Synergies and trade-offs between the greenhouse gas emissions and biodiversity performances of global livestock production

Félix Teillard[1,2,*], Alessandra Falcucci[1], Pierre J. Gerber[1]

[1] Food and Agriculture Organization of the United Nations, Animal Production and Health Division, Rome, Italy
[2] INRA-AgroParisTech, UMR 1048 SAD APT, F-75005 Paris, France
[*] Corresponding author. E-mail: *felix.teillard@fao.org*

ABSTRACT

We provide the first global environmental assessment of livestock production that includes both greenhouse gas (GHG) emissions and biodiversity criteria. We compared performances on these two environmental criteria across scales, commodities (dairy and beef cattle) and production systems (grassland and mixed). To do this, we combined a global model computing the greenhouse gas emissions of livestock with the Mean Species Abundance biodiversity indicator to quantify the biodiversity impact of livestock through land use. Results showed weaker synergies and more trade-offs between environmental criteria in grassland than in mixed production systems. Efficiency in the utilization of feed and their associated land use is likely to drive the synergies in mixed production systems. Grassland systems based on extensive feed land use with high biodiversity values may have contrasted GHG emissions performances. Our global mapping of the relationships between environmental criteria could be used for spatially targeting decisions and actions.

Keywords: Environmental sustainability, Life Cycle Assessment, multi-criteria, intensity

1. Introduction

Livestock production faces a challenge in satisfying an increasing food demand while improving its environmental sustainability. In the next decades, global population growth, urbanization and higher incomes will lead to a 70% growth in the demand for animal products (FAO 2011). On a global scale, livestock are major contributors to food security but also to environmental impacts such as greenhouse gas (GHG) emissions (Gerber et al. 2013), water pollution (Carpenter et al. 1998) or biodiversity loss (Steinfeld et al. 2006). Large scale assessments of the environmental impacts of livestock reveal both hotspots of impact and sustainable systems; they are thus key to effective action and improvement.

Most of the recent global assessments of the livestock environmental performances focused on GHG emissions. They provided consistent results for the emission associated with different types of livestock products and for the hotspots along the supply chain (e.g., review in De Vries and De Boer 2010). Relying on these assessments, technical (Smith et al. 2008; Garnett 2009) and policy (Gerber et al. 2010; Steinfeld and Gerber 2010) options have been proposed to mitigate the contribution of livestock to climate change. Yet, environmental impacts of livestock production are not restricted to GHG emissions and carbon footprint cannot be used as an indicator of the overall environmental impact (e.g., for meat production, Röös et al. 2013).

In particular, livestock have a very strong impact on biodiversity. The main global driver of biodiversity loss is habitat change (MEA 2005; Foley et al. 2005). As major users of land resources, livestock have a strong contribution to this driver. About 30% of the global area is currently dedicated to livestock production through pastures and feed crops (Ramankutty et al. 2008). The Amazonian forest may host up to a quarter of the world's terrestrial species (Dirzo and Raven 2003). Its conversion to pastures (representing 85% of the new agricultural lands, Steinfeld et al. 2006) and soybean crops is an important threat to biodiversity. In Europe, grassland intensification and conversion to cropland have caused an important decline of farmland species (Vickery et al. 2001; Firbank et al. 2008). Overgrazing is an important factor causing habitat degradation through desertification and woody encroachment in arid rangeland system, which leads to decrease in the species richness of plant communities (Milton and Dean 1995; Asner et al. 2004).

Although there is large evidence of the global impact of livestock on biodiversity, very few quantifications exist and including biodiversity impacts in LCAs is still an emerging area of work. A framework (Mila i Canals et al. 2007; Koellner et al. 2013) and several characterization factors (review in Curran et al. 2011) have been proposed to compute biodiversity impacts through land use in LCAs. Most of the characterization factors are available at country to region scale (Koellner and Scholz 2008; Schmidt 2008; Goedkoop et al. 2012). At global scale, Koellner et al. (2013) proposed a standardized land use classification for computing biodiversity characterization factors (as part of the UNEP-SETAC Life Cycle Initiative). De Baan et al. (2013) relied on this classification to quantify the land use impact on biodiversity with a Biodiversity Damage Potential characterization factor. This study did not describe several levels of intensity within the pasture/meadow and annual crop land use classes, which is a limitation for quantifying the impact of livestock production with precision. Alkemade et al. (2009) developed a Mean Species Abundance (MSA) indicator and computed its value at global scale for various land use and intensity classes. Authors did not used the MSA within an LCA perspective but to predict the effect of global socio-economic scenarios on biodiversity, certain scenarios specifically addressing livestock production (Westhoek et al. 2011; Alkemade et al. 2012).

Most large scale environmental assessments have focused on one environmental criteria, and chiefly on GHG emissions. Both synergies and trade-offs are however likely to exist between the performances on GHG emissions and on other environmental impact categories, such as biodiversity. For instance, grassland systems often involve lower feed digestibility and thus higher enteric CH4 emissions (Eckard et al. 2010). However, in some regions grassland systems are crucial for maintaining rich biodiversity habitats, and both intensification and abandonment lead to the loss of a unique pool of species (Bignal and McCracken 2000). Quantifying both GHG emissions and biodiversity impacts is important to reveal how policy options targeting one criteria will involve benefits or conflicts with the other criteria.

The objective of this paper was to compare the GHG emissions and biodiversity impact of livestock production on a global scale. We combined the GLEAM model (Global Livestock Environmental Assessment Model, Gerber et al. 2013; Opio et al. 2013) which computes the global GHG emissions of livestock, with the MSA methodology in order to quantify the global biodiversity impact of livestock through land use. We investigated the relationship between performances on these two environmental criteria among commodities, production systems, and across scales.

2. Methods

2.1. Overview

The methodology was based on GLEAM which models global livestock supply chains in details and computes the GHG emission (Gerber et al. 2013, Section 2.2.). Computing the land use for feed is an intermediary output of the model (Figure 1). We used this intermediary output to develop a new component of GLEAM, which estimated the impact of livestock on biodiversity through land use. For this biodiversity component, we relied on the MSA methodology which provides a biodiversity value (expressed as Mean Species Abundance) for several classes of land use and intensity (Alkemade et al. 2009; 2012, Section 2.3.).

All computations addressed the global scale and were based on GIS raster layers with a resolution of 3 arc minutes (5*5km at the equator). The year of reference was 2005. We focused on a single species (cattle) but we described two commodities (milk and meat) and two production systems (grassland and mixed). For the dairy herd producing meat as a co product, no allocation was performed; environmental impacts (GHG emissions and MSA impact) were expressed by kg of proteins, summing proteins from milk and from meat.

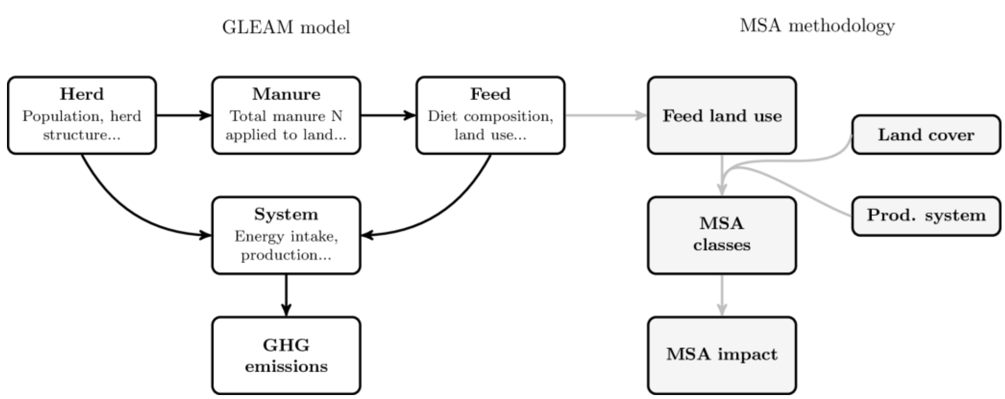


Figure 1. Overview of the modeling procedure used to compute GHG emissions, through the Global Livestock Environmental Assessment Model (GLEAM, Gerber et al. 2013) and the biodiversity impact, through the Mean Species Abundance methodology (MSA, Alkemade et al. 2009; 2012). Prod. = production. Adapted from Gerber et al. 2013.

2.2. GHG emissions: the GLEAM model

GLEAM is a novel modeling framework that enables a comprehensive analysis of the emissions of global livestock production (Gerber et al. 2013). It provides disaggregated estimates of the GHG emissions for the main commodities, production systems and world regions. The main GHGs in the agriculture context – CH_4, N_2O and CO_2 – are accounted for. GLEAM is built on modules reproducing the main elements of livestock supply chain: the herd module, the manure module, the feed module and the system module (Figure 1).

The GHG emissions results included emissions from feed production (main sources of emissions: N_2O from fertilization; CO_2, N_2O and CH_4 from energy use, fertilizer manufacture and land use change related to soybean cultivation) and livestock production (main sources of emissions: CH_4 from enteric fermentation and manure management; N_2O from manure management; CO_2 from on-farm energy use for livestock). For comparison with biodiversity impact through land use, we excluded emissions from manufacture of on-farm building and equipment, and post farmgate emissions. For a detailed description of the GLEAM model, refer to Gerber et al. (2013) and Opio et al. (2013).

2.3. Biodiversity impact: the MSA methodology

In order to compute the MSA values of different land use and intensity classes, Alkemade et al. (2009; 2012) conducted a meta-analysis and selected articles that presented data on species composition in disturbed (occupied) vs undisturbed (reference) land uses. No selection of specific species groups was performed; studies included in the meta-analysis addressed both plants and animals (mainly birds, mammals and insects). For each species *k* within each occupied land use *i*, the ration $R_{i,k}$ was calculated as:

$$R_{i,k} = \begin{cases} \dfrac{n_{i,k}}{n_{ref,k}} & if\ n_{i,k} < n_{ref,k} \\ 1, & otherwise \end{cases} \qquad \text{Eq. 1}$$

where $n_{i,k}$ is the abundance of the species *k* in an occupied land use *i* and $n_{ref,k}$ its abundance in the reference land use. The MSA of any occupied land use MSA_i is then calculated by summing and weighting the ratios $R_{i,k}$ of each species:

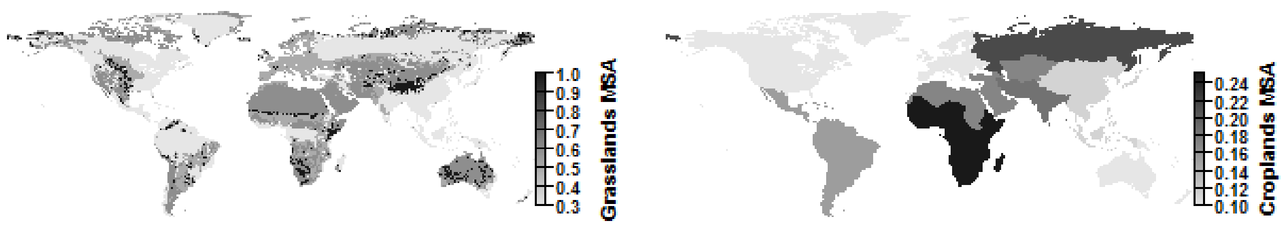
$$MSA_i = \frac{\sum_k \left(R_{i,k} / V_{i,k} \right)}{\sum_k 1/V_{i,k}} \qquad \text{Eq. 2}$$

where $V_{i,k}$ is the variance of the ratios of species abundances for each study and copes for differences between studies. MSA values vary between 0 and 1. MSA = 1 in undisturbed ecosystems where 100% of the original species abundances remains, conversely, MSA = 0 in a destroyed ecosystem with no original species left.

Table 1. Mean Species Abundance (MSA) value of the different land use and intensity classes of rangelands/grasslands, and croplands (Alkemade et al. 2009; 2012).

Land use and intensity classes	MSA value
Rangelands/grasslands	
Natural rangelands	1
Moderately used rangelands	0.6
Intensively used rangelands	0.5
Man-made grasslands	0.3
Croplands	
Low input agriculture	0.3
Intensive agriculture	0.1

Table 1 shows the MSA values of the different land use and intensity classes. Feed land uses as computed by the GLEAM model were translated into MSA land use classes (Figure 1). For each grid cell, we allocated a rangeland/grassland class (Figure 2a), and a cropland class (Figure 2b), corresponding to MSA values. For distinguishing between the different rangeland/grassland classes, we used information from three different layers mapping potential vegetation (ecoregions, Olson et al. 2001); global land cover (GLC 2000) and the distribution of grassland vs. mixed production systems (Gerber et al. 2013). We used the following hierarchic rules. The man-made grasslands class (MSA = 0.3) was allocated to grid cells with forest as potential vegetation, except for Europe where although forest is the potential vegetation, grasslands are sufficiently old to host specifically adapted species and very high biodiversity levels (Bignal and McCracken 1996; Benton et al. 2002). The natural rangelands class (MSA = 1) was allocated to grid cells with herbaceous land cover and grassland production systems. For grid cells with non herbaceous land cover (e.g., crops, crops/grass mosaic), the moderately used rangelands class (MSA = 0.6) was allocated to grassland production systems while the intensively used rangelands class (MSA = 0.5) was allocated to mixed production systems.



(a) (b)

Figure 1. Mean Species Abundance (MSA) value attributed to (a) the grasslands and (b) the croplands of each grid cell.

We further computed a characterization factor illustrating the impact of livestock production on MSA in each grid cell of the global raster:

$$MSA\,impact = \sum_i (1 - MSA_i) \times Area_i \qquad \text{Eq. 3}$$

where *(1-MSA$_i$)* stands for the loss of MSA in land use *i* compared to the reference land use. For instance, an MSA value of 0.6 in extensive grasslands means that the MSA loss *(1-MSA$_i$)* is 0.4, i.e. that 40% of the mean species abundance is lost compared to the reference land use. *Area$_i$* is the area of land use class *i* necessary to produce the feed consumed by cattle in the grid cell. Therefore, the MSA impact of land use for feed is not allocated where feed is produced but where it is consumed. For instance, the MSA impact of soybean cultivated in brazil and consumed by cattle in a given European grid cell will be allocated in this European grid cell. As a consequence, livestock production in a given grid cell could have a land use impact on a larger area than the grid cell area. The MSA impact is expressed as an *MSA loss * km^2* and then divided by the kg of protein produced.

2.4. Spatial analyses

We computed average GHG emissions and MSA impacts across grid cells, at the level of agro-ecological zones (intersection of climate zones and global regions, Fischer2008) and climate zones (arid, humid, temperate).

We investigated the relationship between GHG emissions and MSA impact environmental criteria at local (grid cell) scale. For the two environmental criteria, the environmental impact of a grid cell was compared with the average impact at sub-regional level (moving window average). If the impact on the two environmental criteria were both lower or both higher than the sub-regional average, we considered that there was a synergy between criteria for the grid cell. Conversely, if the impact was higher than the regional average for one environmental criteria and lower for the other, we considered that there was a trade-off between environmental criteria for the grid cell.

All analyses were performed using the R software, version 3.0.3 (R Core Team 2014). We used the raster package (Hijmans 2014) to perform GIS analyses.

3. Results

3.1. Relationship between environmental criteria across agro-ecological zones

As a general trend across agro-ecological zones, there was a synergy between the climate change and biodiversity performances: agro-ecological zones with lower GHG emissions also tended to have lower MSA impacts (Figure 3). This general trend was observed similarly for dairy (Figure 3a) and beef (Figure 3b) cattle, and for grassland and mixed production systems. For both commodities, the correlation between GHG emissions and MSA impact was much lower in grassland production systems (R^2 = 0.2936 and 0.2295 for dairy and beef cattle, respectively) than in mixed production systems (R^2 = 0.8577 and 0.7658 for dairy and beef cattle, respectively).

Figure 3 also shows that environmental impact on the two criteria was higher for beef cattle than for dairy cattle. For the two commodities, grassland production systems had a slightly higher MSA impact than mixed production system. In terms of GHG emissions, the difference between production systems was very small.

Environmental impacts on the two criteria across climate zones were similar for the two commodities (results not show). Environmental impacts were the highest in arid regions and the lowest in temperate regions. Humid regions had GHG emissions levels close to those of arid regions while they had an MSA impact intermediate between those of arid and temperate regions.

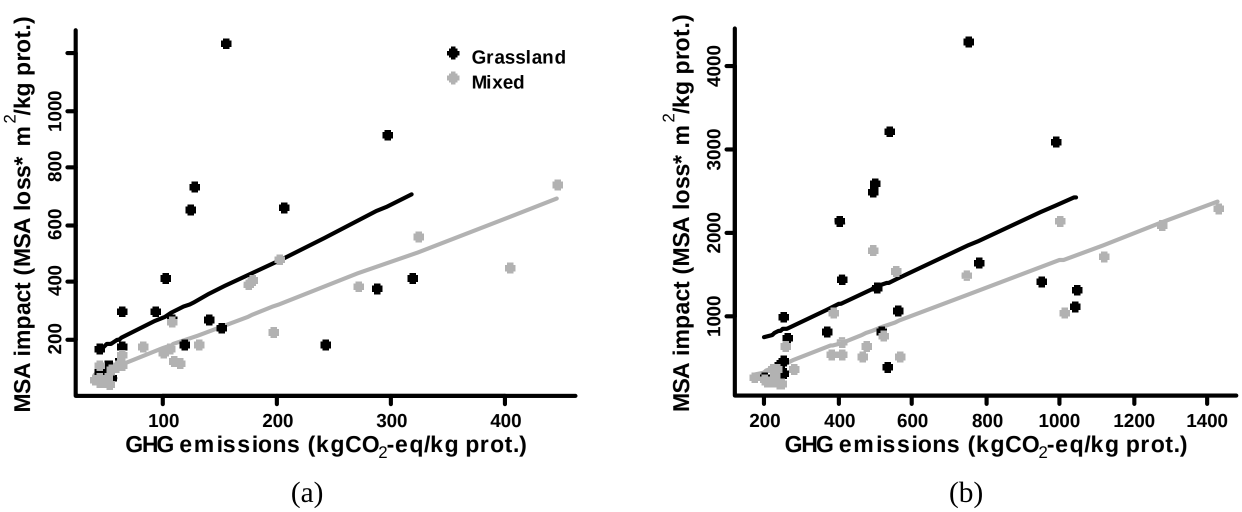
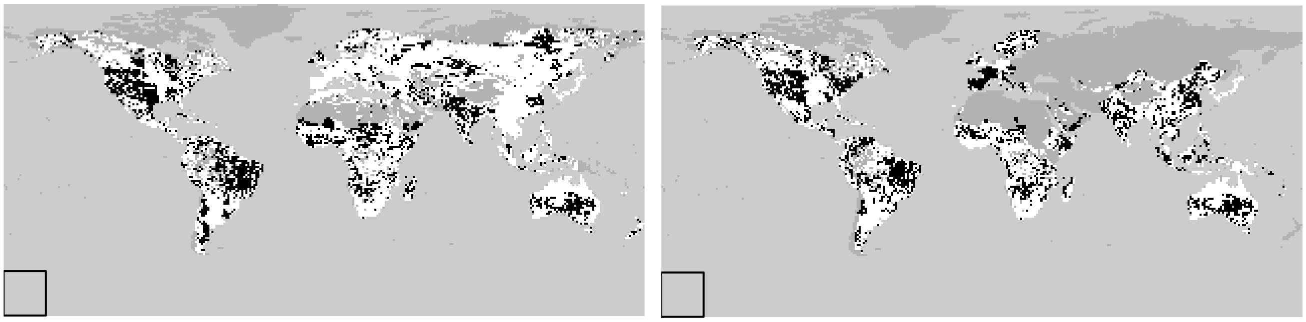


Figure 3. Relationship between GHG emissions and MSA impact per unit of production across agro-ecological zones. (a) Dairy cattle; (b) Beef cattle. Each point stands for the mean GHG emissions and MSA impact of one production system (see legend), within one agro-ecological zone. Regression lines are plotted. Prot. = protein.

3.2. Relationship between environmental criteria at local scale

Figure 4 shows the global distribution of synergies and trade-offs between GHG emissions and MSA impact. For both commodities, regions with a higher concentration of trade-offs included brazil and the western part of North America, as well as India for dairy production and western Europe for beef production. The rest of Europe Asia, and Africa showed more uniform proportions between synergies and trade-offs. Trade-offs tended to be more frequent in arid climate, grassland production systems and areas with high MSA values while synergies tended to be more frequent in temperate systems and area with higher yields.



(a) (b)

Figure 4. Relationship between GHG emissions and MSA impact per unit of production at local (grid cell) scale. (a) Dairy cattle; (b) Beef cattle. White = impact on the two environmental criteria are both lower or both higher than the sub-regional average (synergy). Black = impact is higher than the regional average for one environmental criteria and lower for the other (trade-off). Grey = no production. Black square = size of the moving window for the sub-regional averages.

4. Discussion

4.1. The MSA indicator

The MSA indicator is one of the very few biodiversity characterization factors for land use available at global scale. One limitation of the MSA indicator is that it does not make it possible to account for potential differences in conservation value, at the species and at the ecosystem levels. At the species levels, all species groups were included in the MSA. The MSA indicator is thus expected to be a good proxy for the overall biodiversity, although it is based on published literature where certain taxa are underrepresented (e.g., arthropods). However, common species and patrimonial or red listed species have the same contribution to the MSA. The IUCN red list is a widely recognized system for classifying species according to their risk of extinction and biodiversity indicators based on the red list are a useful tool for targeting conservation actions (Butchart et al. 2004). Characterization factors can be computed separately for common and red-listed species (e.g., in Koellner and Scholz 2008 for central Europe) but this distinction is not made in the MSA. Besides, while most biodiversity characterization factors are based on species richness (e.g., Koellner and Scholz 2008; Michelsen 2008; Schmidt 2008; Goedkoop et al. 2012), the MSA is based on species abundance which does not capture information about species extinction.

At the ecosystem level, the MSA value of each land use and intensity class is global and does not account for regional differences. It means that the biodiversity value of undisturbed forest – or the biodiversity loss following its conversion to pasture – is the same in Siberia and Amazonia for example. Yet, it is recognized that forests in certain specific areas are biodiversity hotspots (e.g., Amazonia, Dirzo and Raven 2003). The framework developed by the UNEP-SETAC Life Cycle Initiative (Koellner et al. 2013) includes a structure for including such regional differences, i.e. for the regionalization of land use elementary flows. Characterization factors computed by De Baan et al. (2013) follow this regionalization structure. The MSA indicator includes more precise land use and intensity classes, specifically adapted to livestock production. However, the absence of regionalization is a limitation. In the context of livestock production, the biodiversity value of grazing lands of varying intensity is very likely to differ between global regions. We accounted for one of these difference by making the assumption of extra MSA value to European grassland (compared to Alkemade et al. 2009; 2012) because they are very old ecosystems, although located in a forest ecoregion. Other differences were not considered; for instance, the management intensity threshold leading to rangeland degradation is lower in humid and arid regions than in temperate regions (Asner et al. 2004). Using information on rangeland productivity and livestock density would be an interesting development to regionalize the MSA values of grasslands and rangelands.

The UNEP-SETAC Life Cycle initiative recognizes that the effects of land use on biodiversity can be divided in three main stages that occur successively over time: land transformation, land occupation and land restoration (Lindeijer 2000; Mila i Canals et al. 2007). In this study, we only focused on occupation impacts. Calculating transformation impacts (e.g., impacts from land use change) and permanent impacts (i.e. impacts that cannot be recovered even after restoration) requires region and ecosystem specific data on the time and success of restoration (De Baan et al. 2013). Such data was not available; moreover the GLEAM model that we used to compute land occupation is a static model and its input datasets are not available as time series.

4.2. Impact categories coverage

The MEA (2005) recognizes five main driver of biodiversity loss at global scale: habitat change, climate change, pollution, invasive species and overexploitation. On a global scale, livestock production contributes directly or indirectly to each of these five drivers (Steinfeld et al. 2006). We focused on the habitat change driver and described it through land use only, which does not cover other of its components such as spatial heterogeneity and habitat fragmentation. This focus on land use leads our results to underestimate the overall biodiversity impacts. It also overemphasizes the role of productivity: productivity gains at feed production and animal level both strongly drive the calculated MSA impact. This results in a bias in favor of high productivity systems. Land use is expected to be the main driver of impact on biodiversity for extensive grassland systems which use large area to generate one unit of product because the conversion of grass to animal protein is rather inefficient (Wirsenius et al. 2011). Mixed systems and intensive grassland systems show higher yields, they need less area which limits their impact on biodiversity through land use, despite lower MSA values. However, they often involve important pollution that causes significant biodiversity impacts which are not captured in our results. This pollution includes two main categories. The first one is nutrient pollution which is associated with animal concentration (Peyraud et al. 2012) and can lead to biodiversity loss through acidification and eutrophication in soils and water (Carpenter et al. 1998). The second one is associated with higher input intensity; it is the release of ecotoxic components in the environment, mainly pesticides (at the feed production stage) and veterinary products (including hormones, at the animal husbandry stage). Hormonally active pesticides cause adverse effects on a wide range of organisms (Colborn *et al.*, 1993) and have recently been pointed as one of the responsible of bee population decline (vanEngelsdorp and Meixner 2010). Several veterinary products used on livestock have also been shown to impact biodiversity, such as anti-inflammatory drugs (Baillie 2004), hormones (Soto et al. 2004) or anthelmintics (Lumaret and Errouissi 2002).

Our results did not show significant differences in GHG emissions between grassland and mixed production systems. Adding the impact of climate change on biodiversity should not lead to important changes in the relative impact of grassland vs. mixed production systems on biodiversity. However, it could reveal new synergies between the two environmental criteria.

Many LCA studies on livestock focused on climate change as a single environmental criteria (see review in De Vries and De Boer IJM 2010). Adding land use impacts on biodiversity to the climate change impact is still far from a complete coverage of all the categories relevant to the environmental performance of livestock production at global scale. Interestingly, most of these additional categories are midpoint impacts which also have an effect on biodiversity. Including their assessment and their effect on biodiversity would be an important further development of our approach. Characterization factors to model their effect on biodiversity in LCA at global scale are still rare. In the MSA methodology, biodiversity characterization factors are also available for the impact of fragmentation and climate change (Alkemade et al. 2009). Other global characterization factors exist for climate change (Schryver et al. 2009), water use (Pfister et al. 2009) and ecotoxicity (Rosenbaum et al. 2008) while country to region characterization factors exist for eutrophication and acidification (Van Zelm et al. 2007; Struijs et al. 2011).

4.3. Implications

Across agro-ecological zones and at local scale, there was a weaker correlation and more trade-offs between GHG emissions and MSA impact in grassland production systems than in mixed production systems. In mixed production systems, feed mainly come from intensive land uses with low MSA values. Efficiency in the utilization of feed is a way to improve both environmental criteria. It makes it possible to use less area of intensive feed land uses with low MSA values, which also decreases GHG emissions associated with feed cultivation. At the same time, production is increased which leads to lower environmental impact per unit of product. In grassland production systems however, another option than efficiency to mitigate biodiversity impacts is to use more extensive feed land uses with high MSA values. This option can have contrasted effects on GHG emissions performances because more extensive systems have lower production levels and can be associated with higher emissions from manure deposition and enteric fermentation (Gerber et al. 2011).

By mapping synergies and trade-off between two environmental criteria on a global scale and at fine resolution, our method could provide a useful tool for spatially targeting interventions, or further investigations of the farming system properties. We reveal different relationships between the GHG emissions and biodiversity performances. The systems where performances on both criteria are higher in the surrounding region could be studied in order to apply beneficial management options to other systems, e.g., to those were performances on both criteria are lower than those of the neighboring systems. Systems were only one environmental criteria performs better than the surrounding region could reveal trade-offs between criteria that needs to be considered when designing interventions. Limitations of the implications of our results at the grid cell level include that certain parameters of the model only have a country resolution (e.g., animal's ration in OECD countries), and that management decisions could have different constraints across grid cells, even within a sub-region.

5. Conclusion

This study provides a tentative global quantitative assessment of the environmental performances of livestock production on two criteria: GHG emissions and biodiversity. It is a first attempt to develop multi-criteria assessments over such large scale, which are key to inform decision and action towards improving the overall sustainability of the livestock sector. Our preliminary results show that both synergies and trade-offs exist between the performances on the GHG emissions and biodiversity criteria. With our approach, more frequent and stronger synergies were found in mixed production systems where efficiency (i.e. decreasing the feed land use area while increasing production) could be a way to improve performances on both GHG emissions and biodiversity criteria. Weaker synergies and more trade offs were found in grassland production systems. Further developments, and testing of our approach are however required before results can be used for decision making. Improvements would include the inclusion of additional and regionalized biodiversity impacts.

6. References

Alkemade R, Oorschot M, Miles L, et al. (2009) GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss. Ecosystems 12:374–390.

Alkemade R, Reid RS, Van den Berg M, et al. (2012) Assessing the impacts of livestock production on biodiversity in rangeland ecosystems. Proceedings of the National Academy of Sciences of the United States of America.

Asner GP, Elmore AJ, Olander LP, et al. (2004) Grazing Systems, Ecosystem Responses, and Global Change. Annual Review of Environment and Resources 29:261–299.

De Baan L, Alkemade R, Koellner T (2013) Land use impacts on biodiversity in LCA: a global approach. The International Journal of Life Cycle Assessment 18:1216–1230.

Baillie J (2004) A 2004 IUCN Red list of threatened species: A Global Species Assessment. Tech. rep., UICN.

Benton TG, Bryant DM, Cole L, Crick HQP (2002) Linking agricultural practice to insect and bird populations: a historical study over three decades. Journal of Applied Ecology 39:673–687.

Bignal EM, McCracken DI (2000) The nature conservation value of European traditional farming systems. Environmental Reviews 8:149–171.

Bignal EM, McCracken DI (1996) Low-Intensity Farming Systems in the Conservation of the Countryside. Journal of Applied Ecology 33:413–424.

Butchart SHM, Stattersfield AJ, Bennun L a, et al. (2004) Measuring global trends in the status of biodiversity: red list indices for birds. PLoS biology 2:e383.

Carpenter S, Caraco N, Correll R, et al. (1998) Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8:559–568.

Colborn T, vom Saal F, Soto A (1993) Developmental effects of endocrine-disrupting chemicals in wildlife and humans. Environmental health perspectives, 101, 378–384.

Curran M, De Baan L, De Schryver AM, et al. (2011) Toward meaningful end points of biodiversity in life cycle assessment. Environmental science & technology 45:70–9.

Dirzo R, Raven PH (2003) Global State of Biodiversity and Loss. Annual Review of Environment and Resources 28:137–167.

Eckard RJ, Grainger C, De Klein C a. M (2010) Options for the abatement of methane and nitrous oxide from ruminant production: A review. Livestock Science 130:47–56.

FAO (2011) World livestock 2011 - livestock in food security. Food and Agriculture Organization of the United Nations (FAO), Rome.

Firbank LG, Petit S, Smart S, et al. (2008) Assessing the impacts of agricultural intensification on biodiversity: a British perspective. Philosophical transactions of the Royal Society of London Series B, Biological sciences 363:777–87.

Foley J a, Defries R, Asner GP, et al. (2005) Global consequences of land use. Science (New York, NY) 309:570–4.

Garnett T (2009) Livestock-related greenhouse gas emissions: impacts and options for policy makers. Environmental Science & Policy 12:491–503.

Gerber P, Key N, Portet F, Steinfeld H (2010) Policy options in addressing livestock's contribution to climate change. Animal : an international journal of animal bioscience 4:393–406.

Gerber P, Steinfeld H, Henderson B, et al. (2013) Tackling climate change through livestock: a global assessment of emissions and mitigation opportunities. Food and Agriculture Organization of the United Nations (FAO), Rome.

Gerber P, Vellinga T, Opio C, Steinfeld H (2011) Productivity gains and greenhouse gas emissions intensity in dairy systems. Livestock Science 139:100–108.

Goedkoop M, Huijbregts M, Heijungs R, et al. (2012) ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First edition (revised).

Hijmans RJ (2014) raster: Geographic data analysis and modeling.

Koellner T, Baan L, Beck T, et al. (2013) Principles for life cycle inventories of land use on a global scale. The International Journal of Life Cycle Assessment 18:1203–1215.

Koellner T, Scholz R (2008) Assessment of land use impacts on the natural environment-Part 2: Generic characterization factors for local species diversity in Central Europe. International Journal of Life Cycle Assessment 13:32–48.

Lindeijer E (2000) Biodiversity and life support impacts of land use in LCA. Journal of Cleaner Production 8:313–319.

Lumaret JP, Errouissi F (2002) Use of anthelmintics in herbivores and evaluation of risks for the non target fauna of pastures. Veterinary Research, 33(5), 547-562.

MEA (2005) Ecosystems and human well-being. Millenium Ecosystem Assessment. Washington.

Michelsen O (2008) Assessment of Land Use Impact on Biodiversity. Proposal of a new methodology exemplified with forestry operations in Norway. International Journal of Life Cycle Assessment 13:22–31.

Mila i Canals L, Bauer C, Depestele J, et al. (2007) Key elements in a framework for land use impact assessment within LCA. International Journal of Life Cycle Assessment 12:5–15.

Milton SJ, Dean WRJ (1995) South Africa's arid and semiarid rangelands: why are they changing and can they be restored? Desertification in Developed Countries. Springer, pp 245–264

Olson DM, Dinerstein E, Wikramanayake ED, et al. (2001) Terrestrial Ecoregions of the World: A New Map of Life on Earth. BioScience 51:933.

Opio C, Gerber P, Mottet A, et al. (2013) Greenhouse gas emissions from ruminant supply chains - A global life cycle assessment. Food and Agriculture Organization of the United Nations (FAO), Rome.

Peyraud JL, Cellier P, Aarts F, et al. (2012) Les flux d'azote liés aux élevage: réduire les pertes, rétablir les équilibres. Expertises Scientifiques collectives, synthèse du rapport, INRA (France).

Pfister S, Koehler A, Hellweg S (2009) Assessing the environmental impacts of freshwater consumption in LCA. Environmental science & technology 43:4098–104.

R Core Team (2014) R: A Language and Environment for Statistical Computing.

Ramankutty N, Evan AT, Monfreda C, Foley J a. (2008) Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. Global Biogeochemical Cycles 22:1–19.

Röös E, Sundberg C, Tidåker P, et al. (2013) Can carbon footprint serve as an indicator of the environmental impact of meat production? Ecological Indicators 24:573–581.

Rosenbaum RK, Bachmann TM, Gold LS, Huijbregts MaJ, Jolliet O, Juraske R, Koehler A, Larsen HF, MacLeod M, Margni M, McKone TE, Payet J, Schuhmacher M, Meent D, Hauschild MZ (2008) USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. The International Journal of Life Cycle Assessment, 13, 532–546.

Schmidt JH (2008) Development of LCIA characterisation factors for land use impacts on biodiversity. Journal of Cleaner Production 16:1929–1942.

Schryver A De, Braakkee K, Goedkoop MJ, Huijbregts M a. J (2009) Characterization factors for global warming in life cycle assessment based on damages to humans and ecosystems. Environmental science & technology 43:1689–1695.

Smith P, Martino D, Cai Z, et al. (2008) Greenhouse gas mitigation in agriculture. Philosophical transactions of the Royal Society of London Series B, Biological sciences 363:789–813.

Soto AM, Calabro JM, Prechtl NV, Yau AY, Orlando EF, Daxenberger A, Kolok AS, Guillette Jr LJ, le Bizec B, Lange IG (2004) Androgenic and estrogenic activity in water bodies receiving cattle feedlot effluent in Eastern Nebraska, USA. Environmental health perspectives, 112, 346.

Steinfeld H, Gerber P (2010) Livestock production and the global environment: consume less or produce better? Proceedings of the National Academy of Sciences of the United States of America 107:18237–8.

Steinfeld H, Gerber P, Wassenaar TD, et al. (2006) Livestock's long shadow: environmental issues and options. Food and Agriculture Organization of the United Nations (FAO), Rome.

Struijs J, Beusen A, De Zwart D, Huijbregts M (2011) Characterization factors for inland water eutrophication at the damage level in life cycle impact assessment. The International Journal of Life Cycle Assessment 16:59–64.

Vickery J a., Tallowin JR, Feber RE, et al. (2001) The management of lowland neutral grasslands in Britain: effects of agricultural practices on birds and their food resources. Journal of Applied Ecology 38:647–664.

De Vries M, De Boer IJM (2010) Comparing environmental impacts for livestock products: A review of life cycle assessments. Livestock Science 128:1–11.

Westhoek H, Rood T, Van de Berg M, et al. (2011) The protein puzzle. The Hague

Wirsenius S, Hedenus F, Mohlin K (2011) Greenhouse gas taxes on animal food products: rationale, tax scheme and climate mitigation effects. Climatic Change 108:159–184.

vanEngelsdorp D, Meixner MD (2010) A historical review of managed honey bee populations in Europe and the United States and the factors that may affect them. Journal of invertebrate pathology, 103, S80-S95.

Van Zelm R, Huijbregts M a. J, Van Jaarsveld AS, et al. (2007) Time horizon dependent characterization factors for acidification in life-cycle assessment based on forest plant species occurrence in Europe. Environmental science & technology 41:922–927.

This paper is from:

Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector

LCA Food 2014

8-10 October 2014 - San Francisco

Rita Schenck and Douglas Huizenga, Editors
American Center for Life Cycle Assessment

The full proceedings document can be found here:
http://lcacenter.org/lcafood2014/proceedings/LCA_Food_2014_Proceedings.pdf

It should be cited as:

Schenck, R., Huizenga, D. (Eds.), 2014. Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2014), 8-10 October 2014, San Francisco, USA. ACLCA, Vashon, WA, USA.

Questions and comments can be addressed to: staff@lcacenter.org

ISBN: 978-0-9882145-7-6