using energy conversion units from Table A.6. Fuels consumed in traction include all agriculture fuels reported by official agricultural statistics, except those employed in irrigation and in modern livestock production (Fig. A5). The latter was estimated by assuming that all non-liquid fuels, except coal during the first half of the 20th century (which was used by threshers and locomobiles), were used in livestock facilities. There is a lack of sources reporting fuel consumption before 1950, so it was assumed to be proportional to the installed power of machinery, taking 1950–1951 data as a reference for machinery fuel consumption.

2.3. Impact assessment

2.3.1. Working animals

Cerutti et al. (2014) pointed out some problems in LCA of working animals, including the variability of power generation and the difficulty of modelling the whole life cycle of animals. The power variability problem is overcome in this study by the scale of analysis, which integrates the power of all agriculture working animals. The whole life cycle of animals has been assessed, based on the herd structure. GHGe from each species of working animal a in year t include N_2O and CH_4 from manure management ($MM_{a,t}$), N_2O from excreta deposited in grassland and cropland ($DG_{a,t}$), CH_4 from enteric fermentation ($EC_{a,t}$), GHGe from feed production ($FP_{a,t}$), and GHGe from breeding ($B_{a,t}$) (Equation (1)).

$$A_t = \sum_a MM_a + DG_{a,t} + EC_{a,t} + FP_{a,t} + B_{a,t} \quad (1)$$

Biogenic emissions from animals were estimated based on the number of animals of each species working in agriculture and their live weights, using the Tier 2 IPCC (2006) approach in most cases. Manure management N_2O emission factors (EFs) (Table A.7) were estimated based on animal excretion and its distribution into grassland and cropland deposition, liquid, solid and daily spreading management systems (Aguilera et al., 2018). We applied a direct N_2O EF of 1.5% for solid storage (Pardo et al., 2015), 0.5% for liquid storage and 0.0% for daily spreading (IPCC, 2006). Indirect N_2O emissions from manure management (Table A.7) were calculated from volatilization losses, using specific factors for each species and management system type from IPCC (2006). We corrected the default N_2O EFs of N deposition from volatilized manure from IPCC (2006) by multiplying it by 0.5, which is the average reduction in the direct N_2O EF under Mediterranean conditions, as compared to the IPCC default value (Cayuela et al., 2017). The same correction was applied to the direct and indirect EFs of grazing excreta from IPCC (2006), resulting in the EFs per animal head shown in Table A.8. The estimation of CH_4 emissions from manure management (Table A.9) followed the IPCC (2006) Tier 2 approach (Appendix A). Tier 1 and Tier 2 IPCC approaches were applied for the estimation of the enteric CH_4 EF of equines and bovines, respectively (Table A.10, see Appendix A). The N-based EF of feed (Table A.12) is multiplied by feed grain, straw and forage ingestion and straw use for bedding (Table A.11) to obtain annual GHGe due to feed production per animal species (Table A.12). Biogenic emissions from working animals were excluded from cropland GHG emissions to avoid double counting. In addition, feed for working animals was taken into account in the production-based indicators by subtracting it from the production. GHGe from the breeding stage includes emissions from the parents and young animals. They were estimated based on the herd structure derived from historical data (JCA, 1920) (Table A.13. See Appendix A).

Finally, total emissions from working animals were allocated to traction work following economic criteria with economic data gathered from JCA (1920). Working animals produced not only work, but also manure, meat and/or other products such as skins, young animals (in the case of non-sterile females), and, in some cases, milk. The allocation shares selected (Table A.14) are based on the annualized income from all these products (Fig. A3), excluding manure, which represents a very small share of the economic output and whose emissions are usually attributed to the animals, not to its final use as fertilizer.

2.3.2. Mechanical traction

The EFs of industrial inputs estimated in this paper are based mainly on the embodied energy data compiled by Aguilera et al. (2015), converted into CO_2 equivalents using typical values of GHGe from fuels combustion (Table A.15). Global EFs were derived for fuel production and transport emissions (Table A.16), including resource extraction, transport of the raw resource, refining and processing, and transport of the refined products to the farm. Fugitive CH_4 emissions from fuel production and transport were also included (see Appendix A). For the estimation of the GWP of the production of industrial inputs in which only embodied energy data were available (without differentiating energy sources), we used the global fuel mix, taken from Koppelaar (2012). The resulting global emission intensities of primary, fossil and renewable energy are shown in Table 1.

Mechanical traction emissions (MT_t) in a given year t include machinery manufacture and maintenance (MP_t) and traction fuels direct (FD_t^T) and production (FP_t^T) emissions (Equation (2)).

$$MT_t = MP_t + FD_t^T + FP_t^T \quad (2)$$

2.3.2.1. Machinery production and maintenance. Machinery embodied emissions are calculated following Equation (3):

$$MP_t = \sum_m \left(\frac{N_{m,t} \cdot P_{m,t} \cdot W_{m,t}^P \cdot EF_{m,t}}{LE_{m,t}} \right) \quad (3)$$

Where$N_{m,t}$ is the farm machinery census (units) for each type of machinery or implement m; $P_{m,t}$ is the rated power of the machinery (KW unit^{-1}); $W_{m,t}^P$ is the specific weight (kg KW^{-1} rated power, in the case of motorized machinery, and kg unit^{-1}, in the case of farm implements); $EF_{m,t}$ is the machinery production EF per unit machinery weight (kg CO_2e kg^{-1}) (Table A.17); and $LE_{m,t}$ is the lifespan of the machinery (years) (Fig. A1). The $EF_{m,t}$ are calculated by summing emissions from raw materials production, manufacture, transport, and maintenance (including tyres and lubricating oil). The inventory analysis of these processes is described in detail in Aguilera et al. (2015).

2.3.2.2. Fuel. Direct emissions from fuel use in traction ($FD_{f,t}^T$) are estimated based on fuel energy consumption ($E_{f,t}^T$) and EFs ($EF_{f,t}^{EP}$) of each type of fuel (Table A.15) (Equation (4)).

$$FD_t^T = \sum_f \left(E_{f,t}^T \cdot EF_{f,t}^{EP} \right) \quad (4)$$

Table 1
Historical evolution of the GHG emission intensity of the global energy mix, 1900–2010 (kg CO_2e GJ^{-1}).

	1900	1910	1920	1930	1940	1950	1960	1970	1980	1990	2000	2010
kg CO_2e/GJ Primary	53.4	61.4	65.5	65.1	65.2	69.3	73.5	75.3	74.7	72.5	70.4	69.2
kg CO_2e/GJ Fossil	114.4	110.4	106.7	104.7	102.7	98.9	95.6	91.1	90.8	90.8	88.5	85.9
kg CO_2e/GJ Renewable	10.4	10.4	10.4	10.4	10.4	10.4	10.4	10.4	10.3	10.3	10.3	10.1

Fuel production emissions (FP_t^T) are calculated in the same way (Equation (5)). They also include transport emissions.

$$FP_t^T = \sum_f \left(E_{f,t}^T \cdot EF_{f,t}^{EP} \right) \quad (5)$$

2.4. Scenarios

GHGe from traction in 2008 was compared to emissions from 4 new scenarios (2 animal-based and 2 mechanical-based) to check the theoretical potential of traction technologies to mitigate GHGe, using land use and crop production data from 2008. In the Scenario Animal 1, the power developed by the machinery in 2008, estimated from fuel consumption using the values of specific fuel consumption under field conditions from Aguilera et al. (2015), was assumed to be supplied by working animals, with the mix of species of 1900 and the EF of feed production in 2008. In Scenario Animal 2, the number of animals per hectare of 1900 were assumed, with the animal mix and the feed EF of the same year. For simplification, the emissions from non-motorized machinery are maintained in the Animal scenarios. Two biofuel scenarios were designed, in which diesel used for traction was assumed to be replaced by ethanol produced from barley. Ethanol production was modeled using inventory data from Spanish facilities and economic allocation (73% to ethanol) (Lechón et al., 2005). The plants used in this study produced distiller dried grain with solubles (DDGS) and electricity from combined heat and power as coproducts. In Ethanol 1 scenario, ethanol is produced using natural gas as the thermal energy source and barley grain with the 2008 EF. In the Ethanol 2 scenario, straw is used as the thermal energy source and the 1900 EF for barley grain. Emissions from straw combustion were modeled using the EF for biomass use in energy industries from IPCC, 2006. The use of ethanol in diesel engines requires additives (ignition improver, denaturants and a corrosion inhibitor), whose energy content was taken from Bernesson et al. (2006). The change in engine fuel efficiency using ethanol was considered to be 0.89 MJ ethanol MJ $diesel^{-1}$ (Fredriksson et al., 2006).

3. Results and discussion

3.1. Inputs consumption

The installed capacity of animal traction grew during the first third of the 20th century, decreased after the Spanish Civil War (1936–1939), and did not grow thereafter (Fig. 1a). The number of horses and donkeys working in agriculture dropped sharply after the war, while the share of cows used for traction increased (Table A2), intensifying the previous trend toward a higher use of bovines. Our estimations of the number of animals in the early 20th century are comparable to those of Kander and Warde (2009). There was a continuous growth in machinery installed power during the studied period (Fig. 1b). Despite their dominance in motorized machinery during the early 20th century, coal-powered locomobiles never represented a significant fraction of total installed power, peaking at 0.4% in 1932. Most of the growth in farm machinery until almost 1930 was dominated by threshers and "other engines", while tractors became dominant in the 1950s. Total machinery power surpassed animal power in 1954, and represented 10 times as much power in 1967, and 100 times as much in 1980. Machinery installed power kept growing until the end of the study period, up to 58 GW in 2010 (Fig. 1b). This is mainly due to the lack of removals from the census, as growth continued despite the number of registrations peaked in the 1970s (Fig. A2). This implies that the average lifespan of registered farm machinery was also growing (Fig. A1). Machinery overcapacity in minifundio areas in Spain was already acknowledged in 1980 (Naredo and Campos, 1980). Machinery rated power increased four-fold from the early 20th century to the end of the studied period (Fig. A3), which contrasts with Canada, where the average tractor power peaked in the early 1970s (Dyer and Desjardins, 2009). Overall, total installed power in Spanish agriculture grew 80-fold (Fig. 1c), with a growth rate that ranged 1–3% during the first third of the 20th century and peaked at 10–23% in the 1960s.

Energy consumption for feeding and bedding ranged 86–113 PJ yr^{-1} during the 1st half of the 20th century, dominated by straw (Fig. 2a). Total fuel direct energy consumption increased three orders of magnitude during the studied period, peaking at 91 PJ yr^{-1} in 1987, and decreasing to 50 PJ yr^{-1} in 2014 (Fig. 2b). Coal dominated fuel consumption in 1900 and was progressively replaced by gasoline during the first third of the 20th century, while gasoline was rapidly replaced by diesel fuel in the 1950s. Thus, the amount of energy contained in the feed used by animal traction was higher than direct fuel energy used by mechanical traction (Fig. 2c).

3.2. Historical greenhouse gas emissions

3.2.1. Total greenhouse gas emissions

Total traction emissions (Fig. 3) grew from 3.1 Tg CO_2e yr^{-1} in 1900 to 12.3 Tg CO_2e yr^{-1} in 1979. Overall, animal traction remained above 90% of total traction GHGe until 1950, despite a 10-fold growth in mechanical traction emissions. Animal emissions dropped fast since 1970, but most of this reduction was offset by the growth in mechanical traction in the following two decades, resulting in total emission levels above 10 Tg CO_2e yr^{-1} until 1988. Traction emissions fell by one third in 1989, linked to a 43% drop in fuel consumption. Consequently, the relative role of machinery increased thereafter, representing about 20% of emissions by the end of the period, the same share as fuel production emissions. Total emissions ranged 7–11 Tg CO_2e yr^{-1} from 1990 until the end of the study period in 2014.

The area-based carbon footprint of traction (Table 2) increased 3-fold during the studied period, peaking in 1980 at 597 kg CO_2eq ha^{-1} yr^{-1}, 13% above 2008 levels. Aside from the peak during the transition period, the comparison of the area-based carbon footprint of working animals and mechanical traction indicates lower emission levels for animal traction, ranging 185–242 kg CO_2e ha^{-1} yr^{-1} in 1900–1933, when the practical totality of traction was animal, and 503–540 kg CO_2e ha^{-1} yr^{-1} in 2000–2008, when traction was almost solely mechanical. The analysis per unit product show different trends (Table 2). The carbon footprint per unit C product increased 13% from the beginning to the end of the study period,

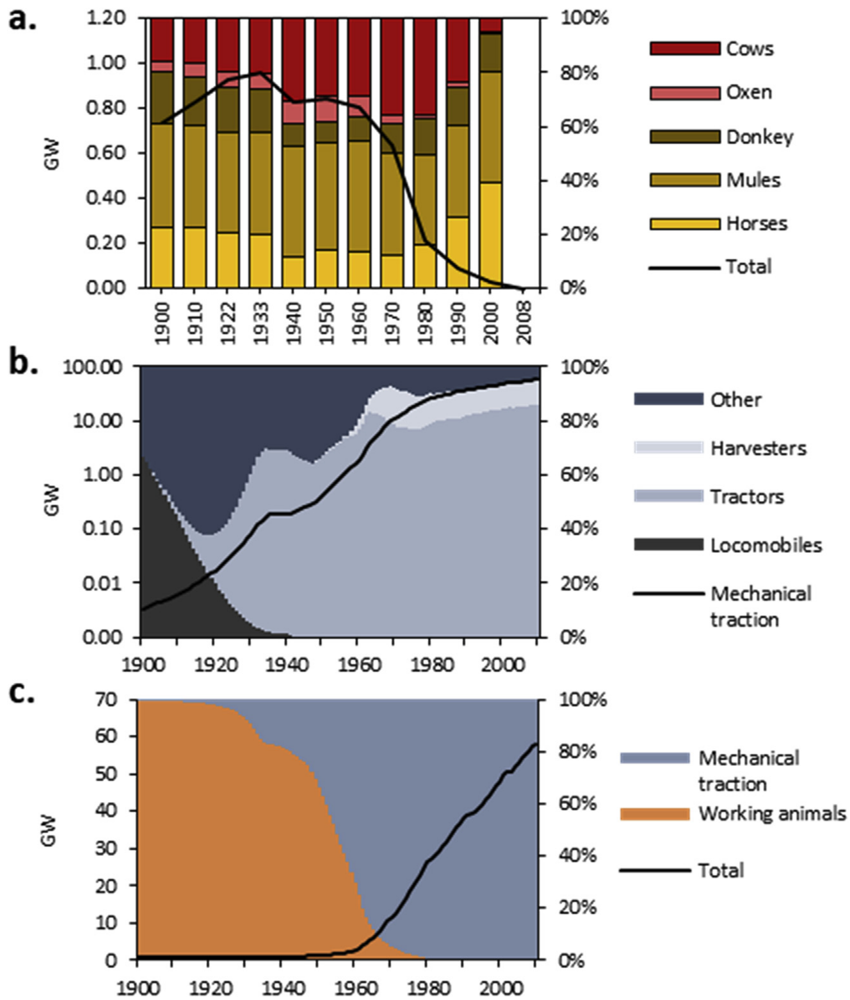

Fig. 1. Installed power in Spanish agriculture (GW). a. Animal power, per animal type; b. Machinery power per type of motorized machinery; c. Total installed power, including animal and mechanical power.

while that per unit N dropped 20%, reflecting a higher N productivity growth. Moreover, a peak in 1970 can be observed for the product-based indicators, roughly doubling the emissions in 2008. In all cases, the peak in the C footprint was reached during the transition period from animal to mechanical traction. The decreasing trend in the last decades is in line with global trends of agricultural GHGe (e.g. Bennetzen et al., 2016), but global historical studies on agricultural C footprints only cover the last decades, starting when the first digital global databases are available, e.g. 1961–1970 for FAOSTAT data (FAO, 2018). Our longer time frame, however, suggest a lower emission intensity in the traditional system preceding the expansion of mechanization.

GHGe from animal and mechanical traction have been compared at the farm level, but we have found no previous assessment performed at wide spatial scales or with a long-term approach. When comparing animal traction emissions to those of light mechanization, Cerutti et al. (2014) found 74% and 94% reductions in GHGe from forest harvesting and tillage, respectively, using animal traction (donkeys and mules), compared to mechanical traction. Gathorne-Hardy (2016) found higher GHGe from ploughing one hectare with bullocks than ploughing with tractor. These results are in line with our historical study, which indicates lower emissions when equines contributed most to the working animals' herd. The long-term analysis presented herein also reveals the influence of large changes in the EFs of each type of working animals (Section 3.2.3), while the scale of analysis precludes the comparison of traction types in each specific task, i.e., we do not know to which extent the differences are due to different efficiency for performing an equivalent task or to changes in the tasks that are performed. In this sense, it is probable that the ploughing depth and frequency increased with mechanization (Infante-Amate, 2014). This has additional implications for climate change, as intensive tillage can foster the loss of soil organic carbon (Gonzalez-Sanchez et al., 2012), although this effect has not been considered in our analysis.

3.2.2. Animal traction emissions

There was an overall increase (55%) in animal traction emissions during the first third of the 20th century (Fig. 4), mainly linked to a 23% expansion in cropland area and a 121% increase in the feed EF, linked to the intensification of crop production. The decrease in animal numbers and the extensification of livestock and crop management after the Civil War did not result in a reduction in animal traction GHGe, due to the increasing role of ruminants in the mix. Cows represented the highest share of animal traction emissions (38–58%), and horses the lowest (4–12%) (Fig. 4a), despite the installed power of horses was higher than that of cows until the Civil War (Fig. 1a). Enteric fermentation was the main process contributing to the carbon footprint of animal work until the 1970, when it was surpassed by animal feed (Fig. 4b). Our results are in

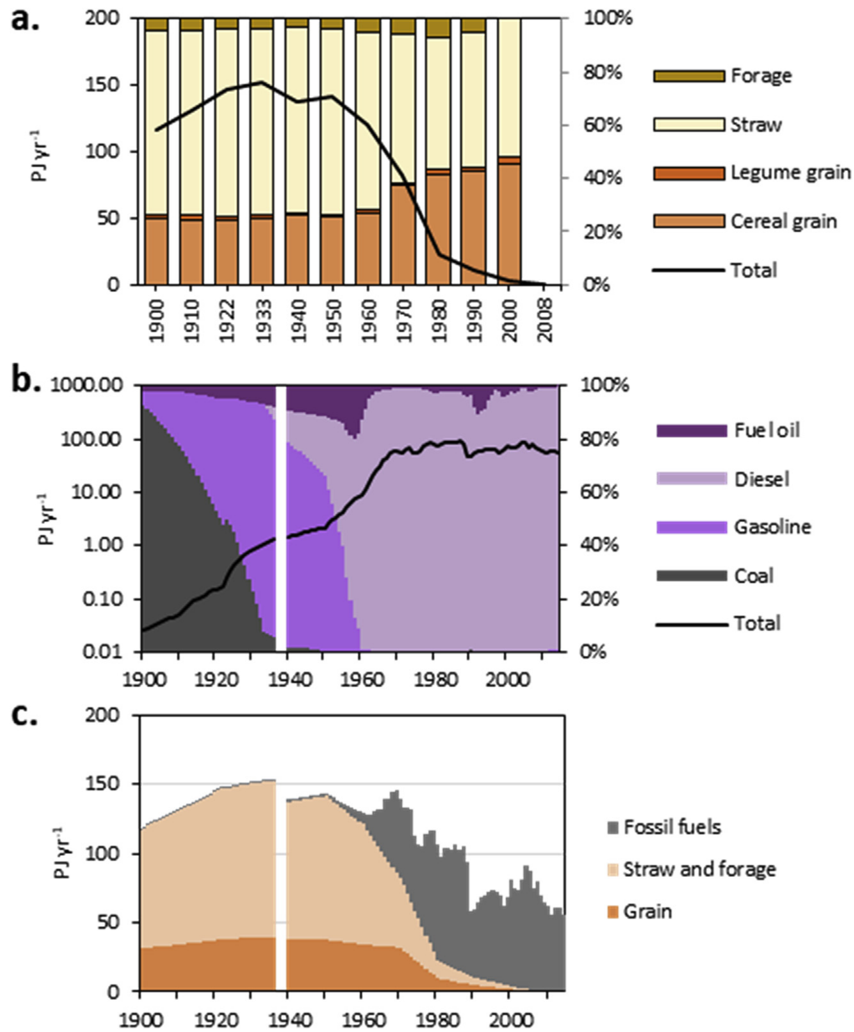

Fig. 2. Direct energy consumption in traction (PJ yr^{-1}), by type of feed used in animal traction (a), by type of fuel used in mechanical traction (b) and total energy consumption as absolute values (c). Grazed pasture is not included in animal feed.

line with other studies, showing a major role of enteric CH_4 emissions in the C footprint of ruminants, and of feed emissions in the C footprint of equines (Cerutti et al., 2014; Spugnoli and Dainelli, 2013). Nonetheless, our diachronic analysis also shows major changes in the composition of the C footprint of working animals, with a growing relevance of feed production for all animal species. Indeed, although playing a smaller role during most of the study period, GHGe from feed production surpassed enteric CH_4 emissions in 1980.

The EFs of working animals (Fig. 5) show marked differences between species and time periods (Fig. 5a). As expected, the highest EFs are for bovine animals but, interestingly, the live weight-related EF of equines since 1980 was only slightly lower than that of bovines in 1900–1950 (Fig. 5c and d). This was due to the growth in emissions from feed production, which was less relevant for bovines than for equines. Bovine animals were valued in traditional agriculture due to their ability to be fed with low-quality biomass such as crop residues and to the additional products they offered, but this also had a cost in terms of GHGe, mainly due to large CH_4 emissions from enteric fermentation (Fig. 5c). The EF per unit installed power of cows and oxen was very similar, but this similarity hides a lower power capacity of cows per unit live weight, which is compensated by a higher share of non-work products in the economic allocation of emissions. The low performance of bovines is exacerbated by the lower amount of working days (207–212 per year), as compared to mules and donkeys (247–251 per year) (JCA, 1920), as has been also observed by Gathorne-Hardy (2016), who estimated that every hour worked by bullocks had an associated 10 h of non-work emissions. In the case of equines, the lower EF of horses is explained by a higher allocation of emissions to offspring, given that a majority of the horses working in agriculture were breeding mares producing mules and young horses to be used for transport. This also had, however, an impact on the number of days they worked (187 per year, on average) (JCA, 1920).

3.2.3. Mechanical traction emissions

Total fuel emissions (Fig. 6a) follow a similar pattern to fuel consumption, growing by three orders of magnitude during the studied period, while machinery production and maintenance emissions (Fig. 6b) grew 57-fold, as the initial value was relatively higher. Machinery emissions were first dominated by tillage implements, and then by threshers until about 1940, while tractors became the dominant source of machinery GHGe by the 1960s. The role of other motorized machinery was relatively minor. Machinery emissions peaked at 1.4 Tg CO_2e yr^{-1} in 1990 and remained stable afterwards despite the growth in the installed power (Fig. 1), due to the increase in its lifespan (Fig. A1). A recent global assessment of energy use in agriculture found that a large share of the uncertainty

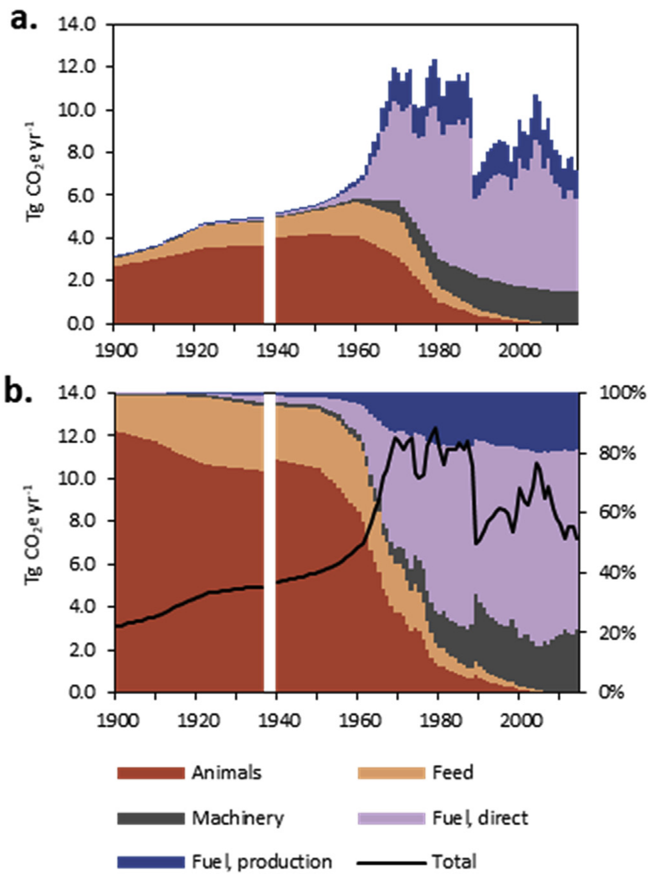

Fig. 3. Total traction emissions (Tg CO_2e yr^{-1}), with its main components expressed as absolute (a) and relative (b) values.

of total energy use was due to the uncertainty in the lifespan of the machinery (Pellegrini and Fernández, 2018). Our assessment overcomes this uncertainty with the use of official statistics data on registrations and removals, which allows us to avoid using an arbitrary lifespan value. The resulting average lifespan of tractors by the 2010s, above 40 years, was above the upper bound of the range (10–30 years) used by Pellegrini and Fernandez (2018), which underlines the gain in accuracy achieved by the use of more detailed country-specific data. Direct fuel emissions dominated emissions from mechanical traction during most of the studied period (Fig. 6c), but machinery emissions (mainly tillage implements) were the dominant emissions source until the 1920s. After the consolidation of tractors as the main traction power technology in the 1950s (e.g. Naredo and Campos, 1980; Martínez Ruiz, 2000), direct diesel fuel emissions became dominant, but the share of fuel and machinery production still remained close to 40% of mechanical traction emissions until the end of the study period.

The weighted average direct EF from fuel use (Fig. 7) declined during the first third of the 20th century due to the decreasing contribution of coal to the fuel mix, which also caused the increase in the indirect EF, as production of oil fuels has a higher emission intensity due to refining processes and fugitive CH_4 emissions. The direct EF remained flat until the end of the study period due to the absence of significant changes in the fuel mix, but the indirect EF increased in the last two decades due to the increasing contribution of unconventional oil resources such as oil sands, extra heavy oil, or shale oil, which require more energy to extract and process (Aguilera et al., 2015).

Most GHGe from mechanical traction are due to fuel consumption. Tillage operations are usually the tasks involving most fuel consumption in cropping systems, so conservation tillage (CT) practices have been proposed to lower fuel consumption and associated GHGe in agricultural operations (e.g. Koga et al., 2003), which has also been observed in Spain (Guardia et al., 2016). CT also has other impacts on the GHGe balance, which seem to be highly context-dependent. For example, it improves soil structure by promoting stratification, but its net effect on the net C storage is not clear (Powlson et al., 2014). CT may also increase N_2O emissions in dry climates, but decrease them in the long term (van Kessel et al., 2013). Other side effect of tillage practices must also be taken into account, such as negative impacts on yields (Pittelkow et al., 2015), a challenge that is even larger for herbaceous crops under organic management, as herbicides cannot be used (Peigne et al., 2007). Some promising approaches to weed management in Mediterranean organic systems are harrowing (Armengot et al., 2014) and cultivating old varieties (Carranza-Gallego et al., 2018). It is also worth noting that fuel consumption could be expected to be reduced in systems with higher SOC contents due to the reduction in the draught force required for tillage operations (Peltre et al., 2015). Higher SOC contents have been observed in organic and reduced tillage systems under Mediterranean conditions (Aguilera et al., 2013). Controlled traffic farming could also help reducing soil compaction and increasing fuel efficiency, by preventing the overlapping between machinery passes (Kroulik et al., 2011).

3.3. Mitigation potential

Biofuels for self-consumption in agriculture represent the modern version of animal traction, using biomass resources from the farm as the source of motive power (Bernesson et al., 2006; Fredriksson et al., 2006), but retaining the labor-saving benefits of mechanical traction and avoiding the need to maintain the animals while they are not working. The results of scenario Ethanol 1 (Fig. 8), however, warn us that a simple shift from diesel to ethanol would actually increase GHGe, due to the high burden of natural gas used for thermal energy and GHGe from barley production. These emissions are greatly reduced in Ethanol 2 scenario, leading to a 64% net GHG saving, with the 1900 EF of barley production, and straw used for thermal energy. In addition, however, these

Table 2
Total greenhouse gas emissions from working animals and mechanical traction in agriculture. Emissions are expressed as CO_2e per cultivated area, per unit C and N product and per unit installed power of each type of traction.

	1900	1910	1922	1933	1940	1950	1960	1970	1980	1990	2000	2008
Area-based emissions (kg CO_2e ha^{-1})												
Animals	185	206	242	232	264	266	279	242	84	31	10	0
Mechanical traction	2	2	4	11	16	23	76	349	513	404	493	540
Total traction	187	208	246	244	280	289	354	590	597	435	503	540
Product-based emissions (Mg CO_2e Mg^{-1})												
C production	0.36	0.37	0.44	0.41	0.54	0.54	0.54	0.73	0.60	0.39	0.39	0.40
N production	14.80	14.82	16.97	16.14	23.47	21.70	16.93	22.49	17.50	10.93	10.99	11.78

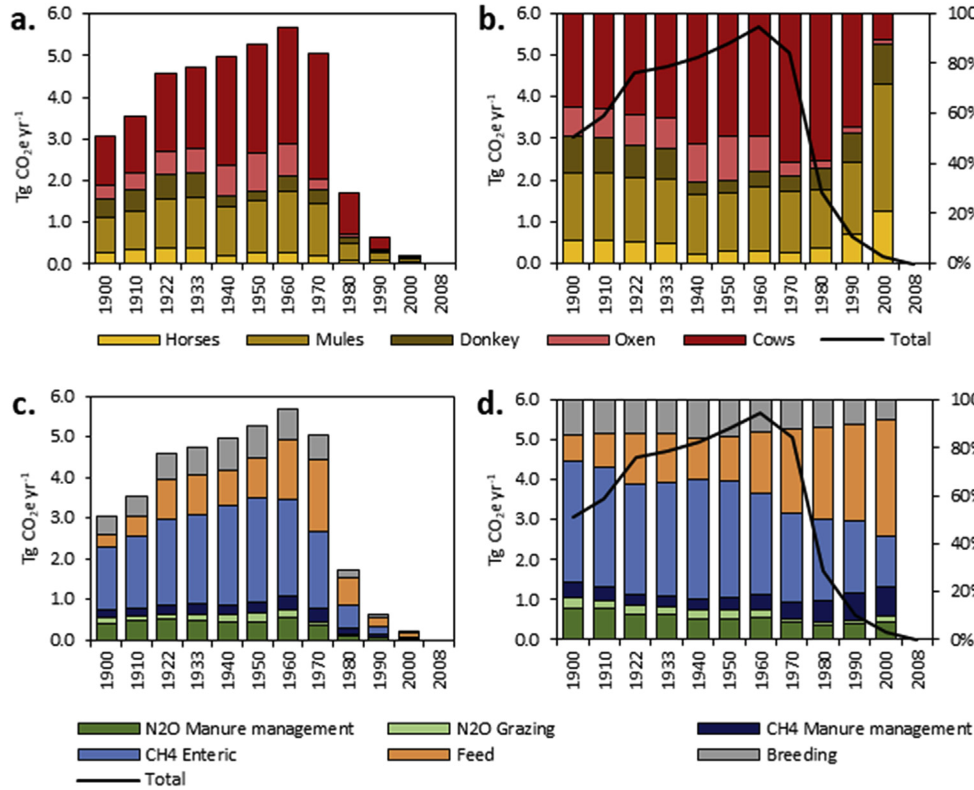

Fig. 4. Greenhouse gas emissions from animal traction in Spanish agriculture, 1900–2008 (Tg CO_2e yr^{-1}), by type of animal (a and b) and type of emission (c and d).

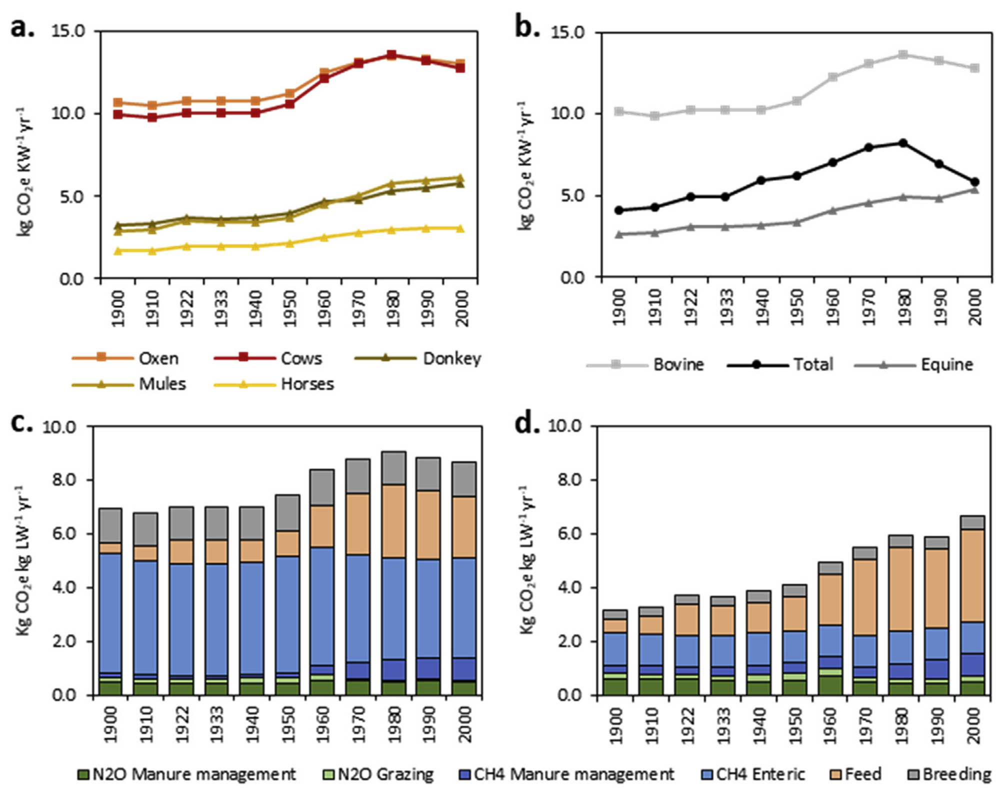

Fig. 5. Emission factors of working animals, per unit animal power (kg CO_2e KW^{-1} yr^{-1}), including total emissions of each animal species (a) and type (b), and per unit live weight (kg CO_2e kg LW^{-1} yr^{-1}), including emissions of bovines (c) and equines (d) by emission source.

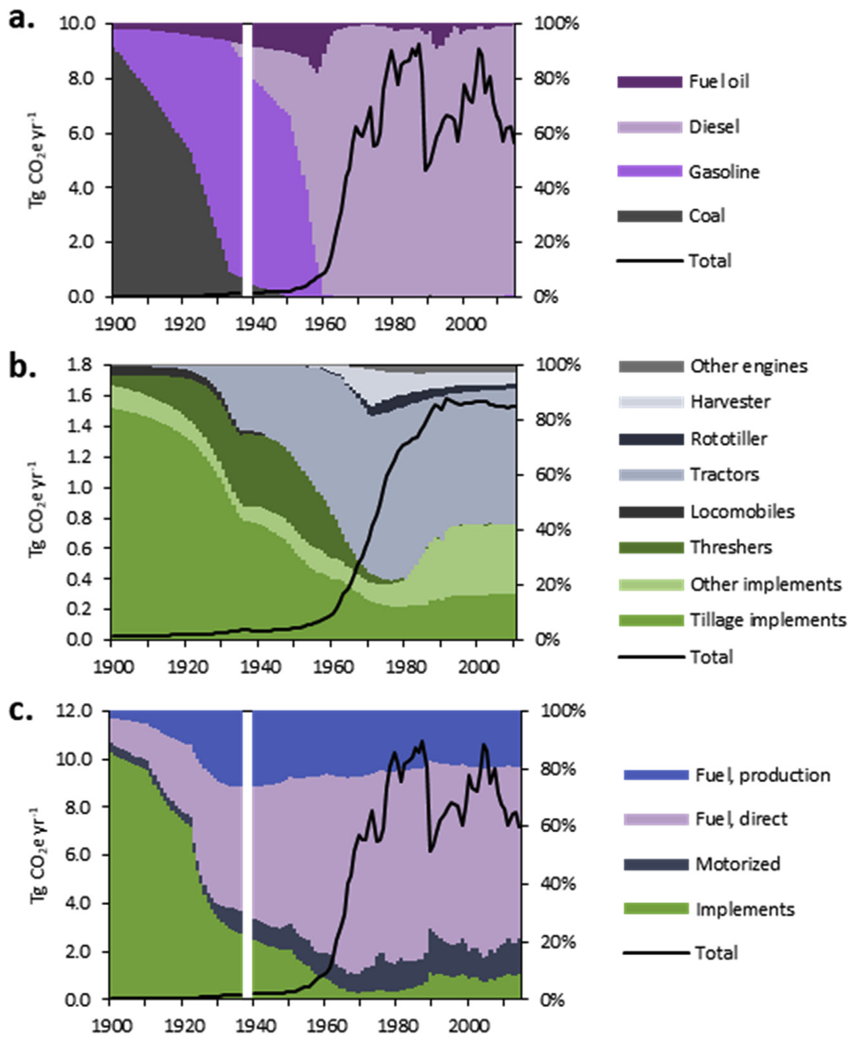

Fig. 6. Greenhouse gas emissions from mechanical traction in Spanish agriculture, including fuels, by fuel type (a), machinery, by machine type (b), and total emissions (c).

scenarios would require using 36% of all the cereal grain produced in Spain, and, in Ethanol 2 scenario, 26% of the straw. They would also produce DDGS and electricity as coproducts, which would decrease the environmental burden of ethanol to a varying degree (27% in our assessment with economic allocation), depending on the criteria used for allocation (Lechon et al., 2009). The high-protein coproduct (DDGS), combined with straw for animal feed, could help minimizing the loss of feeding capacity in systems that use feed products (grain and straw) to produce their own biofuel (Guzmán et al., 2011).

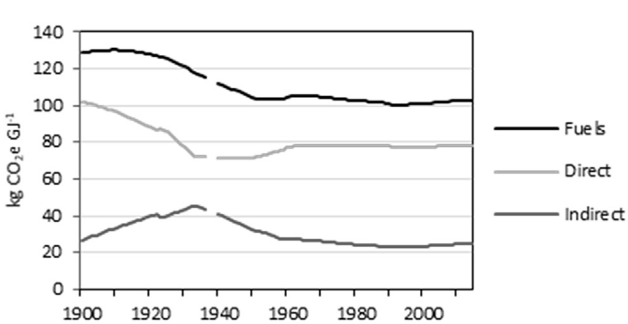

Fig. 7. Implicit emission factors of average fuels.

The disparity between the two situations selected is even higher for animal-based scenarios. The scenario Animal 1 represents a situation in which all traction power currently developed by machines is replaced by animal power. This scenario would roughly double GHGe, while using 87% of the total cereal grain production of the country, and 280% of the straw, which could be only partially mitigated (26–34%) by the animals' coproducts. By contrast, in the Animal 2 scenario only the 1900 area-based traction power is developed, while the feed carbon footprint is also that of 1900. In this situation, traction emissions are reduced by 56%, compared to current emissions, while 24% of the grain and 76% of the straw would be required for animal feed. However, the use of these feedstocks may provoke an increase in imports, potentially offsetting the GHG gains (which has not been modeled in this analysis), unless some structural changes are undergone in the food system, such as a shift to less animal products in the human diet. In any case, a large-scale transition to animal traction is very unlikely due to its labour requirements, given current prices of labour and crop products, while the lower power in the Animal 2 scenario is probably not enough to perform the tasks that support the current productivity. In this sense, the shift back to animal traction would probably only make sense for certain specialized tasks, such as weeding or cultivating in rough and sloping terrains. These large-scale transition scenarios only represent theoretical extreme

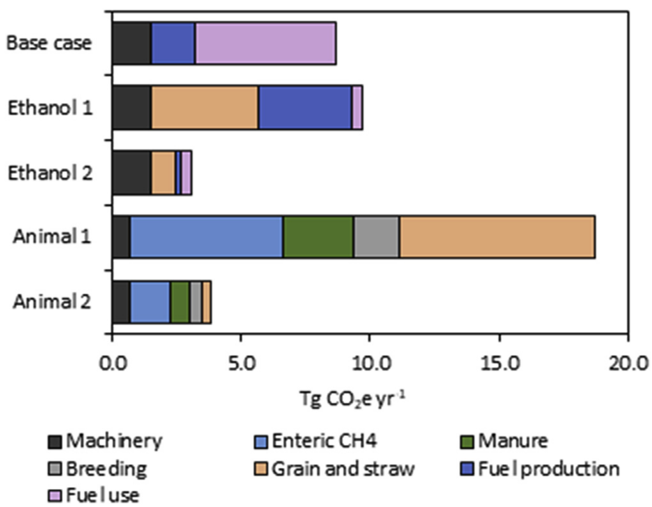

Fig. 8. Greenhouse gas emissions from agricultural traction in alternative scenarios, compared to emissions in 2008 using diesel fuel (Base case). Ethanol 1 represents ethanol production based on natural gas for thermal energy and with the 2008 crop EF; Ethanol 2 uses straw for thermal energy and the 1900 crop EF; in Animal 1, the traction power output of 2008 is supplied by the working animals' mix of 1900, using the 2008 crop EF; in Animal 2, the area-based power output, the animal mix and the crop EF of 1900.

boundaries, but inform us about the biophysical implications of two major alternatives to diesel fuel, namely self-produced biofuel and the recovery of working animals. They indicate that both types of strategy may bring significant net GHG savings, but also that realizing this potential would require major changes in production and consumption, and an appropriate integration of the coproducts in the system. Otherwise, large increases in GHG emissions and other impacts could be promoted. In particular, our results call for the need for reducing fuel consumption in mechanical traction, which would greatly improve the feasibility of the scenarios. It is worth noting the high share of straw required in the Animal scenarios, which reflects the low straw production with modern cereal varieties. The recovery of old wheat varieties could help producing more straw while maintain modern wheat productivity in semi-arid rainfed systems (Carranza-Gallego et al., 2018). Another possibility for increasing traction's sustainability would be electric-based traction, which could perform all the functions of conventional tractors, with a competitive cost for some specific, high quality agricultural processes, although high power operations would still be uneconomical (Bardi et al., 2013).

4. Conclusions

Our results underline the value of long-term (>1 century) studies to understand the environmental impact of socio-ecological transitions, unveiling processes that are not captured by shorter-term analyses. Most historical studies on agricultural C footprints only cover the last decades, when they observe a decrease in the emission intensity of crop production. Our results, however, indicate peak emission levels during transition from traditional organic to industrialized agriculture, and show that the decrease in the last decades have not been enough to reach the low emission levels of traditional cropping systems. They also show a peak in emissions from traditional systems in this transition period, suggesting that industrialization provoked a disruption of the functioning of the remaining traditional systems, which affected their performance. Another major finding of our study is that both working animals and self-produced biofuels could be used today to reduce the carbon footprint of traction, but only if they are accompanied by significant reductions in traction energy use and in the carbon footprint of food production. Structural changes in the agro-food system, for example by reducing feed demand through changes in the human diet, may also be necessary to compensate for the significant amount of biomass used for biofuel production.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2019.02.247.

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