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Exploration and mitigation of greenhouse gas emissions in ruminant and grassland systems

Landbauforschung – *Journal of Sustainable and Organic Agricultural Systems* is a peer-reviewed interdisciplinary journal for scientists concerned with new developments towards sustainable agricultural systems. Of special interest is the further development of agricultural systems to generally fulfil the sustainable development goals of the United Nations' Agenda 2030, and also of organic farming systems.

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Research articles present original new research results. The material should not have been previously published elsewhere. The novelty of results and their possible use in further development of sustainable and organic agricultural systems should be clearly claimed.

Review articles present new overviews generated from existing scientific literature to analyse the current state of knowledge. Conclusions on necessary consequences for further sustainable development of agricultural systems and research needs shall be drawn.

Position Papers present science-based opinions on new, or possibly disruptive, developments in sustainable agricultural systems. Authors should use scientific references to validate and approve arguments for a position. These papers shall allow the reader to understand controversial positions and to find an own position.

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CALL FOR PAPERS FOR THE SPECIAL ISSUE

Exploration and mitigation of greenhouse gas emissions in ruminant and grassland systems

Today the pure numbers of ruminants, their methane emissions, their worldwide feedstuff demand and the related emissions, as well as improper and inefficient use of their excrements make them one of the biggest sources for greenhouse gas emissions in agriculture. We also face grassland systems that are overfertilised, overgrazed and destroyed due to missing concepts for their use under protection of their ecosystem services and soil carbon stocks.

This situation still evokes many scientific questions how to build appropriate and climate friendly ruminant and grassland production systems in different parts of the world. Also we need to find ways to curb the consumers demand, e.g. for milk and meat from ruminants. We have to develop strategies for a rapid worldwide conversion of production and consumption to mitigate greenhouse gas emissions.

We would be very pleased if you could contribute to this issue. We publish (1) original research and review papers and (2) science-based position papers from all science disciplines. We are interested in worldwide experiences and views.

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Editorial

Exploration and mitigation of greenhouse gas emissions in ruminant and grassland systems

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Dear colleagues, authors, reviewers and readers!

In this first issue of Landbauforschung in 2019 we focus on the 'Exploration and mitigation of greenhouse gas emissions in ruminant and grassland systems'. In our *call for papers*, we highlighted the environmental and climate burdens related to methane and nitrogen emissions from worldwide ruminant production systems. We also addressed the role of soil carbon storage in grassland when it is threatened by mismanagement or when it is replaced by arable cultures. And on the other hand we highlighted that, more efficient management offers chances to improve ruminant productivity, to reduce its numbers worldwide and to alleviate its product-related climate burden. We asked scientists from all disciplines to contribute their knowledge on the multi-faceted role of ruminants and grassland and its management in today's international agriculture, and for efficient ways to mitigate their greenhouse gas emissions. Also we got some interesting research contributions from the 'International Conference on Agricultural Greenhouse Gas Emissions and Food Security: Connecting Research to Policy and Practice' that took place from September 10–13, 2018 in Berlin, Germany. Please find also a subsequently included article on pig production and greenhouse gases as accepted by the former scope of the journal.

One initiative of the new Landbauforschung editorial office, beyond the new focus as *Journal of Sustainable and Organic Agricultural Systems* from 2019 on, was to implement five new elements to increase visibility, quality and interest:

- A focus on a special topic announced in advance for each issue.
- A new 'Position Paper' format. This is a shorter scientific communication on ideas and new developments or demands or an innovative summary of developments.
- A strict international and interdisciplinary focus.
- A double-blind international review process with publication of all reviewers' comments on the website.
- Possibilities to place online comments to all forms of articles.

Particularly the new format 'Position Paper' challenged the authors to find focus. It was sometimes hard to claim the exclusivity and innovation of their position with the limited space available and at the same time to satisfy the reviewers' demands for scientific quality. We hope that with transparency in the review process we can show that discussions were already started here. We will also use this new format to stimulate scientific discussion and future topics.

With all mentioned changes it was a long but interesting path to finishing this first issue. We hope you enjoy the articles and position papers selected for publication. We would like to thank all reviewers who helped to select and improve the submissions. With their consent, all participating reviewers are listed at the end of this issue.

Hans Marten Paulsen and Claudia Heidecke

POSITION PAPER

Livestock and climate change: what are the options?

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Henning Steinfeld

KEYWORDS agri-food systems, livestock, climate change, ruminants

1 The tension around livestock

Our food and agriculture systems are both broken. On the food side: hunger, food insecurity and malnutrition are making a resurgence, while excess weight, obesity and diet-related diseases have become a global epidemic (FAO, IFAD, UNICEF, WFP, and WHO, 2019). On the agriculture side: growing resource depletion and rapidly accelerating environmental degradation are breaching planetary boundaries (Rockström et al., 2009; Steffen et al., 2015), most notably in the form of climate change (IPCC, 2019).

Livestock systems, in particular, have been singled out as a major driver of environmental change (FAO, 2006). Specifically, ruminant grazing systems play a major role in land and biomass use (Gerber et al., 2013). The majority of human appropriation of net primary production (HANPP) goes to livestock (Haberl et al., 2007). The sector is responsible for around 2/3 of emissions from agriculture and land use (FAOSTAT, 2016). It uses 80 % of agricultural land, most of which is pasture but around 30 % of total arable land is used for feed production (Mottet et al., 2017). Through habitat occupation and change (Leadley et al., 2010; Mitloehner, 2010), the sector affects biodiversity in numerous direct and indirect ways (Dise et al., 2011; Bobbink et al., 2010). It draws heavily on nutrients: consuming around 65 Tg of nitrogen (Uwizeye, 2019) and some 111,000 km^3 (approximately 10 %) of annual global water flows (Deutsch et al., 2010). Animals are also involved in the emergence and spread of diseases affecting human health (SOFA, 2009).

Currently, a total of six billion metric tons of biomass (dry matter) is needed annually for farmed animals to live and grow (Mottet et al., 2017). Around 3/4 are roughage, made up of grass, leaves, crop residues and cultivated fodder. Grains are responsible for around 13 % of total feed consumption but account for one third of all cereals cultivated – a share that continues to grow (Mottet et al., 2017). Oilseed rape and its by-products make up the rest.

Current projections indicate a continued growth in demand for meat, milk and eggs, driven by population and income growth in low and middle income countries (OECD/FAO, 2019). Livestock systems are dynamic and are engaged in rapid structural change. Productivity growth results from intensification associated with an increased use of concentrate feed, a shift from ruminants to monogastrics (poultry in particular), growing volumes of production and processing, and strong vertical integration. Livestock production has become more geographically concentrated in areas with good access to feed and urban markets. Trade in feed and livestock has grown, implying large volumes of transferred resources and emissions. There are significant regional differences, with increases in demand and future transformation likely to be most prominent in Africa, where demand is projected to triple by 2050 (FAO, 2017).

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2 Emissions and livestock

The transformation of feed into livestock products is associated with direct and indirect emissions of greenhouse gases (GHG), amounting to 7.1 Gt of CO_2eq annually, which equates to around 14.5 % of all anthropogenic emissions (Gerber et al., 2013). Direct emissions are produced from the animal and are associated with biological processes such as enteric fermentation as well as manure and urine excretion. Ruminants produce large amounts of methane – a short-lived climate gas – through enteric fermentation. Methane and N_2O emissions are produced via the nitrification/denitrification of manure and urine. Indirect emissions come from the production of fertiliser for feed production (CO_2), feed production itself (CO_2 and N_2O), manure storage and application (N_2O and CH_4), as well as the processing and transportation of feed, animals and livestock products (CO_2). Comparisons within the system point to large variations in production efficiency and, therefore, to considerable potential for emission reductions through the adoption of best practices.

Emission intensities vary widely among different livestock species and foods. On average, they are highest for ruminant products, especially beef and small ruminant meat (295 and 201 kg of CO_2eq per kg of protein). Cattle milk stands at 86 kg. Emission intensities are lowest for poultry products (eggs at 31 kg and poultry meat at 35 kg) and somewhat higher for pork at 55 kg of CO_2 eq per kg of protein (Gerber et al., 2013).

The scale of the emissions and the abatement potential have drawn livestock, and meat in particular, into the climate debate. On an international level, that debate must take into account the role that livestock play in food security and for the poor. They provide nutritious and appetising food, and play a key role in many rural economies. Livestock are an important buffer in local and national food systems, represent the largest asset for many farmers, and are vital for the poor in rural communities. They provide income and employment, fertiliser (manure), energy (biogas and traction) and other products such as leather, hair and wool. Livestock feature prominently in various cultures and are part of many cultural identities.

The debate also needs to be held in the context of maintaining a healthy diet. Eating habits are changing worldwide, often for the worse, and obesity and diet-related diseases have become global public health concerns, heavily impacting human lives at high costs. Dietary requirements differ a great deal between individuals and population groups. Animal food products convey distinct nutritional advantages to humans because of the quality and availability of key nutrients.

3 What can be done

How, then, can livestock and climate change be reconciled? There are four major ways of alleviating this conflict: increasing efficiency at all levels, creating offsets and other environmental benefits, recycling nutrients and energy, and seeking alternatives across the spectrum.

Firstly, the ongoing process of increasing productivity in livestock systems makes resource use more efficient. In many parts of the world, technological innovations – such as improved feeding, genetics, animal health and information technology – and organisational innovations are driving up productivity and reducing resource use and environmental impact, relative to the amount of livestock produced. There is also considerable scope for greater efficiency in fertiliser production, by using renewable energy, for example, and in its application in feed production, through precision application for instance. This productivity growth has mostly been in response to increasing demand rather than any climate considerations. However, the intensification process could be steered towards low emissions if the appropriate incentives were set. For example, productivity is still stubbornly low in large parts of Sub-Saharan Africa, Latin America and South Asia. It is low because their systems serve purposes other than production, such as asset building in the form of stock (as in Africa and South Asia), or through rising land prices (as in Latin America). In these cases, policies are required that encourage efficiency and better agro-ecological integration, and discourage the keeping of animals for asset accumulation. Extensive, labour-intensive livestock systems with low productivity, prevalent in many low and middle-income countries, are obvious targets for low carbon investments (Mitloehner, 2010).

Secondly, regenerative forms of grazing can generate carbon offsets and other environmental benefits. Well-adapted grazing systems with improved pasture and optimised grazing regimes have the potential to stimulate plant growth and capture soil carbon, particularly in areas where degradation is not yet severe. In particular, the introduction of trees in tropical pastures on previously forested land (silvo-pastoralism) and other forms of agro-ecology (Bonaudo et al., 2014) can help to stabilise productivity and generate multiple social and environmental benefits. Whilst the potential for carbon sequestration and the permanence of such capturing methods are still subject to much debate, the extent of pasture degradation and loss of productivity is such that urgent action is required even if large carbon gains may not be realised in the short term. Regenerative grazing can also contribute to improved biodiversity and water efficiency. Such positive externalities need to be recognised through payments for environmental services. At the same time, slowing down and reversing the expansion of pastures into forests remains the most effective way for grazing systems to contribute to mitigation. The same applies to forest clearance for producing feed crops.

Thirdly, emissions can be reduced by reverting to one of the original reasons for keeping livestock: recycling nutrients and energy. Traditional links between livestock and arable farming have become increasingly severed over the course of intensification, and livestock operations have become concentrated in areas with limited arable land on which to apply manure. This disrupts nutrient cycles and creates depletion upstream as well as excesses downstream. Cycles can operate on various levels, for instance, within farms, on the watershed level or globally. While there are considerable differences in recycling practices, large amounts of potential feed such as crop residues, agro-industrial by-products and food waste are unused, often with direct adverse environmental impact as well as a loss of opportunity for recycling. Similarly, only a fraction of the nutrients contained in animal waste are returned to the land in a useful way. A combination of regulations and spatial planning is required to create opportunities and incentives for recycling, which will reduce the impact on our climate.

Fourthly, there are alternative paths to the one which depends on conventional feed and livestock. Bio-technological innovations are revolutionising the way protein can be produced and used. This includes established practices such as the use of synthetic amino acids, novel techniques involving algal, fungal and microbial proteins, replacing conventional feed protein such as soy, and making its use more efficient. The use of insects has also been growing, both for feed and food.

There is a rapidly growing interest in substitutes for livestock products. Most of them are plant-based imitations of the original product, however, there is a rapidly growing field of application in microbial protein. While their actual environmental impact varies, there can be little doubt that low-carbon alternatives to today's livestock products can be developed rapidly, given the massive start-up investments that are taking place. Plant-based alternatives also appeal to concerns around animal welfare and healthy diets. Efforts are also underway to generate synthetic meat through cellular agriculture based on stem cells. Policies that discourage the consumption of high-emission food products, such as beef, through awareness-building and taxation are also being considered.

Each of these approaches has considerable potential to reduce livestock emissions, and they will be even more powerful in combination, with different approaches being more relevant to different social contexts and food systems.

4 Time to act

Livestock play a large role in natural resource use, and, as such, have taken centre stage in the climate change debate as an obvious target for mitigation. The pressure to reduce emissions will only increase, fuelled by consumer concerns around diet, health and animals. Plant-based alternatives have recently seen a rapid upsurge. Livestock systems will have to adapt, not only to climate change and market demands, but also as a result of upcoming policy changes aimed at low-cost mitigation options. It is only a matter of time before livestock become a direct target of climate change policies.

Ruminant systems, particularly beef, are being challenged the most. Research is underway to reduce enteric methane emissions by manipulating the rumen flora, however, related techniques are not yet practical or cost-effective. For now, the only way to substantially reduce emissions is through offsets from afforestation and soil carbon.

Climate change calls the place of livestock in food and agriculture into question. Finding that place, and renewing the license to operate, is urgent. Such efforts need to be built on transparency and a consensus on methods for measuring emissions and tracking progress. Pricing and regulations must encourage best practice and responsible consumption. Engagement from all stakeholders is required in conjunction with local solutions to tap the potential of livestock systems and contribute to climate action.

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POSITION PAPER

Recognize the high potential of paludiculture on rewetted peat soils to mitigate climate change

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KEYWORDS **peatland, wet agriculture, GHG reduction potential, ecosystem services, carbon credits, food security**

1 Introduction

Draining peat soils leads to oxidation of the peat and soil subsidence. In Germany, drained peatlands account for only 7 % of the agricultural land but are responsible for 37 % of the agricultural greenhouse gas (GHG) emissions (GMC, 2018). Rewetting peat soils appears to be a cost efficient GHG mitigation measure (Röder et al., 2015). The ideal situation would be a natural colonisation with peat forming plants after rewetting and a return to a carbon sequestrating system without harvesting. However, the productive function can often not be relinquished and paludiculture, the practice of productive use of wet and rewetted peatlands, should be considered. In paludiculture, harvesting wet crops for food, fodder, fibre and fuel is combined with the provision of vital ecosystem services (Wichtmann et al., 2016). This concept provides production opportunities for the necessary, fundamental change in land use of drained peatlands to a more sustainable, wetter land use, which should benefit both the regional economy and the climate. Peatlands used for paludiculture maintain a productive function under permanently wet, peat preserving conditions. The average groundwater level in the growing season is 20 cm below the soil surface or higher, and the minimum groundwater level is never more than 40 cm below the soil surface (Geurts and Fritz, 2018). This implies that drained grasslands and croplands can be converted into peat moss lawns, reed and cattail plantations, or wet meadows with grass species adapted to a higher soil moisture content. The biomass can be used for a whole range of products and applications, including human consumption and fodder, or wet grasslands can still be used as pastures (e.g. by light dairy cows or water buffaloes).

2 Paludicrops

There are various types of peatland cultivation systems with crops grown under wet conditions, so-called paludicrops. Many of these are ready to be implemented on a larger scale, including on farms. Biomass yields of 15 to 30 t dry matter per ha are potentially possible (Heinz, 2012; Köbbing et al., 2013; Grosshans, 2014), which is comparable to conventional crops. Paludicrops can be used as fodder, as protein

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source, or as raw material for the production of horticultural growing substrates, or alternatively for bio-energy and as a resource for bio-based materials (insulation, building materials, paper, bioplastics). Paludicrops differ in their soil chemical and hydrological requirements, and growers need to adapt to these requirements (Geurts and Fritz, 2018). *Table 1* lists promising paludicrops, their preferred range in water levels, applications for which they are grown (both on-farm and off-farm), existing pilots and large-scale implementation, and to which extent they have potential for carbon and blue crediting systems (see below). Moreover, usage of biomass for bio-based materials will prolong the lifecycle of carbon, as compared to fodder for ruminants where part of the carbon is rapidly emitted again as CH_4 and CO_2.

3 Payments for ecosystem services

There is a large GHG emission reduction potential when rewetting drained and fertilised peat soils, commonly 40 to 60 t CO_2-eq ha^{-1} a^{-1} for productive and fertilised grasslands. Firstly, CO_2 emissions become lower at higher groundwater levels and approach zero in waterlogged soil. Secondly, emissions of N_2O, a very strong GHG, are reduced as N fertilisation will usually be decreased and N_2, rather than N_2O, will be formed during denitrification when oxygen availability is low in wet conditions (Tiemeyer et al., 2016). In addition to biomass use, the GHG emission reduction creates opportunities for business models based on carbon crediting schemes (e.g. Moorfutures®; Joosten et al., 2015; Günther et al., 2018). The climate mitigation potential is partly counteracted by methane emissions that are largely driven by summer inundation, topsoil chemistry, vegetation type, availability of easily decomposable biomass, and nutrient or carbon input (Couwenberg and Fritz, 2012). Guidelines for low GHG emission (< 10 t CO_2-eq ha^{-1} a^{-1}) production cycles on rewetted peatlands are available (Tiemeyer et al., 2016; Günther et al., 2017; Geurts and Fritz, 2018). In addition, every hectare of drained peatland that is converted to paludiculture is as effective as taking climate mitigation actions on 10 to 100 ha of mineral soils for food production, which would have led to a lower productivity (e.g. lower use of fertilisers).

In addition to climate benefits, paludiculture can reduce nutrients in surface water and reduce flood risks and droughts by acting as temporary water storage areas, and increase biodiversity compared with conventional agriculture. In so-called blue crediting schemes, farmers could be paid for these water management related ecosystem services (Bohlen et al., 2009; Grygoruk et al., 2013). However, these schemes are still in the development stage.

4 Pilot projects

Paludiculture pilots and demonstration sites on a farm-scale already exist in various countries *(Table 1)*. Preliminary results suggest that peat forming paludicrops (e.g. peat moss, reed and alder) grown at groundwater levels 10 cm below the soil surface are the optimal compromise between biomass production, climate mitigation, and peat preservation (Schäfer and Joosten, 2005; Jurasinski et al., 2016; Günther et al., 2017). However, some crops, such as cattail, perform better at water levels 5 to 20 cm above the surface, which may lead to substantial methane emissions in case of adverse circumstances (high carbon input or presence of fresh litter (Couwenberg and Fritz, 2012). Harvesting belowground biomass is not eligible since causing regular soil disturbance conflicts with the preservation of the peat carbon stock as a primary concern of paludiculture. In addition, caution should be taken if using exotic plant species as paludicrops (e.g. wild rice, rice, giant reed, miscanthus), because they may become invasive (Matthews et al., 2015).

5 Opportunities and bottlenecks for implementation

For large-scale implementation of paludiculture, long-term schemes and income security for farmers is required. In this respect, paludicrops need to acquire the general eligibility for agricultural payments in the first and second pillar of the EU's Common Agricultural Policy (CAP) as currently exist for conventional CO_2-intensive crops from drainage-based agriculture (Wichmann, 2018). So far, most paludicrops lack the status of agricultural crops despite centuries of productive use (e.g. reed for thatching, willow for wattle fences). Within the next funding period, any kind of cultivation for food, fibre, or energy on rewetted peat soils should become eligible for direct CAP payments. Furthermore, future public payment schemes need to set a new course by considering the external effects of peatland use, i.e. phasing out any support for drainage-based peatland use, supporting the shift to paludiculture (e.g. investments for planning, planting, special machinery), and paying for reduced GHG emissions and other ecosystem services provided by wet and rewetted peatlands (Wichmann, 2018). Moreover, the application of the 'polluter pays principle' (e.g. used in the Water Framework Directive; Correljé et al., 2007) on drainage-based peatland use may promote CO_2-neutral and economically sustainable production systems on peat such as paludiculture.

An obstacle that still exists is the fact that water management in agricultural areas is usually tailored to serve drainage-based agriculture, which often makes rewetting expensive when surrounding fields are still drained. Furthermore, while special machinery and certain important production chains are already available, the scale of production is currently too small to feed supply chains of e.g. peat moss for bulk growing substrate, and cattail for insulation and building material. As a result, the market for most paludiculture products as raw materials for bioenergy and bio-based materials is not yet functional and business models are still under development.

Next to biomass revenues and harmonised subsidies, ecosystem services should be rewarded and incentives should be developed to stimulate the implementation of paludiculture, including the accounting for reduced GHG emissions (carbon credits), water purification, climate change-related water retention and storage (blue credits), and biodiversity. In the Netherlands, this has already been done for some forms of nature-inclusive agriculture (Runhaar, 2017).

Further steps in implementing paludiculture are being taken in several projects in various European countries (see acknowledgement). Pilot projects are very important to further develop management and harvesting techniques, obtain robust data on environmental benefits (including Life Cycle Analyses (LCA) of land use and associated products), and create markets for products.

TABLE 1

Overview of important paludiculture crops and applications, range in water levels, list of important production areas including pilots and potential areas, potential for carbon credits based on estimates of GHG emission reduction (including biomass use for replacing fossil resources), and potential for blue credits based on suitability for water purification (P) and water storage (S): ++ very high potential, + high potential, 0 little potential, - negative effect. Figures based on references in Wichtmann et al. (2016) and Geurts and Fritz (2018).

Crop	Water level (cm +/- soil surface)	Product	Potential for carbon credits	Potential for blue credits	Important production areas including pilots (in ha) and potential areas (in italics)
Cattail *(Typha sp.)*	0 to +20	insulation and building material	+	P + S +	Kamp (D) 30 Zuiderveen (NL) 4 Peel (NL) 1 Bûtefjild (NL) 0.1 *Danube delta (RO)*
		bedding material	+	P + S +	Peel (NL) 1 Zegveld (NL) 0.4
		extraction of protein, fibres, cellulose	0/+	P ++ S +	*Canada*
		feed for pest-controlling predatory mites	0/+	P ++ S +	Zegveld (NL) 0.4
		fodder	-/+	P ++ S +	Peel (NL) 1 ha Zegveld (NL) 0.4
		combustion	-/+	P + S +	Canada > 500
Reed *(Phragmites australis)*	-20 to +20	thatching, insulation and building material	++	P + S ++	UK 6,500 Netherlands 4,500 Mecklenburg-Vorpommern (D) 550 Poland 8,000 Hungary 7,500 Austria 1,500 Denmark, China *Romania 190,000* *Ukraine >100,000*
		paper	++/+	P + S ++	China > 1 million
		extraction of protein, fibres, cellulose	0/+	P +/++ S ++	*Germany*
		combustion/ biogas	-/+	P +/++ S ++	Italy 0.75 Germany *Belarus & Ukraine: large potential areas*
Peat moss *(Sphagnum sp.)*	-15 to -5	high quality substrate in horticulture	++	P + S 0/+	Hankhausen (D) 14 Twist (D) 10 Ilperveld (NL) 8 Canada 8 Finland, Chile
Grasses like reed canary grass *(Phalaris arundinacea)*	-30 to +10	combustion/ biogas	-/+	P 0 S +	Malchin (D) 200 Denmark, Estonia, Belarus
		fodder	0/+	P 0/+ S +	Mecklenburg-Vorpommern (D)
Alder *(Alnus sp.)*	-40 to +5	wood/timber	++	P 0/+ S ++	Mecklenburg-Vorpommern (D) USA

To convince landowners, producers/farmers, and manufacturers, long-term schemes and certificates for CO_2 and other ecosystem services have to be developed and experiences from existing paludiculture pilots in Europe and large-scale implementation in peat-rich regions in the world should be shared. The second pillar of the CAP already provides some incentives for all steps of implementation that can be used and refined (cf. Wichmann, 2018).

6 Conclusions

- Farm carbon footprints benefit largely from raising water levels to the peat surface resulting in substantial GHG emission reduction.
- Small areas of drained peatlands converted to climate mitigation optimised paludiculture can offset the need to take climate mitigation actions on 10 to 100 times larger areas of mineral soils for food production.
- Sustainable wet agriculture can also be economically viable. New business models are being created, which can often be combined with conventional farming (fodder, bedding material, meat/milk with CO_2 certificate), but high quality off-farm applications also exist already.
- Society is responsible for creating essential preconditions for large-scale peatland rewetting and paludiculture, including the provision of the necessary infrastructure and recognition of the sustainability value of paludiculture.
- Techniques and tools for paludiculture are available and under optimal conditions comparable biomass yields and revenues as in conventional agriculture are potentially possible.
- Water level management, nutrient availability, and crop choice are the main determinants for productivity. Other aspects are GHG emission reduction, costs of implementation, and the provision of other ecosystem services.
- CAP funding schemes need to be revised to facilitate sustainable solutions for wet peatland agriculture.
- Well-documented, long-term pilot projects and the generation of LCAs are very important to gain insight into long-term yields and income from paludiculture and are necessary for innovations and further market development.

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POSITION PAPER

Farm-level digital monitoring of greenhouse gas emissions from livestock systems could facilitate control, optimisation and labelling

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1 Greenhouse gas emissions from livestock

Societal efforts to limit climate change necessitate the participation of all major emitters. Global livestock production of both ruminant and non-ruminant animals contributes annually about 7.1 Gt CO_2-eq (14.5 %) of anthropogenic greenhouse gas (GHG) emissions (Gerber et al., 2013). Unfortunately, diffuse non-point sources make accurate monitoring systems expensive and prevent an efficient implementation of emission regulations in both the crop and livestock sectors. Proposed remedies include subsidies and taxes on management regimes, which are correlated with emissions. The available farm-level GHG calculators comprise automated web-, Excel-, or other software-based calculation tools, which rely on coarse approaches used in national GHG inventories (e.g. IPCC Tier 1 and 2 GHG emission factors) and are therefore too simplistic to depict farm-level emission fluxes in sufficient detail (Denef et al., 2012).

GHG emissions from livestock systems involve up to four distinct categories (*Figure 1*). Firstly, machinery used for land management, operation of livestock facilities, and transportation and processing of livestock commodities requires fuel and electrical power. Also fertilisers, pesticides, buildings, and machinery contain embedded energy. Emissions from fuel and power use are generally easy to monitor because most energy is accounted at power meters or fuel nozzles, whereas embedded energy is more difficult to define and would need agreed tabulated values to enable selection options for famers.

Secondly, enteric fermentation from ruminant animals causes methane emissions. The magnitude of these emissions depends on the breed of animal, feeding regime, and various operational and environmental factors (Hristov et al., 2018). Thirdly, livestock manure leads to methane and nitrous oxide emissions. The breed of livestock, diet, manure storage and handling, and environmental factors affect emission levels (Chadwick et al., 2011). Respiration chambers are the state-of-the-art measuring method for emissions from both enteric fermentation and manure.

Fourthly, carbon dioxide, methane, and nitrous oxide are emitted on pastures and livestock-related croplands. These emissions vary highly across local soil and weather conditions and land management regimes. Vegetation composition, stocking density, applications of manure, mineral fertilisers, and pesticides, and intensity of irrigation affect emissions also on pastures (Bolan et al., 2004). Emissions from croplands are further impacted by the choice of crop rotation and soil tillage.

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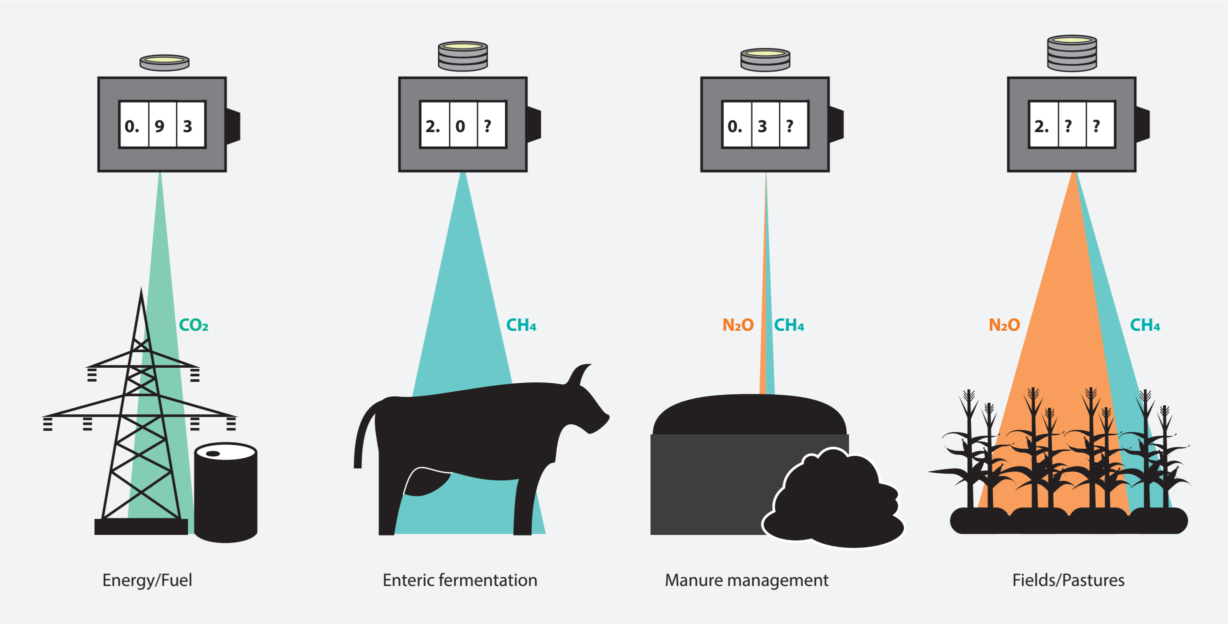

FIGURE 1
Total GHG emissions from livestock systems in 2016 according to FAOSTAT. Values in upper boxes show the global contribution in Gt CO_2-eq. The number of question marks symbolises the variability of emissions. Coin piles depict qualitative differences in measurement costs.

2 Digital monitoring of livestock emissions – Linking detailed farm records and scientific models

Current market transactions can account for on-farm and off-farm emissions from livestock-related energy combustion. However, effective and efficient emission regulations require a comprehensive and accurate accounting of all significant on-farm emission sources and sinks from livestock. To quantify emissions from enteric fermentation, manure and land management, we propose a digital monitoring network where state-of-the-art scientific models take over the tasks of expensive measurement devices or simplistic accounting tools. The digital monitoring network would control information exchange and information processing between farmers, IT enterprises, authorities, scientists, and the public (*Figure 2*).

One network component is a suite of scientific tools supported by scientists for the estimation of on-farm emissions that are difficult to measure. These tools depict agro-ecological processes and estimate emissions from i) enteric fermentation of ruminant animals, ii) manure management, and iii) management of pastures and croplands used for feed production. Emissions from enteric fermentation and manure management can be predicted using detailed empirical or mechanistic models (Rotz, 2018). The latter depict nutrient digestion, absorption, microbial development, and fermentation stoichiometry to determine methane emissions. An important determinant for the accuracy of these predictions is the quality of input data, i.e. data on feed intake and composition, body weight and movement, housing and manure handling and, in the case of dairy, milk yield. Predictions of emissions from pastures and croplands are more challenging. However, over the past decades agricultural scientists have developed ever more detailed biophysical process models to simulate cultivated vegetation on agricultural fields under specific soil and weather regimes (Brilli et al., 2017). State-of-the-art crop models include EPIC (Wang et al., 2012), DayCent (Del Grosso et al., 2005), DSSAT (Jones et al., 2003) and several others. These models operate on a daily time scale and depict all major interactions between vegetation, soil, weather, and land management. Simulated environmental impacts include soil organic matter changes, emissions of trace gases, soil erosion, nutrient leaching, and others. Supported by scientific experiments in many diverse case studies, the representation of relevant biophysical processes has reached a mature stage. Nevertheless, the quality of local model predictions depends strongly on the quality of input data.

A second component of our proposed digital monitoring network consists of livestock farmers. Participating farmers would record and submit detailed information on the number and characteristics of animals, animal feeding and product yields, manure management and date, location, and intensity (e.g. ploughing depth, seed density and fertiliser type and application rate) of pasture and cropland operations. Some or eventually all of this information could be automatically collected through digital devices. Farmers using computerised feeding systems or sensor and satellite supported fertiliser applications could automatically submit high-resolution data.

A third network component is a user-friendly IT platform (server), which controls and organises the exchange and processing of information. Farmers can register on this platform and verify the spatial coordinates for their land ownership. Upon registration, the system would examine existing farm data and request amendments if any necessary data are missing, incomplete, or inadequate. Amendments, e.g. for soil data, would mostly require one-time measurements of particular soil properties. The platform would also examine daily meteorological data from the nearest official weather stations, reanalysis data and climate projections. If available, farmers could submit their own meteorological data from approved on-farm weather stations. Registered farmers could provide or link field specific farm management data and receive a prediction of annual emissions in carbon dioxide equivalents per animal or hectare. Farmers could use the system to plan future livestock management and predict productivities and emissions. The IT platform could be soft or hard linked to existing farm management tools already used by farmers.

A fourth component involves governmental authorities and regulations. Authorities could use the system to verify GHG balances of participating livestock producers. The more farmers who register and participate, the more information about livestock impacts would be available on aggregate regional, national, or even international scales. Authorities could use this information to better plan, design, or amend policies. The fourth component would also include the implementation of data privacy laws to protect non-public data.

Finally, a fifth component addresses specific interest groups and the public. They would be able to access aggregate information, to inform themselves, to play scenarios, and to participate in public debates. A possible application would be the estimation of detailed environmental footprints for crop and livestock commodities.

The proposed digital emission monitoring system would improve emission accounting from crop and livestock production and allow a more efficient regulation of these emissions. Despite public benefits from reduced environmental externalities, there is a question of private cost and benefits. Why should farmers voluntarily register and participate in a digital emission accounting system, spend effort on organising information, and risk adverse consequences from disclosing detailed business information? Firstly, if farmers subjected to climate policies refused to use accurate emission accounting methods, they or the authorities would employ inferior methods, e.g. assign default emission factors. The submission of such more biased or more uncertain emission estimates should result in financial disincentives based on society's risk aversion preferences (Kim et al., 2008). If, on the other hand, farmers did use a detailed and accepted accounting method, they could legally verify their actual emission values and pay fewer emission penalties or, in case of negative emissions, gain higher rewards. Additional benefits from participation include enhanced planning tools for farm management, computation of various environmental footprints, and access to commodity labels.

3 Conclusions

GHG emissions from crop and livestock production are highly variable across fields and animals. Traditional options for accounting and regulating GHG emissions from agricultural operations are either costly or imprecise. Most existing policy proposals include practice-based payment systems with a fairly large uncertainty. We suggest an emission-based payment system with a digital monitoring network, where validated state-of-the-art scientific models eliminate the need

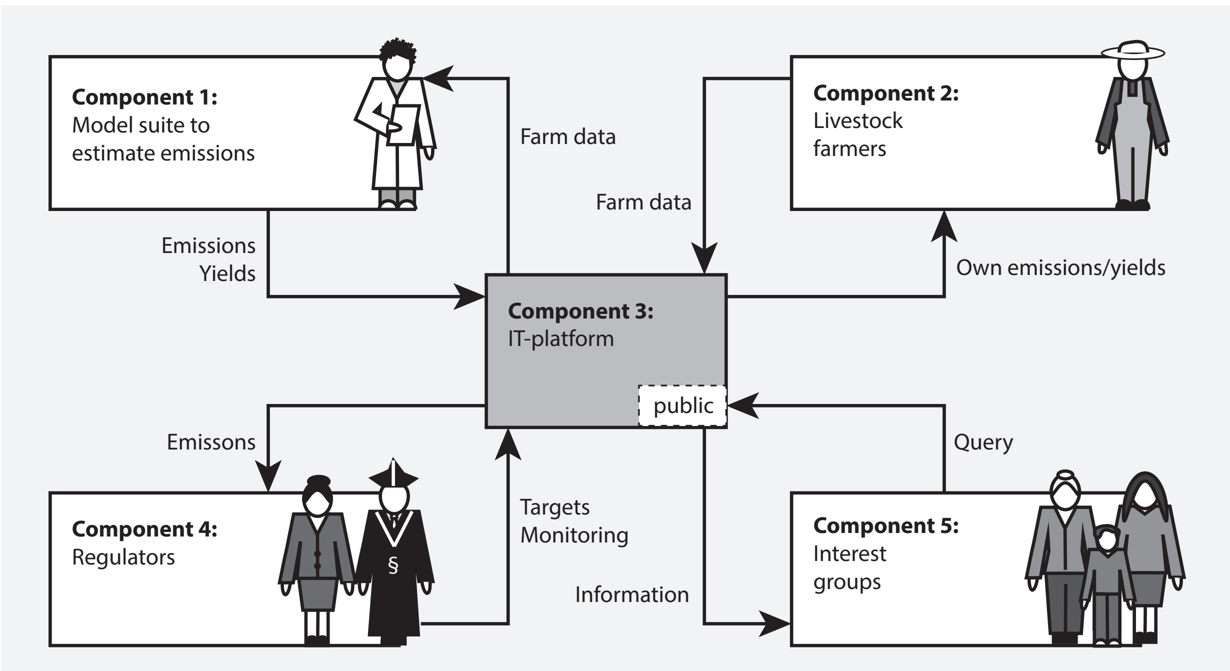

FIGURE 2
Integrated digital emission monitoring system

for costly measurement devices. This network would be applicable to all agricultural operations including specialised crop or livestock businesses and mixed farms.

Emission measurements would still be needed for model validation, however, only at certain intervals on selected sites. Suitable models are already used in scientific assessments and for national GHG inventories. However, the often low quality of input data severely limits the quality of model-based assessments. We therefore propose to combine sophisticated scientific models with detailed and comprehensive management information available at farm level. The reduced uncertainty of otherwise crudely estimated emissions should translate into a financial incentive for farmers to participate. The increasing digitalisation of agricultural operations could facilitate automatic or semi-automatic exchange of data between farmers, scientific tools, authorities, and the public.

The complex modelling system would also permit monitoring of agro-environmental impacts beyond greenhouse gases, including nutrient and pesticide leakages to water bodies and soil erosion. Participating farmers could also benefit from access to new market labels based on ecological footprints rather than on a crude distinction between organic and conventional agriculture.

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RESEARCH ARTICLE

Applying a conceptual framework for effective implementation of on-farm greenhouse gas mitigation: Evaluation of knowledge exchange methods in Wales and Uruguay

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HIGHLIGHTS

- Agricultural knowledge exchange methods used in two countries
- Potential impacts on challenges to greenhouse gas mitigation assessed
- Methods for initial engagement and widening farmers' perspectives highlighted
- Resources for and a renewed research focus on extension systems is vital

KEYWORDS agricultural extension, farming, greenhouse gas mitigation, knowledge exchange, stakeholders

Abstract

Globally, agriculture must tackle many complex challenges to ensure food security for a growing population while safeguarding biodiversity and ecosystem services and contributing to greenhouse gas (GHG) emissions reduction. Effective agricultural knowledge exchange (KE) strategies are vital to implementing GHG emissions mitigation measures. Here, KE activities undertaken by publicly funded extension services in Wales (in the global north) and Uruguay (in the global south) were compared using a previously developed conceptual framework.

The main goals were to assess the utility of the framework and to evaluate KE methods in terms of i) potential challenges to initial engagement, ii) categories of challenge they could address and their potential mode of operation, iii) their potential impacts on non-target stakeholder groups, including iv) the interests and limitations of KE practitioners. Use of the framework highlighted issues including the need to i) tackle initial challenges potentially affecting engagement with mitigation narratives, ii) widen the outlook of stakeholders on climate change and emissions reduction, iii) recognise how KE may affect, and be affected by, non-target stakeholders and, iv) address KE practitioners' needs and outlooks. Priorities for improved implementation of mitigation measures include the use of technical (e.g. modelling) and social (e.g. discussions involving non-food chain actors) KE methods that act on stakeholder interests, with the potential to engage farmers in empowering KE processes for GHG emissions mitigation. A renewed research focus on agricultural extension systems is needed to more effectively apply KE resources to meet sectoral GHG emissions targets.

1 Introduction

The agricultural sector faces the challenge of ensuring food security in the context of a growing world population, requiring increases in food production that can be sustained in the long term, while enhancing ecosystem services and minimising greenhouse gas (GHG) emissions. Transformative change may be needed to achieve these goals (Martin et al., 2013) and solutions must be integrated, recognising impacts and needs across interacting spheres (environmental, economic

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and social) to avoid unintended consequences and identify multi-win solutions (Garibaldi et al., 2017). In the face of these challenges, identification not only of solutions but also of effective implementation strategies is essential if required changes are to be realised on the ground. Effective agricultural knowledge exchange (KE) is therefore vital in the context of challenges to implementing GHG emissions mitigation measures in the agricultural sector (Wreford et al., 2017).

Over time, theory and practice in agricultural extension has shifted, from an emphasis on the top-down transfer of knowledge and regulation (from researchers and policymakers to farmers), to more interactive KE which empowers stakeholders to determine and drive the direction of change (Leeuwis, 2004). This trend reflects a similar change from hard to soft systems approaches to communicating and implementing scientific research (van Paassen et al., 2007). Despite these trends, there remain tensions between local, stakeholder driven processes of change in agricultural practice and pre-existing top-down structures of governance (Colvin et al., 2014).

At the same time, previously unified public provision of KE has fragmented and been replaced with KE provided by a mix of public, private and non-governmental organisations, as market-driven agricultural extension models have become favoured for their assumed efficiency benefits (Knuth and Knierim, 2015). In Europe, these changes have occurred in the context of the Common Agricultural Policy (CAP) which has been criticised for reducing social capital in farming communities through its individualised focus (Leventon et al., 2017).

Against this backdrop of theoretical development and policy change, many practical resources have been developed to support best practice in KE, exemplified by the Global Forum for Rural Advisory Services (GFRAS) Global Good Practice Notes report (Davis et al., 2018) which brings together practical summaries of a wide range of advisory methods to inform KE practitioners, especially those working in the developing world. Still, at different levels agriculture has made slow progress in relation to many of the challenges it faces, such as the need to substantially reduce farming related GHG emissions.

Recent work on implementing GHG mitigation measures on Welsh livestock farms analysed the views of stakeholders on challenges to change and solutions, forming a framework categorising challenges and strategies for improved implementation (Kipling et al., 2019a; Kipling et al., 2019b). This work augmented existing resources which provide information on the practical requirements, strengths and weaknesses of different tools for KE, with a conceptual framework that facilitates critical analysis of the potential impacts of implementation strategies, not only on farmers but also on other rural stakeholders. The main goal of the current study was to apply this conceptual framework to evaluate KE methods used by practitioners in two contrasting countries (Wales, in the global north, and Uruguay in the global south) in order to both, i) test the usefulness of the framework and ii) provide an overview of KE strategies in these countries, their potential to address different challenges, their likely impacts and gaps in capacity.

2 Materials and methods

2.1 Study countries and mitigation measures

Agriculture in Wales falls under the European Union's CAP which provides payments to farmers based on the area of land farmed and adherence to practices aligned with sustainability objectives. Eighty percent of agricultural land in Wales has been classified as 'Less Favoured Areas' for farming (Welsh Government, 2013), reflecting the extent of exposed uplands. A large proportion of farm businesses provide low income levels for an ageing farming population (Morris et al., 2017). The climate is oceanic, with warm winters and wet summers ideal for grass growth. Due to the topography and conditions grass-based sheep and beef production dominate agriculture, with a growing dairy industry in more lowland areas of the country (Morris et al., 2017).

In relation to the challenges facing farming, the Welsh Government is pursuing a target of an 80% reduction in GHG emissions against 1990 levels across the Welsh economy by 2050. However, by 2015 farming emissions had only fallen by 15% (Jones et al., 2017) driving the commissioning of research to improve performance (Kipling et al., 2019b). Studies have shown wide differences between the most and least production-efficient farms, indicating potential to improve efficiency and reduce GHG emissions intensity by spreading best practice (Hyland et al., 2016b). A wide range of mitigation measures have been suggested for livestock systems at the UK level (including Wales) and indicate that no single solution will achieve desired emissions reductions; rather, improvements in practice throughout farm systems are required, focussing on measures that avoid carbon leakage by improving production efficiency without altering production levels (Kipling et al., 2019b).

Uruguay lies within the South American Campos, an ecological region of grasslands and pastures with scattered trees and shrubs. Uruguay's climate is temperate, moderate and rainy. The temperature of the coldest month is between -3 °C and 18 °C and the temperature of the warmest months exceeds 22 °C. Precipitation shows high inter-annual variability with an annual total reaching 1300 mm in the north of the country; according to the Koeppen climate classification Uruguay is classified in the 'Cfa' category (Bidegain and Caffera, 1997).

Livestock production is mainly in the form of extensive grassland-based beef and sheep systems. Due to edaphic and climatic conditions, and specifically low phosphorous levels, these systems face agricultural issues including low productivity resulting from poor nutrient value and digestibility of grasses (Royo Pallarés et al., 2005). Sheep production has fallen over recent decades as a result of factors including declining domestic mutton consumption, falling wool prices and issues with sheep rustling (Royo Pallarés et al., 2005). While cattle numbers have risen from 8.69 million head in 1991 to 11.74 million head in 2017, sheep numbers have fallen from a high of 26.6 million in 1991 to 6.6 million in 2017 (FAO, 2019). An historic trend towards agricultural land concentration has increased in recent years, with changing patterns of ownership, rising land prices, increases in land devoted to cropping and forestry, and associated socio-economic changes (Oyhantçabal and Narbondo, 2019). In particular, a large and growing area of the country is occupied by eucalyptus plantations which are of increasing economic importance (Pozo and Säumel, 2018).

In 2016, GHG emissions from the Uruguayan agricultural sector were 16.1 % higher than 1990 levels, and on average between 1990 and 2016, 63 % of emissions by sector resulted from enteric fermentation, and a further 26.9 % from manure left on pasture (FAO, 2019). However, recent research indicates that Uruguayan livestock systems based on natural grasslands provide a range of ecosystem services and have the potential to deliver economic and environmental 'win-wins' (Modernel et al., 2018). As in Wales, differences in economic and environmental performance between farms suggest a potential to reduce GHG emission intensity through the spread of best practice in livestock production (Becoña et al., 2014). In Uruguay's extensive beef cow-calf production systems, effective GHG mitigation measures focus on improved grazing management (stocking rate, forage allowance and pasture improvement) (Becoña et al., 2014).

In Wales, although policymakers and KE practitioners seek to drive change that can reduce GHG emissions, KE for farmers has mostly focussed on improving economic performance, with GHG emissions mitigation tackled implicitly through a drive for improved production efficiency. In contrast, in Uruguay, there has been a more direct strategy to create awareness of the environmental impacts of livestock systems in order to drive change. Given the common goal in the two countries to reduce GHG emissions and other environmental impacts from livestock systems, and shared pressure for sustainable intensification of production, comparisons of differences in the KE methods applied can be the basis for learning between KE practitioners in the two countries. The importance of the livestock sector in relation to the global challenges facing agriculture means that the grassland livestock systems of Wales and Uruguay also provide a case study of KE strategy with relevance beyond the focus countries.

2.2 Study context and conceptual framework

The current study is part of a longer-term research effort *(Figure 1)*. In previous work, analysis of the views of stakeholders associated with the Welsh livestock production sector produced a conceptual framework categorising challenges and solutions relating to the implementation of on-farm GHG mitigation measures (Kipling et al., 2019a; Kipling et al., 2019b) *(Figure 1: A)*. The categories were tested for their relevance to global barriers to climate friendly farming and potential solutions, as reviewed by the OECD (Wreford et al., 2017), with the outcome indicating their general relevance beyond the Welsh context *(Figure 1: B)*. Here, this conceptual framework is applied to assess KE methods used in Uruguay and Wales *(Figure1: C, D)*. The framework consists of various components, which are listed on the next page and which provide the structure for the analyses described below (see Appendix 1 for detailed summary of each).

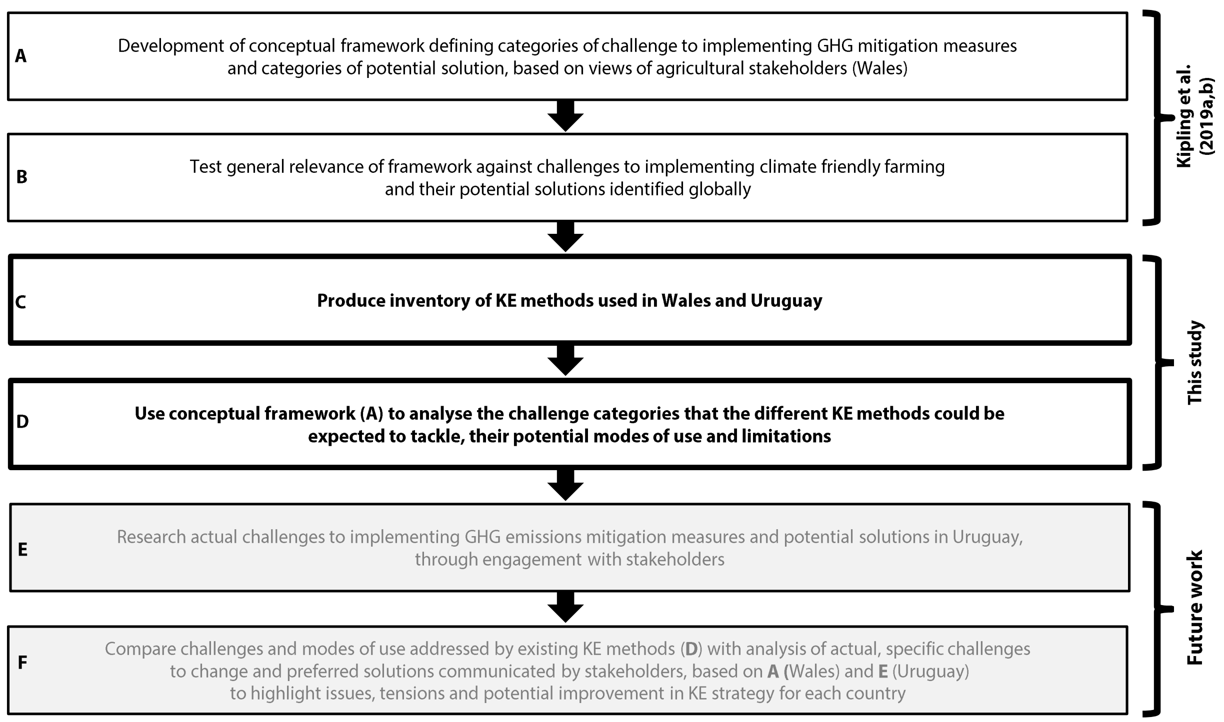

FIGURE 1
Context of current study. Bold text boxes (C, D) = current study, normal text (A, B) = completed research, grey text boxes (E, F) = future work

Four challenge categories:
- Practical limitations
- Knowledge limitations
- Cognitive limitations
- Interests

Three levels of change at which solutions may operate:
- Work around
- Overcome
- Alter

And three approaches to change they may take:
- Accommodate
- Control
- Empower

2.3 Data collection and analysis

Factual descriptions of KE methods utilised were gathered from the two major KE providers in Wales and Uruguay (Farming Connect (FC) https://businesswales.gov.wales/farming-connect and Plan Agropecuario https://www.planagropecuario.org.uy/web, respectively). The providers were asked to return a list of the KE methods they used (e.g. demonstration farms, factsheets), to describe the goals aimed for in their use (e.g. to ensure farm advisors have up to date knowledge), the target groups aimed at (e.g. farm advisors, young farmers, farmers in general) and how target groups were given access to the KE provided (e.g. via a website, promotion at events). Data were either provided via email or drawn from internal documentation shared with the researchers by the organisations. Based on these data, a summary description of each KE method was prepared. The KE providers checked and approved or amended the descriptions, ensuring accuracy.

The following stages of analysis of collected data were undertaken (the outcomes of each are considered in turn in section 3):

1. In order to gain an overview of KE strategies, a grounded theory approach (in which categories are drawn out of the data rather than being imposed a priori by the investigator – to ensure 'grounding' in the dataset) was used to group the KE methods used in Wales and Uruguay into thematic types and classes according to common aspects and roles.
2. KE method descriptions were assessed to compare the methods applied in Wales and Uruguay.

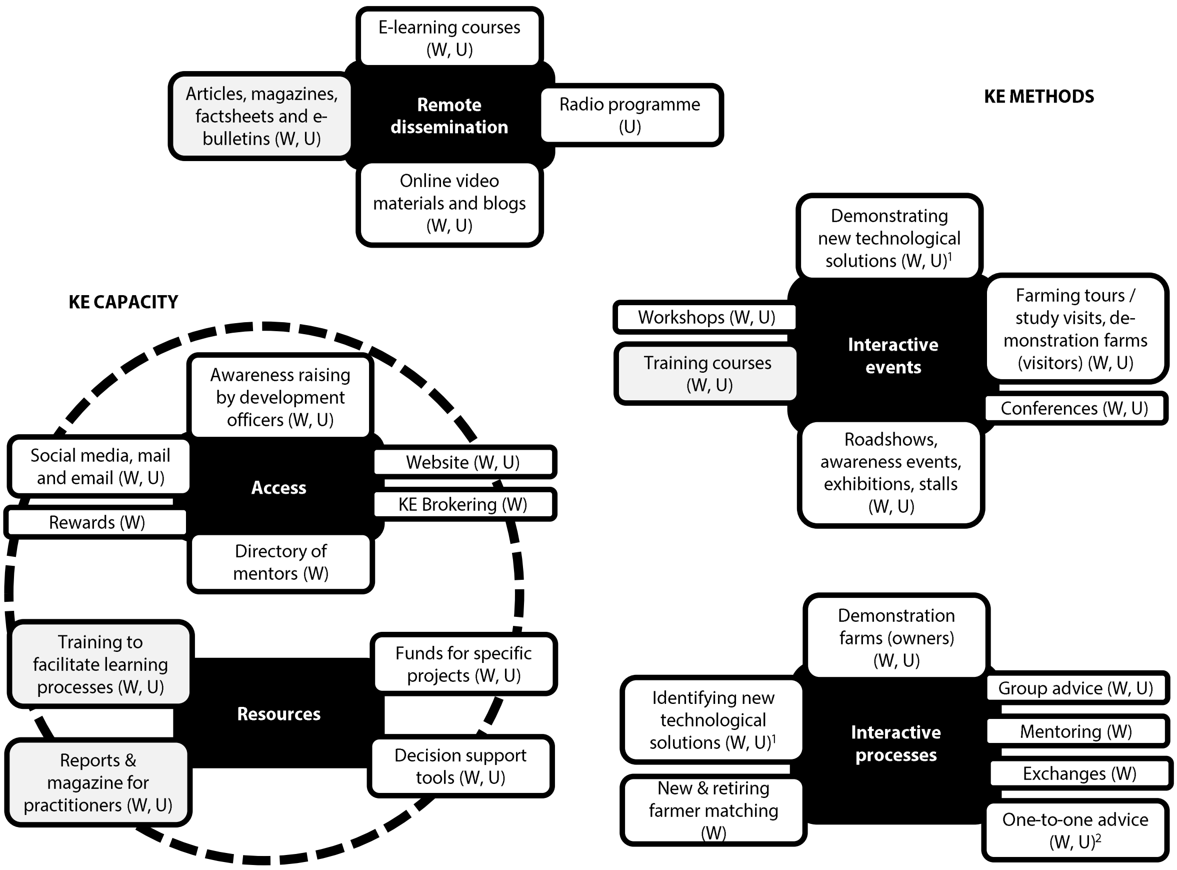

FIGURE 2
Classification of KE methods and KE capacity for Uruguay (U) and Wales (W)
[1]Delivered by the private sector in Uruguay, [2]delivered by the World Bank in Uruguay
Grey boxes = KE methods used both for farmers and as resources for KE practitioners

3. The conceptual framework described above was used to conduct a qualitative assessment of i) potential challenges to initial engagement with KE methods, ii) categories of challenge KE methods could address and their potential mode of operation, iii) the potential impacts of KE methods on non-target stakeholder groups, including iv) the interests and limitations of KE practitioners.
4. The use of the conceptual framework was evaluated.

Assessment of 3 (i) was included as previous research has suggested that individual farmers may not prioritise GHG emissions mitigation, for example due to their perceptions of climate change (Hyland et al., 2016a).

In order to assess 3 (ii), the description of each KE method was considered against the different possible combinations of challenge and solution categories defined in the conceptual framework. For example: a factsheet mainly addresses (by altering – level of change) the challenge categories knowledge limitations and (potentially) cognitive limitations. However, it does not directly address practical limitations and, if the interests of stakeholders are not aligned with its topic, it is not likely to be effective (as an effort is required by the stakeholders to read it). The provision of information is empowering but may be controlling if the aim is to persuade or sell (approach to change). This form of analysis makes explicit and reveals aspects of specific KE methods that might not otherwise be critically evaluated by practitioners. If these aspects are not considered, unintended consequences may occur, or tensions develop between different KE methods applied together (Kipling et al., 2019a). Across the stages of analyses a review of literature was undertaken to ensure theoretical sensitivity. In the following sections, summaries of each element of the analysis are followed by discussions of its implications, grounding in, and relevance to, existing theory.

3 Results and discussion

3.1 Agricultural KE in Wales and Uruguay and challenges to engagement

Data from FC and Plan Agropecuario indicated that a wide range of KE methods are applied in both countries *(Figure 2)*. Information provided fell into two classes: i) KE methods representing resources for stakeholders and, ii) KE capacity initiatives (actions to increase access to KE) and resources underpinning KE activities. Within the class of KE methods, several were grouped as forms of remote dissemination, consisting of a variety of written, verbal or visual forms of information-sharing, while interactive KE methods divided into ongoing KE processes and one-off or short-term events.

In general, potential challenges to farmers engaging with the different KE methods used in Wales and Uruguay were found to be similar although the relative importance of each challenge varied between methods. These challenges are recognised and addressed by KE practitioners in both countries *(Table 1)*. Most access-related KE capacity methods *(Figure 2)* focus on raising awareness of the availability of KE, tackling practical, knowledge and cognitive limitations to engagement; this may be sufficient to encourage engagement when the topics addressed are aligned with farmers' interests. However, research has demonstrated that many farmers lack understanding of how agriculture effects climate change and may be unwilling to accept responsibility for reducing these effects (Hyland et al., 2016a). Therefore, to improve engagement with KE, farmers' interests will need to be addressed at one of the three levels of change: worked around, overcome or altered, to achieve engagement with KE focussed on GHG mitigation. Such interventions will use one of the three approaches to change in the conceptual framework (accommodate, control, empower). A range of

TABLE 1
Challenges to accessing KE methods and solutions employed to address them

Challenge category	Relevance of challenge category to described KE methods	Solutions employed (from *Figure 1*)
Interests	Farmers must have the motivation to engage with resources, so KE methods must work around (accommodate) existing interests. If interests are not fully aligned, partial intake/use is possible, but the expectations of users may not be met. Outcomes may be suboptimal when partial solutions are applied, potentially creating future Interest challenges (loss of trust)	Rewards (KE capacity: Access) for engagement with KE methods provide value to farmers who take part, even if they would not otherwise want to; funded projects (KE capacity: Resources) provide resources to undertake the on-farm changes recommended by KE (helps farmers feel that they can implement what they learn); interaction with others from different backgrounds within engagement, altering interests and reducing interest challenges in relation to future opportunities to engage (KE methods: Interactive events and processes); Information which can be taken in with little effort – working around (accommodating) the challenge – e.g. radio programmes; interactive KE at events already attended by farmers, all of the access-enhancing KE methods
Practical limitations	Farmers need time (and potentially money) to access resources	All access-related tools and the funding to support them (Resources: KE funding)
Knowledge limitations	Farmers need to know the resources exist, and have the skills to access them (e.g. ICT skills)	
Cognitive limitations	Farmers may be overwhelmed with other priorities and not have the mental space to assess the value of engagement	Pro-active access-related tools – e.g. awareness-raising by practitioners approaching farmers; rewards which make the value of engagement immediate and obvious.

combinations of these levels and approaches were identified in methods used to address the challenge category 'interests' in Uruguay and Wales *(Table 1)*.

Accommodating approaches to KE which work around farmers' interests encourage engagement by emphasising the co-benefits of GHG mitigation (e.g. improving production efficiency). Solutions which tackle challenges to engagement through rewards or funding are controlling when applied to overcome interests not aligned with KE topics. Unless the KE method itself can then alter these interests or unless policy forces change, engagement is not likely to stimulate implementation of mitigation measures. Highlighting the dangers of climate change may alter interests and stimulate engagement in a controlling or empowering way (depending on the nature of the intervention) with some research suggesting that farmers may be more willing to implement mitigation measures if they believe climate change will negatively affect their farm (Haden et al., 2012). Finally, KE methods with few challenges to engagement (such as radio programmes) and work on KE capacity to improve access – such as awareness-raising by development officers *(Figure 2)* – may be used to alter interests in an empowering or controlling way. This can be achieved by using these methods i) to disseminate information about climate change and its impacts or ii) as conduits via which more controlling or accommodating advertising of KE activities can be delivered.

Given that KE methods differ in terms of the category and size of challenges to stakeholder engagement with them in different contexts, the use of a mix of KE methods in Uruguay and Wales identified here is one strategy to ensure that different groups of farmers have access to the types of KE best suited to them. This nuanced approach is recognised in the targeted nature of much KE provision in the two countries (e.g. courses for new entrants to farming in Uruguay, one-to-one advice and clinics for harder to reach businesses in Wales). Previous studies have recognised that farmers differ in their preference for advice provision and that communication preferences also differ in relation to different topics – for example with financial matters one-to-one advice may be preferred over group discussions (Hilkens et al., 2018).

Using a mix of KE methods also helps ensure that the quality of information being spread within stakeholder communities, for example through farmer to farmer dissemination, is maintained: interactive events and processes can be backed up by knowledge provided through remote dissemination KE methods. The information provided via these 'one-way' channels can be explained and explored through the use of more interactive KE methods. Again, this strategy is consciously applied in the use of the KE methods examined in this study, one example being the online resources that back up information provided to farmers at events held by FC in Wales. Given the diversity of individual challenges to implementing GHG mitigation measures, a mixed KE strategy may be effective, although identifying which categories of challenge are most important in relation to the uptake of specific measures could help improve the choice of solutions (Kipling et al., 2019b), in this case improving KE provision efficiency.

3.2 Differences between KE methods applied in Wales and Uruguay

Despite large socio-economic and agricultural differences between the two countries, the types of KE methods applied in each were similar at the level of analysis presented. This high level of overlap may reflect similarities in categories of challenge to change faced by agriculture, which appear to be relevant across systems and countries (Kipling et al., 2019b) despite wide diversity in specific challenges (e.g. a farmer in Uruguay may need very different knowledge to a farmer in Wales but knowledge limitations will have relevance to both). The wealth of practical knowledge about KE shared globally through bodies such as the GFRAS (Davis et al., 2018) also offers many resources for those facing similar categories of challenge to learn about and apply relevant KE method to address them.

However, there were also some differences between KE methods applied in Wales and Uruguay. One was the inclusion of radio as a remote dissemination KE method in Uruguay. Although not reported by FC as a KE method used in Wales, at the UK and Welsh national levels radio programmes provide information and discussion on farming and related topics, including the BBC's 'Farming Today' programme (www.bbc.co.uk/programmes/b006qj8q). TV broadcasts also carry programmes for rural communities, such as the weekly farming and countryside magazine series 'Ffermio' (www.ffermio.tv). Given increasing experience of and research into the use of farm radio in the developing world (Oswald, 2019) there may now be lessons to learn for the global north in relation to more tailored use of radio especially on climate change related topics. Using radio may be useful for harder-to-reach farm businesses, in terms of benefits such as the low barriers to farmers engaging with content and the potential for delivering localised, targeted content to remote areas (Oswald, 2019). Modern IT (e.g. mobile phones) facilitates interactive forms of radio KE (e.g. non-response voting, phone-ins) and may help address limitations related to one-way remote dissemination KE methods. At the same time, Gilberds and Myers (2012) emphasise the need for more research to understand the issues related to radio broadcasters as knowledge intermediaries.

The types of interactive KE events reported were the same between Uruguay and Wales, with some differences in the context of their use (e.g. only in Uruguay were courses specifically provided for new entrants to farming). However, in the category of interactive processes, exchanges, mentoring and new and retiring farmer matching, were only used in Wales, while in Uruguay one-to-one advice was provided using World Bank funding rather than being provided by the KE service *(Figure 2)*. These differences may reflect differences in the social context of Wales and Uruguay: in Europe, the CAP has been criticised for the negative impact of its individualised focus on the capacity of farmers to work together (Leventon et al., 2017) while social isolation amongst farmers has been identified as a major issue affecting farmers (Truchot and Andela, 2018). Against this backdrop, KE strategies focussing on bringing together individuals with complementary interests or needs may be particularly beneficial. In more general terms the social nature of learning makes interaction between farmers in networks a vital element of KE (Klerkx et al., 2010) highlighting the importance of group-based KE methods. As described, efforts to alter stakeholder interests (in this case through exchanging views with others) are likely to be of particular value in relation to GHG mitigation measures which farmers may not initially wish to prioritise.

3.3 Categories of challenge KE methods could address and their potential mode of operation

Analysis with the conceptual framework highlighted how different KE methods might be expected to address the four different categories of challenge to the implementation of GHG mitigation measures described above. Across all KE method types identified, the analysis indicated that providing new knowledge can give insights to stakeholders that order and simplify previous understanding but that provision can also have a potentially negative impact on cognitive limitations by adding complexity to the understanding of participants *(Table 2)* (Kipling et al., 2019b). Studies on the uptake of agro-environment schemes in the UK have previously suggested that both the mechanism and timing of knowledge sharing can be key to the effects of new knowledge, with the provision of large amounts of complex information in response to initial inquiries often overwhelming farmers (Morris et al., 2000). Considering the need for (and challenges relating to) synthesising and applying knowledge, suggests the importance of thinking beyond the provision of knowledge to how knowledge should be put into practice in given contexts. This insight supports the view of Coquil et al. (2018) who describe how, as more transformative changes (e.g. towards agro-ecological farming practices) are undertaken by farmers, the learning process, the roles of KE practitioners and farmers, and their understanding of the system, can all alter; emphasis moves from making knowledge available towards supporting the learning process of farmers as their perspectives and practice change.

Knowledge limitations may be altered by KE methods in an empowering way or, controlled by pressurising or selling approaches *(Table 2)*. Formats in which providers and participants interact have the potential to reduce controlling elements of KE by offering the opportunity for knowledge sharing and enabling practitioners to shape activities to the needs of participants in real time, highlighting learning as a social process (Klerkx et al., 2010). Additionally, it has long been recognised that the environment in which learning interactions take place can have an important impact on learning processes and outcomes. Environment may facilitate different forms of persuasion which may be purposefully

TABLE 2
Summary of expected impacts of KE methods used in Uruguay and Wales on the four categories of challenge to implementing GHG mitigation measures from the conceptual framework. KE method types as in Figure 1

KE method	Practical limitations (PL)	Knowledge limitations (KL)	Cognitive limitations (CL)	Interests (I)
Remote dissemination (for farmers)		Alter (empower) by direct knowledge provision; controlling element possible in choice of which information to share.	Alter (empower) by provision of new management knowledge/knowledge that simplifies practice. Accommodate (control) if messages are 'sold'. Knowledge may increase perceived complexity, increasing CL. If information is not trusted, it will be evaluated further, again increasing CL.	Alter (empower) through new knowledge and perspectives or overcome/alter (control) if content uses sales approach to re-package old facts or selectively represent new ones.
Interactive events	No direct change, but alteration of KLs and CLs may reveal ways of addressing PLs that were in fact based on issues of knowledge or understanding	As for remote dissemination for farmers but knowledge may also grow through interaction with others. Controlling elements may decrease (vs. approaches without interaction) due to chances to question or increase due to physical context and expression of power relations.	As for remote dissemination for farmers but context and practical demonstration can be used to tackle issues of perceived complexity – Alter (empower) – but may also be used to enhance a sales approach – accommodating (control). Presence of other participants may facilitate synthesis of knowledge/evaluation of messages in an empowering way or be another source of control.	As for remote dissemination for farmers but these processes may occur through direct interaction with others as well as with materials. Physical context and the power relations between individuals may have additional effects that control or empower participants.
Interactive processes			Alter (empower) as ongoing processes enable i) difficulties to be identified and addressed, ii) solutions which simplify rather than adding complexity to be developed, and iii) trust to build between those involved – groups acting as networks for ongoing learning.	Alter (empower) through interaction with others. Overcome/Alter (control) if ideas are 'sold' or if there are power inequalities. If participants are like-minded, may reinforce existing interests (I) – work around (accommodate).

arranged, (e.g. KE providers occupying a raised stage to maintain a separate, controlling position or facilitating empowerment by holding the event in a farm environment that participants feel comfortable in). However, control or empowerment of different groups can also happen accidentally, with positive or negative effects on the goals of the event.

3.4 Potential impacts of methods on non-target stakeholder groups

The conceptual framework focused attention on how the implementation of KE methods can affect the interests and limitations of stakeholders beyond those directly engaged. For the types of KE method defined in *Figure 2*, *Table 3* summarises the nature of these potential impacts.

In relation to the influence of non-target stakeholders on KE, some differences were found between the organisations delivering particular forms of KE in Wales and Uruguay. Within Wales, data on KE methods were collected from FC and KE supplied by other providers (e.g. non-governmental organisations, farm suppliers, veterinarians) were not included, while in response to the shared information from Wales, the Uruguayan KE provider indicated that some of the methods applied in Wales were also available in Uruguay but were provided by other bodies. This mixture of provision brings to the fore the issue of how other stakeholder groups (in this case other KE providers) interact with KE provision.

In particular analysis using the conceptual framework highlighted the potential influence of other stakeholders on KE resulting from both their interests and their limitations *(Table 3)*. This may result in the use of 'controlling' approaches involving pressure to implement or selling of particular solutions to farmers *(Table 2)*. This reveals another challenge for KE practitioners, reflecting the previously recognised complexity of their role in diversified farm advisory systems (Vrain and Lovett, 2016) – the need to identify, understand and manage how other stakeholders influence the scope, content and delivery of KE within the context created by different types of KE methods. In this respect, Uruguayan KE practitioners might draw lessons from the efforts by FC in Wales to avoid sales-type approaches to disseminating information about new technology, including choosing which technologies to highlight based on the views of panels of farmers and KE practitioners before engaging with the companies involved.

A more positive aspect of the influence of non-target stakeholders in relation to GHG mitigation focussed KE activities, is that pressure from customers may drive retailers to try to reduce carbon footprints associated with their suppliers (farmers) (Poore and Nemecek, 2018). In this way, KE processes involving supply chain actors may present opportunities to drive change. Depending on how such drivers act, retailers' pressure for change may represent control over the interests of farmers or an empowering alteration of farmers' interests that enables them to gain higher prices from lower-emissions products. However, consumer preferences for low carbon food products may not always translate into substantial changes in consumption patterns (Kemp et al., 2010) suggesting limits to this driver for change.

TABLE 3
Summary of potential impacts of KE methods on stakeholders not directly engaged.
PL = Practical limitations, KL = Knowledge limitations, CL = Cognitive limitations, I = Interests

KE method	Impacts on non-target stakeholders
Remote dissemination	Other farmers: Empowering alteration of the KLs / CLs of wider groups could arise through the spread of information from those initially engaged. However, there is potential for misinformation / partial information to spread due to the CLs / Is of those passing it on – this may negatively affect: the CLs / KLs of others, the PLs of others if poor knowledge is acted upon, and the trust (Is) of others in future engagement. Information may be spread in a way that seeks to control others' actions, outside the influence of the initial communicator. However, value may be added to knowledge shared by the addition of accumulating experiences of application as information spreads between stakeholders – this may increase levels of trust (or overcome distrust) in external knowledge within the community. Supply chain, research and rural stakeholders: Changes in farmers' KL / CL may affect how they interact with suppliers, customers and those affected by farming activities, including appreciation of their Is, limitations and needs. This may in turn affect the behaviour of those other stakeholders, including their motivation to influence the information farmers receive.
Interactive events	The same issues as identified for remote dissemination apply, plus: Other farmers: Interactions provide opportunities for misunderstandings to be identified and resolved before information is spread further, including weaknesses in the information itself. Facilitated learning / events taking place in a farm context (e.g. demonstration farm visits) may help participants develop a fuller understanding of new knowledge, increasing the likely accuracy of the information they pass on to others. Trust of, and rapport with, KE practitioners built through interactions may motivate more accurate knowledge sharing. KE practitioners: Interactions are likely to alter the limitations and Is of the KE practitioners (and any researchers) involved, potentially improving their understanding of and effectiveness in delivering KE practice (PL, KL, CL) as well as their priorities and motivation (I). However, if only certain groups of stakeholders are engaged (representing particular interests) the outlook (Is) understanding (CL) and knowledge (KL) of KE practitioners may become skewed towards what works for that stakeholder group or towards the Is of that stakeholder group. This may have implications for the style of KE and the content of knowledge shared, and for access to KE by other stakeholders. Supply chain, research and rural stakeholders: Stakeholders with their own Is and limitations may be motivated to shape content, delivery and outcome of interactive events.
Interactive processes	The same issues as identified for interactive events apply but with a decreased likelihood of misinformation or partial information being spread due to the longer-term interaction and growth of understanding within an interactive process (vs. a one-off event). However, influence by specific stakeholders within more involved processes may be deeper. Such influences may increase the possibility that the Is and limitations of KE practitioners become aligned with those of a specific group of stakeholders.

Another revealed potential impact of KE methods on non-target stakeholders *(Table 3)* is how peer to peer spread of shared information could lead to issues with the quality of knowledge being shared. The use of more interactive KE methods and ongoing KE processes could be expected to reduce such problems by providing interactions with providers and opportunities to clarify or question information given. This benefit of KE processes may be particularly relevant in relation to information about GHG mitigation measures, which farmers may not prioritise without engagement in interest-altering interactive activities.

3.5 Interests and limitations of KE practitioners

In both Uruguay and Wales, some KE methods were used as resources for KE practitioners (*Figure 2*: KE capacity: Resources). This recognises the need to support KE practitioners given their key role in how KE methods will be applied and the subsequent outcomes. Previous studies have found that the climate change perspectives of farm advisors can feed through into the advice they give to farmers (Church et al., 2018) suggesting the need to address the interests and limitations of KE practitioners when considering how to improve on-farm implementation of GHG mitigation measures.

In this context, while the use of a mixture of KE methods in Wales and Uruguay may reflect a conscious choice to fulfil strategic goals (see section 3.1) it may also be a pragmatic response to KE practitioners' interests and limitations. In relation to practical limitations (practitioners' time and resources) the in-depth and therefore expensive nature of KE methods (grouped as interactive processes, *Figure 2*) may limit their use in the context of the withdrawal of government funds from KE provision over recent decades (Vrain and Lovett, 2016), as may a lack of skills in facilitating such processes (knowledge limitations). Given the already highly complex role of KE practitioners in diversified advisory landscapes (Vrain and Lovett, 2016) cognitive limitations may affect the extent to which they consider the importance of, learn and use more involved KE methods. Finally, the influence of factors such as the professional self-image of KE practitioners (interests) may also affect the types of KE method made available to farmers.

Considering KE practitioners' interests and limitations highlights that they face challenges in changing their practice. In this study it was observed that, in Wales and Uruguay, KE methods involving more ambitious levels of interaction were reported in discrete, funded, projects (*Figure 2:* KE capacity: Resources) or with limited capacity, relative to broadly available remote dissemination resources. This suggests limitations in the capacity of KE providers to roll out such interactive processes more widely. Addressing these issues, Nettle et al. (2018) examined factors that could support the adoption of novel techniques by KE practitioners, emphasising the need for a supportive context for learning and the importance of processes of experimentation.

Just as potential challenges to farmers accessing KE were identified, analysis of KE methods used as resources for KE practitioners revealed similar potential challenges to engagement by practitioners. However, some unique aspects were revealed. Firstly, the provision of technical information to KE practitioners (*Figure 2:* KE capacity: Reports and magazine for practitioners) highlights the need to effectively bridge the gap between the knowledge domains of researchers and KE practitioners in order to facilitate the integration and co-creation of knowledge from research and practice (Paschen et al., 2018). Differences in the communities or networks of practice (Tagliaventi and Mattarelli, 2006) of these groups can be expected to affect exchange and understanding between them, just as arises between KE practitioners and farmers. This point emphasises the importance of sharing solutions across the research disciplines involved in analysing both research-practitioner and practitioner-stakeholder relationships.

Secondly, analysis using the framework drew attention to how the type and content of KE provided by practitioners may be influenced by the demands of KE recipients (farmers) and other stakeholders, in turn affecting the nature of the topics practitioners themselves demand and engage with for their own development. Such influences may not be conducive to the more transformative changes required to achieve significant reductions in agricultural GHG emissions, given that processes of engagement in which stakeholders are more empowered most often deliver incremental change (Martin et al., 2013). However, KE processes in which farmers are expected to implement externally-derived policies or directions (such as GHG emissions reduction) are more likely to be characterised by bias in power towards the KE practitioner and prescriptiveness in their role, which can have negative consequences on stakeholder attitudes and outcomes (Hilkens et al., 2018; Vrain and Lovett, 2016). A farmer's trust in KE practitioners and the feeling that they are acting in their interests can be vital to the relationship (Ingram, 2008) and this may well be undermined under such circumstances. This tension in KE provision is played out in the way that the privatisation of KE services has led to gaps in provision (Nettle et al., 2017) with demand from farmers (and therefore the supply by KE practitioners reliant on their patronage) not necessarily aligned with policy agendas such as GHG emissions reduction and sustainability.

One potential solution for reducing tensions between the KE topics demanded by stakeholders and societal requirements for agricultural KE came from a specific project in Uruguay. This involved advisors providing farmers with information about the impacts of farming practices on other stakeholders to give the farmers a better understanding of the environmental consequences of their actions and induce them to make changes. Providing open platforms for exchanges between different types of stakeholder has been recommended within processes aimed at developing hybrid, co-generated knowledge to tackle challenges related to agriculture (Nguyen et al., 2014). In this respect, the Uruguayan example may both represent a way to empower bottom-up change towards lower GHG emissions practices by altering farmers' perceptions and enable KE practitioners facilitating such processes to maintain a balanced view of issues without losing the trust of farmers. However, multi-stakeholder interactions must be carefully planned and managed to avoid the damaging consequences of processes in which farmers feel outnumbered, and to address the challenges of developing trust between farmers and other stakeholder groups (Inman et al., 2018). The application of such techniques as a widely used KE method require changes to be made by farm advisors in terms of their skills and practice, highlighting the need for the provision of carefully designed resources for KE practitioners, including the development of networks for the development and sharing of new knowledge and practice (Nettle et al., 2018).

In addition to KE methods used to provide information and training to KE practitioners, other resources can support improvements in KE practice, including the use of decision support tools to facilitate effective interactions with farmers, for example as 'boundary objects' in social learning processes (Eastwood et al., 2012). Given that one of the advantages of modelling is to make invisible processes visible (van Paassen et al., 2007) they have a clear role in helping tackle issues relating to farmers' understanding of how their systems contribute to GHG emissions (Hyland et al., 2016a). Modelling is used in KE in Wales and Uruguay (*Figure 2:* KE capacity: Resources). However, while in Wales modelling used by FC within its KE programme mainly supports improved farm economic performance, in Uruguay it is being directly applied to investigate how farmers might best reduce emissions through the 'Evaluación Medio Ambiental Ganadera' (EMAG) model (https://www.planagropecuario.org.uy/web/102/contenido/evaluación -medio-ambiental-ganadera.html) (Becoña et al., under review). Participatory modelling of Uruguayan farming systems has also been used to inform best practice in climate change adaptation, demonstrating that in these extensive systems adaptive management rather than rigid prescriptions are most likely to be economically resilient (Dieguez Cameroni et al., 2014). In terms of the conceptual framework applied here, this finding reinforces the importance of empowering KE approaches which build the capacity of stakeholders themselves to manage change and (through this) the need to alter farmer interests in relation to mitigation, rather than simply controlling them. Therefore, if any initial interest-related challenges to engagement with KE methods can be overcome, modelling provides an important resource to support KE. However, processes involving modelling must be transparent about limitations and assumptions in their characterisation of systems, in order to ensure findings are appropriately interpreted and used.

3.6 Use of the conceptual framework

The conceptual framework used here facilitated a systematic appraisal of KE methods used in Uruguay and Wales in terms of their capacity to tackle different categories of potential challenge to the implementation of GHG emissions mitigation measures on livestock farms, including consideration of impacts on non-target stakeholder groups and the challenges to farmer engagement associated with each method. This use of the framework represents a technique for systematically organising the thoughts of the implementers of KE strategies including, forcing them to address aspects of proposed actions that would otherwise have remained implicit or unexplored. Combining this form of analysis of KE methods with an exploration of the actual challenges to change and preferred solutions in a specific location (or for a specific GHG mitigation measure) can facilitate the development of effective KE tailored to specific circumstances. In the context of KE in Wales and Uruguay, further exploration of specific applications of KE methods in each country is also important in order to draw lessons from subtle differences in how the KE methods examined here are actually implemented on the ground. Despite these limitations, this study has highlighted important issues to be addressed by practitioners and researchers in relation to the KE methods reviewed, their strengths and limitations, and has explored differences between the two countries in terms of the KE methods they apply.

4 Conclusions

Analysis of KE methods used in Wales and Uruguay using the conceptual framework highlighted i) the focus of current KE methods in terms of the categories of challenge they are likely to address most effectively, and their different modes of working, ii) the need to recognise how non-target stakeholders may affect the use of (and outcomes associated with) KE methods and, iii) the importance of recognising the particular challenges of delivering KE on GHG emissions mitigation measures versus delivering advice on other topics. KE professionals in the two countries may be able to learn from differences in the KE methods they use and how they are applied (such as, in Uruguay: the use of processes in which farmers engage with non-agricultural stakeholders or the use of modelling that demonstrates to farmers the emissions impacts of their practices and, in Wales: the use of exchanges to share knowledge). This study indicated the utility of the conceptual framework in facilitating critical evaluation of KE methods, going beyond an assessment of their practical efficacy to explore the ways that they could be used to drive change, their limitations and the likely impacts of their application, both on farmers and non-target stakeholder groups. Taking these factors into account can support more effective and efficient KE strategies for on-farm GHG emissions mitigation. It forms the basis for aligning the use of KE methods to the actual mix of challenges experienced in particular locations or environments.

Acknowledgements

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Appendix 1

Summary of challenge and solution types within the conceptual framework described by Kipling et al. (2019a); challenges fully described in Kipling et al. (2019b). Descriptions focus on farmers but the challenge categories apply to all stakeholder groups.

A 1.1 Categories of challenge to change

Practical limitations:

A range of challenges relating to resources, including the availability of finance, time limitations and the practicality of adjusting existing systems to allow the adoption of changed practices or equipment (e.g. relating to infrastructure or to the environmental context of the farm).

Knowledge limitations:

Relating to stakeholders' awareness of novel management options or technological solutions, level of knowledge about the risks and benefits of a change, and whether they had the skills to implement them.

Cognitive limitations:

Refers to how the complexity of the farm system and the economic, social, environmental and policy pressures on farmers can restrict the mental space available to weigh up the benefits of and synthesise new information about technology or changes to management. Complexity can be added to when information is not trusted (requiring additional evaluation) and when there are many sources of information.

Interests:

All aspects of decision making relating to what stakeholders want to do: an umbrella for widely researched areas relating to the many influences on decision-making.

A 1.2 Solutions Categories

Levels of Change:

Three levels of change were identified:

i) Work around: solutions which do not change or seek to overcome a challenge, but instead avoid it (e.g. aligning all actions with the existing interests of farmers, rather than seeking to change them),

ii) Overcome: solutions which do not remove or reduce a challenge, but give stakeholders the ability (or force them) to overcome it (e.g. providing funds to buy expensive new equipment),

iii) Alter: solutions that actually alter a particular challenge (e.g. new technology may make a particular task much less time consuming, reducing the practical challenge to its implementation).

Approaches to Change:

Three approaches to change were identified, relating to how a specific solution is implemented:

i) Accommodate: accepts that a challenge exists and takes it into account when making changes (e.g. bringing in new roles incrementally to give time for practical changes to be made),

ii) Control: forces or directs change (e.g. regulation to overcome interests that are not aligned to implementation, or providing resources for only certain types of activity),

iii) Empower: enables the stakeholder to take control of the situation and drive change (e.g. providing training in strategic decision making to reduce cognitive limitations and help the stakeholder achieve what they want to).

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POSITION PAPER

Reducing ruminant numbers and consumption of animal source foods are aligned with environmental and public health demands

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1 The environmental burden of ruminants

There are many ways for efficient management of ruminant systems to provide more food with less environmental impact, such as to improve feeding quality, avoid overgrazing, introduce silvopasture, control parasites, or even feed specific ingredients that reduce the emissions of climate-heating methane (Lemaire et al., 2014; Schader et al., 2015; Landholm et al., 2019). The technical potential for climate change mitigation of these options ranges from 0.2 to 2.4 Gt $CO_{2\text{-eq}}$ yr^{-1} in comparison to the current emissions of 4.1±1.2 Gt $CO_{2\text{-eq}}$ yr^{-1} from the livestock sector (Mbow et al., 2019). Therefore, even the best ruminant production systems cannot avoid putting pressure on the environment. Ruminants inevitably produce methane in their rumens, require land for their feed, and their excretion leads to emissions of ammonia, nitrate, and nitrous oxide, responsible for air pollution, water pollution, and global warming (Steinfeld et al., 2006). Producing 1 kg of boneless ruminant meat requires an average of 2.8 kg human-edible feed that varies between 0.1 to 9.4 kg human-edible feed depending on region and intensity of production – e.g. ruminants in grazing and mixed systems mainly consume roughages (about 90 %; Mottet et al., 2017). However, despite the relatively large environmental impact, ruminant systems produce a relatively modest 18 % of the per capita protein supply in comparison to 60 % from crops (FAO, 2019).

The dimension of the global ruminant livestock production system further amplifies its already high per-product impact. The global ruminant livestock population of around 4 billion in 2017, consisting of 38 % cattle, 31 % sheep, 26 % goats, and 5 % buffaloes (FAO, 2019), has a bodyweight that is more than 10 times the bodyweight of all wild mammals (Bar-On et al., 2018). Their feed requirements and nutrient excretion are exceeding the absorption capacity of natural systems, even when fed sustainably. The environmental footprint of diets containing livestock products is considerably higher than those of plant-based diets (Poore and Nemecek, 2018). The ruminant supply chain emit 5.7 Gt $CO_{2\text{-eq}}$ yr^{-1} (Opio et al., 2013), which is roughly one-tenth of global greenhouse gas emissions. Even if the most efficient and currently available management practices were adopted in the entire agricultural sector, a food system with high levels of animal source foods in general, and ruminant meat and milk in particular, would risk to exceed key planetary boundaries (Springmann et al., 2018a). These include those for climate change, land use, freshwater extraction, nitrogen and

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phosphorus. Model-based analyses showed that the food system has a chance of staying within planetary boundaries only when efficient management practices were combined with dietary changes towards less animal source foods and less food waste (Springmann et al., 2018a).

From an environmental perspective, reducing animal source foods, in particular ruminant-source ones, is imperative in most regions to meet emission reduction targets and other environmental concerns. However, as such calls become louder, they are also facing several prevalent counter-narratives. Here, we provide a novel discussion on three of these narratives that relate to the social, economic, and environmental threats of reducing animal source foods, including i) food and nutrition security, ii) development and livelihoods, and iii) conservation of biodiversity and cultural landscapes. For each point, we highlight how the reduction of animal source foods, in particular ruminant-source ones, can go hand-in-hand with concomitant improvements rather than threats. Our discussion is focused on ruminant systems because ruminants emit higher amounts of greenhouse gases and often have a higher environmental impact than monogastric animals; however, our arguments also hold for monogastric systems.

2 Food and nutrition security

Globally, 821 million people are facing hunger and undernourishment (FAO et al., 2019). Animal source foods can provide protein and micronutrients in food-insecure countries and help diversify mainly starch-based diets (Willett et al., 2019). In large parts of Sub-Saharan Africa, an increase in animal source foods could contribute to improving nutritional status and reduce stunting, in particular for children (Neumann et al., 2002; Bwibo and Neumann, 2003). However, this can also be achieved by environmentally sustainable options that do not include livestock. Bhutta et al. (2013) showed in a comprehensive review that micronutrient supplementation programmes together with the promotion of breastfeeding are the most cost-effective options for improving maternal and child nutrition in low-income countries. But also the broad category of complementary food supplementation has a role to play. Among food-based interventions, the role of home gardening, optionally expanded by some backyard animal husbandry or fish ponds, has been widely discussed as a promising option for improving dietary diversity and strengthening the women's role in the household (Darnton-Hill, 2014). Other interventions, such as conditional cash transfers, have also shown effectiveness in some instances (Lagarde et al., 2009; Pega et al., 2015). Thus, a wide range of options exist for improving maternal and child health, many of which can be considered less environmentally intensive than the promotion of ruminant-source foods.

An additional benefit of promoting more holistic food system options is for long-term health. In 2015, the International Agency for Research on Cancer (IARC), the cancer agency of the World Health Organisation, classified the consumption of red meat, which includes beef, lamb, and pork, as 'carcinogenic to humans' if eaten in processed form, and as 'probably carcinogenic to humans' if eaten unprocessed (Bouvard et al., 2015). In addition to being linked with cancer, the consumption of red and processed meat has also been associated with increased rates of coronary heart disease (Micha et al., 2010), stroke (Chen et al., 2013), type 2 diabetes mellitus (Feskens et al., 2013), and overall mortality (Sinha et al., 2010; Larsson and Orsini, 2014). Although some researchers have questioned the need for recommending reductions in red and processed meat consumption (Johnston et al, 2019), such opinions are not shared by the public health and nutrition community[1], nor by the available epidemiological evidence[2]. At a population level, the public health impacts of red and processed meat consumption are large (GBD 2017 Dietary Collaborators, 2019) and carry a substantial cost burden, in particular in countries with high consumption (Springmann et al., 2018b).

The consumption of red and processed meat exceeds recommended levels in most high and middle-income countries and increasingly in several low-income countries (*Figure 1a*). By 2030, the average consumption of red meat in low-income countries is projected to exceed values recommended on health grounds by the EAT-Lancet Commission on Healthy Diets from Sustainable Food Systems (Springmann et al., 2018c). Conversely, the consumption of fruits and vegetables, which is consistently associated with reduction in chronic disease mortality (Aune et al., 2017), is often too low in low-income countries (*Figure 1b*). Thus, also from a health perspective, a focus on promoting nutritious plant-based foods, such as fruits, vegetables, legumes, and nuts, has arguably greater prospects for contributing to food and nutrition security in the medium- and long-term than the promotion of ruminant-source foods.

3 Development and livelihoods

Livestock creates income and livelihoods for the poorest of the world, with about two-thirds of households in developing countries receiving part of their income from livestock farming, and with almost two thirds of poor livestock keepers being rural women (Davis et al., 2010; Herrero et al., 2013a). In addition to income, animals are often used to provide traction, for asset formation, or as insurance, and their manure can transfer nutrients from grassland into smallholder arable systems (Herrero et al., 2013a). However, such statistics deserve to be put into perspective. Livestock contributes a lower share of income than cropping (Davis et al., 2010), and a dietary transition from animal source foods towards healthier, more plant-based diets may create opportunities that could be more beneficial for smallholders than the foregone income from livestock farming. However, these opportunities may not hold for regions where farmers have limited possibilities for alternative agricultural activities besides livestock farming (e.g. pastoralism in Mongolia, Himalaya,

1 https://www.sciencemediacentre.org/expert-reaction-to-new-papers-looking-at-red-and-processed-meat-consumption-and-health/

2 https://www.hsph.harvard.edu/nutritionsource/2019/09/30/flawed-guidelines-red-processed-meat/

the European Alps, etc.). Horticultural production of fruits, vegetables, legumes, and nuts often accounts for higher net farm income than conventional cropping (Weinberger and Lumpkin, 2005). In 2014, livestock contributed 35 % to the global value of agriculture production of 2.55 trillion international dollars, while cropping systems contributed the remaining 65 %, which included 23 % from fruits and vegetables (FAO, 2019). Increasing the production of fruits and vegetables in line with recommendations would require a massive upscaling of the horticultural sector (see *Figure 1b*) that could be of benefit for livelihoods. Economic land productivities of horticulture are often larger than that of cereals. This offers potential for income growth also to small-scale land-owners shifting from conventional cropping to the horticulture sector without the need to convert pastures or natural forests (Weinberger and Lumpkin, 2005). In general, global agro-ecological zones show that arable land suitable for cereal productions is also suitable for horticulture (IIASA/FAO, 2012). Reducing the consumption of animal source food also decreases the demand for human-edible feed for livestock production (Muller et al., 2017), making conversion of staple cropping for food and feed to the horticulture sector a plausible option. Additionally, labour intensity is much higher in this sector that could trigger high employment effects; moreover, horticultural production in urban and peri-urban areas may also benefit the urban poor (Weinberger and Lumpkin, 2005; Jaenicke and Virchow, 2018). However, the horticultural sector also needs to expand sustainably by avoiding environmentally intensive production and distribution systems, such as heated greenhouses and transport using air cargo (Clark and Tilman, 2017).

Given that most of the growth in the livestock sector occurs in industrialised systems, which not only show poor environmental performance but also low contribution to poor livelihoods (Herrero et al., 2013a), a shift in development aid and research priorities is appropriate (USAID, 2005). Instead of trying only to improve the environmental performance of the industrial livestock sector by subsidies and development aid, which consolidate and promote the livestock sector as such, priority should rather be given to supporting horticultural systems and their value chains. Horticulture currently receives only a minor share of both development aid and research funding (USAID, 2005), despite its critical importance to healthy and sustainable diets.

4 Biodiversity and cultural landscapes

It is argued that ruminants play an important role in maintaining cultural landscapes in many parts of the world, which are shaped by a long tradition of livestock grazing. In comparison to natural land, these semi-natural grasslands can have a higher diversity of plant species (Dahlström et al., 2006; Yuan et al., 2016). However, on the one hand, such grasslands are rich in biodiversity when they are sustainably managed with low-input and appropriate stocking density. As soon as the grasslands are intensely fertilised, the number of species is strongly reduced (Hautier et al., 2009). Overgrazing also has negative effects on biodiversity. Globally, grassland systems

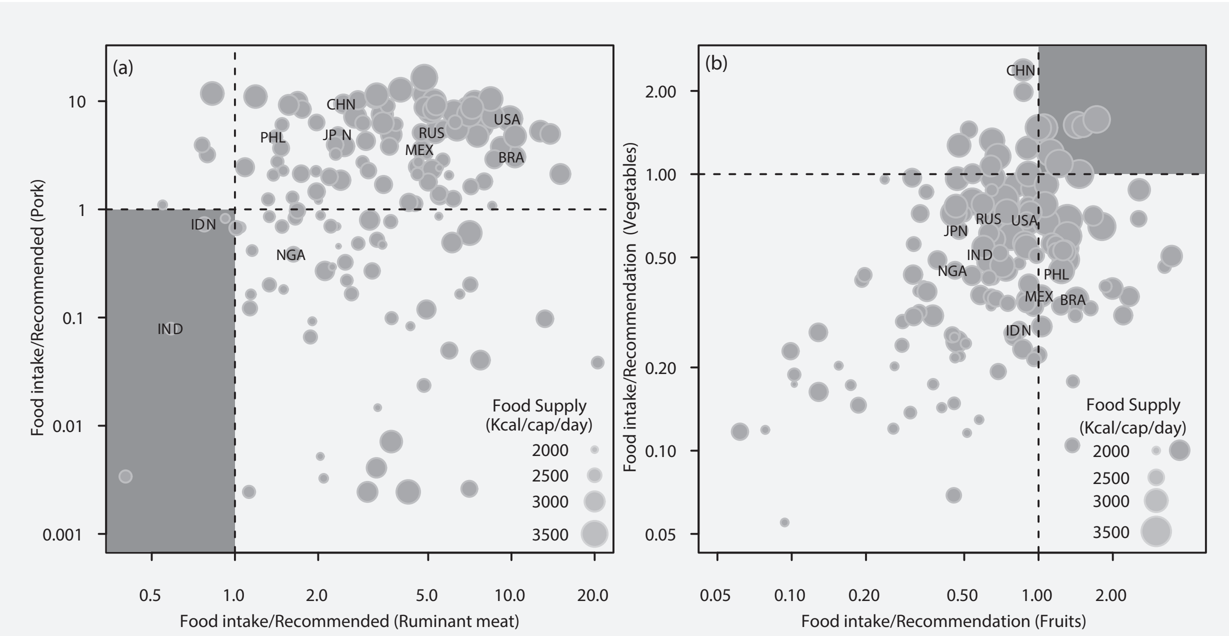

FIGURE 1
(a) Intake of pork and ruminant meat (bovine meat, mutton, lamb) for 2010 across the world in comparison to the maximum intake of 1 serving per week as recommended by the EAT-Lancet report
(b) Intake of fruits and vegetables for 2010 across the world in comparison to the recommended minimum intake (Willett et al., 2019). Circles indicate the food supply of the countries (FAO, 2019). ISO codes of the ten world's most populous countries are displayed on the respective circles. The dark grey areas represent the zone within which the consumption of the respective food is desirable for sustainable and healthy diets (Willett et al., 2019)

produce relatively low amounts of animal source foods in comparison to other systems. Mixed crop-livestock systems are the most important ruminant production systems in both developed and developing countries, producing 69 % of milk and 61 % of ruminant meat globally (Herrero et al., 2013b). Interestingly, the industrial livestock system or overgrazing often dominates in those regions that argue for the need of livestock for maintaining their cultural landscapes (Herrero et al., 2013b). Hence, a much-reduced number of ruminants with sustainable grassland management would be sufficient to maintain cultural landscapes across the world. Such landscape maintenance can be guided by policies to preserve cultural and biodiverse pasture landscapes, producing animal source foods in the meantime.

On the other hand, multiple studies have shown that substitution of animal source foods by plant products in diets would reduce deforestation and the expansion of croplands due to a reduced demand for feed crops (e.g. Weindl et al., 2017; Stehfest et al., 2009; Alexander et al., 2016; Kastner et al., 2012). Additionally, Weindl et al. (2017) showed that low consumption of animal source foods has clear positive net-impact on the carbon stocks by avoiding land use changes from forests to pasture and from pasture to cropland for livestock feeding. There is a vast potential to create different cultural landscapes through afforestation and to increase biodiversity through rewilding (Bakker and Svenning, 2018), which would provide additional climate benefits through carbon sequestering (Bastin et al., 2019). A recent study shows that 205 gigatonnes of carbon can be stored by afforesting areas that would naturally support forest growth, except current agricultural and urban areas (Bastin et al., 2019). However, high level of reforestation, forest restoration, and afforestation can have moderate negative impacts on food security (IPCC, 2019). Nevertheless, the current expansion of ruminant systems is a major driver of deforestation worldwide (Gibbs et al., 2010; Curtis et al., 2018), being responsible for around 70 % of deforestation in South America in 2017 (De Sy et al., 2015). This deforestation is widely associated with negative impacts on climate, biodiversity, and ecosystem services, rather than with the appraisal of new cultural landscapes.

5 Conclusion

The evidence provided shows that the counter-narratives presented for discussion do not offer pertinent arguments against a drastic reduction in animal source foods, in particular from ruminants, as recommended for planetary and public health (Willett et al., 2019). Instead, dietary change towards plant-based diets with a limited amount of animal source foods presents major opportunities for climate change mitigation and adaptation with human health co-benefits (IPCC, 2019). While a world without any livestock production could indeed have negative trade-offs, the current scale of livestock production and consumption of animal source foods in the large majority of world regions exceeds the amounts appropriate for good health food security, development, biodiversity, and cultural landscapes.

Public perception may be misled by world views dating back several decades, when obesity and diabetes was not yet an issue in developing countries, the world population was smaller, environmental pollution from livestock farming was not so pervasive, and ruminants in developed countries were mostly grassland-based. These world views have to be updated, future situations have to be anticipated, and the inertia of the system has to be considered. Today's world population of almost eight billion people cannot sustainably feed four billion ruminant animals. Encouraging intensive livestock systems in many countries may not be farsighted when considering the high growth rates of animal source foods that are already inherent.

Importantly, we do not argue that ruminants and grassland systems cannot be made more productive and sustainable. The innovation here is indeed needed. But we argue that priority should be given to a shift from animal source foods to more healthy and sustainable plant-based foods. Such a shift in priorities implies, for example, that sustainable ruminant systems are incentivised by taxes rather than by subsidies, and that development cooperation, is realigned from supporting the ruminant industry towards promoting horticulture, in line with the shift from undernutrition to overconsumption.

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POSITION PAPER

Greenhouse gases from pastoral farming – a New Zealand perspective

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1 Introduction

New Zealand has a unique greenhouse gas (GHG) profile amongst developed countries in that around half of the emissions in carbon dioxide equivalents (CO_2-eq), some 38 Mt, consist of methane and nitrous oxide (N_2O) originating from agriculture. Since 1990, farming has undergone significant changes with sheep numbers more than halving (now 27 million) and dairy cattle numbers doubling to 6.5 million (including young stock). The geography of New Zealand dictates farming systems. Much of the North Island is rolling to steep upland (known as hill country), suitable only for sheep and beef farming. Along its western side, a chain of mountains runs the length of the South Island. About one quarter of the total agricultural area is sufficiently flat and at a low enough altitude to allow pastoral dairy farming. Eastern areas of both islands are prone to summer drought. In recent years, much lowland has been converted from sheep and beef to dairy farming accompanied by irrigation and an intensive pastoral system utilising substantial fertiliser inputs. New Zealand exports 95 % of its dairy products which supply 30 % of those traded on the world market.

Average herd size is 430 milking cows per 150 ha (2.86 cows per ha). Total effective dairying area is 1.76 million ha with another 0.6 million ha devoted to dairy support, i.e. grazing of young stock, off-farm grazing of dry cows, and cropping. Approximately half the cow population is Holstein-Friesian/Jersey crossbred, one-third Holstein-Friesian and 10 % Jersey. Farmers are paid on kg of milk solids (MS). A separate dollar value is assigned to fat and protein with a minor penalty for milk volume. Most farms milk seasonally, with all cows calving in the spring (July, August, September) and being dried off by the end of May. Average production varies with seasonal weather patterns but is typically 380 kg MS per cow or 1,080 kg MS per ha (approximately equivalent to 9,800 kg fat and protein corrected (FPC) milk)[2]. DairyNZ defines farming systems by numbering from 1 to 5, with System 1 consisting of all-grass home-grown feed through to System 5, where 30 to 50 % of feed is grown off-farm (LIC and DairyNZ, 2018). The majority of farms fall somewhere between these two extremes, but there is a considerable year-to-year variation depending on weather and milk price.

Much sheep and beef farming is now carried out on poorer producing hill country in the North Island and sub-alpine areas in the South Island. These are the core breeding flocks and herds which provide young animals for finishing on better producing, lower altitude farms. Feedlots are rare; the vast majority of animals destined for the meat industry are finished on pasture. The total area used for grazing animals for meat industry is about three times that of dairying. Grain growing and horticulture are practiced only on a small scale, totaling some 70,000 ha.

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2 Definition International Dairy Foods Association, Washington, D.C: Milk is approximately 87 % water and 13 % solids. As it comes from the cow, the solids portion of milk contains approximately 3.7 % fat and 9 % solids-not-fat.

The only government support of agriculture consists of a relatively small amount of funding for research. In 2003, the Pastoral Greenhouse Gas Research Consortium was established as a joint venture between research organisations, the government and farming groups, to determine practical methods of reducing agricultural emissions. Much research has concentrated on methane since it is the largest contributor to New Zealands agricultural greenhouse gas emissions. This also had a political objective as under terms of the Kyoto Protocol and Paris Agreement any decline in methane can be credited against CO_2 from fossil sources.

There have been no legislated controls on CO_2 emissions from the energy, industrial, and transport sectors. These sectors account for 70 % of the increase in national emissions since 1990, the remaining 30 % originating from agriculture. The growth in methane emissions since 1990 has been considerably smaller than the growth in CO_2 emissions from the energy sector. A reduction in methane emissions due to the decline in sheep numbers have partly compensated for the increased methane emissions related to the rise in dairy cattle numbers. It is N_2O, along with direct CO_2 emissions from the application of urea fertiliser, primarily by dairy farmers, that is the largest contributor to direct farm emissions growth from 1990 to 2017 (*Figure 1*; Ministry for the Environment, 2019). Emissions from on-farm fuel use and electricity consumption are insignificant.

The impact of methane emissions on warming has been the subject of some debate. Methane's short lifetime (some 12 years) means that stabilising emissions will result in a decline in atmospheric concentration. In contrast, the emission of long-lived gases such as CO_2 and N_2O need to be reduced to zero in order just to stabilise concentrations. Also subject to debate has been the reasons atmospheric methane concentration stabilised for a period in the early 2000s but has resumed an upward trend since 2009. Evidence suggests much of this additional methane is of fossil origin (Worden et al., 2017).

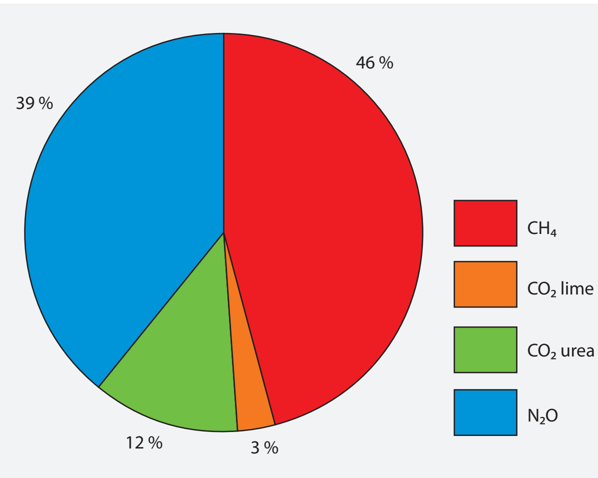

FIGURE 1
Growth in agricultural greenhouse gas emissions from New Zealand 1990–2017. Total growth is 600 kt CO_2-eq y^{-1} estimated by applying linear regression to annual data for each gas. CH_4: methane (Global Warming Potential, GWP 25).
CO_2 lime: direct emissions due to lime application.
CO_2 urea: direct emissions by soil organisms due to urea application. N_2O: nitrous oxide (GWP, 298).

2 Options for methane reduction

To date, no practical, cost-effective methods for reducing methane emissions from pasture-based systems have been developed. This is not surprising as enteric *Archaea* that produce methane (methanogens) have proved very resilient to attempts to manipulate their populations. The following is a summary of research findings.

Breeding

It has been determined that there is individual variation in ruminants in terms of their methane production in pastoral systems. In sheep, a flock reduction of 4 to 6 % can potentially be achieved by targeted breeding (Goopy et al., 2014). Currently, the Livestock Improvement Corporation is carrying out genetic screening for low emission traits in dairy cattle. However, populating a significant proportion of the national herd/flock with low emitters would be a long term process and may conflict with other breeding objectives.

Feed manipulation

Methane production is linked to the fibre content of the feed. Feeding a carbohydrate-rich, highly digestible diet results in a relatively low emissions intensity measured as kg CO_2-eq per kg FPC milk. However, the growing of crops necessary for the carbohydrate component of feed leads to a net increase in the GHG footprint (Williams et al., 2007), especially if feeding total mixed rations (Van der Nagel et al., 2003).

Feed additives under New Zealand conditions have so far been demonstrated either to have only temporary or no effect, or have turned out to be impractical to administer. Procedures have been developed to screen large numbers of compounds rapidly. The ideal compound (such as 3-nitroxypropanol) would inhibit the metabolic pathway for methane production rather than inhibiting the organism itself. The main issue remains administration in a pastoral situation. Although slow-release intraruminal mechanisms are available, they are not designed for long-term use.

Certain forages may have a significant effect, at least in sheep, e.g. forage rape (*Brassica napus*). Work in all these areas is ongoing.

Vaccination

Some progress has been made in developing a vaccine with antibodies being passed in saliva. So far, this has not resulted in significant inhibition of *Archaea* but work is continuing. Large scale clinical trials are still some time away (well into the 2020s), and although eventual efficacy is unknown, it seems unlikely to exceed a 20 % reduction (PCE, 2016).

Effect of stock numbers

The single most significant influence on national methane emissions is stock numbers. Sheep numbers have halved since 1990, and both beef cattle and deer populations have decreased substantially since 2004. Thus, a growing dairy cow population has largely driven the increase in emissions since 2008.

3 GHG reduction by farm management changes

A number of studies on how best management practices affect the environmental impact of pastoral farming have been published. There is an emerging consensus that, despite lower milk production per hectare, the reduction of cow numbers combined with nitrogen input limitation has a neutral to positive effect on farm profitability. Reliance on feed imported from off-farm, typically maize silage or palm kernel expeller, can have a serious negative effect on profitability at times of low milk prices. (Dewes and Death, 2015; Fraser et al., 2014).

It is becoming clear that maximising milk production is poorly correlated to maximising operating profit. Many farms reach a point where marginal costs of producing extra milk exceed marginal returns. After this point, profitability declines; however, this decline may not be obvious if a simplistic analysis which only averages costs and returns is carried out. A more sophisticated analysis, which allows identification of the point of maximum profitability for different levels of milk price, different management strategies, and associated risk, indicates the most resilient system. Evidence suggests the system most consistently profitable over a wide range of economic circumstances is one where stocking rates are low to moderate and cows are predominantly grass-fed with home-grown feed (Beukes et al., 2009; Anderson and Ridler, 2010; Anderson and Ridler, 2017; DairyNZ, 2019). A model constructed by Groot et al. (2012) has demonstrated similar findings on a largely pasture-based organic farm in the Netherlands where decreasing cow numbers and increasing forage intake results in improved environmental performance and profitability.

A significant influence on efficiency is the replacement rate of breeding stock. Animal wastage means both an expense to farmers and an unnecessary source of GHG emissions as it requires the rearing of an excessive number of young stock. A reduction of replacement rate should be achievable by combining better management with improved disease control.

There is a school of thought which proposes that since methane emissions are linked to dry matter intake, there will be little effect of reducing cow numbers if the same amount of feed is consumed. However, the data does not support this. Improved feed conversion efficiency and minimal cow wastage are characteristic of high-efficiency low-input farms, resulting in a lower proportion of consumed feed being devoted to maintenance of animals (Macdonald et al., 2014).

Currently, only around 5 % of dairy farmers operate a high-efficiency low-input system (Dewes and Death, 2015). If such strategies were applied nationwide, reductions in cow numbers and nitrogen usage would likely have a substantial effect on agricultural emissions. However, efficacy is highly dependent on the quality of management, and a proportion of farmers will not be capable of taking advantage of changes without training.

Modelling

Measuring changes at the farm level is not straightforward. Overseer® is a software package most commonly used by farmers and advisors for tracking nutrient flows and losses. Although it can also track GHGs, it is not primarily set up to do so. DairyNZ's Whole Farm Model can be linked to more sophisticated models in order to provide a more accurate picture of emissions for various scenarios at the farm level. However, like most international models, it is primarily used for research purposes. Dynes et al. (2018) have carried out a review of available emission models, many of which are limited in scope.

Soil carbon

There is increasing global interest in the potential for grassland soils to absorb large quantities of carbon of atmospheric origin if managed appropriately. It is well documented that continuous cropping results in substantial carbon loss to the atmosphere (e.g. Sparling et al., 1992). Return to pasture allows recovery; however, unless there is just a single cropping event, this may take some time.

Careful pasture management can build soil carbon, especially in degraded areas. There have been sizeable losses from some New Zealand soil types (peat, allophanic, gley), but many lowland soils have a high carbon content by international measures (Schipper et al., 2017). New Zealand does not report soil carbon losses except for those arising from changes in land use. Consequently, there is a paucity of data.

The role of nitrogen

The use of nitrogen (N) fertilisers has largely driven the intensification of dairying. N not only accounts for the increase in dry matter production of pasture but also the protein content of grass. Bacterial degradation of urine in the top layer of soil is the prime source of both leached nitrate and N_2O emissions, the amounts depending on a product of dry matter consumed and its protein percentage. The rise in N_2O emissions has paralleled the increase in the application of synthetic urea since 1990 (*Figure 2*).

4 Options for nitrous oxide reduction

4.1 Management changes

Reduction of N inputs is an obvious practical strategy. Before 1990, the nitrogen cycle of dairy pasture was driven by atmospheric fixation by clover. Studies have shown a marked reduction in N leached from farms where fertiliser N inputs have been reduced or eliminated. In parallel, reductions in the quantity and concentration of urea excreted in urine and thus N_2O formation can be expected (e.g. Thatcher et al., 2017).

Managing effects of cropping and pasture renewal can have a significant influence on N_2O. Herbicide use speeds up denitrification processes, and soil disturbance affects the potential for both N_2O and CO_2 emissions (Luo et al., 2017). Compaction of soils in wet conditions as a result of high

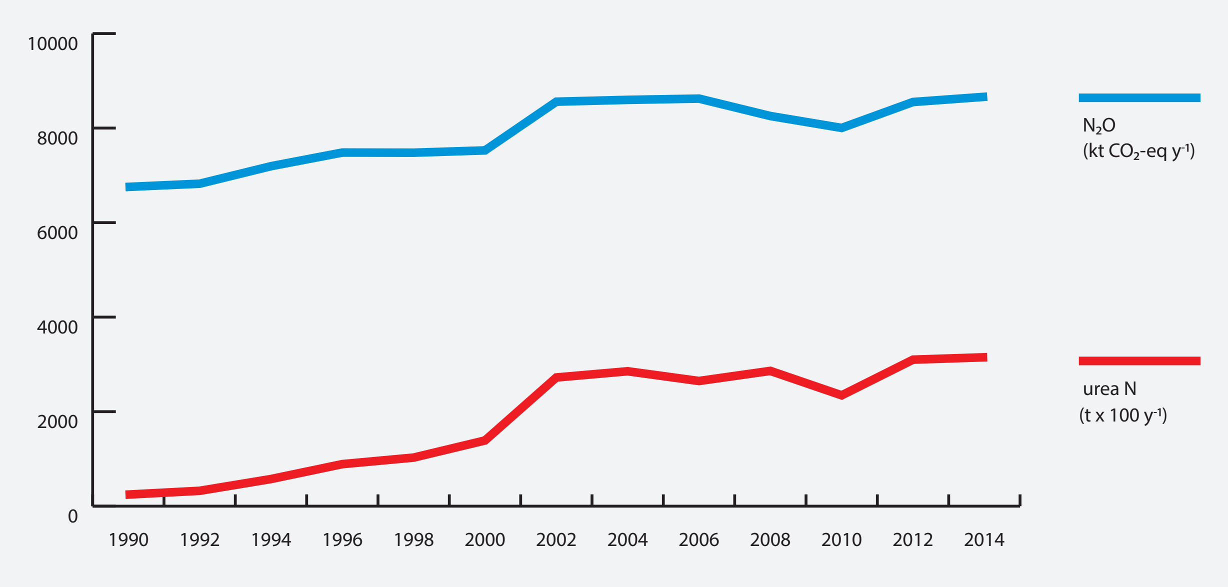

FIGURE 2
N_2O emissions from New Zealand agriculture and the use of urea N fertiliser 1990–2015. The dip of 2007–2009 was due to a spike in the price of urea (Data source: Ministry for Primary Industries; Ministry for the Environment)

stocking rates combined with concentrated deposition of urine and dung can lead to very high rates of N_2O production (Bhandral et al., 2007). Reduced stocking rates, smaller cows, maintenance of soil carbon, and better management of pastures can all help address this issue.

Manure management

On most dairy farms, over 90 % of manure is deposited directly on paddocks. Effluent is only generated when cows are yarded for milking. The housing of dairy cattle is rare in New Zealand and, except at shearing time, non-existent on sheep farms. More common are open concrete yards used for 'standing off' dairy cows temporarily during wet weather in order to prevent pasture damage. Such yards may also be used for feeding supplement. Manure is regularly scraped off and typically stockpiled before being returned to paddocks. Although emissions from this source are relatively minor, simple measures, such as covering stored solid manure to prevent water entry, can make a difference.

Raw effluent washed off milking yards is usually returned to paddocks by spraying. Emissions depend on prior storage and soil conditions at the time of dispersal. Additionally, most farms have 'hot spots' (gateways, troughs, laneways) where dung and urine concentrate, leading to a higher risk of N_2O formation. Ammonia volatilisation is a minor issue and only of any significance when urine or N fertilisers are applied to very dry soils.

Whatever methods of effluent collection and dispersal are employed, the potential amount of N_2O released is linked to the total amount of N cycling through the farm system. Precision testing would allow farmers to make better decisions as to timing and the amounts of N fertilizer to apply, but the most practical option for reducing emissions is to simply apply less.

4.2 Dietary manipulation

There is evidence that the addition of herbs such as chicory (Cichorium intybus) and plantain (cultivars of *Plantago lanceolata*) to pastures substantially reduces urinary urea (Box et al., 2016) and may also reduce methane with no effect on milk production. Plantain, in particular, seems to have an additional diuretic effect, and its roots exude a compound, yet to be identified, which appears to further limit nitrate leaching (DairyNZ, 2019). The effect on N_2O production has yet to be quantified, and work in this area is ongoing.

Reducing the protein intake of cows can be achieved by increasing the carbohydrate content of the diet, typically by supplementing it with a harvested crop such as maize, which reduces grass intake by substitution. As mentioned above, this increases the GHG footprint. High-sugar ryegrasses have been shown to be ineffective at reducing urine urea levels.

4.3 Soil amendments

Dicyandiamide (DCD, in the form of coated urea pellets) was introduced primarily to reduce nitrate leaching with the added effect of reducing N_2O emissions. However, it was withdrawn from the market in 2013 when residues were detected in milk. The urease inhibitor n-butylthiophosphoric triamide (nBTPT) reduces the conversion of urea to NH_3 gas, especially under dry conditions. Currently being spread on some 200,000 ha, its effect on N_2O production is minor (< 1 %).

5 Resource efficiency – comparing sheep and cows

Emissions intensity, expressed as kg CO_2-eq emitted per unit of product, is commonly quoted as a means of assessing the efficiency of a process and may be used to justify an increase in emissions. Thus, unless emissions actually decline, such figures can be misleading when considering the overall environmental impact.

Despite the marked drop in sheep numbers, from 1990 to 2016 total sheep meat exports declined by only 2 %. These figures reflect improved breeding, management, and feeding of a reduced number of animals. Total emissions from the sheep sector fell by 6 Mt CO_2-eq from 1990 to 2015. Emissions intensity at the farm gate fell from 45.2 to 26.6 kg CO_2-eq per kg meat exported. Farm profitability increased during this time in real terms by 110 % (Beef+Lamb NZ, 2018).

Over the same period, national MS production per cow increased by 41 % and production per ha by 72 %, largely as a result of improved genetics and feeding (LIC and DairyNZ, 2018). Methane emissions rose by 7.7 Mt CO_2-eq (129 %; Ministry for the Environment, 2019). The change in emissions intensity from 12.2 to 10.5 kg CO_2-eq per kg MS was relatively modest. Comparisons with international figures are difficult due to the variety of assumptions across different systems and significant uncertainties in the measurement of gases, although it appears New Zealand is likely to be at the lower end of the range of intensities.

Thus, although both the sheep meat and dairy sectors have shown an improvement in emissions intensity, only the production of sheep meat has shown an actual decline in emissions. In contrast, the rise in emissions from dairying has been substantial despite the modest decline in emissions intensity.

6 Conclusions

The most substantial growth in agricultural emissions from New Zealand since 1990 has been due to N_2O plus associated CO_2 released as a result of soil hydrolysis of urea. This is closely tied to the growth in the application of N fertilisers to dairy pastures, which in turn has allowed increased stocking rates. The most practical options currently available for reducing overall agricultural emissions involve a combination of dairy farm management strategies to tackle all three GHGs, particularly focusing on N_2O. A programme including a financial incentive to reduce the use of N fertilisers combined with farmer education would seem the most likely to produce results.

Unless a practical feed additive is developed, the only likely option for reducing methane emissions from ruminants on pasture in the long term is vaccination (PCE, 2016). The success of such interventions and the extent to which they may reduce overall emissions is by no means guaranteed. However, what is certain is that reducing stock numbers will reduce methane emissions. The sheep meat sector has demonstrated that productivity can be maintained despite a dramatic decline in stock numbers, and there is increasing evidence that the profitability of dairy farming can be sustained while reducing stocking rates and N inputs. If widely adopted, such strategies could produce a significant reduction in emissions. A conservative analysis suggests this would come at a zero cost to farmers, but for many it may lead to improved profitability.

6.1 International implications

Pasture-based dairy farming has a number of advantages over total-mixed-ration feeding of permanently housed cows:

- Grass is always the cheapest feed and, if well managed, the most profitable.
- In comparison to other feedstuffs, grazing has the least greenhouse gas emissions if nitrogen inputs are kept to a minimum. Appropriate management of legumes is important to maintain the nitrogen cycle.
- Pastures have the potential to convert atmospheric CO_2 to soil carbon, if well managed.
- Mixed pastures that include herb species appear to have some environmental advantages over a pure grass sward.

Cropping for animal feed as an adjunct to grazing may have a place if the following considerations are taken into account.

- Minimising soil disturbance
- Minimising herbicide use
- Cropping an area once then regrassing immediately after harvest will rebuild soil carbon. It is important not to leave a field bare of vegetation over winter.

Since a considerable fraction of dairy production in the northern hemisphere is not required for the fresh milk market, it would seem likely to be profitable for a proportion of farmers to switch to a seasonal pasture-based system in climatically suitable areas. If these farmers were paid on a milk solids basis, it would promote a diversity of breeds more suited to both a grazing system and the manufacture of butter and cheese.

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RESEARCH ARTICLE

Modelling greenhouse gas emissions from organic and conventional dairy farms

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HIGHLIGHTS

- Our new greenhouse gas (GHG) accounting model includes all relevant farm-scale GHG flows
- The model is sensitive enough to take into account management changes
- Our new model is able to identify the causes of high GHG emissions
- The dairy farms analysed showed an enormous variability in GHG flows
- GHG reduction in dairy farming requires farm-specific optimisation approaches
- C sequestration and land-use change affect GHG footprints

KEYWORDS energy balance, methane emissions, modelling, system approach, land-use change, carbon sequestration, mitigation strategy

Abstract

Dairy farming is a major source of greenhouse gas (GHG) emissions in agriculture. There are numerous scientific studies analysing GHG flows and testing GHG reduction methods in dairy farming, yet very few scientific papers cover all the relevant GHG flows. GHG flows that are difficult to quantify, such as C sequestration in soils, the effects of land-use change (LUC) or the energy input used to produce capital equipment, are not always considered.

This paper describes the development and application of a model for energy and GHG accounting in dairy farming. This new model enables all relevant nutrient, energy and GHG flows to be modelled at farm level. This then forms the basis for system analysis and derivation of GHG mitigation strategies. The model was used on 18 organic and 18 conventional farms in Germany. Calculated CO_2-eq emissions per kg of Energy Corrected Milk (ECM) were 995 g on average for organic farms (org) and 1,048 g on average for conventional farms (con). The largest contribution (55 % (org) and 43 % (con)) to total GHG emissions came from enteric methane emissions (549 g CO_2-eq (kg ECM)$^{-1}$ (org) and 449 g CO_2-eq (kg ECM)$^{-1}$ (con)). On the organic dairy farms, there was an increase in soil humus and therefore carbon storage and sequestration in soils, whereas the GHG emissions for the conventional farms included CO_2 emissions from LUC due to soybean usage. The significantly higher energy input in the conventional systems resulted from the production of energy-intensive concentrates, mineral fertilisers and pesticides, and transportation (imported feed).

This study shows that there are many factors that influence GHG emissions in dairy farming, and that these factors often interact with each other. An increase in productivity is one of several optimisation strategies; however, it must not be at the expense of productive lifetime or require an extremely high amount of concentrates. GHG reduction in dairy farming requires farm-specific optimisation approaches due to the heterogeneity of production systems.

1 Introduction[2]

1.1 Problem description and research gap

Dairy farming is a major source of greenhouse gas (GHG) emissions in agriculture, both nationally and globally (FAO, 2006), and is the focus of public debate on the climate impacts of livestock farming, mainly due to methane emissions.

There are numerous scientific studies which analyse GHG flows and test GHG reduction methods in dairy farming (Thomassen et al., 2008; FAO, 2010; Bell et al., 2011;

[2] This article is based on results published as part of the German research report Frank et al., 2015: Energy and greenhouse gas footprints of dairy farming – Research in the pilot farm network, doi:10.3220/REP_29_2015. Compared with the German research report, the number of farms and years analysed for this paper has been significantly increased, which scientifically substantiates our results and conclusions.

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Hörtenhuber et al., 2011; Vellinga et al., 2011; Zehetmeier et al., 2012; Schueler et al., 2018, Grandl et al., 2019). Studies often focus on methane emissions in relation to feed and milk yield (Kirchgeßner et al., 1991; Jentsch et al., 2007). Few scientific papers claim to quantify all relevant GHG flows in dairy farming; most GHG emission calculations are incomplete. For example, the impact of dairy farming on soil C sequestration and the effects of land-use change for soy production have been included in only a few GHG emission calculations for milk production. Although fossil energy use in dairy systems has been analysed (see Refsgaard et al., 1998; Kraatz 2009), the CO_2 emissions associated with energy input have only rarely been included in GHG accounting for dairy farming.

Although there are systemic differences between organic and conventional dairy cattle farming, it is still unclear which system produces milk in a more climate-friendly way, as studies show differing results. Initial comparison studies focused on enteric methane emissions and concluded that organic dairy farming had higher product-related GHG emissions due to lower milk yields. However, this is a rather superficial conclusion and does not take into account important aspects such as differences in the productive lifetime of dairy cows, feed rations and animal husbandry. Many studies comparing the GHG footprints of organic and conventional milk production exist, but the results are contradictory and inconsistent; a valid assessment is not yet possible (Weckenbrock et al., 2019).

The energy and GHG footprints available for dairy farming are usually based on a small number of experimental or model farms (e.g. Refsgaard et al., 1998; Cederberg and Mattson, 2000; Haas et al., 2001; Thomassen and de Boer, 2005; Kraatz, 2009), or have only been calculated for individual cows (Grandl et al., 2019). A systematic investigation of GHG flows in dairy farming has only been carried out on farms with different structures and production intensities to a limited extent, in part due to a lack of suitable models.

1.2 Description and aims of this study

In this study, we describe a model we developed that can be used to analyse the nutrient (nitrogen, phosphorus and potassium), energy and GHG flows of dairy farms. The aim when developing this model was to record all relevant nutrient, energy and GHG flows related to milk production and to merge them into a system analysis. The model is designed to be applicable to organic and conventional dairy cattle farms. It is largely based on available farm data (field records, feed ration balances, livestock management systems, milk yield tests) and therefore relatively little effort is required for data collection on farms.

In order to compare the two systems, our model for calculating nutrient, energy and GHG footprints was used on 18 organic and 18 conventional dairy farms from four agricultural regions in Germany[3]. The goal was to analyse the individual variability of GHG flows taking into account site conditions, farm structure, feed, milk yield and other determining factors. Ultimately, applying the model should show whether significant GHG reductions are possible at the farm level, which interactions occur and what trade-offs are necessary. We then discuss whether advisory tools based on the model can help to effectively reduce GHG emissions in practice.

2 Material and methods

The calculation of GHG emissions from dairy farming was based on a process analysis comprising the following components and process steps: (1) feed production and feed purchase, (2) feed storage, (3) housing system[4], (4) enteric emissions, (5) milking system, (6) manure storage and (7) heifer production (*Table 3*). All relevant fossil energy inputs in dairy farming related to primary energy usage were included in the calculation of energy balances; solar energy and human labour were not included in the process analysis (*Figure 1*). Each process step is described in a module. The modules are cross-linked, with subsequent modules using input data from previous modules. The CO_2, CH_4 and N_2O flows were quantified, converted into CO_2-eq (CO_2 equivalents)[5] and reported in relation to the products produced (Frank, 2014). The results were then merged into an "Allocation" module; energy and GHG flows were allocated to the products produced (milk, cull cows and calves) according to defined allocation rules based on physical parameters (related to the energy output of the products (calorific value)). The modelling of the individual process steps is described in detail in Frank (2014).

The following GHG flows were included in the model:

- Process-related GHG emissions from the use of fossil energy: Based on a new method for analysing energy fluxes in dairy farming systems (Frank, 2014), GHG emissions from the use of fossil energy on dairy farms (direct emissions) and the production of operating and capital equipment (indirect emissions) were determined.
- GHG emissions related to land use: N_2O emissions were calculated according to IPCC (2006) as a function of nitrogen input using emission factors according to Dämmgen et al. (2007). Using the REPRO model (Hülsbergen, 2003), CO_2 emissions and CO_2 sequestration due to changes in soil humus stocks were calculated based on soil humus and C balances depending on site conditions, crops, cultivation methods, yields and fertilisation. GHG emissions due to land-use change in soybean production were taken into account (FAO, 2010) and values per unit of soybean meal were used according to Hörtenhuber et al. (2011).

[3] This study took place as part of the following research projects: "Ecological sustainability and greenhouse gas emissions of organic and conventional farms – analyses in a network of pilot farms" (Hülsbergen and Rahmann, 2013), and "Increasing resource efficiency by optimising farm crop and milk production taking into account animal welfare quality aspects", funded by the Federal Office of Agriculture and Food (BLE), Germany.

[4] "Housing system" includes animal housing (buildings and installations, bedding and manure removal systems) as well as straw used in farmyard manure systems.

[5] All emissions were converted to CO_2 equivalents [CO_2-eq] using their specific global warming potential (GWP). The GWP index is defined as the cumulative radiative forcing between the present and a selected time in the future, caused by a unit mass of gas emitted now. The GWP (with a time span of 100 years) of CO_2, CH_4 and N_2O is 1, 23 and 296, respectively (IPCC 1997).

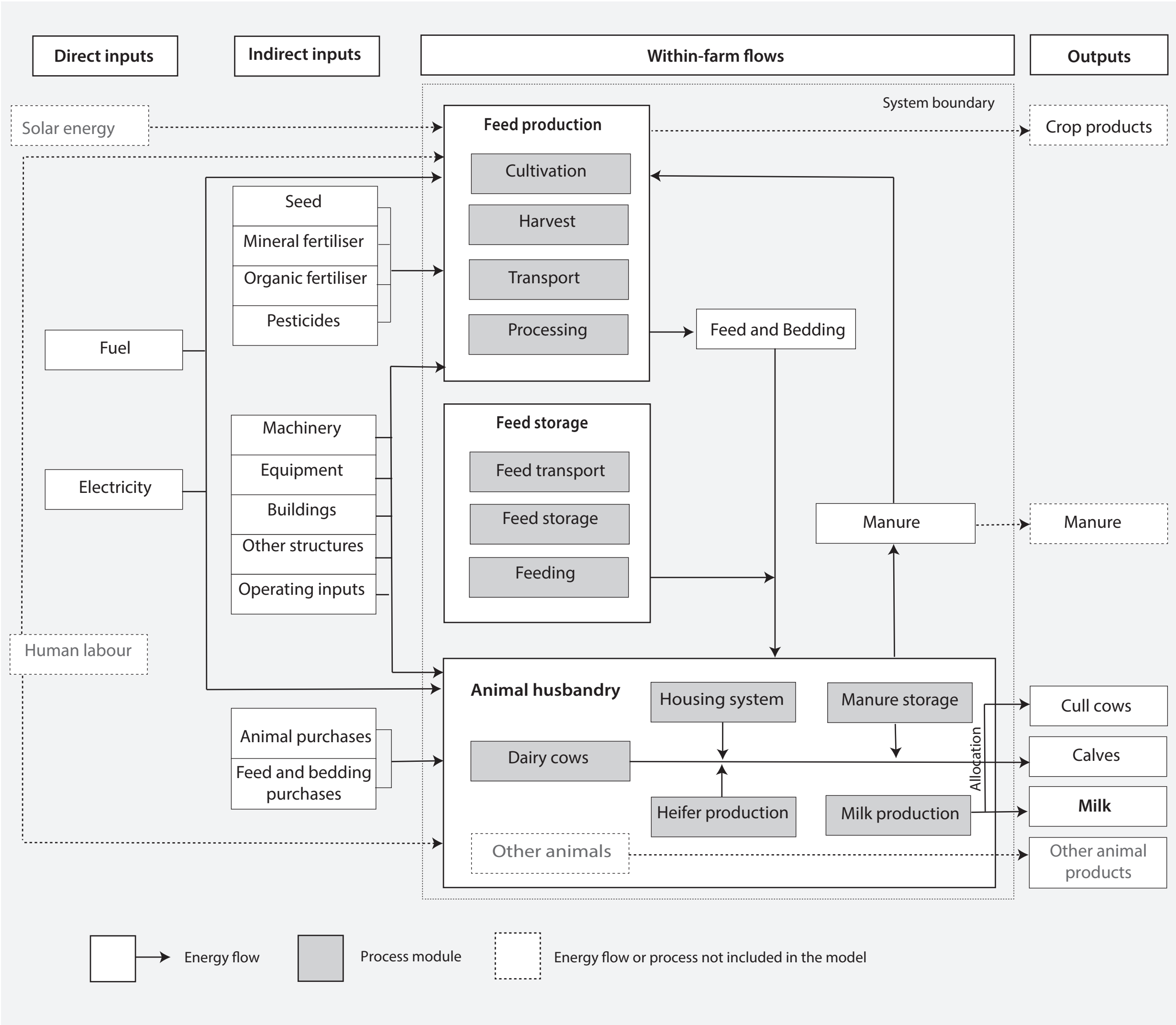

FIGURE 1
Model for calculating energy balances in dairy farming, showing the encompassed system boundaries and nutrient and energy flows

- Enteric GHG emissions: Methane emissions from enteric fermentation in the digestive tract of ruminants were calculated according to Ellis et al. (2007) based on the dry matter intake of cattle.
- GHG emissions from manure treatment and storage: An adjusted version of equation 10.23 according to IPCC (2006) was used to calculate these emissions.

The following amounts of embodied energy were used to calculate energy balances (selected inputs, mean values): diesel: 39.60 MJ l^{-1}, biodiesel: 14.10 MJ l^{-1}, electricity: 11.45 MJ kWh^{-1}, machinery: 108 MJ kg^{-1}, maize seed: 14.62 MJ kg^{-1}, mineral N fertiliser: 35.30 MJ (kg N)$^{-1}$, mineral P fertiliser: 36.20 MJ (kg P)$^{-1}$, herbicides: 259 MJ (kg active substance)$^{-1}$, fungicides: 177 MJ (kg active substance)$^{-1}$, insecticides: 237 MJ (kg active substance)$^{-1}$. The following GHG emission factors were used to calculate GHG emissions: diesel: 3.44 kg CO_2-eq l^{-1}, biodiesel: 0.78 kg CO_2-eq MJ l^{-1}, electricity: 0.72 kg CO_2-eq kWh^{-1}, machinery: 7.76 CO_2-eq kg^{-1}, maize seed: 0.91 CO_2-eq kg^{-1}, mineral N fertiliser: 6.95 CO_2-eq (kg N)$^{-1}$, mineral P fertiliser: 0.70 CO_2-eq (kg P)$^{-1}$, herbicides: 8.33 CO_2-eq (kg active substance)$^{-1}$, fungicides: 5.34 CO_2-eq (kg active substance)$^{-1}$, insecticides: 10.05 CO_2-eq (kg active substance)$^{-1}$. Data sources: Kaltschmitt and Reinhardt (1997), Kalk and Hülsbergen (1996), Hülsbergen (2003), Öko-Institut (2007), Salig and Kölsch (2008), GEMIS (2010), Frank (2014).

In order to quantify energy flows, an energy usage model was developed based on methodology and rules from the REPRO model (Hülsbergen, 2003). A farm is divided into subsystems linked by material and energy flows. In the production process, the output of a subsystem is the input of the following subsystem. To date, only energetic analyses of crop production and/or feed production have been possible using REPRO (Hülsbergen et al., 2001), however, the whole dairy farming system can now be modelled using this new dairy model.

The most important direct energy inputs on dairy farms are fuel and electricity. Indirect energy use includes the energy input required for the production, maintenance and disposal of inputs and capital equipment[6] (Kalk and Hülsbergen, 1996; Hülsbergen et al., 2001; Frank, 2014). The most important indirect energy inputs are machinery and equipment, animal housing and other buildings or structures, and inputs such as seed, fertilisers and pesticides, as well as the purchase of animals and feed. The outputs of a dairy farm are milk, cull cows, calves and heifers, manure and, if applicable, feed. Energy inputs and outputs are assessed using energy equivalents (Gaillard et al, 1997; Kalk and Hülsbergen, 1996; Hülsbergen et al., 2001; Frank, 2014). The energy equivalents used have been adjusted to represent the latest figures.

Our model was used on 18 organic and 18 conventional dairy farms in southern, western, eastern and northern Germany[7], all forming a pilot farm network. Farm selection was based on the following criteria: affiliation with a study region, good data documentation, willingness to actively participate in the project. Farms were also selected based on location; organic farms were paired with a conventional farm in the immediate vicinity (and vice versa), in order to ensure comparable soil and climatic conditions. The modelled energy balances and GHG emissions were evaluated together with the farmers in the pilot farm network, the causes of high emissions were discussed and options for reducing emissions were derived in regional optimisation workshops.

The farm data presented in *Table 1* are mean values for the study years 2009 to 2012. The farms included in the study represent a wide range of soil and climatic conditions, farm sizes and farm structures. The average milk yield (Energy Corrected Milk[8], ECM) of the organic farms (6,491 kg a^{-1}) was significantly lower than that of the conventional farms (8,555 kg a^{-1}). Dairy cows had a longer productive lifetime on the organic dairy farms. The proportion of roughage and forage from pasture in the feed ration was significantly higher in organic than in conventional dairy farming. There were also differences in manure systems, e.g. higher proportions of solid manure systems and grazing on pasture for cows and heifers in organic dairy farming.

[6] An average useful life for machinery and capital equipment was assumed; the actual useful life on the farms could differ. In order to reduce data collection complexity, buildings and structures (e.g. animal housing and feed storage) were categorised to define storage and housing categories. The cumulative energy input for construction, particularly for producing steel, concrete and other building materials, was determined for each building category, and this cumulative energy input was allotted according to useful life. The animal housing used on the dairy farms was allocated to the appropriate category. This pragmatic approach has proven to be appropriate; collecting more detailed information about the building materials used or the actual useful life of animal housing is not possible in practice. For dairy cattle housing, the energy input was calculated per cubicle according to the values calculated by Kraatz (2009), taking into account a useful life of 25 years (see Kalk and Hülsbergen 1996).

[7] Scientists, farmers and farm advisors have been collaborating as part of the pilot farm network since 2009.

[8] Energy Corrected Milk (ECM): values for Energy Corrected Milk (ECM) were determined based on the milk yield and milk constituents in relation to standard milk with 4.0 % fat and 3.4 % protein according to the equation: ECM (kg) = Milk (kg) x [0.38 x (Fat %) + 0.21 x (Protein %) + 1.05] / 3.28

3 Results

Mean CO_2-eq emissions per kg of ECM (delivered milk) calculated using the model were 995 g in the organic farms (org) and 1,048 g in the conventional farms (*Table 2*).

Methane emissions (dairy cows, including replacement calves and heifers) calculated based on milk yield and feed ration made up the largest share of total GHG emissions, with an average of 549 g CO_2-eq (kg ECM)$^{-1}$ (org) and 449 g CO_2-eq (kg ECM)$^{-1}$ (con) (55 % and 43 %, respectively). Methane emissions per kg of ECM from conventional farms were significantly lower than from organic farms, mainly due to higher milk yields and feed rations with a lower proportion of fibre. Methane emissions from manure storage were much lower than emissions from enteric fermentation and did not differ between the two systems (org: 85, con: 77 g CO_2-eq (kg ECM)$^{-1}$).

The N_2O emissions calculated for crop cultivation (soil emissions) and from manure storage are the second most important source of GHG emissions. Emissions were similar for both systems, 253 (org) and 248 (con) g CO_2-eq (kg ECM)$^{-1}$.

There were significant differences in CO_2 fluxes on farms with organic and conventional milk production due to differences in C sequestration and land-use change. According to our calculations, there was C sequestration on the organic farms on average (-57 g CO_2-eq (kg ECM)$^{-1}$) due to an increase in soil humus (attributable to the use of pastures, clover grass leys and fertilisation with farmyard manure). There were also no changes in land use (e.g. no conversion of pasture to arable, no imported soybeans were used). On the conventional dairy farms, on the other hand, CO_2 emissions were calculated as being 82 g CO_2-eq (kg ECM)$^{-1}$) mainly due to LUC, related to the use of soybeans. However, there was mostly no change in humus stocks (*see Table 3*).

GHG emissions from conventional dairy farming associated with the use of fossil energy (192 g CO_2-eq (kg ECM)$^{-1}$) significantly exceeded the GHG emissions from organic dairy farming (165 g CO_2-eq (kg ECM)$^{-1}$). Energy input in milk production was high due to high electricity consumption and the materials needed for milking systems.

Table 3 shows the calculated values for the most important GHG flows for different processes on a dairy farm. The GHG emissions from feed production differed significantly between organic and conventional dairy farming (org: 123 g CO_2-eq (kg ECM)$^{-1}$, con: 308 g CO_2-eq (kg ECM)$^{-1}$), which applies to feed production and feed purchases. The significantly higher energy input in the conventional systems resulted primarily from the use of energy-intensive concentrates (including e.g. soybean or rapeseed meal), as well as from the use of mineral fertilisers and pesticides. On conventional farms, the share of GHG emissions from purchased feed was 11 % (including LUC). There was a higher proportion of energy-efficient pasture (mainly low-input feed production systems) on the organic farms. In addition, ley production (particularly clover grass) was energy efficient. However, the variability of energy utilisation in feed production between individual farms was very high due to the different yield potentials of the various sites and large differences in feed production systems (e.g. harvest frequency

TABLE 1
Pilot farm data: mean values for the study years 2009–2012

	Unit	Organic				Conventional				t-test
		Mean	Min	Max	SD	Mean	Min	Max	SD	
Site conditions										
Elevation	m	**256**	3	780	263	**258**	1	780	262	n.s.
Annual precipitation	mm	**852**	536	1,507	247	**854**	536	1,507	245	n.s.
Mean temperature	°C	**8.5**	6.9	10.8	1.0	**8.5**	6.9	10.8	1.0	n.s.
Average soil quality [a]		**43**	21	54	9	**48**	31	68	10	n.s.
Farm structure										
Agricultural area	ha	**159**	30	1,346	300	**144**	30	973	222	n.s.
Grassland	% FL [b]	**46**	5	100	30	**43**	10	100	30	n.s.
Clover grass	% CL [c]	**36**	0	81	22	**10**	0	46	12	*
Silage maize	% CL [c]	**4**	0	19	5	**24**	0	72	20	*
Grain	% CL [c]	**36**	0	68	21	**40**	0	69	22	n.s.
Stocking density	LU ha^{-1}	**0.94**	0.27	1.56	0.50	**1.64**	0.74	2.72	0.60	*
Dairy farming										
Dairy cows	No.	**52**	19	228	47	**87**	27	452	103	n.s.
Milk yield per cow	kg ECM a^{-1}	**6,491**	4,236	8,840	1,305	**8,555**	6,273	10,275	1,142	*
Age at first calving	months	**30**	27	35	3	**29**	23	34	3	*
Productive lifetime	months	**41**	27	81	14	**30**	25	38	4	*
Calving interval	days	**402**	368	464	23	**406**	367	437	19	n.s.
Feed										
Roughage	% DM [d]	**90**	77	100	7	**71**	51	93	11	*
Pasture	% DM [d]	**26**	1	48	15	**7**	0	34	11	*
Concentrates	% DM [d]	**10**	0	23	7	**29**	7	49	11	*
Soybean meal	% DM [d]	**0**	0	0	0	**3**	0	9	3	*
Manure system										
Manure	%	**44**				**17**				
Slurry	%	**56**				**83**				

* significant at level $p \leq 0.05$, t-test

[a] Soil value, determined using the German system of soil evaluation: a soil value of 100 = highest soil quality

[b] % FL: % farmland
[c] % CL: % crop land
[d] % DM: % dry matter

and forage conservation methods such as silage and hay production).

N_2O emissions in feed production contributed, with 149 g CO_2-eq (kg ECM)$^{-1}$ on the organic farms and 129 g CO_2-eq (kg ECM)$^{-1}$ on the conventional farms, to total emissions. The N_2O emissions per kg of ECM were dependent on the N input (mineral N, N from organic fertilisers or nitrogen fixation by legumes) per hectare of feed, feed yield, feed ration and milk yield. The conventional farms had a significantly higher fertiliser N input than the organic farms, but due to higher yields this did not result in higher product-related N_2O emissions.

With regard to animal housing, the organic farms had a higher product-related energy input due to the high proportion of solid manure systems requiring large amounts of straw. Hence, there were also GHG emissions from straw production. Different requirements in terms of access to pasture and exercise areas also affected GHG emissions. Although housing on the organic farms often had a lower energy input due to its design, this was offset by the bedding required. There were no differences between the systems in terms of manure removal and fertiliser storage.

Total GHG emissions from raising heifers for herd replacement were comparable in both systems (org: 251 g CO_2-eq (kg ECM)$^{-1}$, con: 233 g CO_2-eq (kg ECM)$^{-1}$). Raising replacement heifers mainly generated GHG emissions from the use of fossil energys, enteric CH_4 emissions and N_2O emissions from feed production and fertiliser storage. The heifers raised on organic farms were older at first calving (*Table 1*), but dairy cows had a longer productive lifetime and a higher number of lactations than cows on conventional farms, meaning fewer heifers were needed for herd replacement. The high variability of emissions between farms shows the significant influence farm management and local conditions had and, to some extent, the potential for reductions in GHG emissions.

TABLE 2

GHG emissions from dairy farming per kg ECM including replacement heifers (g CO_2-eq (kg ECM)$^{-1}$), pilot farms (2009–2012)

Process, source	GHG	Organic g CO_2-eq (kg ECM)$^{-1}$				Conventional g CO_2-eq (kg ECM)$^{-1}$				t-test
		Mean	Min	Max	SD	**Mean**	Min	Max	SD	
Energy input [a]	CO_2	**165**	133	218	25	**192**	165	222	19	*
C sequestration, LUC [b]	CO_2	**-57**	-171	38	56	**82**	-71	235	71	*
Crop cultivation [c]	N_2O	**192**	156	263	29	**191**	140	247	30	n.s.
Enteric fermentation [d]	CH_4	**549**	473	706	71	**449**	392	574	46	*
Manure storage [e]	N_2O	**61**	33	95	16	**57**	36	90	13	n.s.
Manure storage [f]	CH_4	**85**	34	151	28	**77**	18	127	30	n.s.
Total GHG emissions	GHG	**995**	**835**	**1,397**	**149**	**1,048**	**901**	**1,269**	**88**	**n.s.**

* significant at level $p \leq 0.05$, t-test

[a] CO_2 emissions from the use of fossil (primary) energy (direct emissions and indirect emissions)

[b] CO_2 emissions due to changes in soil humus stocks and land-use change

[c] N_2O emissions from fertiliser and soils (feed production for cows including heifers)

[d] CH_4 emissions from enteric fermentation (cows including replacement heifers)

[e] N_2O emissions from manure treatment and storage (cows including heifer production)

[f] CH_4 emissions from manure treatment and storage (cows including heifer production)

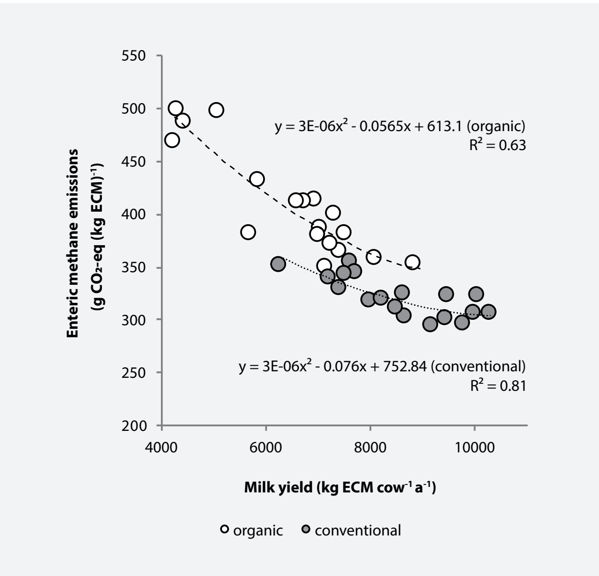

FIGURE 2

Enteric methane emissions of dairy cows in relation to milk yield per cow (without heifer production); Y = enteric methane emission, x = milk yield

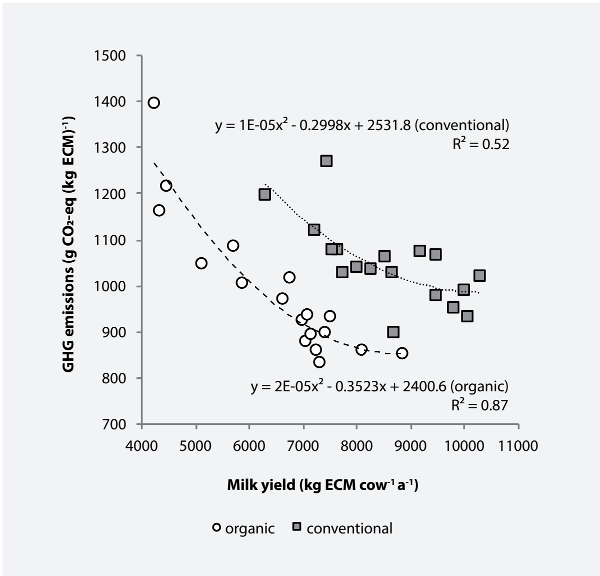

FIGURE 3

Total GHG emissions of milk production in relation to milk yield per cow; Y = GHG emission, x = milk yield

A key factor influencing the amount of CH_4 emissions and total GHG emissions was milk yield. Enteric CH_4 emissions decreased with higher milk yield (*Figure 2*). For the same milk yield (e.g. 8,000 kg ECM per cow), product-related CH_4 emissions from the organic farms were approximately 50 g CO_2-eq (kg ECM)$^{-1}$ higher than CH_4 emissions from conventional farms.

As yields increased, total GHG emissions decreased (*Figure 3*). For the same milk yield (e.g. 8,000 kg ECM per cow), product-related GHG emissions from the organic farms are approximately 200 g CO_2-eq (kg ECM)$^{-1}$ lower than from conventional farms (calculated using regression functions). This value is higher than the mean difference of 53 g CO_2-eq (kg ECM)$^{-1}$ for product-related GHG emissions (org: 995 g CO_2-eq (kg ECM)$^{-1}$ vs con: 1,048 g CO_2-eq (kg ECM)$^{-1}$, see *Table 2*) due to the different mean milk yields (org: 6,491 kg ECM per cow, con: 8,555 kg ECM per cow, see *Table 1*).

The slope of the regression curves shows that significant GHG reductions can be achieved by increasing yields if

TABLE 3
GHG emissions for each process (g CO_2-eq (kg ECM)$^{-1}$), pilot farms (2009–2012)

No.	Process, source	GHG	Organic g CO_2-eq (kg ECM)$^{-1}$				Conventional g CO_2-eq (kg ECM)$^{-1}$				t-test
			Mean	Min	Max	SD	**Mean**	Min	Max	SD	
1.	**Feed production** [a]	**GHG**	**123**	**52**	**237**	**50**	**308**	**197**	**406**	**59**	*
1.1	**On-farm**	**GHG**	**102**	**14**	**183**	**52**	**196**	**109**	**287**	**49**	*
	Energy input	CO_2	39	18	65	13	59	32	105	22	*
	Fertilisation, soils	N_2O	149	109	193	23	129	77	210	36	*
	C sequestration	CO_2	-86	-199	0	57	8	-89	104	47	*
1.2	**Purchase (off-farm)**	**GHG**	**21**	**1**	**79**	**23**	**112**	**7**	**224**	**59**	*
	Energy input	CO_2	7	2	23	6	30	4	54	15	*
	Fertilisation, soils	N_2O	6	0	33	8	25	0	50	15	*
	C sequestration	CO_2	8	-4	53	14	16	0	60	14	*
	LUC (soy)	CO_2	0	0	0	0	41	0	115	41	*
2.	**Feed storage**	**GHG**	**11**	**3**	**22**	**6**	**13**	**6**	**18**	**3**	**n.s.**
3.	**Housing system**	**GHG**	**23**	**8**	**54**	**15**	**11**	**5**	**27**	**5**	*
	Energy input	CO_2	15	7	26	7	9	5	21	4	*
	Fertilisation, soils	N_2O	1	0	10	2	2	0	7	2	n.s.
	C sequestration	CO_2	7	-3	28	9	0	-12	3	3	*
4.	**Enteric fermentation**	**CH_4**	**410**	**349**	**498**	**49**	**321**	**294**	**355**	**19**	*
5.	**Milking system**	**CO_2**	**46**	**34**	**60**	**5**	**44**	**42**	**48**	**2**	**n.s.**
6.	**Manure storage**	**GHG**	**131**	**77**	**227**	**35**	**118**	**47**	**163**	**34**	**n.s.**
	Energy input	CO_2	14	8	28	6	11	5	16	3	*
	Manure storage	N_2O	35	12	67	12	33	9	48	10	n.s.
	Manure storage	CH_4	82	30	149	28	74	16	124	30	n.s.
7.	**Heifer production**	**GHG**	**251**	**132**	**423**	**73**	**233**	**164**	**437**	**68**	**n.s.**
	Energy input	CO_2	33	17	56	10	26	18	51	8	*
	Fertilisation, soils	N_2O	36	19	60	10	35	23	69	12	n.s.
	C sequestration	CO_2	14	6	29	6	13	3	35	9	n.s.
	LUC	CO_2	0	0	0	0	4	0	12	3	*
	Enteric emissions	CH_4	139	74	230	40	128	87	224	32	n.s.
	Fertiliser storage	N_2O	26	14	43	7	24	16	42	6	n.s.
	Fertiliser storage	CH_4	3	2	5	1	3	2	5	1	n.s.

* significant at level $p \leq 0.05$, t-test

[a] Feed production and feed purchase for dairy cows

the initial yield level is relatively low. For example, doubling the annual milk yield from 4,000 to 8,000 kg ECM on organic dairy farms would lead to a reduction of about 450 g CO_2-eq (kg ECM)$^{-1}$ (about 33 %). However, at even higher milk yields, the potential for GHG reductions is much smaller. Further increases in yield require a higher proportion of concentrates in the feed ration (with the associated high energy input and GHG emissions from feed production) and cow productive lifetime decreases (requiring more herd replacement).

The organic farms had the lowest GHG emissions at around 8,000 kg ECM, whereas none of the conventional farms achieved the theoretical minimum of product-related GHG emissions, even at 11,000 kg ECM.

4 Discussion

4.1 Discussion of methods

Our new model for GHG accounting in dairy farming is capable of modelling different types of farms (for example, organic and conventional), farm sizes and site conditions. This is shown by the application of the model on the 36 pilot farms, all with very different production conditions. Model sensitivity is such that changes in management can also be simulated, e.g. in forage production and housing systems. All model calculations are based on the same methodology, namely process analysis, as well as the algorithms and parameters specified in the model, so that the results for different farms are comparable with each other.

Our model is closely linked with the REPRO environmental management model (Hülsbergen, 2003). The REPRO model analyses crop production, i.e. feed production and energy balance in crop production (Hülsbergen et al., 2001), soil humus dynamics (Brock et al., 2012; Leithold et al., 2015) and farm nutrient cycles (Lin et al., 2016). In the REPRO model, feed production processes are analysed for each field and include the use of organic fertiliser along with its resulting GHG flows (NH_3, N_2O and CO_2 emissions, and C sequestration). These results are included in the calculation of GHG emissions for dairy farming (see *Table 3*, process 1.1).

Our dairy model uses relevant data from REPRO, however, the process steps – feed storage, housing system, metabolism, milk production and manure storage – are modelled using the new dairy model. By combining both models, all the relevant GHG flows in dairy farming can be simulated in detail.

Modelling dairy systems is challenging due to the extremely complex and numerous subsystems, processes and interactions in dairy farming. In addition, animal housing and technical systems are highly variable, and are often specially designed for each individual farm. Therefore, simplifications were required to make the model applicable. For example, structures for feed storage and animal housing were grouped into categories, and corresponding parameters were derived for each of these storage and housing categories, such as the energy input and GHG emissions required for production. Buildings and structures on the pilot farms were assigned to these storage and housing categories. Comparable methodological approaches were used by Kraatz (2009) and Dux et al. (2009) to calculate energy input in dairy farming. Defined standard procedures were also used to simplify the analysis of heifer production, whereby a reduction in accuracy was expected. Using exact, farm-specific data would have been extremely complex and fraught with uncertainties.

Modelling the GHG flows in dairy farming requires the collection of operating data from farms, and thus good data documentation and cooperation from farm managers. To minimise the effort required for data acquisition, less significant subprocesses can be simplified and aggregated. However, processes that are critical to the energy and GHG footprints, such as feed production, require detailed modelling. Our model is designed for use on farms and to process operational data. Despite some uncertainties, our model can calculate complete energy and GHG footprints for dairy farms. The model was designed to enable a comparison of results.

Uncertainties and errors in the model result from

- (a) inaccuracies in the collection of production data on the farms. For example, the grassland (pasture) feed yield can only be estimated based on feed intake and checked for feasibility using feed balances
- (b) errors in calculating nutrient and energy balances. For example, energy balances assume average energy equivalents that do not correspond exactly to operational or regional conditions. Due to the complexity of animal husbandry systems (buildings for animal housing and milking systems) and the required model simplifications, farm-specific conditions can only be approximated by the model. The humus balance can only indicate approximate C sequestration values, since only the most important drivers are included
- (c) GHG accounting using GHG emission factors and algorithms that are a drastic simplification of complex conversion processes
- (d) including LUC and the modelling of the associated GHG flows is highly controversial; there are different methodological approaches for the quantification of GHG emissions caused by LUC.

Overall, it should be noted that the new dairy farming model is a compromise between the scientific goal of describing all GHG flows as completely and accurately as possible, and practicality, which necessitates simplifications of complex milk production systems. Sensitivity analyses and error analyses of the individual model components can be found in Hülsbergen (2003) and Frank (2014).

4.2 Discussion of results

The analysis of GHG flows in dairy farming shows that many interacting factors determine GHG emissions. An increase in productivity is one of several optimisation strategies; however, it must not be at the expense of productive lifetime (number of lactations, effort required for herd replacement) or require an extremely high proportion of concentrate in the feed ration. On the farms we analysed, organic farms with milk yields of 7,000 to 9,000 kg ECM a^{-1} had the lowest GHG emissions of 800 to 900 g CO_2-eq (kg ECM)$^{-1}$. On the other hand, conventional farms with an output of 9,000 to 10,500 kg ECM a^{-1} had GHG emissions of 900 to 1,050 g CO_2-eq (kg ECM)$^{-1}$.

As frequently described in the literature (e.g. Flachowsky and Brade, 2007), an increase in milk yield per cow results in a decrease in enteric methane emissions per kg ECM. An increase from 4,000 to 8,000 kg of ECM cow^{-1} a^{-1} resulted in a CH_4 reduction of around 100 g CO_2-eq (kg ECM)$^{-1}$ for the organic pilot farms. For the conventional pilot farms, the potential for reducing CH_4 if output were to be increased from 7,000 to 10,000 kg of ECM cow^{-1} a^{-1} was only around 30 g CO_2-eq (kg ECM)$^{-1}$. Methane emissions can be reduced by changing feed quality and feed composition (Flachowsky and Brade, 2007), however, this may only be possible to a limited extent due to specific site and production conditions (e.g. regions where permanent pasture is dominant), or due to certain requirements in organic farming. However, our research also shows that increasing milk yield is just one of many GHG mitigation strategies and that an increase in performance is neither possible nor plausible for every farm. Among other things, it could conflict with other goals, such as replacing roughage produced in an extensive system with concentrates that require a lot of energy to produce, or negative effects on productive lifetime and animal health. Intensification of feed production and grassland should also not be exaggerated in order to avoid negative environmental effects, such as a reduction in biodiversity. The pilot farms network gives us the opportunity to study the trade-offs between the intensity of milk production systems, product-related GHG emissions, and other environmental effects.

Feed production contributes significantly to energy use and greenhouse gas emissions from milk production (see *Table 3*). Although higher amounts of nitrogen fertiliser are used on conventional than on organic farms (Hülsbergen and Rahmann, 2013), when higher forage yields and milk yields are taken into account, the product-related N_2O emissions from feed production are at about the same level (*Table 3*). The farms studied did not show significant over-fertilisation of feed production areas, which is due in part to moderate stocking rates (livestock farming based on available land area) (see *Table 1*).

There was enormous variability in the GHG flows within individual processes and in the product-related GHG total emissions for the pilot farms. One reason was the wide variety of site conditions and milk production systems on the farms (*Table 1*). Farm management also had a significant impact. Although systemic differences between organic and conventional dairy farming were found in some GHG flows (*Table 2* and *Table 3*), the differences between farms within each system were much greater. In future, system comparisons between organic and conventional agriculture should take this variability in results, as well as uncertainties and possible errors, better into consideration. A simple comparison between organic and conventional farming without taking variability into account could lead to incorrect assessments.

In order to identify the site-specific productivity optimisations necessary to achieve the largest possible reduction in GHG emissions, additional farms and locations need to be analysed and included. Model calculations and sensitivity analyses (Frank, 2014), in which the influencing parameters are varied and a wide range of productivity values are analysed, could supplement the farm analysis, since insignificant and random farm-specific factors are eliminated from the analysis.

5 Conclusion

Our investigations show that a GHG reduction in dairy farming requires farm-specific optimisation approaches due to the heterogeneity of production and operating systems. A one-size-fits-all approach is not particularly effective. Our new model is able to identify the causes of high GHG emissions and to compare farms (see, for example, benchmarking in *Figures 2* and *3*). Within the pilot farm network, measures for reducing GHG emissions were derived during optimisation workshops with the farmers, and their effects on GHG footprints were analysed using the model. It has often been shown that individual measures (for example, increasing milk yield to the maximum) do not solve the problem because they can have a negative impact elsewhere (such as higher concentrate requirements and decreasing cow productive lifetime).

As our study confirms, organic dairy farming can increase soil humus and contribute to soil carbon sequestration. Dairy cattle can use grassland biomass and therefore contribute to the conservation of ecologically valuable grassland. Overall optimisation which takes into account interactions between feed production, animal husbandry, fertilisation, humus and nutrient management, among others, is required. It should also be emphasised that the assessment and optimisation of environmental sustainability in dairy farming should include other relevant environmental areas, such as soil protection and the preservation of potable water sources and biodiversity, in addition to GHG flows and impacts on the climate.

Our experience with the pilot farms shows that farm managers are increasingly interested in implementing climate change mitigation measures in dairy farming. Our model should therefore be developed further so that it can be used successfully, not only for scientific research, but also by farm advisory services.

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RESEARCH ARTICLE

Modified approach to estimating daily methane emissions of dairy cows by measuring filtered eructations during milking

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HIGHLIGHTS

- **Maximum eructation measured by peak height or peak amplitude can provide reliable metrics to estimate daily CH_4 emissions**
- **We recommend using peak amplitude as it removes any background emissions**

KEYWORDS **dairy cow, enteric methane, spot measurement, sampling**

Abstract

The aim of this study was to compare metrics for quantifying enteric methane (CH_4) emissions from individual cows during milking using frequent spot measurements and peak analysis methods. An infrared gas analyser was used to measure the CH_4 emitted by cows, and eructation peaks were identified using a Signal Processing Toolbox provided by Matlab. CH_4 emissions were quantified by gas peak height, peak amplitude and average concentration, and were expressed in grams per day and CH_4 yield (grams per kilogram of dry matter intake (DMI)). Peak analysis measurements of CH_4 were obtained from 36 cows during 2,474 milkings, during which cows were fed a ration containing between 39 and 70 % forage. Spot measurements of CH_4 were compared to a separate dataset of 196 chamber CH_4 records from another group of 105 cows, which were fed a ration containing between 25 and 80 % forage. The results showed that the metrics of CH_4 peak height and CH_4 peak amplitude demonstrated similar positive relationships between daily CH_4 emissions and DMI (both $r = 0.37$), and a negative relationship between CH_4 yield and DMI ($r = -0.43$ and -0.38 respectively) as observed in the chamber measurements ($r = 0.57$ for daily emissions and $r = -0.40$ for CH_4 yield). The CH_4 metrics of peak height and peak amplitude were highly repeatable (ranging from 0.76 to 0.81), comparable to the high repeatability of production traits (ranging from 0.63 to 0.99) and were more repeatable than chamber CH_4 measurements (0.31 for daily emissions and 0.03 for CH_4 yield). This study recommends quantifying CH_4 emissions from the maximum amplitude of an eructation.

1 Introduction

The process by which ruminants convert plant material into useful products such as meat and milk through rumen fermentation results in a loss of energy in the form of CH_4 emissions. The animal removes CH_4 building up in its rumen by repeated eructations of gas through its mouth and nostrils. Globally, dairy farming contributes to 20 % of total greenhouse gas emissions coming from the livestock sector, with enteric CH_4 being the largest source of dairy emissions (Gerber et al., 2013). Historically, CH_4 produced by livestock was regarded as wasted dietary energy and an inefficiency in feed utilisation. This is still the case, but CH_4 is also now seen as a pollutant and potent greenhouse gas. Although a large proportion of the variation in CH_4 emissions can be explained by diet composition and feed intake (Bell and Eckard, 2012; Niu et al., 2018), there is additional variation among animals, which may allow selective breeding (de Haas et al., 2011; Garnsworthy et al., 2012; Breider et al., 2019).

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Historically, most studies to assess CH_4 emissions from cattle have been performed using respiration chambers (Blaxter and Clapperton, 1965; Mills et al., 2003; Ellis et al., 2007; Yan et al., 2010), which is seen as the 'gold' standard for measuring emissions. However, respiration chambers are impractical for estimating emissions from individual animals on a large scale in national populations and on commercial farms. Approaches such as the sniffer method to measure enteric CH_4 emissions from individual animals on commercial farms are being developed (Bell et al., 2014a; Lassen and Løvendahl, 2016) now that more portable gas analysis equipment is available, and that frequent gas sampling at the robotic milking station feed bin whilst individual cows are being milked has been found to correlate (r = 0.89) with the chamber measurements of total CH_4 production from the same cows (Garnsworthy et al., 2012). This approach of taking frequent 'spot' measurements of CH_4 within a day (expressed in various units that were measured such as the CH_4 emission rate calculated from the area under CH_4 peaks, average concentration and the ratio of CH_4 to carbon dioxide) has been found to be a repeatable measure (Huhtanen et al., 2015, Bell et al., 2014b; Negussie et al., 2017). However, to be used as a reliable measure, the data requires processing to account for error sources such as cow head position (Huhtanen et al., 2015) and the number and timing of measurements (Cottle et al., 2015; Hammond et al., 2016). The location of the animal's head relative to the gas sampling tube can be determined using a proximity sensor (Huhtanen et al., 2015), or alternatively using data filtering methods to identify CH_4 eructation peaks (Garnsworthy et al., 2012) as investigated in the current study. The current study reanalysed the dataset by Bell et al. (2014b). The hypothesis was that enhanced filtering of eructation spot measurements (i.e. individual or clusters of peaks) within a milking period could improve the reliability and repeatability of measurements used to estimate the daily CH_4 emissions of individual cows.

The objective of the current study was to compare different metrics for quantifying the CH_4 emissions of individual cows during milking using frequent 'spot' measurements and peak analysis methods. Results were compared to chamber CH_4 records for different dairy cows, as chamber measurements are considered to be the gold standard for measuring daily emissions.

2 Materials and methods

2.1 Breath sampling data

Enteric CH_4 emitted from the mouth and nostrils of 36 Holstein Friesian dairy cows was measured during milking at Nottingham University Dairy Centre (Sutton Bonington, Leicestershire, UK). The dataset covered wide ranges of milk yield (14 to 55 kg day^{-1}), lactation number (1 to 5), stage of lactation (15 to 409 days in milk) and live weight (473 to 805 kg) (*Table 1*). Cows were group housed in a freestall barn and milked individually at an automatic (robotic) milking station (Lely Astronaut A3; Lely UK Ltd., St Neots, UK). Gas concentrations (v/v) in air sampled from the milking station feed bin were measured continuously by an infrared gas analyser (Guardian Plus; Edinburgh Instruments Ltd., Livingston, UK) during 2,474 individual cow milkings throughout a sampling period of 28 days. For a full description of the study see Bell et al. (2014b), who estimated cow CH_4 emissions by calculating the area under the eructation peaks that were measured during a whole milking rather than selected peaks within a milking as in the current study. The spot sampling technique is described briefly below.

The CH_4 concentration (v/v) was logged at onesecond intervals on data loggers (Simex SRD-99; Simex Sp. z o.o., Gdańsk, Poland) and visualised using logging software (Loggy Soft; Simex Sp. z o.o.). The CH_4 analyser was calibrated at the start of the study using standard mixtures of CH_4 in nitrogen (0.0, 0.25, 0.50, 0.75 and 1.0 % CH_4, Thames Restek UK Ltd., Saunderton, UK). The CH_4 concentration in the gases emitted during milking was recorded in parts per million (v/v). The CH_4 concentration data measured every second were then extracted from the time-series signal using the peak analysis tools in the MatLab Signal Processing Toolbox (version R2018a; The MathWorks, Inc., Natick, United States. See https://uk.mathworks.com/help/signal/examples/peak-analysis.html for metrics). The peak analysis tools were used to identify clusters of CH_4 eructation peaks during one milking (*Figure 1*) from raw logger data, using the findpeak function. The findpeak function is a tool for extracting local maxima from two-dimensional signals. This MatLab function can be parameterised using constraints such as the number of peaks allowed, peak height, width or prominence, and the distance between peaks. The data by Garnsworthy et al. (2012) comparing chamber CH_4 measurements with spot measurements for the same cows showed that the CH_4 emission rate (g min^{-1}) and total CH_4 production (g day^{-1}) were highly correlated to CH_4 peak height (r = 0.91), CH_4 peak amplitude (r = 0.89), and less so to peak frequency (r = 0.29). Therefore, values for the following metrics were derived:

- maximum peak height (ppm)
- maximum peak amplitude (ppm)
- average CH_4 concentration (ppm)

To identify individual and clusters of peaks in CH_4 emissions from within one milking, the program extracted the data based on the following filtering criteria:

- three or more consecutive peaks (clusters)
- minimum time between peaks of 20 seconds
- minimum peak height and amplitude of 200 ppm

The average rise time for peaks (applied to the average CH_4 concentration measure) and the maximum rise time for maximum peak height and amplitude in seconds were obtained using peak analysis for each milking. The background CH_4 concentration was subtracted from measures of peak height and the average concentration during milking, with the background level assumed to be the minimum value measured. With all three metrics in ppm based on the analyser recording every second, the values were converted to emission rate in grams per minute by multiplying by 60 and assuming a CH_4 density of 0.656×10^{-6} g L^{-1}. This assumes

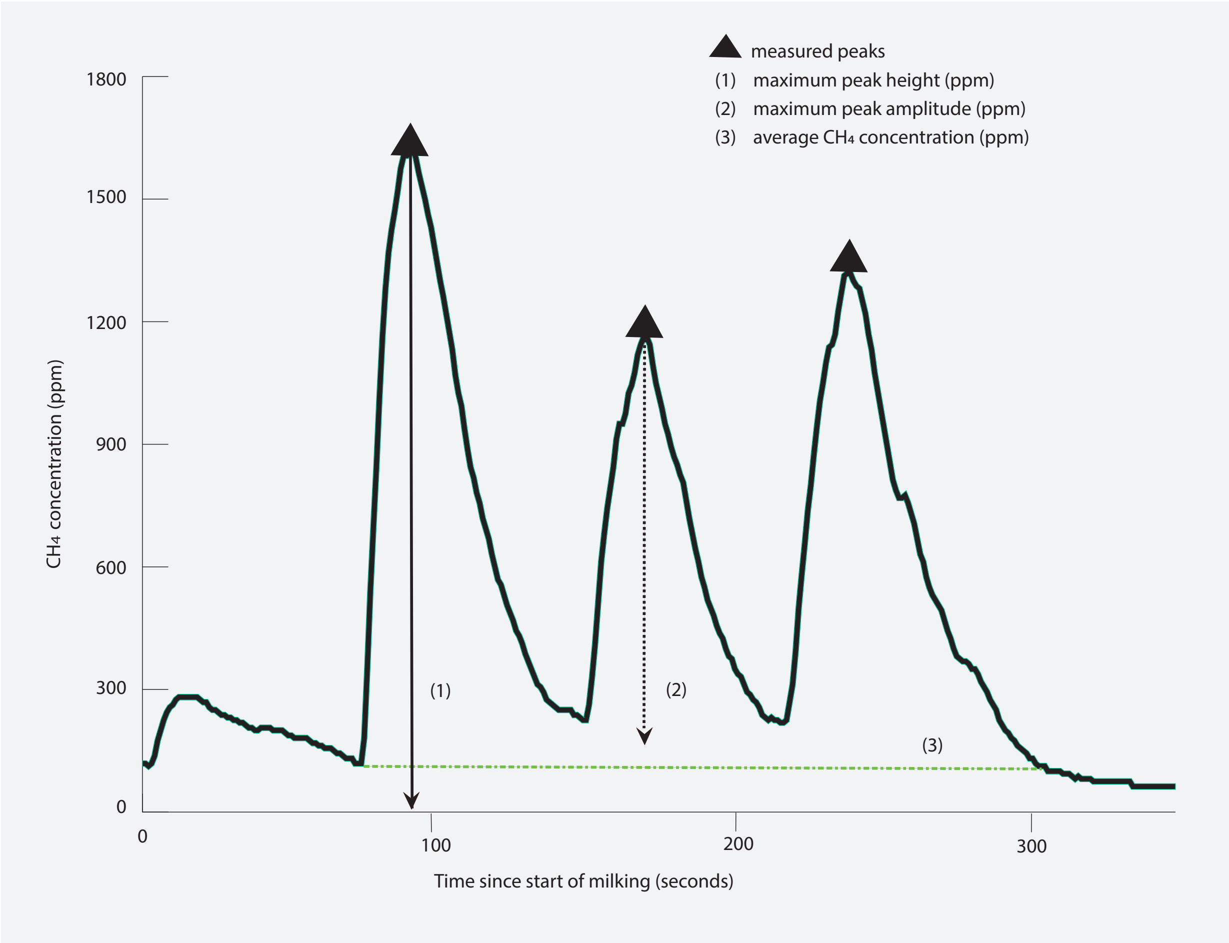

FIGURE 1
The CH_4 concentration profile in eructated gas for cow 2158 during milking showing measured peaks, maximum peak height measurements ((1) solid black line with arrow) and maximum peak amplitude measurements ((2) dotted black line with arrow). Data on the background CH_4 concentrations were extracted within each milking period ((3) dashed green line) to obtain the average concentration during this time and the minimum (i.e. background) concentration.

the analyser is sampling air at a flow rate of 1 L min^{-1}. The exponential response curve for the gas concentration measured was determined in a previous study by Garnsworthy et al. (2012). This response curve was found to relate peak rise time and amplitude to known amounts of released gas for dilution tests. All extracted emission rates (in grams per minute) during milking were scaled to the estimated emissions based on the exponential increase in gas concentration and the extracted rise time for eructation peaks using equation [1], given it takes 60 seconds for the analyser to reach the 'true' peak asymptote and fully process the gas sample:

Estimated CH_4 emission rate (g min^{-1}) = (CH_4 concentration ppm–background CH_4) [1- EXP (-(peak rise time in seconds / 60))]$^{-1}$ x 60 x 0.656 x 10^{-6} [1]

The maximum peak height and maximum peak amplitude metrics used their associated peak rise time, whereas the average concentration metric used the calculated average rise time for peaks sampled during each milking. The estimated emission rate was converted to grams per day by multiplying by 1,440. Emission values were not adjusted for potential dilution of eructated CH_4, as the study aimed to assess the potential of advanced peak analysis filtering methods to replace to need to adjust values due to gas dilution.

Measurements of enteric CH_4 during milking were conducted during two consecutive feeding periods of 14 days, during which cows were fed either a grass or maize silage partial mixed ration (PMR) *ad libitum* plus concentrates at milking. A 14-day crossover design experiment during which cows were each fed a diet containing between 39 and 70 % forage (*Table 1*) was conducted. A concentrate was dispensed into the feed bin throughout the milking period, which helped to keep the cow's head within suitable proximity of the gas sampling tube. Daily concentrate allowance fed during milking was 1.5 kg plus 0.16 kg per litre of milk yield above 23 L d^{-1}. Total daily DMI of concentrate from the PMR and AMS combined was calculated. Milk yield and live weight were recorded automatically at each milking. Feed intake was recorded automatically by electronic feeders (Roughage Intake Control feeders; Fullwood Ltd., Ellesmere, UK).

TABLE 1
Average production values and composition of diets fed to cows (n = 36) with spot samples and cows (n = 105) with chamber methane (CH_4) measurements

Component	Units	Spot		Chamber	
		Mean (sd)	Range	Mean (sd)	Range
Observations		*n=72*		*n=196*	
Forage	%	54 (10)	39–70	50 (13)	25–80
Dry matter intake (DMI)	kg day^{-1}	19.7 (3.2)	12.4–26.1	18.0 (2.8)	11.4–24.5
Forage DMI	kg day^{-1}	10.5 (2.2)	6.6–16.0	8.8 (2.6)	2.9–15.2
Concentrate DMI	kg day^{-1}	9.2 (2.3)	5.1–13.0	9.2 (2.8)	3.6–16.9
Milk yield	kg day^{-1}	33.3 (9.4)	14.4–55.2	25.9 (6.9)	14.1–49.1
Live weight	kg	646 (68)	473–805	572 (60)	385–733
Crude protein	g kg^{-1} DM	170 (1.1)	166–173	188 (21)	127–250
Ether extract (oil)	g kg^{-1} DM	22.4 (2.5)	17.9–27.5	55.2 (8.4)	25.2–63.5
Starch	g kg^{-1} DM	187 (0.9)	158–206	129 (30)	72–216
Sugar	g kg^{-1} DM	40.9 (3.2)	34.9–47.7	56.5 (16.7)	39–137
Neutral detergent fibre	g kg^{-1} DM	313 (12)	292–337	390 (50)	264–554
Ash	g kg^{-1} DM	14.5 (0.8)	13.1–16.3	84.4 (8.6)	57–111
Metabolisable energy	MJ kg^{-1} DM	12.1 (0.03)	12.0–12.2	12.1 (0.7)	10.3–14.4
Milkings per day		3.3 (0.8)	1.8–6.0	2	–
Milking duration	s	395 (100)	242–778	–	–
Average peak rise time	s	10.4 (3.0)	6.5–15.6		
Maximum peak rise time	s	15.5 (2.3)	10.8–20.3		
Minimum CH_4 concentration	ppm	185 (33)	113–251		
Maximum CH_4 height	ppm	1,253 (208)	744–1,736		
Maximum CH_4 amplitude	ppm	1,042 (208)	535–1,497		
Average CH_4 concentration	ppm	568 (91)	344–814		
Daily CH_4 production					
Peak CH_4 height	g day^{-1}	288 (59)	152–431		
Peak CH_4 amplitude	g day^{-1}	282 (63)	135–431		
Average CH_4 concentration	g day^{-1}	177 (72)	66–344	387 (64)	202–541
CH_4 yield					
Peak CH_4 height	g kg^{-1} DM	14.8 (3.1)	9.1–23.4	–	–
Peak CH_4 amplitude	g kg^{-1} DM	14.5 (3.2)	8.8–23.3	–	–
Average CH_4 concentration	g kg^{-1} DM	9.0 (3.2)	3.6–17.9	21.8 (3.4)	13.8–33.5

2.2 Chamber data

The chamber dataset consisted of a total of 196 measurements from 105 lactating dairy cows of different breeds (Holstein Friesian, Norwegian Red and Jersey Holstein) taken during energy metabolism studies conducted at the Agri-Food and Biosciences Institute (Yan et al., 2010) and Ellinbank (Williams et al., 2013) research centres. The dataset covered wide ranges of milk yield (14.1 to 49.1 kg day^{-1}), lactation number (1 to 9), stage of lactation (early to late) and live weight (385 to 733 kg) (*Table 1*). All cows were offered a diet of between 25 and 80 % forage (either fresh cut grass, grass silage or alfalfa hay) *ad libitum*. The concentrate portion of the diet was offered either as part of a complete diet mixed with the forage or as a separate feed, and when the concentrates were fed they consisted of cereal grains (barley, wheat or maize), by-products (maize gluten meal, molassed or unmolassed sugar-beet pulp, citrus pulp or molasses) or protein supplements (fish meal, soybean meal or rapeseed meal). Prior to commencing CH_4 measurements, all cows were offered experimental diets for at least two weeks. In the metabolism unit, each cow spent at least four days in metabolism stalls followed by three days in a chamber (indirect open-circuit calorimeter) for CH_4 measurements, with the CH_4 measurements from the final 48 h period being used for analysis.

2.3 Statistical analysis

Data from spot sampling and chamber measurements were analysed using a linear mixed model in Genstat Version 19.1 (Lawes Agricultural Trust, 2018). Average emissions per day and average CH_4 yield (grams per kilogram DMI) were calculated for each cow during each feeding period (two-weeks for spot measurement values and two days for chamber values) and used in the analysis. Equation [2] was used to calculate variance components for feed intake (DMI, forage DMI and concentrate DMI), milk production, live weight and various metrics for CH_4 per individual cow:

$$y_{ijkl} = \mu + P_i + D_j + L_k + L_k.C_l + E_{ijkl} \qquad [2]$$

where y_{ijk} is the dependent variable; μ = overall mean; P_i = fixed effect of measurement period; D_j = fixed effect of diet; L_k = fixed effect of lactation number (k = 1, 2 or 3 and more); $L_k.C_l$ = random effect of individual cow; E_{ijkl} = random error term.

Repeatability of animal production variables and gas emission measures were assessed by σ^2 animal (σ^2 animal + σ^2 residual)$^{-1}$, where σ^2 is the variance. The between-cow and residual coefficient of variation (CV) were calculated from variance components as root mean square error divided by the mean. The Pearson correlation coefficient was used to assess the association between CH_4 emission metrics and total DMI, forage DMI, concentrate DMI, milk yield and live weight across all individual cow records. The results for the three metrics of CH_4 from peak analysis (peak height, peak amplitude and average concentration) were compared with each other after converting to daily emissions and CH_4 yield, which allowed comparison to CH_4 emissions from chamber measurements. Significance was attributed at $P<0.05$. Equation 1 was validated on peak analysis data from spot measurements and chamber measurements from the same ten cows from the study by Garnsworthy et al. (2012). The maximum peak amplitude (mean ± sd of 1054 ± 313 ppm and ranging from 625 to 1592 ppm) and peak rise time (mean ± sd of 10.9 ± 0.4 seconds and ranging from 10.2 to 11.5 seconds) were derived within milking periods and the total daily CH_4 production (mean ± sd of 370 ± 28 g day^{-1} and ranging from 332 to 407 g/day) whilst in the chamber. For this data, the Pearson correlation coefficient (*r*), Lin's bias correction factor (C_b) and concordance correlation coefficient (*CCC*) were used to test the association between total CH_4 production estimated from spot measurements using Equation 1 and chamber measurements from the same cows. Coefficient *r* was multiplied by Lin's bias correction factor (C_b), which measures how far the best-fit line deviates from the 45° line through the origin, in order to derive the *CCC* (Lin, 1989).

3 Results and discussion

3.1 Methane and its association with production traits

After filtering spot measurements for peaks in emissions during milking, ranges of 66 to 431 g CH_4 day^{-1} and 3.6 to 23.4 g CH_4 kg^{-1} DM were observed across CH_4 metrics from peak analysis (*Table 1*). The average CH_4 concentration values (177 g day^{-1} and 9 g kg^{-1} DM) were lower than those for peak height (288 g day^{-1} and 14.8 g kg^{-1} DM) and peak amplitude (282 g day^{-1} and 14.8 g kg^{-1} DM) metrics, which were all lower than the average CH_4 emissions measured for dairy cows in the chamber data (387 g day^{-1} and 21.8 g kg^{-1} DM). After deriving CH_4 emission metrics it is noticeable that the peak height and peak amplitude metrics produce similar results. Both metrics have been found to be associated with total CH_4 production (Garnsworthy et al., 2012). Furthermore, using input data of maximum peak amplitude and peak rise time from the study by Garnsworthy et al. (2012) in Equation 1 of the current study, found that estimates of total CH_4 production (mean ± sd of 388 ± 31 g day^{-1} and ranging from 334 to 430 g day^{-1}) are correlated to chamber CH_4 values (r = 0.75 and *CCC* = 0.62; mean ± sd of 370 ± 28 g day^{-1} and ranging from 332 to 407 g day^{-1}) (*Figure 2*).

Although peak analysis can help to identify when the animal's head is in close proximity to the gas sampling tube (i.e. from maximum peak height and peak amplitude during one milking), the difference in average daily CH_4 emissions (*Figure 3*) and CH_4 yield (*Figure 4*) between spot measurements and chamber measurements would suggest that some dilution or loss of spot measurement CH_4 emissions occurred between the emissions being expelled by the cow and sampled by the gas analyser. Metrics for spot measurement CH_4 were not adjusted for any dilution effect. Further refinement of the breath sampling approach to capture more of the eructation produced by the animal may improve estimates and is worth comparing to the current proposed approach.

This study found a positive relationship between total DMI (*Figure 3*), forage DMI and CH_4 emissions per day (*Table 2*), and a negative relationship between DMI and CH_4 yield (*Figure 4* and *Table 2*) estimated from peak height and peak amplitude. The magnitude of the correlation between DMI and CH_4 yield estimated from peak height (r = -0.43) and peak amplitude (r = -0.38) were noticeably similar to the correlation between DMI and CH_4 yield from chamber measurements (r = -0.40). As observed in chamber measurements, CH_4 yield declined with increasing concentrate DMI but not forage DMI for metrics of peak height and peak amplitude.

When a highly energy-dense diet is formulated to meet the nutrient requirements of a high milk yielding animal (with spot sampled cows averaging 33 kg milk day^{-1} compared to 26 kg milk day^{-1} for chamber cows), often through feeding a higher proportion of concentrates in the diet, the CH_4 yield can be 19 g kg^{-1} DMI or less (Mills et al., 2003 and *Figure 3* for cows with high DMI). The CH_4 yield would be expected to be higher (21 g kg^{-1} DM or more, see Moate et al, 2011) for predominantly forage-based diets. Bell and Eckard (2012) found that in lactating dairy cows fed a diet with a high or low proportion of forage content, the relationship between CH_4 production and DMI appears to be linear up to an average intake of 15 kg DMI day^{-1}. Above this level of intake (as the majority of cows in this study), the CH_4 yield declines, with the lower CH_4 yield for spot measurements potentially being influenced by the allocation of concentrates during milking (*Figure 3*).

The improved relationship between DMI, forage DMI and CH_4 emissions found in the current research compared to the results published in a previous study by Bell et al. (2014b) (r = 0.19 to 0.29) can be attributed to the extraction and

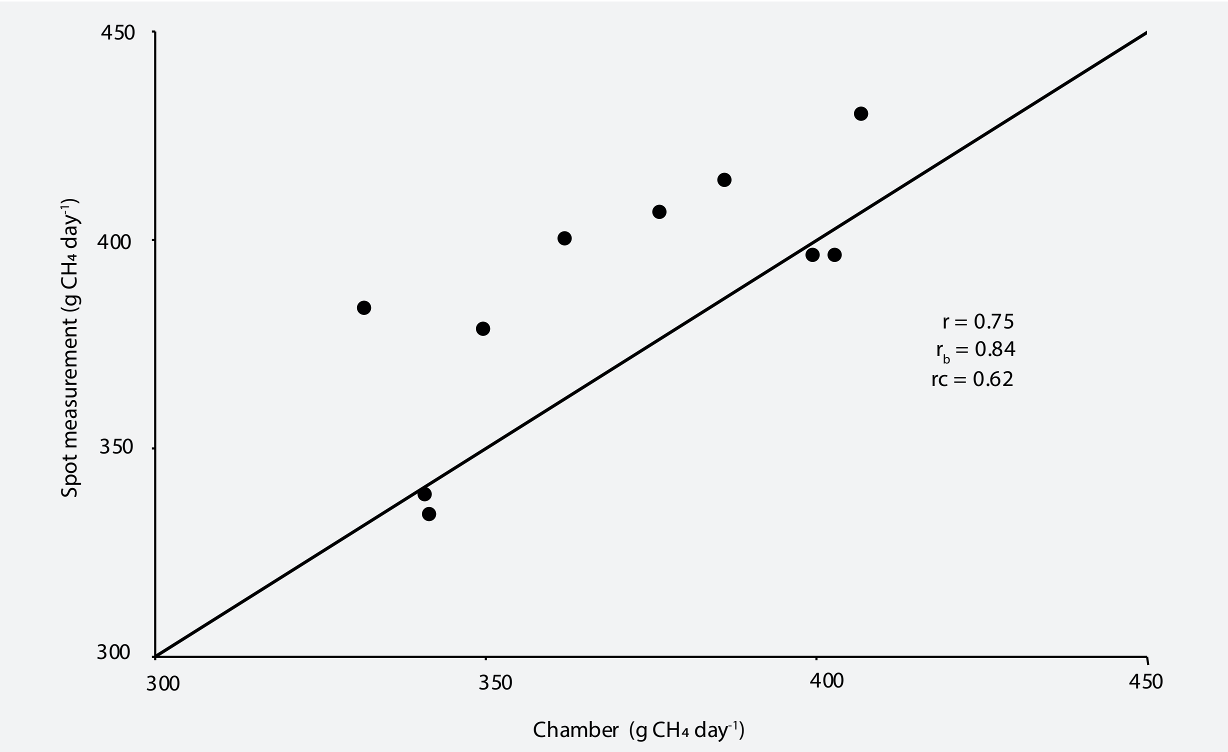

FIGURE 2
Relationship between spot measurements of CH_4 emissions estimated using Equation 1 and chamber measurements for the same cows

improved identification of clusters and of individual eructation peaks in CH_4 emissions during each milking, rather than extracting measurements from across the whole milking period as before. The benefit of extracting the amplitude of eructation peaks is the potential to easily remove background emissions and any buildup in gas that may occur in the feed bin during milking. Milk yield was negatively associated with CH_4 yield from both spot measurements and chamber measurements (*Table 3*). The allocation of concentrate was different between the spot sampled cows and cows in chambers. The high correlation ($r=0.833$ for spot sampled cows compared to $r=0.609$ for chamber cows) between milk yield and concentrate DMI for spot sampled cows may explain the lower CH_4 yield in these cows compared to the CH_4 yield of cows in chambers. Increased intake of more digestible feeds such as concentrate results in a reduction in CH_4 yield (Yan et al., 2010). There was no association between liveweight and CH_4 yield from spot or chamber measurements (*Table 2* and *Table 3*), but daily chamber CH_4 emissions were positively associated with live weight.

3.2 Repeatability and variability of methane measures

The CH_4 metrics from peak analysis were highly repeatable for metrics of peak height and peak amplitude (ranging from 0.76 to 0.81), and comparable to the high repeatability of production traits for the same cows (ranging from 0.63 to 0.99) (*Table 4*). These instances of high repeatability for CH_4 emissions from spot measurements have been observed in several other studies (0.72 to 0.87 by Huhtanen et al., 2015) and confirm findings from our previous work (0.74 to 0.75 by Bell et al., 2014b). There was little difference in the residual CV observed for the CH_4 metrics derived from peak height and peak amplitude (ranging from 8 to 9 % for daily CH_4 emissions and CH_4 yield) compared to chamber CH_4 measurements (11 %), and in the feed intake traits for spot sampled cows and cows in chambers (CV ranged from 7 % to 15 %). These findings are consistent with the results of Huhtanen et al. (2013), and the modified approach used in the current research to identify eructation peaks within each milking – rather than throughout the whole milking – has improved the reliability of the technique compared to our previous research (Bell et al., 2014b).

The between-cow CV for both daily emissions and CH_4 yield derived from peak analysis metrics in the current study were within the range of 3 to 34 % found in studies using respiration chambers to measure emissions in research herds (Grainger et al., 2007; Ellis et al., 2007; Yan et al., 2010). The between-cow CVs ranged from 16 % to 18 % across peak analysis metrics for CH_4 (*Table 4*) and were higher than the values observed for chamber between-cow CVs of 8 % for daily emissions and 2 % for CH_4 yield. The approach of extracting eructation peak height and peak amplitude to quantify daily CH_4 emissions and CH_4 yield resulted in similar variation between-cows (CV ranging from 16 to 18 %), residual variation (CV ranging from 8 to 9 %) and repeatability (ranging from 0.76 to 0.81) for spot measurements compared to variation between-cow (CV = 12 % for spot but CV = 8 % for chamber

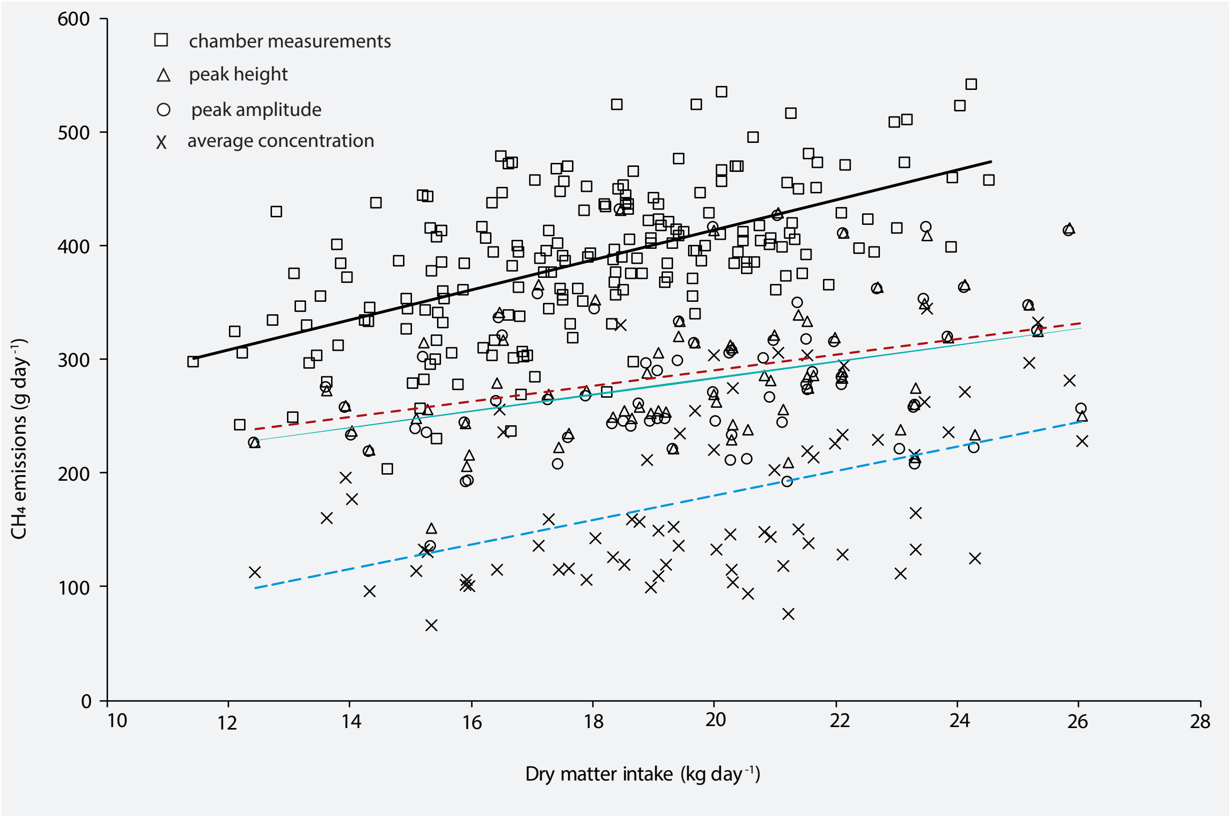

FIGURE 3
Observed dry matter intake and CH_4 emissions per day for chamber measurements and spot sample CH_4 metrics of peak height, peak amplitude and average concentration. The line of best-fit is shown for chamber measurements (black solid line, $y = 149 + 13.2x$, $r = 0.57$, $P < 0.001$) and CH_4 metrics of peak height (red dashed line, $y = 153 + 6.8x$, $r = 0.37$, $P < 0.01$), peak amplitude (green solid line, $y = 138 + 7.3x$, $r = 0.37$, $P < 0.01$) and average concentration (blue long dashed line, $y = -33.8 + 10.7x$, $r = 0.47$, $P < 0.001$).

TABLE 2
Pearson correlation coefficients and significance of the relationship between feed dry matter intake (DMI), milk production, live weight and daily methane emissions as estimated from peak related parameters (below diagonal) and methane yield as estimated from peak related parameters (above diagonal) for cows measured using spot sampling.

Variable[1]	Units	DMI	Forage DMI	Concentrate DMI	Milk yield	Live weight	Peak height	Peak amplitude	Average concentration
		kg day^{-1}				kg	g CH_4 kg^{-1} DMI		
DMI	kg day^{-1}	1					-0.432 (<0.001)	-0.380 (<0.001)	0.056 (0.640)
Forage DMI	kg day^{-1}	0.698 (<0.001)	1			-0.015 (0.901)	0.043 (0.718)	0.431 (<0.001)	
Concentrate DMI	kg day^{-1}	0.726 (<0.001)	0.015 (0.903)	1		-0.588 (<0.001)	-0.572 (<0.001)	-0.335 (<0.01)	
Milk yield	kg day^{-1}	0.618 (<0.001)	0.031 (0.799)	0.833 (<0.001)	1	-0.524 (<0.001)	-0.515 (<0.001)	0.300 (<0.05)	
Live weight	kg	-0.03 (0.800)	0.355 (<0.01)	-0.383 (<0.001)	-0.186 (0.118)	1	0.011 (0.929)	0.0002 (0.999)	0.007 (0.953)
Peak height	g CH_4 day^{-1}	0.366 (<0.01)	0.550 (<0.001)	-0.017 (0.891)	-0.056 (0.638)	-0.028 (0.818)	1		
Peak amplitude	g CH_4 day^{-1}	0.367 (<0.01)	0.568 (<0.001)	-0.033 (0.785)	-0.074 (0.539)	-0.037 (0.758)	0.993 (<0.001)	1	
Average concentration	g CH_4 day^{-1}	0.470 (<0.001)	0.691 (<0.001)	-0.007 (0.950)	-0.031 (0.798)	-0.011 (0.927)	0.792 (<0.001)	0.814 (<0.001)	1

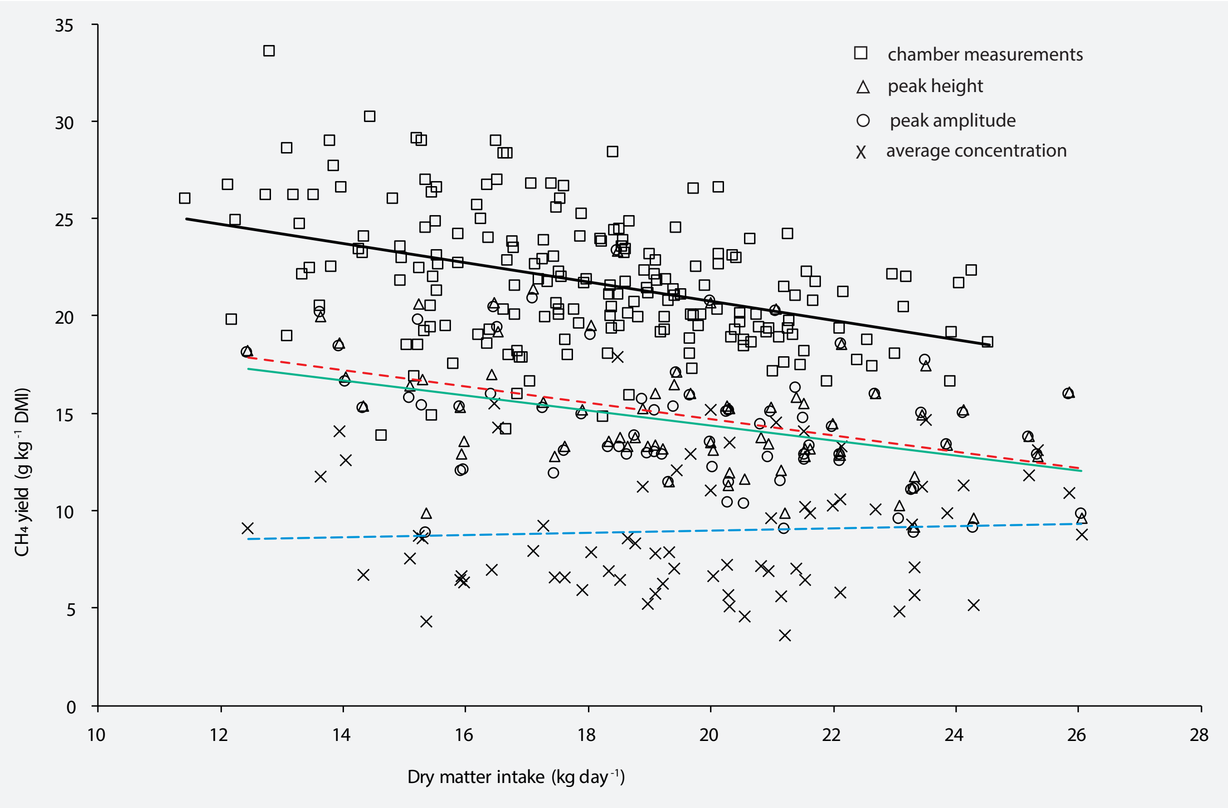

FIGURE 4

Observed dry matter intake and CH_4 yield for chamber measurements and CH_4 metrics of peak height, peak amplitude and average concentration. The line of best-fit is shown for chamber measurements (black solid line, $y = 30.5 - 0.49x$; $r = -0.40$, $P < 0.001$) and CH_4 metrics of peak height (red dashed line, $y = 23.1 - 0.42x$; $r = -0.43$, $P < 0.001$), peak amplitude (green solid line, $y = 22.1 - 0.39x$; $r = -0.38$, $P < 0.001$) and average concentration (blue long dashed line, $y = 7.8 + 0.06x$, $r = 0.06$, $P=0.640$).

measurements), residual variation (CV = 8 % for spot and CV = 9 % for chamber measurements) and repeatability (0.70 for spot but different at 0.40 for chamber measurements) for DMI, which was also found by Huhtanen et al. (2013) using a spot sampling approach.

The frequent 'spot' sampling of enteric CH_4 emissions from cows has come about due to the need to measure CH_4 emissions from large numbers of commercial animals for farm benchmarking, improving national greenhouse gas inventories and for selecting low CH_4 producing animals. Methods that are more mobile, non-invasive to the animal and can fit into the animal's normal environment are of great interest, such as the technique used in this study. Furthermore, identification of eructation peaks and clusters of peaks can provide a repeatable and reliable metric that is consistent with cow chamber records, which is the gold standard measure for measuring CH_4 emissions. Cows in the current study were milked on average three times per day and had spot measurements of CH_4 recorded for two weeks during two feeding periods to obtain individual cow enteric CH_4 emission rates. Duration of spot sampling depends on the frequency and number of spot measurements being obtained (Cottle et al., 2015). This approach of taking spot measurements over at least a week is longer than the three days animals spend in a chamber to measure CH_4 emissions. However, this approach can be implemented on commercial farms unlike the use of chambers.

In conclusion, this study showed that quantifying enteric CH_4 emissions using eructation peaks (maximum peak height or maximum peak amplitude) detected within a milking can provide a highly repeatable metric for quantifying daily CH_4 emissions and daily CH_4 yields. The association between DMI and metrics for estimating methane emissions derived from peak height and peak amplitude were similar for cows studied using spot sampling and in a respiration chamber. The extraction of eructation CH_4 peaks can provide a repeatable and reliable method for quantifying CH_4 emissions and assessing variation among cows. We recommend estimating daily CH_4 emissions by measuring the maximum peak amplitude of an eructation during one milking.

TABLE 3
Pearson correlation coefficients and significance of the relationship between feed dry matter intake (DMI), milk production, live weight and daily methane emissions and methane yield for cows measured in chambers.

Variable[1]	Units	DMI	Forage DMI	Concentrate DMI	Milk yield	Live weight	CH_4 yield
		kg day^{-1}				kg	g CH_4 kg^{-1} DMI
DMI	kg day^{-1}	1					-0.402 (<0.001)
Forage DMI	kg day^{-1}	0.445 (<0.001)	1				-0.018 (0.808)
Concentrate DMI	kg day^{-1}	0.578 (<0.001)	-0.474 (<0.001)	1			-0.379 (<0.001)
Milk yield	kg day^{-1}	0.583 (<0.001)	-0.039 (0.588)	0.609 (<0.001)	1		-0.439 (<0.001)
Live weight	kg	0.454 (<0.001)	0.262 (<0.001)	0.207 (<0.01)	0.142 (0.05)	1	0.088 (0.219)
CH_4 emissions	g CH_4 day^{-1}	0.574 (<0.001)	0.425 (<0.001)	0.177 (<0.05)	0.146 (<0.05)	0.509 (<0.001)	0.509 (<0.001)

TABLE 4
Variability and repeatability (standard error in parentheses) of dry matter intake (DMI), milk production, live weight and methane (CH_4) emissions, and of yields as derived from peak analysis for spot sampled cows and by traditional methods for cows in chambers.

Variable	Units	Spot			Chamber		
		Between-cow CV (%)	Residual CV (%)	Repeatability	Between-cow CV (%)	Residual CV (%)	Repeatability
DMI	kg day^{-1}	12.0	7.8	0.70 (0.21)	7.5	9.2	0.40 (0.16)
Forage DMI	kg day^{-1}	13.7	10.5	0.63 (0.21)	8.2	15.4	0.22 (0.14)
Concentrate DMI	kg day^{-1}	23.1	7.3	0.91 (0.24)	4.0	10.0	0.14 (0.18)
Milk yield	kg day^{-1}	23.7	10.0	0.85 (0.23)	16.4	10.4	0.71 (0.15)
Live weight	kg day^{-1}	9.8	1.1	0.99 (0.24)	9.3	3.6	0.87 (0.15)
Daily CH_4 production[1]							
Peak CH_4 height	g day^{-1}	16.1	9.1	0.76 (0.22)	–	–	–
Peak CH_4 amplitude	g day^{-1}	17.6	8.6	0.81 (0.22)	–	–	–
Average CH_4 concentration	g day^{-1}	16.4	18.0	0.45 (0.19)	7.8	11.7	0.31 (0.15)
CH_4 yield[1]							
Peak CH_4 height	g kg^{-1} DMI	16.9	8.2	0.81 (0.22)	–	–	–
Peak CH_4 amplitude	g kg^{-1} DMI	17.7	8.8	0.80 (0.22)	–	–	–
Average CH_4 concentration	g kg^{-1} DMI	16.5	14.6	0.56 (0.20)	1.8	11.4	0.03 (0.12)

[1] Spot sampling metrics for daily emissions and CH_4 yield were estimated from emission rate.

Acknowledgements

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RESEARCH ARTICLE

Pork production in Thuringia – management effects on ammonia and greenhouse gas emissions. 2. Reduction potentials and projections

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HIGHLIGHTS

- **Analysis of the mass flows within the entire pork production chain reveals several minor emission reduction potentials for ammonia**
- **Diets with reduced protein content improve health and reduce ammonia emissions significantly without additional costs**

KEYWORDS **pork production, ammonia, greenhouse gases, area under cultivation, fertilising, energy, water**

Abstract

Measures to reduce emissions from pork production have been evaluated for fattening pigs in Thuringia, where fattening dominates emissions. Next, an expert team provided data sets for emission scenarios for the entire pork production chain (including breeding, piglet production, fattening as well as feed production, fertiliser use and production, provision of water and energy) in 2020 and 2025.

Moderate increases in performance and reduction of animal losses had almost no effect. Substantial emission reductions were found for feeds with reduced protein contents, filtering exhaust air from buildings through scrubbers and reduced emission slurry application procedures. Manure systems using solid farmyard manure emit greater quantities than slurry based systems.

A combination of the measures anticipated for 2025 in a comprehensive (fictive) reference enterprise could result in a NH_3 emission reduction by about one fifth as compared to 2015. A minor reduction of greenhouse gas emissions is a welcome side effect.

1 Introduction

Compared with other German regions, pork production in Thuringia (Thüringen) is characterised by a low livestock density (expressed as pigs per unit of productive land). Major changes occurred due to the restructuring of agricultural production after the German unification. Currently about 750.000 animal places with about 320.000 fattening places can be regarded as standard (StatBA, 2017).

During the past two decades, numerous new livestock buildings have been erected that comply with the regulations on best available techniques, including measures to reduce environmental pollution. Thuringian production units are larger than the German mean (StatBA, 2017) which contributes to the competitiveness of its respective enterprises. Hence, pork production will have a promising future within Thuringian agricultural production.

However, pork production will have to adapt to restrictions imposed by German and European legislation on atmospheric emissions and ground water pollution, such as EU (2016) or the Thuringian enactment on air scrubbers (TMfUEN, 2016). At present, German administrations are reluctant to enforce these regulations. For agriculture, the overall nitrogen

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problem (eutrophication and acidification of ecosystems, nitrate pollution of drinking water sources) will remain of immense importance. If a reduction of livestock numbers is to be avoided, these ambitious emission reduction goals can only be achieved by introduction of greatly advanced techniques not only in livestock husbandry itself, but also in feed production and the provision of water and energy.

Pork production is a complicated multi-stage process. Earlier investigations showed that the entire production chain has to be analysed in order to identify and assess reduction potentials (e.g. Dämmgen et al., 2016). The preceding paper (Dämmgen et al., 2018a) elucidated that in pork production ammonia (NH_3) from livestock buildings and from feed production has to be addressed with priority. In comparison, emission reductions of greenhouse gases (GHG) from pork production are minor. However, their reduction is a welcome side effect.

This paper reports a detailed systematic analysis of those factors that are related to herd management, with a clear emphasis on fattening. Productive lifetime and fertility of sows were the subject of a separate paper (Dämmgen et al., 2018b). Estimates of future emissions related to pork were estimated using information provided by a Thuringian expert team.[6]

2 Methods

Investigations make use of a fictive 'reference enterprise' which comprises the fattening of pigs, raising of piglets and weaners, basic production (boars) and pure breeding (altogether named 'the herd') as well as production of feed and fertilisers and the provision of water and energy.

2.1 The herd

1,000 pigs (30 to 122 kg pig^{-1}) are fattened at a time (all in all out). Piglet production supplies the right number of piglets at the right time with the necessary number of sows (as a function of the number of piglets weaned per sow). Basic production and pure breeding are taken into account to provide the sows and breeding boars.

As a whole, the example of a comprehensive pork production enterprise reflects the mean Thuringian situation. For details see section 4.2 and Dämmgen et al. (2018b).

2.2 Emission modelling

The quantification of emissions relies on mass flow modelling. Internationally accepted methods (EMEP, 2016; IPCC, 2006) are used to generate comparable results. In addition, national approaches deal with the determination of livestock excretion rates as a function of livestock performance and feed properties. For German pork production these can be found in Haenel et al. (2011) and Dämmgen et al. (2011, 2012, 2017). The work at hand makes use of many data describing non-agricultural processes. Data and methods were described in Dämmgen et al. (2016). The Thuringian data set used was described in detail in the first paper of this series (Dämmgen et al., 2018a).

3 Identifying and assessing reduction potentials – a systematic analysis

The rearing of fattening pigs (fattening hybrids) dominates both NH_3 and GHG emissions in Thuringia (Dämmgen et al., 2018b). Hence, the following detailed examination of reduction potentials is restricted to fattening, including the related direct emissions from feed and fertiliser production as well as indirect emissions resulting from the deposition of reactive N species emitted during this part of the entire production chain. It should be kept in mind that any reduction in NH_3 emissions results in reduced requirements for N fertilisers, and thus at the same time in less emissions from fertiliser production and application.

Reduction potentials are discussed for each single aspect of the production process. They are then compared with the respective projections made by the Thuringian expert team for 2020 and 2025. Drawings contain the absolute emissions for the entire herd of fattening pigs (fp-herd) and the emissions per unit of carcass produced.

3.1 Assumptions for a baseline

Assumptions are similar to the state of pork production in Thuringia in 2015 using statistically available data for animal performance and losses, as well as information provided by the expert panel. However, figures are rounded, and numbers of options are reduced (e.g. for feed, housing, spreading and incorporation).

Animal performance:
- daily weight gain 845 g pig^{-1} d^{-1}, start weight 30 kg pig^{-1}, final weight 122 kg pig^{-1}, carcass dressing percentage 79 %

Animal losses:
- 4 % of fattening pigs housed initially

Feed:
- standard feed only

Housing:
- fully slatted floor only, no exhaust air scrubbers

Storage:
- conventional round tank without cover or natural crust, no fermentation for biogas

Slurry spreading and incorporation:
- trailing hose only; 50 % to bare soil, incorporation within 4 h, rest to short vegetation

N lost to surface and ground waters:
- 5 % of the amount actually available

3.2 Structure of figures

Figures 1 to 10 show the effect of systematic changes of input parameters, such as weight gain, on the left hand side, and the emissions resulting from the mix of parameters for 2015, 2020 and 2025 on the right hand side. The situation for 2015 is *not* the baseline.

[6] Thuringian State Institute for Agriculture (TLL), Germany: Thomas Bauer, Simone Müller, Jürgen Müller, Gerd Reinhold, Hubert Schröter, Wilfried Zorn. Thuringian Ministry for Infrastructure and Agriculture: Michael Mußlick. Qnetics GmbH: Brigitte Neues

3.3 Effects of animal performance

Usually, enhanced performance and reduced final weights help reduce product related emissions. Cumulative energy requirements and thus feed intake rates increase with decreasing daily weight gains, as the requirements for maintenance (energy, nutrients) increase. This affects emissions, as can be seen from *Figures 1* and *2* (left columns: weight gains in g pig^{-1} d^{-1}, right columns projections for 2015, 2020 and 2025 as in Table 1). The Thuringian expert team expect a very limited increase of daily weight gains in the coming decade. The present final weights remain unchanged.

TABLE 1

Animal performance as proposed by the expert team

performance parameter	unit	year		
		2015	2020	2025
daily weight gain	g pig^{-1} d^{-1}	845	845	850
final live weight	kg pig^{-1}	122	122	122

Table 1 summarizes the assumptions with respect to the development of animal performance.

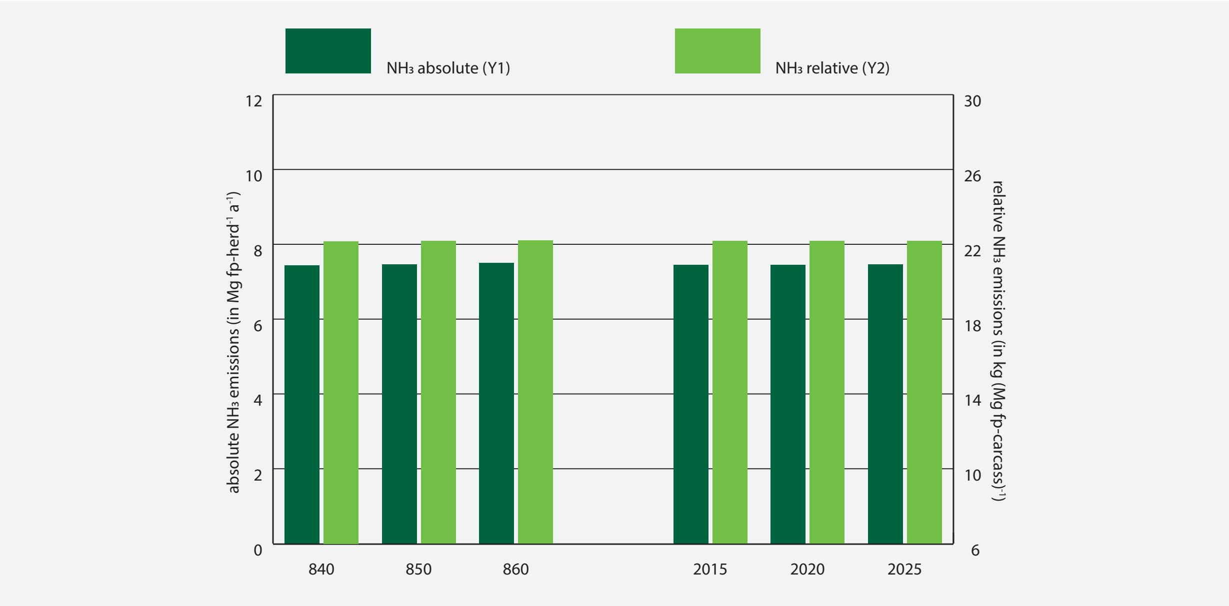


FIGURE 1

Impact of varying daily weight gain on NH_3 emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy)

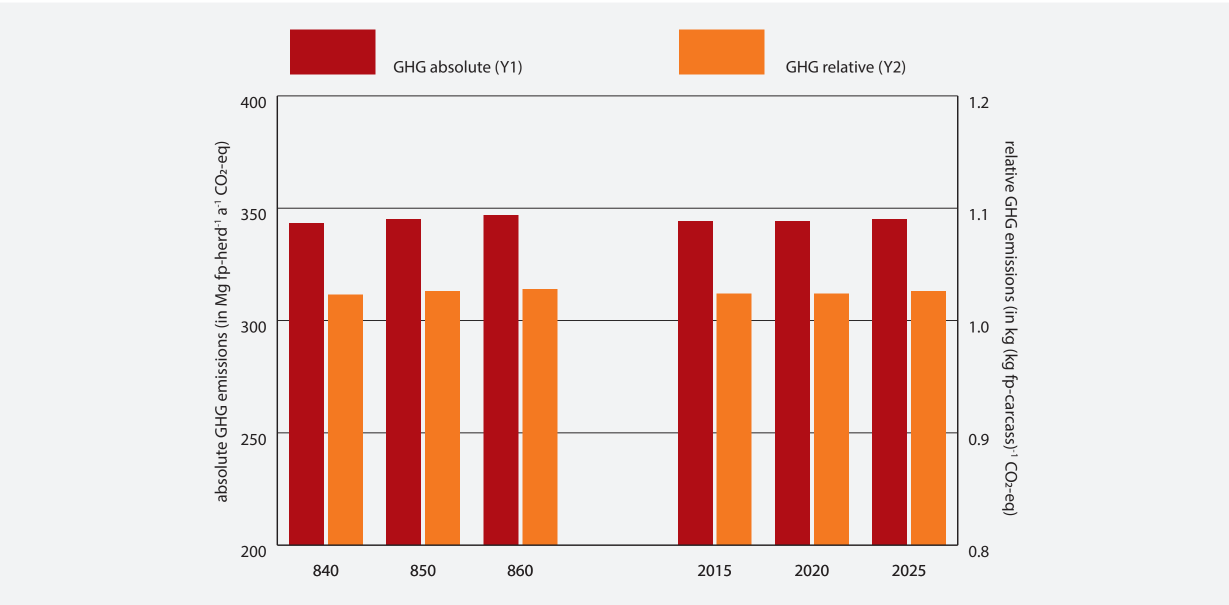


FIGURE 2

Impact of varying daily weight gain on GHG emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy)

The minor changes in animal weight gains have no visible effect on emissions of NH_3 or GHG. Higher daily weight gains result in reduced cumulative energy requirements for maintenance, hence less feed and less excretions. They also result in increased number of animal rounds and thus increased carcass weights per place and year.

Absolute emissions increase slightly whereas relative emissions decrease. Overall emission reductions due to the reduced emission per animal produced are partly compensated by the effect of increased number of animal rounds per year.

Minor changes in daily weight gain can be ruled out as effective measures in emission reduction.

3.4 Effects of animal health

Improved animal health and welfare result in decreased losses of animals whose carcasses cannot be marketed. Our calculations differentiate between those pigs that can be sold at the end of their lives, and those that go to the knacker's yard. For the latter we assume that they have to be fed until half way through their intended lifespan, as we presuppose stochastic deaths over the production period.

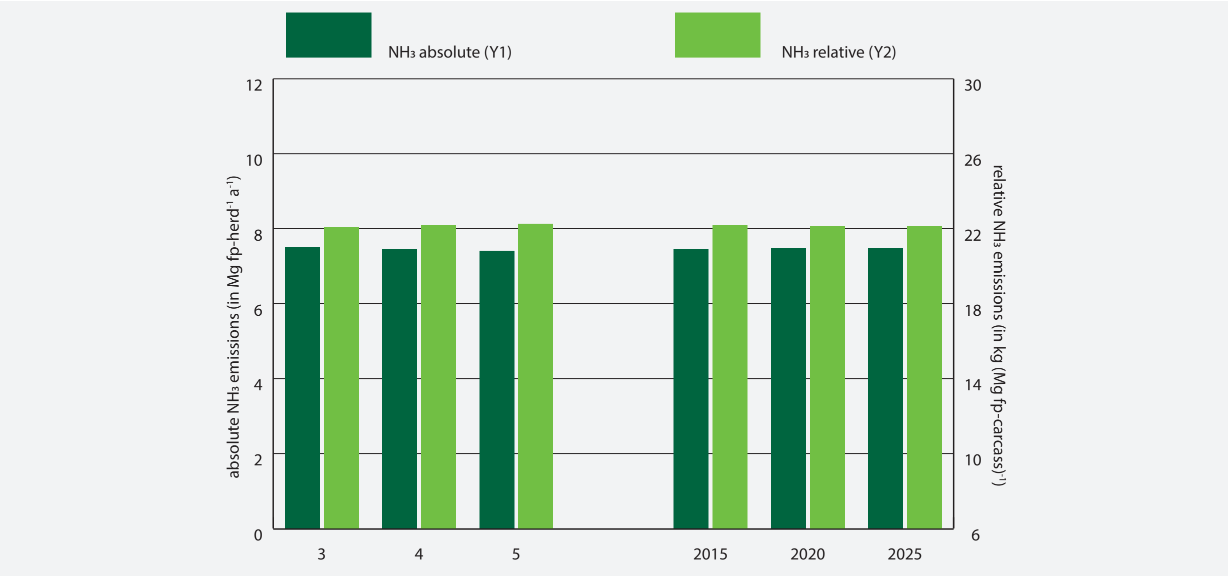


FIGURE 3
Impact of animal losses on NH_3 emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy). Left columns: animal losses in %, right columns projections for 2015, 2020 and 2025 as in Table 2

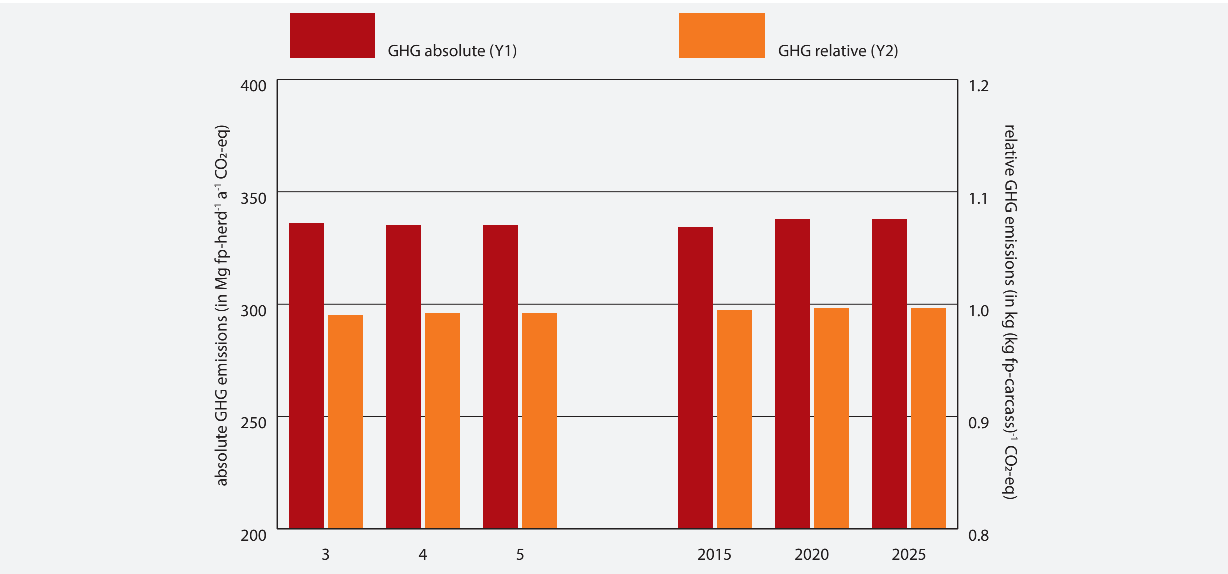


FIGURE 4
Impact of animal losses on GHG emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy). Left columns: animal losses in %, right columns projections for 2015, 2020 and 2025 as in Table 2

TABLE 2
Animal losses proposed by the expert team

loss parameter	unit	year		
		2015	2020	2025
losses of fatteners	% *	4.0	3.5	3.5
* of pigs housed initially				

Figures 3 and *4* indicate that absolute emissions decrease with increasing losses; less animals have to be fed to the end of their lives. However, relative emissions increase with increasing losses, again due to the decreasing number of useful carcasses.

Small improvements of animal welfare and health have no noticeable effect on emissions.

3.5 Effects of feed composition

At present standard feed and N P reduced feeds are taken into consideration. The use of a special feed improving animal welfare ('Gesundfutter') with reduced protein contents and increased amounts of fibre has been discussed. However, no projections could be made with respect to its use.

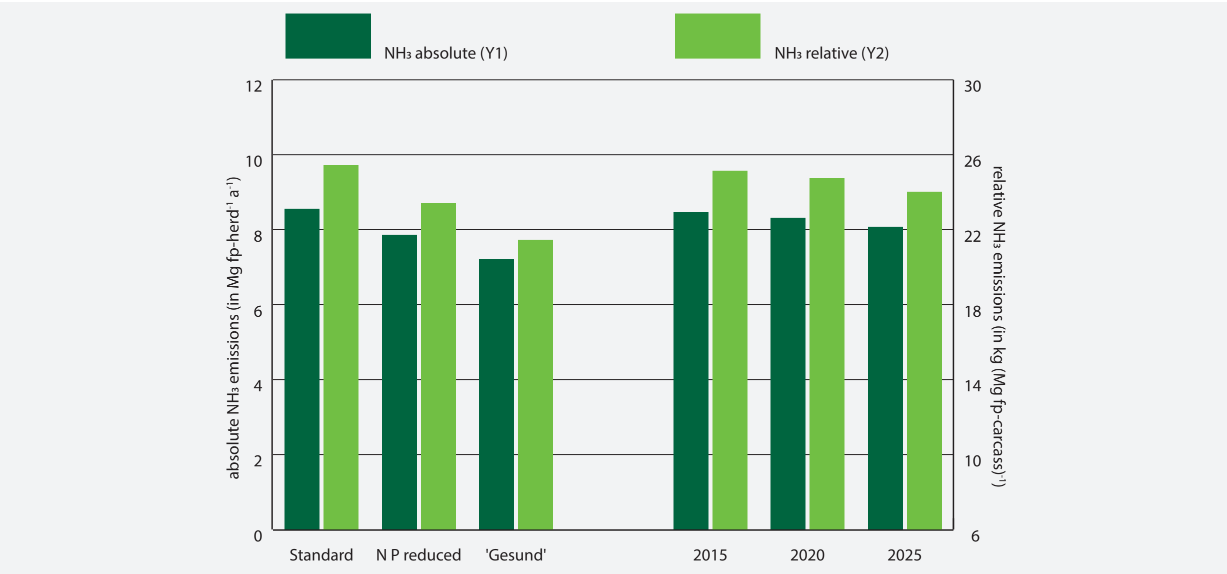


FIGURE 5
Impact of varying feed on NH_3 emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy). Left: 100 % of respective feed, right: proportions of standard and N P reduced feed for 2015, 2020 and 2025 as in Table 3

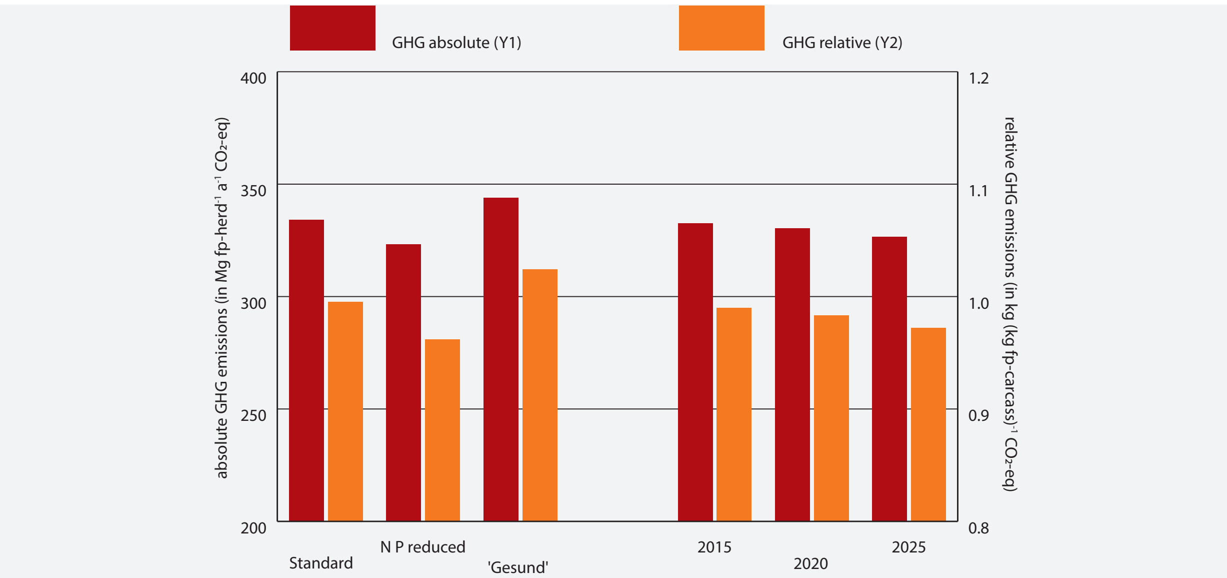


FIGURE 6
Impact of varying feed on GHG emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy). Left: 100 % of respective feed, right: proportions of standard and N P reduced feed for 2015, 2020 and 2025 as in Table 3

TABLE 3
Feeding of fatteners as proposed by the expert team

feed	unit	year		
		2015	2020	2025
standard	% of pigs fed	85	65	30
N P reduced	% of pigs fed	15	35	70

With less crude protein in the diet[7], feeding N P reduced and healthy ('Gesund') diets yield considerable reductions in NH_3 emissions from manure management and from fertiliser application and production occur. Also the fibre rich 'Gesund' diet leads to increased CH_4 emissions from enteric fermentation and from storage (*Figures 5* and *6*).

The reduction of emissions with increasing shares of N P reduced feed is obvious. (NH_3 N P reduced 8 %, 'Gesund' 16 %; GHG N P reduced 3 %, 'Gesund' – 1 %, as compared to standard, absolute and relative reductions).

Changing to feeds with reduced protein contents is definitely a useful tool for NH_3 reduction and is likely to be applied in future.

3.6 Effects of housing

Fully slatted floors have been state of the art for decades. However, they are considered inferior with respect to animal health. Partially slatted floors are assumed to be more animal friendly. Different emission factors were used for both types, assuming a reduction of 20 % for partially slatted floors (judgement of the expert team, based on a literature review described in Dämmgen et al., 2018c, Annex 5.4.) The overall frequency of partially slatted floors is assumed to be constant.

[7] Three phase feeding with crude protein contents in standard feed: 175, 165 and 155 g kg^{-1}, in N P reduced feed: 170, 150, and 140 g kg^{-1}, in 'Gesund' feed 155, 145 and 140 g kg^{-1} for feeding stages 1, 2 and 3, respectively.

For the same reason, fatteners should have more space than provided at present. This will result in larger soiled areas and increased NH_3 emissions. (The expert team assumes 25 % more emissions than 'normal' partially slatted floors on extended partially slatted floors. For details of this decision we refer to Dämmgen et al., 2018c, Annex 5.4). However, no assumptions could be made for their future frequency.

A small proportion of pigs are kept in straw based systems, mainly in organic pork production. Their share is assumed to increase slightly.

Air scrubbers are to be installed in bigger livestock buildings (> 1500 places for fatteners, > 560 places for sows and > 4500 places for weaners (TMfUEN, 2016)). An efficiency of 80 % for NH_3 reduction was used in this study (Dämmgen et al., 2010).

TABLE 4
Housing of fatteners as proposed by the expert team

housing	unit	year		
		2015	2020	2025
fully slatted floors	% of places	65	64	63
partially slatted floors	% of places	30	30	30
plane floor with bedding	% of places	5	6	7
air scrubbers	% of places	18	30	60

For NH_3 and GHG, partially slatted floors reduce absolute emissions by 11 and 2 % respectively, as related to fully slatted floors. However, smaller emissions in the building increase the emission potential in the subsequent processes. The use of farmyard manure (FYM) reduces GHG emissions by 4 %,

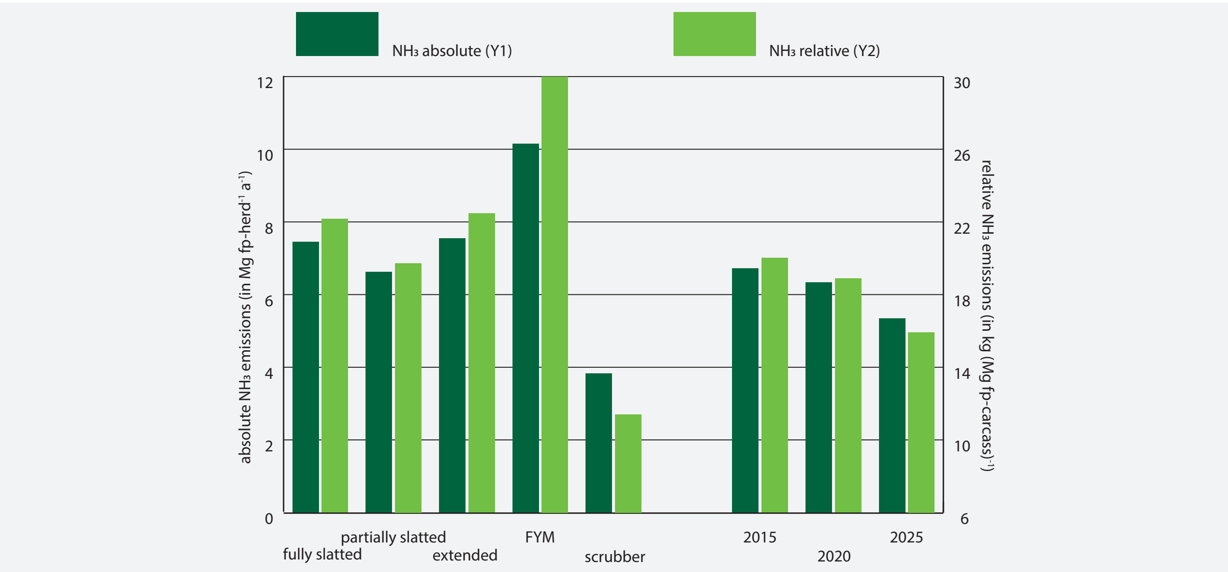


FIGURE 7
Impact of housing systems on NH_3 emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy). Left columns: frequency 100 %, respectively (FYM: farmyard manure), right columns with proportions of housing systems for 2015, 2020 and 2025 as in Table 4

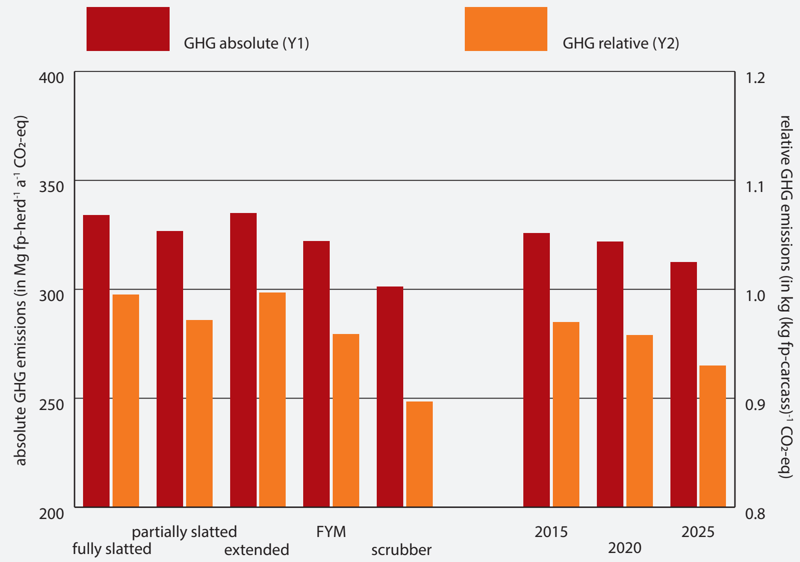


FIGURE 8
Impact of housing systems on GHG emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy). Left columns: frequency 100 %, respectively (FYM: farmyard manure), right columns with proportions of housing systems for 2015, 2020 and 2025 as in Table 4

but increases NH_3 emissions by 36 %. Scrubbers remove N from the system which reduces emissions from the house; the scrubbed N is fed into the slurry system immediately before spreading (see *Figures 7* and *8*.)

The projections for 2020 and 2025 reflect the increase in animal places equipped with active scrubbers.

If fully slatted floors are replaced by partially slatted floors, a considerable emission reduction can be achieved for NH_3. Scrubbers are a very effective (and expensive) means of NH_3 reduction. It is likely that this option is used in future.

3.7 Effects of storage

In Thuringia most slurry is stored in tanks covered with granules wherever slurry is not fermented in biogas plants. Tanks covered with plastic film have the same emission factor as covering with granules.

The expert team agreed that no changes can be anticipated at present. No projections were available for future shares of biogas installations (see *Table 5*). Calculations used the 2015 data for 2020 and 2025.

Changes in storage systems from the prevailing stores covered with granules are not meaningful. The reduction obtained by using solid covers is expensive and results in just a few percents reduction. Obviously fermentation producing biogas is the option to strive for with respect to GHG emissions. For NH_3, the net mineralization of slurry N increases the TAN[8] content of biogas slurry. The fermentation also results in an increased pH, and thus in an increased NH_3 vapour pressure (see *Figures 9* and *10*.)

TABLE 5
Storage of pig slurry as proposed by the expert team

storage facility	unit	year		
		2015	2020	2025
conventional tank without cover	% of slurry N	0	0	0
conventional tank, granules	% of slurry N	100	100	100
conventional tank, floating plastic film	% of slurry N	0	0	0
biogas tanks (gas tight)	% of slurry N	38	no estimate	no estimate

[8] TAN: total ammoniacal nitrogen, N in urine

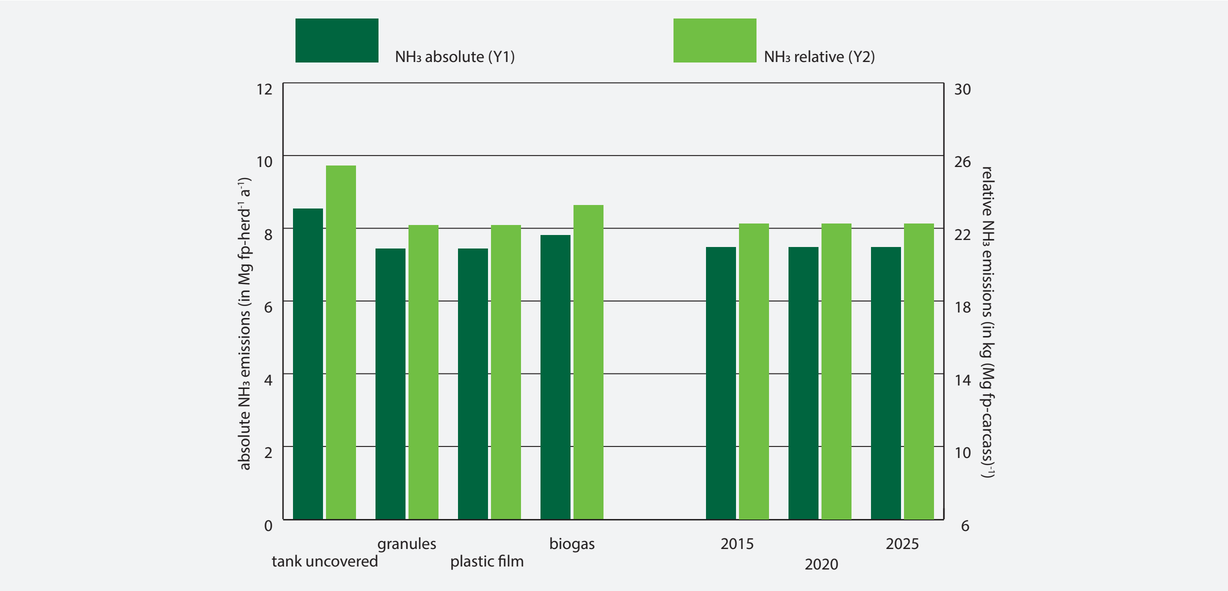


FIGURE 9
Impact of varying storage system on NH_3 emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy). Left columns: frequency of covers or biogas 100 %, respectively, right columns with proportions of storage system for 2015, 2020 and 2025 as in Table 5

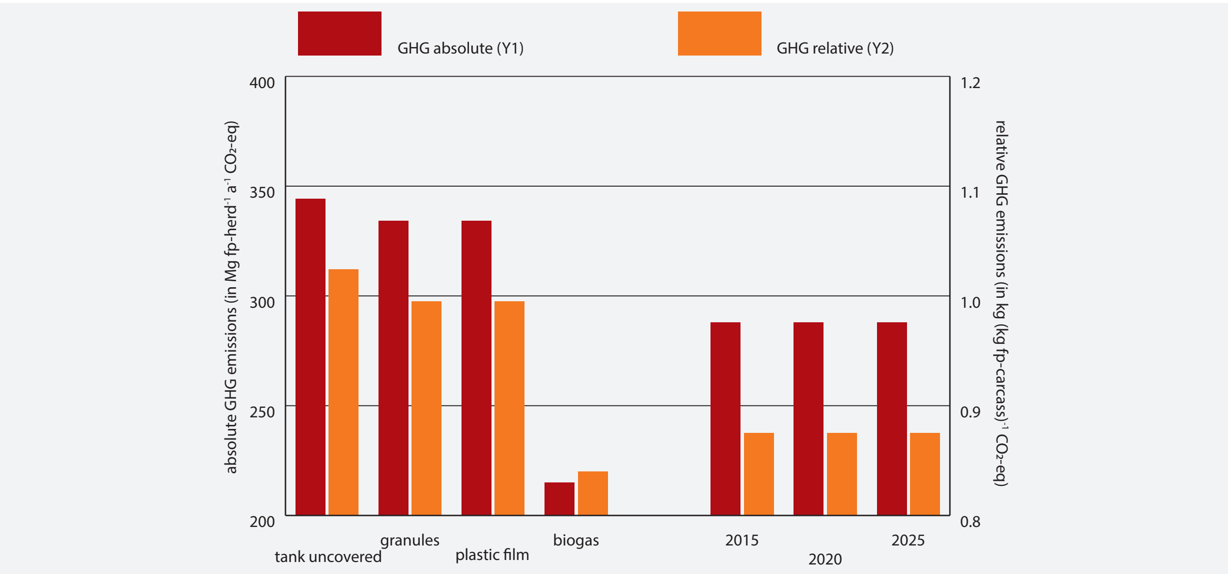


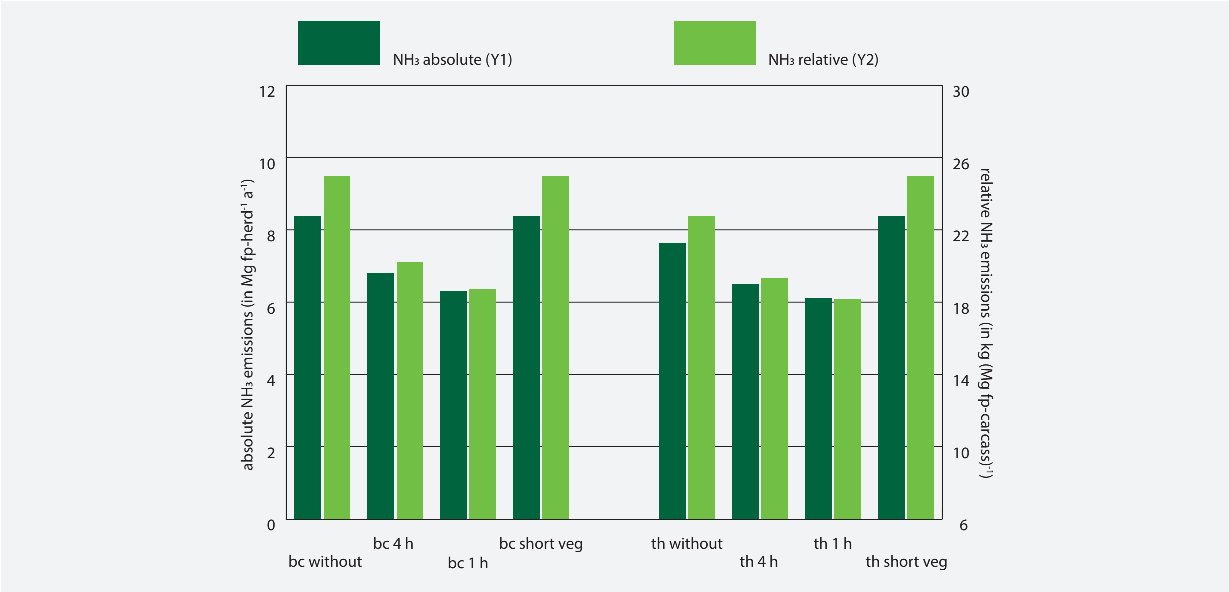
FIGURE 10
Impact varying housing system on GHG emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy). Left columns: frequency of covers or biogas 100 %, respectively, right columns with proportions of storage system for 2015, 2020 and 2025 as in Table 5

3.8 Effects of application techniques and time before incorporation of slurry

For NH_3 emissions, the surface of slurry exposed to the atmosphere controls the speed with which NH_3 is emitted per unit of area. The second important parameter is the duration of exposure.

Emission reduction aims at optimising both parameters. Injection is almost free from emissions whereas the old-fashioned broadcast application without incorporation loses almost all NH_3 to the atmosphere. As shown in *Figures 11* and *12*, NH_3 emissions during and after application of slurry differ greatly with the technique and the times before incorporation. However, in this analysis the overall effect on emission reduction is smaller than expected, as only small quantities of N and TAN are left after housing and storage losses. GHG emissions are also affected. Reductions are calculated for emissions from plant production (less mineral fertiliser) and fertiliser production as well as for indirect emissions.

The experts expect only small future changes. Increased share of injection remains an option.



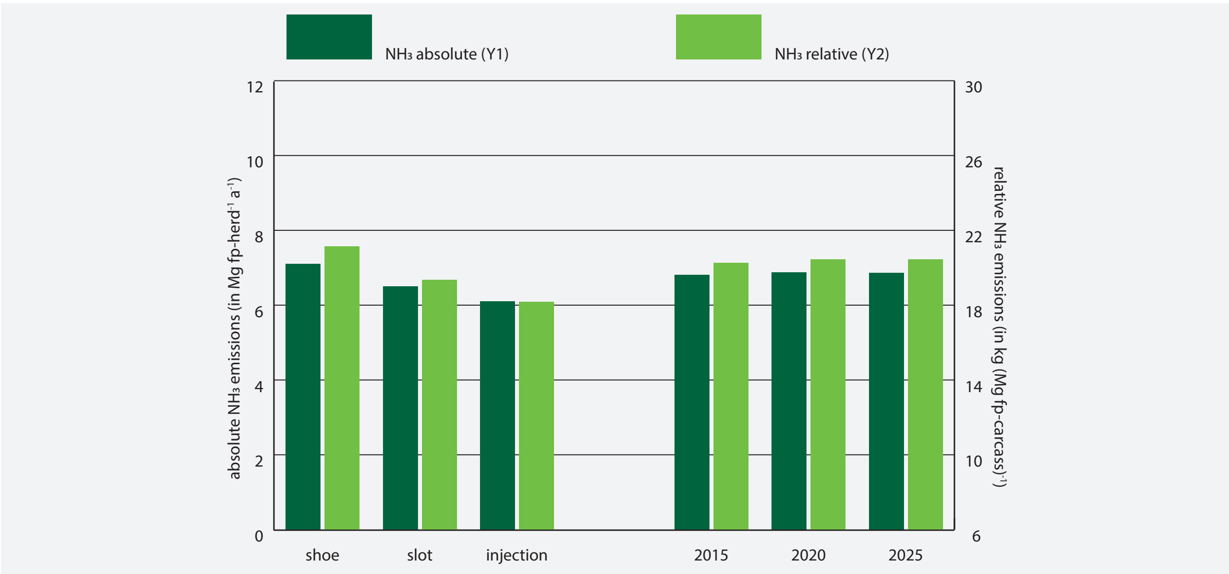


FIGURE 11
Impact of varying application techniques and speed of incorporation on NH_3 emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy).
Top: bc without: broadcast on bare soil without incorporation; bc 4 h: broadcast, incorporation within 4 h; bc 1 h: broadcast, incorporation within 1 h; bc short veg: broadcast on short vegetation, th: trailing hose
Bottom left: shoe: trailing shoe in short vegetation; slot: open slot;
bottom right: columns with proportions of application systems for 2015, 2020 and 2025 as in Tables 6 to 8

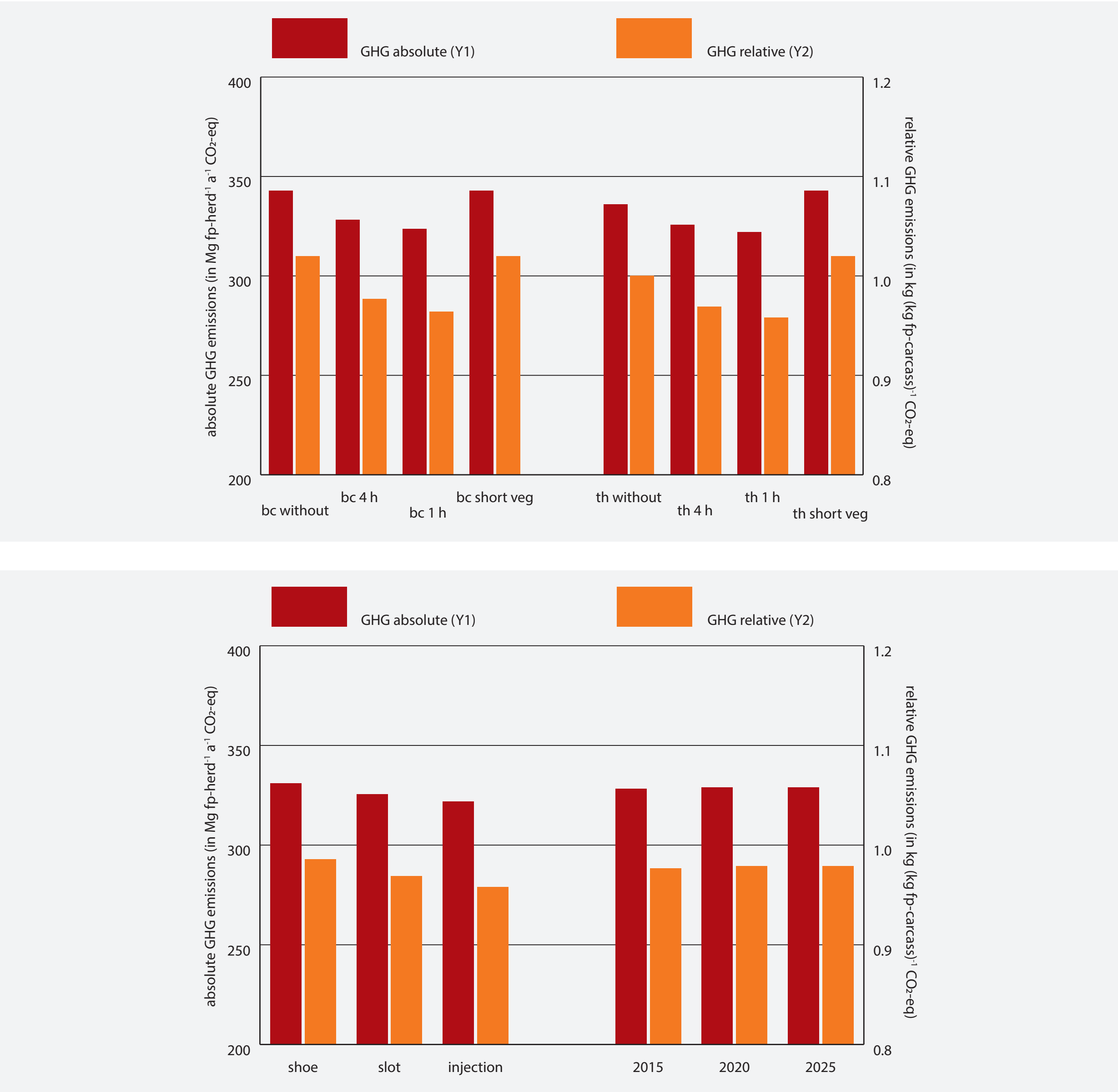


FIGURE 12

Impact of varying application techniques and speed of incorporation on GHG emissions from the herd of fattening pigs (including emissions from feed and fertiliser production, provision of water and energy).

Top: bc without: broadcast on bare soil without incorporation; bc 4 h: broadcast, incorporation within 4 h; bc 1 h: broadcast, incorporation within 1 h; bc short veg: broadcast on short vegetation, th: trailing hose

Bottom left: shoe: trailing shoe in short vegetation; slot: open slot;

bottom right: columns with proportions of application systems for 2015, 2020 and 2025 as in Tables 6 to 8

TABLE 6
Slurry application 1. Broadcast

location and incorporation	unit	year		
		2015	2020	2025
bare soil or stubbles, without incorporation	% of slurry N	0.0	0.0	0.0
bare soil or stubbles, incorporation within ≤ 1 h	% of slurry N	2.5	0.0	0.0
bare soil or stubbles, incorporation within ≤ 4 h	% of slurry N	1.9	0.0	0.0
short vegetation	% of slurry N	2.6	0.0	0.0
subtotal	% of slurry N	7.0	0.0	0.0

TABLE 7
Slurry application 2. Techniques with reduced emission

technique, location and incorporation	unit	year		
		2015	2020	2025
trailing hose				
bare soil, stubbles, without incorporation	% of slurry N	0	0	0
" , incorporation ≤ 1 h	% of slurry N	9	10	10
" , incorporation ≤ 4 h	% of slurry N	6	5	4
short vegetation	% of slurry N	24	30	30
trailing shoe	% of slurry N	1	2	2
open slot	% of slurry N	10	10	10
injection	% of slurry N	43	43	44
subtotal	% of slurry N	93	100	100

TABLE 8
FYM application, broadcast

location and incorporation	unit	year		
		2015	2020	2025
without incorporation	% of FYM N	60	50	40
bare soil, stubbles, incorporation ≤ 4 h	% of FYM N	10	10	10
bare soil, stubbles, incorporation ≤ 8 h	% of FYM N	30	40	50

3.9 Assessment of reduction potentials for fattening pigs

Some of the emission reduction potentials in single links of the production chain discussed above are promising, in particular for diet design in feeding, for the livestock building and for storage. Changing feed properties is a low or even no cost option. The equipment of livestock buildings with scrubbers is legally binding. There is no doubt that the other measures are at least partly feasible, although some of them will mean investments that restrict them to newly built livestock buildings or substantial refurbishments. Subsidies are likely to play a crucial role.

On the other hand, any new livestock building will be built according to modern standards. The experts' estimation is conservative in assuming that new houses are an unlikely option at present.

However, all single measures discussed above add up to considerable overall reductions. *Figure 13* illustrates the results, showing a reduction of almost 26 % for NH_3 and about 6 % for GHG for fattening pigs (absolute values).

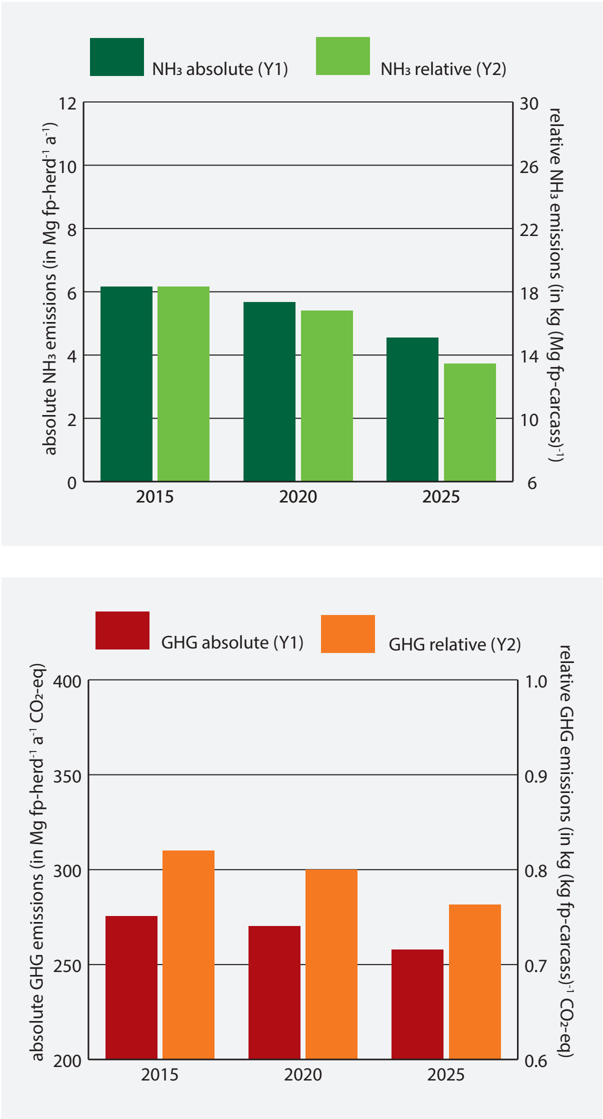


FIGURE 13
NH_3 and GHG emissions taking into account the 2015 data set and the expert projections for 2020 and 2025, fattening pigs only

4 Emission reduction in Thuringian pork production as anticipated for 2020 and 2025

In the following evaluations the scope is widened and covers all emissions from the entire production chain.

4.1 Assumptions

The assumptions of the expert team with respect to fattening pigs are listed in the tables above.

4.1.1 Piglet production

Weaners in piglet production (8 to 30 kg weaner $^{-1}$, *Table 9*) are kept on flat decks. The properties and composition of feeds used in their 3-stage diet are listed in Dämmgen et al. (2018a).

TABLE 9
Daily weight gains and animal losses of weaners as proposed by the expert team

parameter	unit	year		
		2015	2020	2025
daily weight gain	g weaner $^{-1}$ d $^{-1}$	428	440	455
losses	% of weaners housed initially	2	2	2

Breeding sows: No expert judgement could be obtained for the frequency of housing system for sows. Our calculations suppose that all are kept in slurry based houses. Feeding differentiates between lactating and gestating animals (for properties and composition of the feeds see Dämmgen et al., 2018b). Animal weights are taken into account (see *Table 10* and Dämmgen et al., 2018a). The number of piglets weaned and the share of losses are treated as variables (*Tables 11* and *12*). The fraction of stillborn piglets is constant and assumed to be 5 %.

TABLE 10
Animal weights at the beginning and the end of a production cycle (Dämmgen et al., 2018a)

live weight	unit	litter number							
		1	2	3	4	5	6	7	8
beginning of cycle	kg sow $^{-1}$	158	189	215	238	254	266	274	277
end of cycle	kg sow $^{-1}$	189	215	238	254	266	274	277	277

TABLE 11
Performance of breeding sows as proposed by the expert team

performance	unit	year		
		2015	2020	2025
piglets weaned (mean)	piglet sow $^{-1}$ a $^{-1}$	28.1	29.9	29.9

TABLE 12
Piglet losses as proposed by the expert team

year	unit	litter number							
		1	2	3	4	5	6	7	8
2015	% of piglets born live	16	13	12	12	13	14	15	15
2020	% of piglets born live	16	13	12	12	13	14	15	15
2025	% of piglets born live	15	12	11	11	12	13	14	14

Young sows for breeding have a constant daily weight gain of 700 g sow $^{-1}$ d $^{-1}$. Losses of 2 % are taken into account. Animals are raised on bedding and fed a special set of diets. Those suitable for breeding are fed to the start weight of breeding sows. The rest are slaughtered.

Barrows are fed standard diets as for fattening pigs. House and manure management reflect those of fattening pigs.

Weaners in basic production and pure breeding are fed the same diets as weaners for fattening. However, they are kept in bedded systems. Manure management is identical with that for young sows.

4.2 Results

4.2.1 Animal numbers and cumulative carcass weights

Changes in daily weight gains and losses for fatteners and weaners for fattening (*Tables 1, 2* and *9*), changes in the performance of sows as well as piglet losses result in changes for almost all livestock numbers (*Table 13*). Increased daily weight gains for fatteners lead to increased overall carcass weights. However, changes in emissions are almost negligible.

TABLE 13
Numbers of animals fed and cumulative carcass weights (rounded values)

animal category	number of animals fed animal herd^{-1} a^{-1}			carcass weight Mg herd^{-1} a^{-1}		
	2015	2020	2025	2015	2020	2025
fattening						
fattening pigs						
standard feed, slaughtered	2,340.9	1,799.5	834.9	286	220	102
standard feed, knackers	97.5	65.3	30.3			
N P reduced feed, slaughtered	413.1	968.4	1,948.1	50	118	238
N P reduced feed, knackers	17.2	35.1	70.7			
subtotal	*2,868.8*	*2,868.8*	*2,883.9*	*336*	*338*	*340*
weaners						
used for fattening	2,868.8	2,868.8	2,883.9			
knackers	57.4	57.4	57.7			
subtotal	2,926.2	2,926.2	2,941.6			
piglet production						
breeding sows	104.1	97.9	98.4	10.7	10.1	10.1
young sows fattened	68.7	64.5	64.9	1.2	1.1	1.1
barrows fattened	68.8	65.0	65.0	6.5	6.1	6.1
weaners	139.7	131.2	131.9			
subtotal				*18.4*	*17.3*	*17.3*
provision of boars for artificial insemination (AI boars)						
AI boars	0.2	0.2	0.2	0.0	0.0	0.0
teaser boars	0.5	0.5	0.5	0.0	0.0	0.0
young boars	1.8	1.6	1.7	0.1	0.1	0.1
sows fattened	1.8	1.6	1.7	0.2	0.2	0.2
weaners	3.5	3.3	3.3			
subtotal				*0.4*	*0.4*	*0.4*
pure breeding						
breeding sows	14.5	13.6	13.7	1.2	1.2	1.2
young sows	44.3	41.6	41.8			
surplus sows fattened	4.4	4.2	4.2	0.4	0.4	0.4
breeding boars	1.4	1.4	1.4	0.4	0.3	0.3
young boars	4.4	4.2	4.2			
barrows fattened	44.3	41.6	41.8	4.2	3.9	4.0
weaners	97.4	91.6	92.1			
subtotal				*6.2*	*5.9*	*6.0*
total				361	362	364

4.2.2 Emissions

Tables 14 to *16* collate emissions for the years 2015, 2020 and 2025, respectively. In order to improve clarity, the absolute totals and the carcass related emissions are listed in *Tables 17* and *18*. For some sources, emissions of GHG are not reported as N_2O, CH_4 or CO_2. Instead the overall figure is given and referred to as 'GHG' in *Tables 14* to *17*.

Tables 17 and *18* indicate that under the given assumptions a considerable emission reduction for NH_3 can be expected. Keeping in mind the importance of agricultural NH_3 emissions this is a major step forward and close to the target reduction of 29 % in 2030 (EU, 2016). The reduction of GHG is considered a welcome by-product.

TABLE 14
Overall emissions 2015 (values rounded)

emissions of	NH_3	N_2O	CH_4	CO_2	GHG*	total GHG
unit	kg herd^{-1} a^{-1}				Mg herd^{-1} a^{-1} CO_2-eq	
fattening						
fattening pigs	6,157	338	3,813	34,317	45.5	275.4
weaners	656	52	794	5,930	17.9	59.1
subtotal	*6,812*	*390*	*4,608*	*40,247*	*63.4*	*334.4*
piglet production						
breeding sows	977	141	680	3,668	32.9	95.6
young sows for breeding	141	12	33	930	3.1	8.6
surplus young sows fattened	31	2	4	126	0.7	1.6
barrows fattened	131	6	77	681	3.3	7.6
weaners	32	4	8	212	6.0	7.6
subtotal	*1,312*	*166*	*802*	*5,618*	*45.9*	*120.9*
provision of boars for artificial insemination (AI boars)						
AI boars	3	0	0	16	0.0	0.1
teaser boars	5	0	1	26	0.0	0.2
young boars	6	1	1	48	0.1	0.3
sows fattened	5	0	2	27	0.1	0.2
weaners	1	0	0	5	0.0	0.1
subtotal	*19*	*1*	*4*	*121*	*0.2*	*0.9*
pure breeding						
weaners	42	10	7	179	4.2	7.5
young boars	14	2	2	129	0.1	0.7
breeding boars	20	1	2	92	0.1	0.7
surplus barrows fattened	84	4	58	516	0.3	3.4
young sows	187	16	26	818	2.4	8.6
breeding sows	64	10	148	816	5.8	13.2
surplus sows fattened	24	1	6	53	0.0	0.6
subtotal	*434*	*43*	*250*	*2,603*	*12.9*	*34.7*

* Some sources do not report single GHGs (N_2O, CH_4, CO_2), but the respective sum. This column contains such emissions *reported* as GHG, whereas total GHG is the sum of the weighted emissions of CO_2 (global warming potential GWP 1 kg kg^{-1}), CH_4 (GWP 25 kg kg^{-1}), N_2O (GWP 298 kg kg^{-1}) and GHG (GWP 1 kg kg^{-1})

TABLE 15
Overall emissions anticipated for 2020 (values rounded)

emissions of	NH_3	N_2O	CH_4	CO_2	GHG	total GHG
unit	kg herd^{-1} a^{-1}				Mg herd^{-1} a^{-1} CO_2-eq	
fattening						
fattening pigs	5,673	333	3,797	31,205	45.3	270.3
weaners	640	51	786	5,865	17.6	58.3
subtotal	*6,313*	*385*	*4,583*	*37,070*	*62.9*	*328.6*
piglet production						
breeding sows	934	140	639	3,668	31.2	92.7
young sows for breeding	140	12	33	875	3.1	8.4
surplus young sows fattened	31	2	4	118	0.6	1.5
barrows fattened	120	5	76	639	3.3	7.4
weaners	32	4	8	200	6.0	7.6
subtotal	*1,258*	*164*	*761*	*5,500*	*44.1*	*117.5*
provision of boars for artificial insemination (AI boars)						
AI boars	3	0	0	16	0.0	0.1
teaser boars	5	0	1	26	0.0	0.2
young boars	6	1	1	48	0.1	0.4
sows fattened	4	0	2	25	0.1	0.2
weaners	1	0	0	5	0.0	0.1
subtotal	*18*	*2*	*4*	*119*	*0.2*	*1.0*
pure breeding						
weaners	34	3	6	167	3.9	5.2
young boars	13	1	2	121	0.1	0.7
breeding boars	20	1	2	87	0.1	0.7
surplus barrows fattened	86	4	57	485	0.3	3.3
young sows	177	15	25	769	2.3	8.1
breeding sows	63	9	144	801	5.7	12.9
surplus sows fattened	24	1	6	50	0.0	0.6
subtotal	*417*	*35*	*242*	*248*	*12.4*	*31.4*

TABLE 16
Overall emissions anticipated for 2025

emissions of	NH_3	N_2O	CH_4	CO_2	GHG	total GHG
unit	kg herd^{-1} a^{-1}				Mg herd^{-1} a^{-1} CO_2-eq	
fattening						
fattening pigs	4,559	317	3,783	25,719	44.3	258.5
weaners	629	50	780	5,811	17.2	57.4
subtotal	*5,188*	*367*	*4,564*	*31,530*	*61.5*	*315.9*
piglet production						
breeding sows	917	140	643	3,643	31.3	92.8
young sows for breeding	140	12	33	879	3.1	8.4
surplus young sows fattened	32	2	4	119	0.6	1.6
barrows fattened	95	5	76	638	3.2	7.2
weaners	33	4	8	201	6.0	7.6
subtotal	*1,216*	*164*	*764*	*5,480*	*44.2*	*117.5*
provision of boars for artificial insemination (AI boars)						
AI boars	3	0	0	16	0.0	0.1
teaser boars	5	0	1	26	0.0	0.2
young boars	6	1	1	48	0.1	0.4
sows fattened	3	0	2	25	0.1	0.2
weaners	1	0	0	5	0.0	0.1
subtotal	*18*	*2*	*4*	*120*	*0.2*	*1.0*
pure breeding						
weaners	34	3	6	168	3.9	5.2
young boars	14	1	2	121	0.1	0.7
breeding boars	20	1	2	87	0.1	0.7
surplus barrows fattened	72	3	56	485	0.3	3.2
young sows	179	15	25	773	2.3	8.1
breeding sows	56	9	145	804	5.7	12.9
surplus sows fattened	23	1	6	50	0.0	0.5
subtotal	*397*	*35*	*242*	*2,489*	*12.4*	*31.3*

TABLE 17
Compilation of subtotals and totals

emissions of	NH_3	N_2O	CH_4	CO_2	GHG	total GHG
unit	kg herd^{-1} a^{-1}				Mg herd^{-1} a^{-1} CO_2-eq	
2015						
fattening	6,812	390	4,608	40,246	63	334
piglet production	1,312	166	802	5618	46	121
provision of boars	19	1	4	121	0	1
pure breeding	434	43	250	2,603	13	35
total	*8,577*	*600*	*5,664*	*48,588*	*122*	*491*
2020						
fattening	6,313	385	4,583	37,070	63	329
piglet production	1,258	164	761	5,500	44	118
provision of boars	18	2	4	119	0	1
pure breeding	417	35	242	2,479	12	31
total	*8,006*	*586*	*5,590*	*45,168*	*120*	*478*
% of 2015	93	98	99	93	98	97
2025						
fattening	5,188	367	4,564	31,530	62	316
piglet production	1,216	164	764	5,480	44	118
provision of boars	18	2	4	120	0	1
pure breeding	397	35	242	2,489	12	31
total	*6,819*	*567*	*5,574*	*39,619*	*118*	*466*
% of 2015	80	94	98	82	97	95

TABLE 18
Carcass related NH_3 and GHG emissions

gas	unit	year		
		2015	2020	2025
NH_3	kg (Mg carcass)$^{-1}$ NH_3	30.1	28.0	23.7
GHG	kg (kg carcass)$^{-1}$ CO_2-eq	1.72	1.67	1.62

5 Discussion

5.1 General remarks

Future agriculture will have to face a host of problems. However, agriculture is the vitally essential food producer. More people have to be fed from a shrinking agriculturally usable land area. An increasing demand for meat and milk reduces the overall efficiency of agricultural production, i.e. the ratio of output to input of energy. Restrictions are in force or planned that aim to reduce agriculture's impact on the environment and to improve animal health and welfare. The obvious solution to many constraints is an increase in plant and animal performance and increased efficiency in the use of resources. Improving performances in every link of the chain is indispensable, which applies to increased daily weight gains in particular.

This paper is to a large extent based on expert projections. One might call the experts' team's proposals cautious, conservative or even unambitious. It is definitively not describing maximum technical feasibility, but reflects the potential social feasibility in a densely populated area. And: agricultural enterprises have to be profitable. This work could provide a methodical tool to look for serious compromises and proposals to further improve the efficiency of pork production with reduced environmental impact.

5.2 Methods

Pork production is a complex process. Its description mainly reflects the energy needs of animals, coupled with the fluxes of nutrients and water. Energy is also used in the entire production chain. However, energy requirements other than in farm management such as for the construction of buildings and machines or for transport are not treated as variables in this paper.

The description of energy and matter fluxes also forms the base of emission reporting to the various international bodies. A complex way of interlinking the various calculation procedures provided there had to be found that was able to depict pork production in Thuringia correctly – at least in principle. Some models had to be improved or refined to achieve the tool needed to quantify and assess those emission reductions which are in the scope of the livestock farmer.

In most cases, the methods provided in the respective guidebooks are best approximations. In some cases they are 'rules of thumb'. However, it is better to use them than not to use them; they are at least internationally accepted tools.

5.3 Uncertainties

General remarks on the uncertainty of model calculations of emissions can be found in Part 1 of this work (Dämmgen et al., 2018a). The number of digits in the above tables does not reflect the uncertainty. It allows for an easy comparison of the emissions originating from the various animal categories.

5.4 Comparability and comparative data

As shown in the respective Chapter in Part 1 (Chapter 4.2), the results obtained in this work are in line with most other similar investigations. However, a direct intercomparison suffers from inadequate information on details. For example, this paper uses the official German recommendations for the application of mineral fertilisers. However, the basis for these recommendations has not been fully documented. For example, there is no mention of the impacts on atmospheric deposition of N and no adjustments in the recommendations according to the risks of run-off and leaching.

Furthermore, the Thuringian results for 2015, 2020 and 2025 illustrate the range of potential variations. This is what this paper wants to emphasize: changes to management practices in order to reduce emissions are feasible, and they are likely to be required in order to meet commitments to reduce emissions.

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