Food and Agriculture
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DRAFT FOR PUBLIC REVIEW

Methane emissions in livestock and rice systems

Sources, quantification, mitigation and metrics

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FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
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Foreword

[To be completed after public review]

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Development process of methane report

This comprehensive report is a product of the FAO Livestock Environmental Assessment and Performance (LEAP) Partnership. It was developed by the Technical Advisory Group (TAG) on Methane. The TAG was established in January 2021 and met virtually in two workshops: the first in February 2020 and the second in June 2021. Between the workshops, the members of the TAG worked via several online meetings, communications and teleconferences.

Technical Advisory Group on methane

The Technical Advisory Group (TAG) on methane conducted the background research, review and developed the core technical content. The methane TAG was composed of more than 60 experts from 50 entities and was led by Ermias Kebreab (co-chair, University of California, Davis, United States of America), Michelle Cain (co-chair, Cranfield Environment Centre, Cranfield University, United Kingdom), Jun Murase (co-chair, Graduate School of Bioagricultural Sciences, Nagoya University, Japan.) and Aimable Uwizeye (FAO LEAP Secretariat, Animal Production and Health Division, Food and Agriculture Organization of the United Nations, Italy).

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The LEAP Secretariat

The LEAP Secretariat coordinated and facilitated the work of the methane TAG, guided and contributed to the content development and ensured coherence between the various guidelines. The LEAP Secretariat, hosted at FAO, was composed of: Camillo De Camillis (Technical officer and LEAP manager, until 30 April 2022), Aimable Uwizeye (Technical officer and FAO LEAP Coordinator, since August 2022), Monica Rulli (Technical Data Specialist) Emmie Wachira (Programme and outreach specialist, Until 31 December 2021), Sara Giuliani (Programme and outreach specialist, Since March 2022), Tim Robinson (FAO LEAP Coordinator, until 31 July 2022, FAO LEAP Co-Chair, since August 2022) and Henning Steinfeld (FAO LEAP Co-chair, Until July 2022). Aimable Uwizeye and Camillo De Camillis (until April 2022) coordinated technical inputs and development process of this report.

LEAP Steering Committee

The LEAP Steering Committee provided overall guidance for the activities of the Partnership and facilitated review and clearance of the guidelines for public release. Steering Committee members: Angeline Munzara (World Vision, since November 2016, South Africa), Richard de Mooij (European Livestock and Meat Trading Union – EUCBV; International Meat Secretariat – IMS), Don Syme (Embassy of New Zealand, Italy, Until September 2022), Benjamin Ross (First Secretary (Agriculture), Australian Embassy, Rome Italy), Alessandro Aduso (Ministry for Primary Industries, New Zealand, since 2018), Peter Ettema (Ministry for Primary Industries, New Zealand), Hsin Huang (IMS, France, LEAP chair 2022), Gaelle Thyriou (Beef + Lamb New Zealand, IMS), Jean- Pierre Biber (International Union for Conservation of Nature – IUCN, Switzerland), María Sánchez Mainar (International Dairy Federation – IDF, Belgium, since January 2018), Caroline Emond (IDF, Belgium, since January 2018, LEAP chair 2019), Pablo Manzano (IUCN, Kenya), Nicolas Martin (European Feed Manufacturers' Federation – FEFAC, Belgium; International Feed Industry Federation – IFIF), Frank Mitloehner (University of California, Davis, IFIF, United States of America), Anne-Marie Neeteson-van Nieuwenhoven (International Poultry Council – IPC, the Netherlands, until May 2018), Edwina Love (Department of Agriculture, Food and the Marine – DAFM, Ireland), Frank O'Mara (Agriculture and Food Development Authority – Teagasc, Ireland), Nicoló Cinotti (IPC, Italy, since May 2018), Marilia Rangel Campos (IPC, Brazil), Alexandra de Athayde (IFIF, Germany), Julian Madeley (International Egg Commission – IEC, United Kingdom), Jurgen Preugschas (Canadian Pork Council, Canada, IMS), Kim van Seeters (Ministry of Agriculture, the Netherlands, since July 2018), Gianina Müller Pozzebon (Permanent Representative of Brazil to FAO, since March 2018), Felipe Heimburguer (Division of Basic Commodities, Ministry of Foreign Affairs, Brazil, since September 2017), Tim McAllister (Agriculture and Agri-Food Canada), Julius Mutua (State Department of Livestock, Kenya), Mauricio Chacón Navarro (Ministry of Agriculture and Livestock, Costa Rica), Walter Oyhantcabal (Ministry of Livestock, Agriculture and Fisheries, Uruguay), , Alwin Kopse (Swiss Federal Office for Agriculture – FOAG, Switzerland), Jeanine Volken (FOAG, Switzerland), Jennifer Fellows (Permanent Representative of Canada to FAO), Emmanuel Coste (Interbev, France, IMS), Dalena White (IWTO, Belgium), Paul Swan (IWTO, Australia, since March 2018), Pablo Frere (World Alliance of Mobile Indigenous Peoples – WAMIP, Argentina), Henning Steinfeld (FAO, LEAP co-chair), Camillo De Camillis

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Multi-step review process

The initial draft report developed by the methane TAG from March to November 2021. It was peer-reviewed in January 2022 and has been revised before being submitted for public review. The peer-review was conducted by Nico Peiren (International Dairy Federation; Belgium), Alexandra de Athayde (International Feed Industry Federation, Germany), Sabine Van Cauwenberghe (DSM, Switzerland), Carlos Alberto Ramirez Restrepo (CR Eco-efficient Agriculture Consultancy, Australia), Maria Sanchez Mainar (International Dairy Federation, Belgium), Claudia Arndt (International Livestock Research Institute), Peyraud Jean Louis (INRAE, France), Pablo Manzano (IUCN, Spain), Boucher Olivier (CNRS/Sorbonne Universté, France), Bruno Notarnicola (University of Bari Aldo Moro, Italy), and xxx (Wageningen University and Research). FAO LEAP Secretariat has also reviewed the report. The LEAP Steering Committee provided the oversight of the development of the report and provided clearance for public review.

The public review is launched in October 2022 for one month. All the contributors to the public review will be acknowledge in the report.

Livestock Environmental Assessment and Performance (LEAP) Partnership

The FAO LEAP is a multi-stakeholder initiative launched in July 2012 with the goal of improving the environmental performance of livestock supply chains. Hosted by the Food and Agriculture Organization of the United Nations (FAO), LEAP brings together the private sector, governments, academia, civil society representatives and leading experts who have a direct interest in the development of science-based, transparent and pragmatic guidance to measure and improve the environmental performance of livestock products. The first phase of the Partnership (2013–15) focused mainly on the development of guidelines to quantify the greenhouse gas (GHG) emissions, energy use and land occupation from feed and animal supply chains and to illustrate the principles for biodiversity assessment. The second phase (2016–18), known as LEAP+, broadened the scope, focusing on, for example, water footprinting, nutrient flows and impact assessment, soil carbon stock changes, quantification of the impact of livestock on biodiversity and the impact of feed additives. The third phase (2019-2021), known as LEAP3, focused on the road-testing of LEAP guidelines by collaborating with different entities that used the guidelines and provided feedback and case studies that are published in FAO LEAP Catalogue of applications. This phase also initiated the development of the comprehensive report on methane emissions in livestock and rice systems. The fourth phase (2022-2024), known as LEAP4, focuses on the development of guidelines on circular bio-economy, ecosystem services, and leather. It also aims to disseminate LEAP guidelines through the development of e-learning programmes and regional workshops.

The FAO LEAP Partnership provides state-of-the-arts methods and metrics required to assess the environmental impacts and benchmarking performance across livestock supply chains. This is necessary because the projected growth of the livestock sector in the coming decades places significant pressure on livestock stakeholders to adopt sustainable development practices mitigate climate change and build resilience.[1]

[1] More background information on the Partnership can be found at: www.fao.org/partnerships/leap/en/.

Abbreviations and acronyms

3-NOP	3-nitrooxypropanol
AD	anaerobic digestion
BCM	2-bromochloromethane
BES	bromoethanesulfonate
CH_4	methane
CO_2	carbon dioxide
CO_2eq	carbon dioxide equivalent
CP	crude protein
CT	condensed tannins
DM	dry matter
DMI	dry matter intake
EF	emission factor
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GE	gross energy
GEI	gross energy intake
GHG	greenhouse gases
GIT	gastrointestinal tract
GTP	global temperture potential
GWP	global warming potential
HT	hydrolysable tannins
IPCC	Intergovernmental Panel on Climate Change
LCA	life cycle assessment
LEAP	FAO Livestock Environmental Assessment and Performance Partnership
LUC	land use change
MCFA	medium-chain fatty acids
MCR	methyl-coenzyme M reductase
NDF	neutral detergent fiber
N_2O	nitrous oxide
OM	organic matter
PUFA	polyunsaturated fatty acids
SF_6	sulfur hexafluoride tracer gas
SPS	silvopastoral system
TAG	Technical Advisory Group
VFA	volatile fatty acid
VFA	volatile fatty acids
WSC	water soluble carbohydrates
Ym	methane conversion factor (percent)

Executive summary

[To be completed after the public review]

Introduction

Achieving the sustainability of agrifood systems is urgent as the global community is expecting each sector of the economy to undertake necessary transformative actions. This is still a challenge in agrifood systems because of the current production of food, in particular livestock products, and nutrition to a growing population in the context of climate change and other environmental impacts. Agrifood systems, including agriculture, forestry and land-use (AFOLU), are responsible for 23 percent of total anthropogenic greenhouse gas (GHG) emissions globally in 2017 assessed using a global warming potential (GWP) of a 100-year horizon (IPCC, 2019b). Livestock supply chains alone play an important role in climate change representing 14.5 percent of human induced GHG emissions. The share of the livestock sector in GHG emissions is region-specific and depends on the magnitude of other economic sectors, mainly the energy sector. For instance, the United States of America's Environmental Protection Agency (EPA) reports that although agrifood systems are responsible for 9 to 10 percent of total GHG emissions, livestock contribute less than 4 percent of direct emissions (Dillon et al., 2021; Tedeschi, 2022). Most of emissions from AFOLU are in form of methane (CH_4) originating from livestock systems (enteric fermentation and manure management systems, 32 percent) and flooded paddy rice production (8 percent) (UNEP and CCAC, 2020). According to FAOSTAT (2017), the world ruminant population increased by 66 percent from 1960 to 2017, whereas the population of non-ruminants has increased even more rapidly by 435 percent over the same period. Both ruminant and non-ruminant populations are projected to further increase, which will further exacerbate GHG emissions, in particular CH_4, from livestock systems (FAO, 2018). Meat and milk from ruminant livestock provide an important source of protein and other nutrients for human consumption. Although ruminants have a unique advantage of being able to consume forages and grazing lands not suitable for arable cropping, 2 to 12 percent of the gross energy (GE) consumed is converted to enteric CH_4 during ruminal digestion. More CH_4 is also emitted during manure management systems.

Over 122 countries and supporters have signed the Global Methane Pledge (www.globalmethanepledge.org), which is a voluntary commitment initiated by the European Union and United States of America, to decrease CH_4 emissions collectively by 30 percent from 2020 levels by 2030. Reducing CH4 by 30 percent would eliminate over 0.2 °C of average global temperature increase by 2050. Due to the relatively short life of CH_4 in the atmosphere and its high global warming potential, reducing CH_4 emissions is seen as a rapid way to help limit global warming to 1.5 °C above preindustrial levels.

The FAO Livestock Environmental Assessment and Performance Partnership (FAO LEAP Partnership) commissioned this report, which is developed by an international group of scientists and experts working on sources and sinks of CH_4, quantification of CH_4 emissions, and related mitigations and climate metrics. This report aims to provide a comprehensive review and analysis of sources and sinks of CH_4 in agrifood systems, existing and innovative mitigation solutions and metrics to quantify the impacts of CH_4 emissions on climate. This report provides a comprehensive scientific information that can be used by different stakeholders including public, private sector, non-states entities and producers' organizations, to design and implement technical mitigation strategies and programmes to cut CH_4

emissions in livestock and rice systems in the context of the Global Methane Pledge in order to achieve the Paris Agreement goals. It also provides information necessary to advance policy work that will enhance national climate actions. This report complements also previous FAO LEAP guidelines with detailed information required to conduct mitigation scenario analysis using highest tier from IPCC guidelines. This will continuously improve the accuracy, transparency, consistency, comparability, and completeness of estimates of greenhouse gas inventory, including CH_4, as well as the monitoring of mitigation programmes in livestock, thus providing greater transparency.

This report is divided into four parts as follows:

- Part 1: Sources and sinks of methane in agriculture
- Part 2: Quantification of methane emissions
- Part 3: Mitigation of methane emissions
- Part 4: Metrics for quantifying impacts of methane emissions

Part 1 and 2 were published as Tedeschi et al. 2022. *Quantification of methane emitted by ruminants: a review of methods*. Journal of Animal Science, Volume 100, Issue 7, July 2022, skac197, https://doi.org/10.1093/jas/skac197. Part 3 was published as Beauchemin et al. 2022. *Invited review: Current enteric methane mitigation options.* J. Dairy Sci. TBC:1–30. https://doi.org/10.3168/jds.2022-22091.

Part 1. Sources and sinks of methane emissions in agriculture

1. Sources of methane

The provision of quality human food in the context of a growing world population and the need for sustainable food production systems is a major challenge. Indeed, by 2050 the global demand for animal products is projected to increase by 60 percent to 70 percent, with developing countries accounting for the [2]majority of this increase (Makkar, 2018). Global warming as a consequence of anthropogenic emissions of greenhouse gases (GHGs) has become a major challenge to humanity in recent years. Agriculturally derived GHG emissions, and in particular methane (CH_4), primarily result from enteric fermentation and, to a lesser extent storage of manure, of ruminant livestock. The livestock sector is the largest land-use system on earth, occupying between 30 percent (Herrero et al., 2013) to 60 percent (Manzano 2015) of the world's ice-free surface. Livestock supply chains are estimated to account for 14.5 percent of total human-induced GHG emissions (Gerber et al., 2013) and it is estimated that about 80 percent of the GHG from livestock, and 90 percent of CH_4 emissions, is derived from ruminant livestock* (Scholtz et al., 2020). The world ruminant population increased by 66 percent from 1960 to 2017, whereas the population of non-ruminants has increased even more rapidly by 435 percent over the same period (FAOSTAT, 2017). Both ruminant and non-ruminant populations are projected to further increase, which will further exacerbate GHG emissions from animal agriculture. Meat and milk from ruminant livestock provide an important source of protein and other nutrients for human consumption. Although ruminants have a unique advantage of being able to consume forages and grazing lands not suitable for arable cropping, 2 to 12 percent of the gross energy (GE) consumed is converted to enteric CH_4 during ruminal digestion, contributing to approximately 6 percent of global anthropogenic GHG emissions (Beauchemin et al., 2020). Globally, the production, processing and transport of feed account for 45 percent of the GHG emissions from the livestock sector, and contribution of enteric CH_4 is 39 percent (Gerber et al., 2013), which in itself is heavily influenced by the type of feed offered to livestock.

Dairy production systems are estimated to contribute around 2.3 gigatonnes of carbon dioxide (CO_2) equivalent (CO_2eq) annually, representing more than 30 percent of livestock emissions. Nearly 60

[2] Here percentages are computed after weighting the different GHG emissions by their respective Global Warming Potential for a time horizon of 100 years. See Part 4.

percent of these emissions are in the form of CH_4, arising from enteric fermentation while approximately 30 percent are the result of fertilizer and manure application to produce feed, principally in the form of CO_2 and N_2O, with manure management systems accounting for the remaining 10 percent, in the form of CH_4 and N_2O.

1.1 Ruminant livestock and enteric methanogenesis

The majority of CH_4 emissions from the agricultural sector are a consequence of microbial mediated enteric fermentative processes in ruminant livestock. Within ruminants, cattle are the highest daily emitters on a per animal basis, followed by sheep, with goats and buffalos having similar emissions (Seijan et al., 2011). Emissions of GHG, including CH_4, produced by both large herbivorous nonruminants and the large population of small farm animals, such as swine, remain substantial (Patra, 2014). Indeed, Clauss et al. (2020) stated that the CH_4 emissions from some nonruminants, when expressed in intensity terms, remains comparable to that of ruminants. Misiukiewicz et al. (2021) recently provided a comprehensive review of methanogens living in the gastrointestinal tract (GIT) of various nonruminants, such as swine, horses, donkeys, rabbits, and poultry.

Inter-animal variation in enteric CH_4 emissions can be substantial even within the same species with increasing evidence that host genetics play an important role (see later section). Notwithstanding these environmental and management factors, chemical composition of the diet, feeding level, inclusion of certain feed additives etc. have a greater influence on individual animal emissions than genetic makeup per se. Indeed, manipulation of such environmental factors can be harnessed as mitigation strategies to reduce CH_4 emissions (see mitigation strategy section).

The rumen is a complex ecosystem composed of bacteria, fungi, protozoa, methanogens and bacteriophages, all of which contribute to dietary energy harvesting and resultant nutrient supply to the host (Abbott et al., 2020). These microbes interact closely to break down plant material that cannot be digested by humans and also by other animals, whilst providing metabolic energy to the host and, in the case of archaea, producing CH_4 (Huws et al., 2018). Methane is produced from released hydrogen, being utilised by the methanogens, which belong to the domain Archaea. Methane produced in the rumen accounts for up to 90 percent of ruminant enteric CH_4 emissions, with microbial fermentation in the large intestine accounting for the remainder of emissions. Large intestine fermentation is also a characteristic of nonruminants such as swine and hind-gut fermenters such as horses, which also produce CH_4, though to a much lower extent.

1.2 Biochemistry of methane production in microbial anaerobic ecosystems

In anaerobic environments with low oxygen concentrations and limited mineral electron acceptors, fermentation can provide Gibbs energy to generate ATP necessary for microbial maintenance and growth. Fermentation is an incomplete oxidation in which carbon compounds formed in the process itself are the ultimate electron acceptors (Ungerfeld, 2020). The following section focuses on three anaerobic microbial ecosystems in which CH_4 is a principal electron sink: the rumen, manure, and rice soils.

1.2.1 Rumen methanogenesis

Methane production is a ubiquitous, apparently unavoidable side effect of fermentative fibre digestion by symbiotic microbiota in mammalian herbivores (Clauss et al., 2020). Structural and non-structural carbohydrates are the main source of energy and carbon for ruminants. In the rumen, polymers such as cellulose, hemicellulose and starch are digested by a complex consortium of bacteria, protozoa and fungi, and the resulting monomers are metabolized to volatile fatty acids (VFA; mainly acetate, propionate and butyrate), CO_2 and CH_4 as main final products of fermentation, with dihydrogen, formate, lactate and succinate as important electron carrier intermediates (Figure 1; Russell and Wallace, 1997; Ungerfeld, 2020).

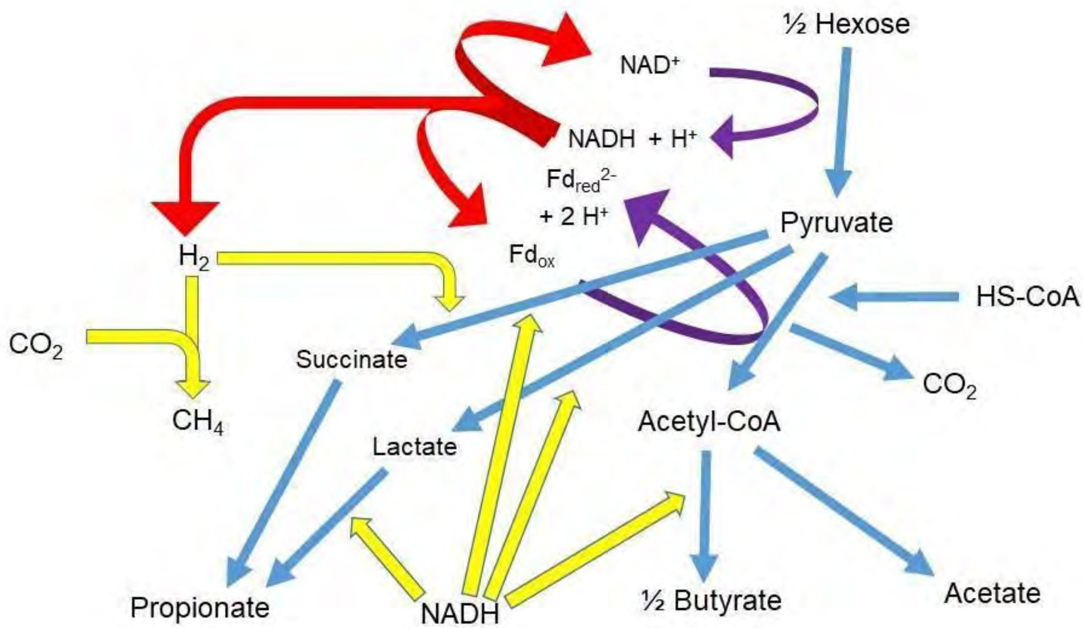

Figure 1. Main biochemical pathways in rumen fermentation.

In blue, transformations of carbon compounds. Note that smaller flows of volatile fatty acids (VFA) interconversion also exist (see summary by Ungerfeld and Kohn (2006) and posterior work by Markantonatos et al. (2008, 2009), Markantonatos and Varga (2017), Nolan et al. (2014) and Gleason et al. (2022)). In purple, semi-reactions of cofactor reduction. In red, dihydrogen production via oxidation of reduced ferredoxins by prototypical or Ech hydrogenases or via flavin-based electron confurcation (Ungerfeld and Hackmann, 2020). Formate may be formed instead of CO_2 and dihydrogen if pyruvate oxidative decarboxylation is catalyzed by pyruvate-formate lyases (Russell and Wallace, 1997). In yellow, incorporation of metabolic hydrogen in reduced cofactors or dihydrogen (or formate, not shown) into methanogenesis, propionate formation via randomizing (succinate) and non-randomizing (acrylate) pathways, and butyrate formation. Metabolic hydrogen in NADH at a particular point in time may or may not have been part of the dihydrogen pool previous to its incorporation into a particular pathway, therefore NADH shown as direct electron donor in propionate and butyrate formation may or may not have been formed through reduction of NAD^+ with H_2 in bifurcation with Fd_{ox}.

Similar to other anaerobic microbial ecosystems, syntrophic interactions in the microbial community are key to rumen metabolism. Central to rumen fermentation is the transfer of metabolic hydrogen, in particular as the interspecies transfer of dihydrogen. In glycolysis and pyruvate oxidative decarboxylation, electrons are transferred to oxidized cofactors (mainly NAD^+ and oxidized ferredoxin; Ungerfeld and Hackmann, 2020). The resulting reduced cofactors must be re-oxidized for fermentation to continue (Wolin et al., 1997). Cofactors re-oxidation occurs mostly through hydrogen-evolving hydrogenases which transfer electrons to protons to form dihydrogen (Frey et al., 2002) and formate (Russell and Wallace, 1997). Greening et al. (2019) showed a pivotal role in rumen fermentation of flavin-based electron confurcation and bifurcation (Buckel and Thauer, 2013; 2018a,b) in the formation and incorporation of dihydrogen, respectively.

Dihydrogen does not accumulate in the rumen because it is transferred to methanogens and other hydrogenotrophic microorganisms. Methanogens utilize dihydrogen to reduce CO_2 to CH_4, which is the main electron sink in rumen fermentation. The consumption of dihydrogen by methanogenesis and other dihydrogen-incorporating pathways keeps dihydrogen concentration low and thermodynamically favors re-oxidation of reduced cofactors, and thus allows fermentation to continue (Wolin et al., 1997). The role of the interspecies transfer of dihydrogen in shaping rumen fermentation was illustrated by elegant experiments demonstrating how pure cultures of rumen microorganisms stopped or decreased producing intermediates of rumen fermentation such as dihydrogen itself, formate, and ethanol, as final fermentation products, when co-cultivated with methanogens or other hydrogenotrophs (e.g., Marvin-Sikkema et al., 1990). Close proximity between hydrogenogens and hydrogenotrophs favors the kinetics of dihydrogen transfer and its rapid utilization, as in microbial biofilms (Leng, 2014) and in the protozoal-methanogen symbiosis (Newbold et al, 2015).

Methane is typically the most important, but not the only electron sink in rumen fermentation. Propionate formation from carbohydrates via the randomizing and non-randomizing pathways results in a net uptake of metabolic hydrogen. Butyrate formation from carbohydrates releases metabolic hydrogen, although there are two reactions incorporating metabolic hydrogen in the conversion of acetyl-CoA to butyrate, the reductions of acetoacetyl-CoA to β-hydroxybutyryl-CoA and of crotonyl-CoA to butyryl-CoA (Ungerfeld and Hackmann, 2020). Microbial biomass is more reduced than fermented substrate and constitutes another electron sink. Mineral electron acceptors such as nitrate and sulfate thermodynamically outcompete methanogenesis, although their availability with most diets limits metabolic hydrogen incorporation in their reduction (Ungerfeld, 2020). Reductive acetogenesis, the reduction of CO_2 with dihydrogen to acetate and water, was estimated to be thermodynamically unfeasible in the rumen (Ungerfeld and Kohn, 2006), although more recent studies found it to be a minor electron sink (Raju, 2016). The presence of genes (Denman et al., 2015) and transcripts (Greening et al., 2019) of hydrogenases involved in reductive acetogenesis has also been reported to occur in the rumen.

Most rumen CH_4 is produced through the reduction of CO_2 with dihydrogen (Hungate, 1967), with formate being the second electron donor in importance (Hungate et al., 1970). Formate must be oxidized to CO_2 and dihydrogen by methanogens or bacteria before dihydrogen released from formate oxidation serves as an electron donor for methanogenesis (Thauer et al., 2008). Apart from hydrogenotrophic methanogenesis, methylotrophic methanogenesis utilizes as substrates methanol, methylamines and methylated sulfur compounds, which can accumulate in the rumen following the ingestion of some diets, e.g., those containing pectin (Söllinger et al., 2018).

The production of acetate and to a lesser extent, butyrate, result in the net release of metabolic hydrogen and resulting formation of dihydrogen. Therefore, acetate production is associated with methanogenesis. The replacement of roughages with concentrates typically decreases CH_4 formed per unit of organic matter (OM) fermented (not necessarily the total amount of CH_4 produced, as intake of rumen fermented OM often increases when feeding concentrates), and shifts rumen fermentation from acetate to propionate. The mechanism explaining this fermentation shift has been proposed by Janssen (2010) to relate to methanogens actual and theoretical maximal growth rates, according to the Monod function of microbial growth, which relates actual and theoretical maximal microbial growth rate to concentration of the most limiting substrate for microorganisms, which is dihydrogen for most rumen methanogens, which are hydrogenotrophs. As concentrates form an increasing part of a ruminant diet, passage rate increases, resulting in increased growth rate of methanogens that are not washed out and continue to produce CH_4. Greater growth rate in turn results in elevated dihydrogen concentration according to the Monod function. Similarly, feeding concentrates generally results in rapid fermentation and rumen pH decreases, resulting in decreased theoretical maximal growth rates of methanogens. According to the Monod function, this results again in an increase in the concentration of dihydrogen, which in turn thermodynamically inhibits acetate and favors propionate production (Janssen, 2010).

Likewise, when methanogenesis is inhibited by chemical compounds, methanogens maximal growth rate decreases and dihydrogen accumulates (Janssen, 2010). In this regard, it is important to consider that inhibition of methanogenesis is not an isolated intervention in rumen fermentation and will have profound consequences on the flows of metabolic hydrogen. Therefore, when strategies to mitigate CH_4 emissions through the use of chemical inhibitors are considered, inhibiting CH_4 production should not be viewed as the sole objective of the intervention, and the redirection of metabolic hydrogen towards pathways that could benefit the nutrition of the host ruminant animal might be sought. For example, depending on the type of physicochemical control, it may be possible to incorporate part of dihydrogen typically accumulating when methanogenesis is inhibited to VFA production through the addition of electron acceptors intermediate of VFA formation or particular microbial additives (Ungerfeld, 2020).

1.2.2 Manure

Anaerobic digestion (AD) of animal manure and other organic wastes to CO_2 and CH_4 also involves complex metabolic interactions between microbial groups. Whilst the physicochemical principles controlling both systems are the same, conditions such temperature, fractional outflow rates and types of substrates differ, resulting in some differences. Anaerobic digestion starts with the hydrolysis of complex carbohydrates such as cellulose and hemicellulose to monosaccharides (Figure 2). Monosaccharides are then fermented to VFA and alcohols, which are subsequently oxidized to acetate, CO_2 and dihydrogen. Finally, acetate and methyl-containing one-carbon compounds are reduced to CH_4 by acetoclastic and methylotrophic methanogens, respectively, and CO_2 is reduced with dihydrogen or formate to CH_4 by hydrogenotrophic methanogens. Acetate is also oxidized to CO_2 and dihydrogen, which serve as substrates for hydrogenotrophic methanogenesis. If present, sulfate and nitrate also serve as electron acceptors (Alvarado et al., 2014; Ferry, 2015) and thermodynamically outcompete methanogenesis if they are present at high concentration.

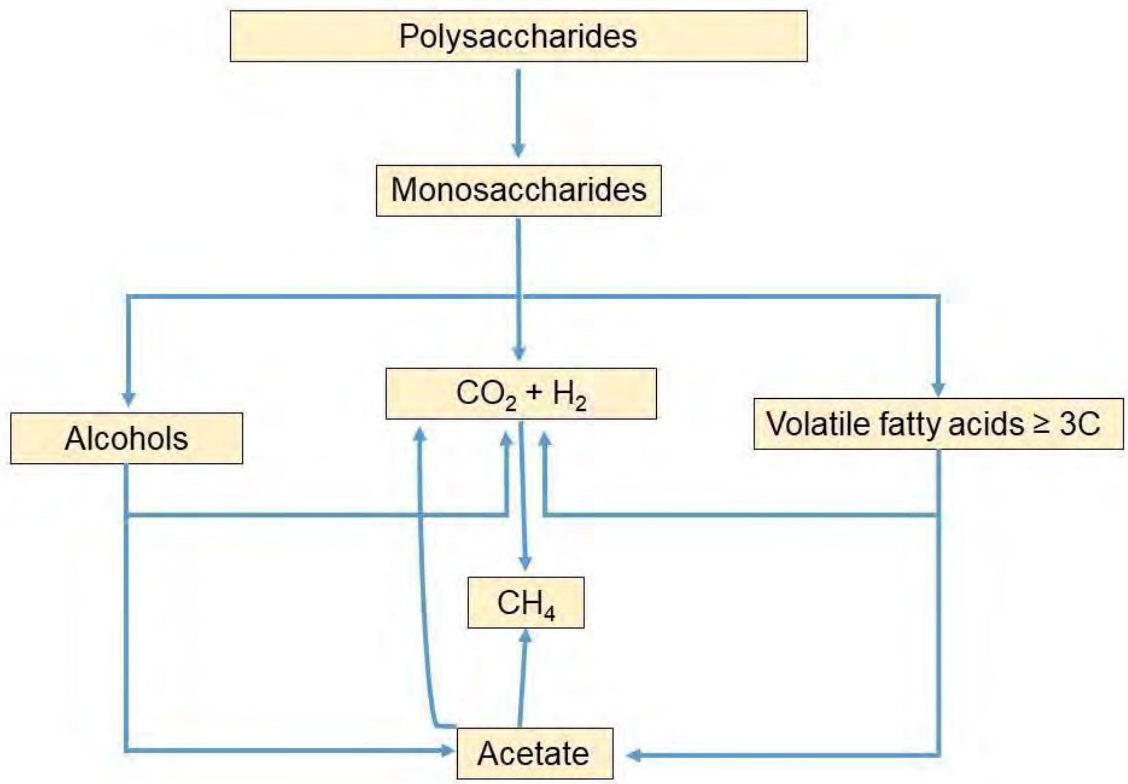

Figure 2. Simplified scheme of main pathways of anaerobic digestion.

Hydrolysis of carbohydrates is a relatively slow process carried out by a very diverse group of bacteria. Bacteria fermenting hydrolyzed monomers predominantly belong to genera *Clostridium*, *Eubacterium* and *Bacteroides* in biodigesters fed with cattle manure (Alvarado et al., 2014). Biochemical reactions in anaerobic degradation are close to thermodynamic equilibrium, and syntrophy is crucial to keep concentrations of reaction products low and processes thermodynamically feasible (Schink, 1997). Imbalance between fermentation and syntrophy can result in increased concentration of VFA and acidification, which is inhibitory to fermentation. For the oxidation of VFA longer than two carbons to acetate to be thermodynamically feasible, concentration of dihydrogen has to be kept very low, for which functional methanogenesis is necessary (Schink, 1997; Ferry, 2011; Alvarado et al., 2014). Low concentration of dihydrogen (as well as pH of less than 7 and high temperatures) is also necessary for bacterial homoacetogens to dissociate acetate into CO_2 and dihydrogen, instead of conducting the reverse process, reductive acetogenesis (Thauer et al., 2008).

The stability of anaerobic digestion is therefore sensitive to its last step, methanogenesis. Methanogens are less diverse than other microbial groups and highly specialized. Methanogen orders Methanobacteriales, Methanomicrobiales and Methanosarcinales are found in anaerobic digesters. Dihydrogen is used as an electron donor by Methanobacteriales and Methanomicrobiales, along with formate, ethanol and isopropanol in some species. Except for the genus *Methanosphaera* (Methanobacteriales), Methanobacteriales and Methanomicrobiales cannot use acetate as substrate for methanogenesis. Methanosarcinales can also use methanol, methylamines and other methylated compounds, with the family Methanotrichaceae (formerly Methanosaetaceae) including acetoclastic methanogens (Alvarado et al., 2014; Conrad, 2020a).

The different one-carbon reduction pathways in methanogenesis from different substrates have their last step in common, the reduction of methyl-coenzyme M to CH_4. In case of acetoclastic methanogenesis, the methyl group in acetyl-CoA is transferred to coenzyme M by methyltetrahydromethanopterin or methyltetrahydrosarcinapterin. The carbonyl group in acetyl-CoA donates via coenzyme B the electron pair necessary for demethylating methyl-coenzyme M and producing CH_4 (Ferry, 1999, 2015).

Generation of ATP in methanogenesis is coupled to transmembrane electrochemical gradients. Methanogens possessing cytochromes can generate more ATP per mole of CH_4 produced than those which do not. However, methanogens with cytochromes have a greater hydrogen threshold and they cannot grow at very low hydrogen concentration (Thauer et al., 2008). Synthropic methanogenic fermentation of VFA to CH_4 is associated with a Gibbs energy value of close to zero, which allows for little ATP generated for anabolic processes and very slow microbial growth rates (Schink, 1997). This explains why acetate and long chain VFA are not metabolized to CH_4 in the rumen, as turnover rates are much faster in the rumen than is the case for anaerobic digesters, where rumen organisms need to generate ATP at greater yield and achieve a faster rate of growth in order to match rumen outflow rates.

1.2.3 Soil

Rice is a major crop for the human population globally, with an increasing demand. Understanding the mechanisms controlling CH_4 production and oxidation in rice soils is thus important to design CH_4 mitigation strategies for the cultivation of this crop (Liesack et al., 2000). Knowledge of the control of major flows of carbon and metabolic hydrogen can facilitate the design of more appropriate interventions and cultivation practices to mitigate the emissions of CH_4 from soils.

The availability of oxygen in the soil is greatly affected by the degree of soil water content or level of saturation. Methane production in soils is affected by the availability of oxygen and other electron acceptors, carbon substrates, water, redox potential, and pH. This section will focus mainly on rice field soils, which are seasonally flooded, alternating oxidizing and reducing conditions. Rice paddies are estimated to contribute 5 percent of total anthropogenic CH_4 emissions (Knief, 2019). Similar to other anaerobic environments in which CH_4 is a predominant electron sink, anaerobic degradation in soils is conducted by a complex microbial community of fermenting bacteria and methanogenic archaea (Conrad, 2020b); apart from methanogenic archaea, soil fungi have also been reported to produce CH_4 from methionine metabolism (Knief, 2019). In addition, as in other anaerobic environments, degradation of polymers, mainly polysaccharides such as cellulose and hemicellulose, is the first step to release fermentable monomers. Rice straw is ploughed under the soil after harvest and degradation of polysaccharides starts (Liesack et al., 2000). Between 80 and 90 percent of rice straw is degraded within the first growth season (Conrad, 2020a). The amount of methane emitted from rice straw – either after soil incorporation or from open field burning -- will depend on the type of management (see below). Moreover, OM provided by the roots of rice plants is always the primary carbon source of CH_4 produced in rice field soil (Kimura et al. 2004).

After flooding of rice paddies, oxygen is rapidly consumed by aerobic bacteria and abiotic chemical reactions. Immediately after flooding, a high concentration of the oxidized forms of inorganic oxidants can maintain reductants dihydrogen and acetate at too low concentration for methanogenesis to be thermodynamically feasible (Conrad, 2020b). Organic matter is sequentially oxidized by available electron acceptors based on their redox potential: nitrate > manganese oxide > ferric iron > sulfate > CO_2. Differences in redox potential of each electron acceptor, including oxygen at oxic interphases, give rise to microscale spatial-temporal chemical gradients of aerobic bacteria, nitrate reducers, manganese reducers, iron reducers, sulfate reducers, and fermenting bacteria and methanogens. Nitrate is formed in the oxic parts of the rice paddy, such as water and the water:soil interphase, from ammonium released from urea added as a fertilizer. Nitrate can be reduced to dinitrogen, nitrite or ammonia. In rice soils, iron content is usually high enough to prevent the accumulation of hydrogen sulfide (Liesack et al., 2000). Importantly, reduced inorganic ions are re-oxidized when the soil is aerated, or if a strong inorganic electron acceptor is added to the soil (e.g., addition of nitrate will regenerate ferric iron and sulfate; Conrad, 2020b). Also, re-oxidation of reduced electron acceptors occurs at the water: soil and the soil: rhizosphere interphases. The depth and the concentration of oxygen in these oxic interphases increases during the day with photosynthesis (Liesack et al., 2000).

Once all of the inorganic electron acceptors are reduced to such an extent that each process reaches thermodynamic equilibrium, anaerobic degradation to CO_2 and CH_4 proceeds through fermentation, and acetoclastic and hydrogenotrophic methanogenesis (Figure 2 and Figure 3). Lignin and xylans can also contribute methanol, which can serve as a minor substrate for methanogenesis, particularly considering that lignin anaerobic degradation is slow and incomplete (Benner et al., 1984). Degradation of rice straw releases phenylacetate and phenylpropionate as minor products, which are degraded exclusively via hydrogenotrophic methanogenesis, as they are metabolized to benzoate, CO_2 and dihydrogen, but not acetate. As degradation of rice straw progresses, the proportion of CH_4 produced via hydrogenotrophic methanogenesis increases and acetoclastic methanogenesis decreases, as more recalcitrant OM is degraded to CO_2 and dihydrogen with no acetate being produced (Liesack et al., 2000; Conrad, 1999, 2020, a, b).

As in other anaerobic environments, dihydrogen turnover is very high. Low concentration of dihydrogen approaching the thermodynamic threshold of methanogenesis allows the thermodynamic feasibility of dihydrogen-releasing reactions (Conrad, 1999). Microorganisms involved in the different phases of the anaerobic degradation oxidation of OM must accommodate their activities to the thermodynamic feasibility of processes, but they are important to greatly accelerate the kinetics of thermodynamically feasible processes. Theoretically, anaerobic degradation of cellulose would result in equimolar amounts of CH_4 and CO_2 and more than two-thirds of CH_4 formed from acetate and less than one third from hydrogenotrophic methanogenesis. However, the products of degradation can be modified by acetate oxidation followed by hydrogenotrophic methanogenesis and acetate oxidation by soil organic compounds, and by reductive acetogenesis. Other organic and inorganic electron donors, acceptors and carriers present also further influence the stoichiometry of final products of anaerobic digestion in soils (Conrad, 1999, 2020a, b).

Temperature influences the predominant substrates for methanogenesis, with a relative increase in acetate and a decrease in dihydrogen at low temperatures (Chin and Conrad, 1995; Conrad, 2020a). Furthermore, at low temperatures, acetate production increases relative to CO_2 and dihydrogen as a consequence of reductive acetogenesis is favored over hydrogenotrophic methanogenesis. Hydrogenotrophic methanogens begin to be outcompeted by reductive acetogens at low temperatures because of greater fluidity of bacterial ester lipids than archaeal ether lipids at low temperatures, and because dihydrogen production and acetate oxidation become thermodynamically less favorable (Conrad, 2020a). Methanogenesis is also negatively affected by decreases in soil pH, although the ratio of hydrogenotrophic to acetoclastic methanogenesis is not affected (Conrad, 2020a).

Soil methanogenesis is strongly inhibited by oxygen, however, soil methanogens have evolved to adapt to succeeding events of flooding and desiccation, and can tolerate the presence of oxygen, although they do not form spores or cysts. Methanogens have generally been found to decline, but not disappear, with soil desiccation (Conrad, 2020b). Methanogenesis has been reported to occur even in anoxic microniches in oxic soils. Methanogens of orders Methanocellales, Methanomicrobiales and Methanosarcinales have been reported to carry genes involved in resistance to oxygen (Knief, 2019). In this regard, soil methanogens may differ from methanogens in other environments such as the rumen or anaerobic digesters, which live under more stable, anoxic conditions.

Natural wetlands, landfills and rice paddies all contribute to CH_4 emissions to the atmosphere, but bacterial methanotrophs in well aerated soils oxidize about 4 percent of atmospheric CH_4. The activities of CH_4-cycling microorganisms determine the net production or consumption of CH_4 in soils (Knief, 2019). In most dry soils, however, the atmospheric concentration of CH_4 is too low to induce aerobic CH_4 oxidizing activity (Conrad, 2020b). However, aerobic methanotrophic bacteria situated in oxic/anoxic interphases can oxidize up to 80 percent of soil-produced CH_4 before it is released to the atmosphere (Knief, 2019). The presence of oxygen in the rhizosphere of rice or other aquatic plants enables CH_4 oxidation to occur, especially during daytime when the extension of the oxic interphases increases due to photosynthesis. Notwithstanding this, much CH_4 escapes to the atmosphere as bubbles and especially through the plants aerenchyma (Liesack et al., 2000).

Anaerobic CH_4 oxidation is conducted by both bacteria and archaea and can also eliminate substantial amounts of CH_4 formed in some soils before it gets to the atmosphere. Methane oxidation coupled to sulfate reduction is important in marine sediments, but may be important in terrestrial soils subjected to cycles of sulfur reduction and oxidation. Nitrate and nitrite, ferric iron and manganese can also act as electron acceptors in CH_4 oxidation (Knief, 2019).

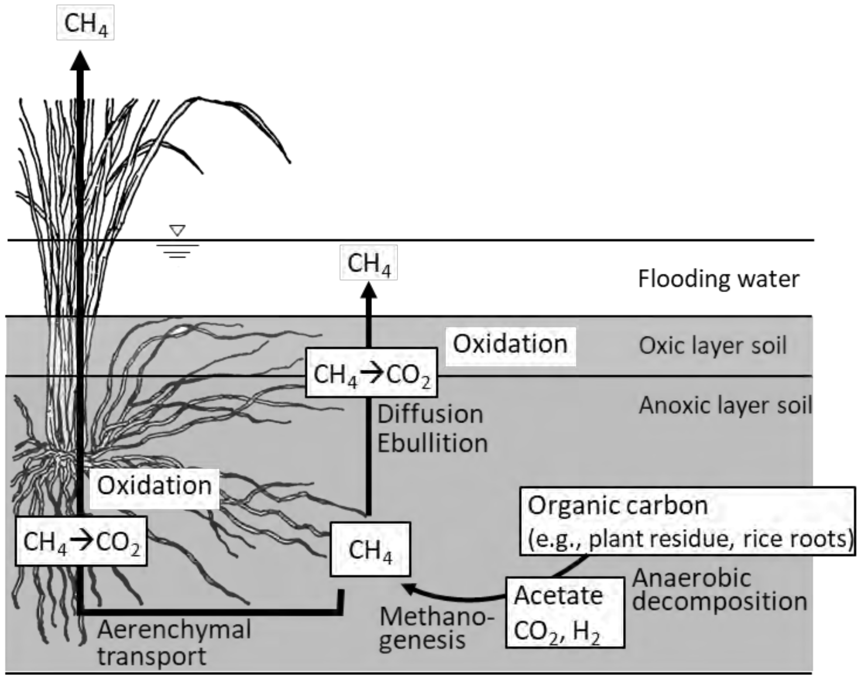

Figure 3. Methane dynamics in flooded rice soil.

1.3 Methane emissions during storage of manure

Methane emissions from manure management are an important contributor to the GHG budget for the farms, and the agricultural sector (Cluett et al., 2020). Manure management from livestock (ruminants and non-ruminants) has been estimated globally at 2.52 Gt CO_2eq, the main source of emission being manure storage and, particularly liquid manure storage where anaerobic conditions are maintained. Ruminant manure contributed 2.3 Gt CO_2eq and swine 0.2 Gt CO_2eq; total CH_4 production from livestock manure was estimated at 17.5 million tonnes per year, in comparison to 85.6 million tonnes per year of enteric methane (Steinfeld et al., 2006). In the US, the Environmental Protection Agency (EPA, 2006) estimated the CH_4 emission from manure at between 470 and 523 Mt CO_2eq per year. For EU-28 manure CH_4 emissions represent 44 Mt CO_2eq (EuroStat, 2018).

Methane is produced under anaerobic condition by archaea mainly during storage conditions, using the OM present in the animal excreta. Hence, the production of CH_4 from manure mainly occurs in slurry and liquid manure. Chianese et al. (2009) indicate an average CH_4 emission from covered slurry of 6.5 kg/m^3 per year, while this is reduced in uncovered slurry to 5.4 kg/m^3, while the emissions from stacked manure is estimated at 2.3 kg/m^3 with emissions varying with both ambient temperature and time in storage (Hristov et al., 2013).

The magnitude of CH_4 emissions generated during storage of manure are principally dependent on duration of manure storage, storage system used, temperature and manure composition (Dennehi et al, 2017; Philippe et al, 2015). For instance, Petersen et al. (2013) found that the cumulative CH_4 emissions from stored pig manure in summer were over 100 times greater than those in winter. Emissions of N_2O are typically from <1 percent to 4.3 percent of the total N in stored cattle and pig farmyard manure heaps, but emissions as high as 9.8 percent have been reported (Chadwick et al., 2011).

1.4 Methane emissions following application of manure

While the preponderance of manure derived CH_4 emissions emanate from stored material, there has been some interest in quantifying emissions following application to soil. Bourdin et al (2014) investigated the impact of slurry DM content, application technique and timing of application on the overall GHG balance from cattle slurry applied to grassland soils. The treatments on plots were a control, calcium ammonium nitrate and cattle slurry derived from either grass-based or maize-based diets and with varying DM contents, applied by mimicking trailing shoe and splash plate application. The DM contents were varied in the study by mixing different ratios of faeces and urine. While, ammonia (NH_3) volatilisation losses were substantially increased on slurry spread plots, cumulative direct N_2O emissions, and corresponding emission factors, were significantly higher when applying calcium ammonium nitrate. In terms of GHG field balance, the potential decrease in indirect N_2O emissions, calculated from a reduction of NH_3 volatilisation losses using trailing shoe as opposed to splash plate, could be easily offset by an increase in direct N_2O emissions and ecosystem respiration. Switching from summer to spring application was much more efficient for mitigation of both NH_3 and GHG emissions, due to favourable soil and climatic factors, which enhanced crop growth. The authors concluded that any potential trade-off between NH_3 and N_2O emissions was cancelled, leading to an overall positive effect on reactive nitrogen losses and offering agronomic benefits to farmers. However, a recent analysis of a large dataset of CH_4 fluxes from agricultural sites across Ireland and the UK indicate that these soils are small net emitters of CH_4 rather than sinks, with fluxes occurring particularly following animal manure applications (Cowan et al., 2020).

There have been many reported studies of N_2O emissions following manure spreading. Emission factors (i.e., cumulative N_2O -N loss as a proportion of total N applied in the manure) can range from <0.1 to 3 percent. Higher emissions (7.3–13.9 percent) have been measured during land application of pig slurry (Velthof et al., 2003). The range in N_2O emission factors following slurry and solid manure applications reflects differences in soil type, soil conditions (i.e., temperature, water filled pore space), manure composition (i.e., NH_4 +-N, C content and form) and measurement period.

Other investigators have examined the potential CH_4 oxidizing capacity of soils, particularly in relation to removing CH_4 emissions from manure and digestate slurry. The magnitude of the effect depends on the chemical and physical properties of the soil that frame the living conditions of the methanotrophic bacteria as well as on the time and intensity of exposition to CH_4 (Oonk et al., 2015).

1.5 Trade-off between GHG and other gaseous emissions

In their review of CH_4 emission factors, O'Brien and Shalloo (2016) stated that several countries recognise that CH_4 emissions from cattle and livestock manure are linked to other GHG emissions from manure, for example N_2O. For consistency, some countries use a comprehensive model that simultaneously quantifies GHG and NH_3 emissions from livestock. Reduced NH_3 losses from manure spreading are likely to increase N availability in agricultural soils and this may, in turn, affect the production and release of N_2O (Brink et al., 2001). As N_2O is a much more potent GHG (IPCC, 2007), this could be regarded as pollution swapping (Stevens and Quinton, 2009) where attempts to abate the release of one ecologically harmful gas result in an increase in the emissions of another.

An integrated assessment of the effects of mitigation measures on NH_3, CH_4 and (direct and indirect) N_2O emissions is warranted from the whole manure management chain. In their meta-analysis, Hou et al. (2015) found that lowering the CP content of feed and acidifying slurry are strategies that consistently reduce NH_3 and GHG emissions in the whole chain. The potential for a trade-off between NH_3 and N_2O emissions following application of cattle slurry was cancelled where climatic and soil conditions favoured good nutrient uptake and crop growth, leading to an overall positive effect on reactive N losses and offering agronomic benefits to farmers (Boudin et al., 2014).

1.6 Spatio-temporal variation in methane emissions

Reliable and high-resolution spatio-temporal inventories of CH_4 emission from livestock production systems are required for accurate and equitable national inventory preparation. For example, the balance between enteric and manure emissions will be influenced by season (particularly for pastoral based production systems), the prevalent livestock species, and vectoral zonation of the production systems (for both pastoral and mixed systems). As a consequence of temporal and spatial variability of emissions from livestock housing and manure management, measurements and monitoring need to occur over longer periods of time that cover both the daily and seasonal variations in emissions in order to accurately reflect annual emissions (NASEM, 2018). Herrero et al. (2015) highlighted the problems associated with quantifying emissions from livestock production. The authors emphasise the large spatial variations of emissions due to variation in soil type, climatic parameters and water conditions, or indeed different soil fertilisation and manure management practices and composition. Detailed discussion on quantification methods is given in section 2. In addition, the conditions for manure are far less well controlled than with enteric emissions, with the physiological regulatory mechanisms of the ruminant in place.

1.7 Contribution of human food and animal feed waste to methane emissions

Food wastage is a global issue intrinsically linked with the growing challenges of food security, resource and environmental sustainability, and climate change. In developed economies, the largest food wastage stream occurs in the consumption stage at the end of the food chain. Du et al. (2018) highlight that historically, livestock animals functioned as bio-processors, converting human-inedible materials into nutritious meat, eggs, and milk. The authors suggest that contemporary treatment technologies can assist in the conversion of food into safe, nutritious, and value-added feed products, instead of wasting it, and that recovering unconsumed food for animal feeding is a viable solution that simultaneously addresses the reduction of food waste, food security, resource conservation, and pollution and climate-change mitigation.

Reduction and sustainable management of food waste is a fundamental tenet of the circular bioeconomy concept. Globally, around 1.3×10^9 tonnes of food waste are disposed of in landfills (Hao et al., 2015). About 13.8 percent of food produced in 2016 was lost from farm to fork, excluding the retail and household stages of the global food supply chain (FAO, 2019). In addition, global annual generation of food loss and waste is estimated at 4.4 GT CO_2eq, or approximately 8 percent of total anthropogenic GHG emissions (Tiffany et al., 2020). The recycling of food waste that could not be reduced by a more efficient food supply chain and informed consumers' behavior provides employment, reduces GHG emissions, decreases disposal costs, mitigates the negative environmental impacts and supports sustainable waste management practices under the bio-circular economy concept. Compared with traditional disposal methods (i.e., landfilling, incineration, and composting), anaerobic digestion followed by CH_4 usage as biogas is a promising technology for food waste management, but has not yet been fully applied due to a number of technical and social challenges (Xu et al., 2018). Indeed, in the U.S., less than 2 percent of food waste is anaerobically digested. The management of food waste by biological processing is a more environmentally sustainable approach than thermo-chemical conversion or landfilling. The composition and CH_4 generating potential of some common food waste streams has been summarised by Xu et al. (2018). However, the composition, physico-chemical and biological characteristics of food waste can affect the overall biological process in terms of product yield and degradation rate. The pre-treatment (i.e., grinding or drying) of food waste in advance of anaerobic digestion has been proposed in order to overcome this major bottle-neck in the system. Co-digestion of food waste with manure, sewage sludge, and lignocellulosic biomass could be beneficial due to dilution of toxic chemicals, enhanced balance of nutrients, and synergistic effects of microorganisms.

There is a major variation in legislation towards the inclusion of unconsumed human food into the diets of livestock between different countries/continents. This variation goes from juxtaposition between reducing food waste entering landfill versus fears over health implications for inclusion in livestock diets.

1.8 Anaerobic digestion

Methane gas has been identified as a promising alternative in the global effort to replace fossil fuels with more environmentally sustainable and renewable energy sources. This has led to a rapid increase in the construction of biogas plants worldwide. In addition, the potential of anaerobic digestion (AD) to mitigate GHG emissions has gained attention. Under the EU Renewable Energy Directive bioenergy pathways must meet minimum GHG emissions savings thresholds in order to count towards renewable targets and to be eligible for public support (Giuntoli et al., 2017).

The feedstock being used has important implications for the overall sustainability of the AD system. The production of CH_4 through biological processes (biogas) has the advantage of using lignocellulosic agricultural and livestock derived byproducts, which following biologically based processing, are converted to electrical, heat, and power energies through relatively easy-to-manage process in small industrial/agricultural units (Antoni et al., 2007). Digester designs vary widely in size, function and operational parameters and have been reviewed within the context of different production systems by Hristov et al. (2013). While strongly recommending the use of anaerobic manure digesters as a CH_4-mitigation strategy for the agriculture sector, Gerber et al. (2013) caution that careful management is necessary, so that they do not become emitters of CH_4 to the atmosphere. The authors suggest that adoption of this type of technology on farms of all sizes may not be widely applicable and will heavily depend on financial and technical capacity, climatic conditions and availability of alternative sources of energy.

When livestock (i.e., cattle and pig) derived slurries are used in AD, there is generally an improved environmental performance compared to traditional manure management (Vadenbo et al., 2014, Hamelin et al., 2011, 2014). This is largely due to the emissions that are mitigated from traditional manure storage and application (Hamelin et al., 2014). As a result, the use of animal wastes such as manures and slurry in AD is encouraged, with some studies suggesting that policy should prioritise the digestion of manures to maximise GHG mitigation (Styles et al., 2016). While manures have a lower biomethane potential compared to other feedstocks, it has been suggested that focussing on smaller biogas plants with lower energy conversion efficiency may be preferable in terms of waste management, rather than renewable energy generation where there are more efficient alternatives in terms of cost and land requirements, such as wind and solar (Styles et al., 2016).

The methodology applied by the Joint Research Centre (JRC) of the European Commission to calculate the GHG emissions associated with bioenergy pathways under Renewable Energy Directive is a simplified attributional LCA (Giuntoli et al., 2017). According to the JRC, biogas produced from manure can receive emission credits due to emissions avoided from the traditional management of manure, including CH_4 and N_2O, if manure is not stored for too long. Using manure in AD systems is considered an improved agricultural management technique and the avoided emissions from the management of the raw manure are credited to the bioenergy pathway. The value of the credit is equal to -45 g CO_2eq/MJ of manure used (Giuntoli et al., 2017). However, the JRC recognises that the credits are not an intrinsic property of the biogas pathway but the result of a common, although less than optimal, agricultural practice (Giuntoli et al., 2017). Furthermore, it is acknowledged that if gas-tight storage of raw manure becomes a standard practice in agriculture, the credit for manure in the biogas pathway would cease to exist.

Anaerobic digestion utilizes the energetic potential contained in the manure for the production of heat and electricity, reduces N_2O emission during the treatment through a relatively closed system, and results in biogas digestate which makes for a valuable fertilizer that still contains most of the nitrogen (Kreidenweis et al., 2021).

1.8.1 Leakage of methane from anaerobic digestion facilities

Methane from the waste sector accounts for around 3 percent of global anthropogenic GHG emissions (Bogner et al., 2008), and about 12 percent of total global anthropogenic CH_4 emissions for the 2008 – 2017 period. Bakkaloglu et al. (2021) suggest that CH_4 emissions from biogas generation may be between 0.4 and 3.8 percent, and could account for 1.9 percent of the total CH_4 emissions in the UK excluding sewage sludge biogas plants.

Scheutz and Fredenslund (2019) recently measured total losses of CH_4 from 23 biogas plants by applying a tracer gas dispersion method across plants that varied in size, substrates used and biogas utilisation. Methane emission rates varied between 0.4 and 14.9 percent of biogas production with an average loss of 4.6 percent. Methane losses from the larger biogas plants were generally lower compared to those from the smaller facilities. In general, CH_4 losses were higher from wastewater treatment biogas plants

(7.5 percent in average) in comparison to agricultural biogas plants (2.4 percent in average). The authors concluded that fugitive CH_4 loss may constitute the largest negative environmental impact on the carbon footprint of biogas production.

2. Methane sinks

Global CH_4 emissions are largely offset by the atmospheric and soil CH_4 sinks. The atmospheric sink occurs though chemical degradation of CH_4 by hydroxyl (OH) and chlorine (Cl) radicals in the troposphere and stratosphere (IPCC, 2007) and is responsible for 90-96 percent of global CH_4 sink (Wuebbles and Hayhoe, 2002; Shukla et al., 2013; Saunois et al., 2019) equivalent to 550 Tg/year. The soil accounts for about 4-10 percent of the CH_4 degraded (Born et al., 1990; Duxbury and Mosier, 1993; Saunois et al., 2019). The ocean acts as a small CH_4 sink for atmospheric CH_4 of about 4 Tg/year (Shukla et al., 2013).

2.1 Soil methane sink

The most important soil sink for CH_4 is upland soil, accounting for 6 percent of the total CH_4 consumption, equivalent to 30 Tg/year (Houghton et al., 2001; Knief et al., 2003; Tian et al., 2016) with an uncertainty of 11 to 49 Tg/year (Tian et al., 2016; Saunois et al., 2019). The bacterial group responsible for the CH_4 sink activity in the soils are specialized members of eubacteria, called methanotrophs and ammonium oxidizing bacteria (Shukla et al., 2013). The kinetics of this process is an aerobic reaction with the enzyme CH_4 monooxygenase in which CH_4 is oxidized as an energy and carbon source (Bender and Conrad, 1992; Roslev et al., 1997).

Among upland soils, forest soils are the most efficient CH_4 sink in both temperate and tropical regions (Henckel et al., 2000; Steinkamp et al., 2001, Singh et al., 1997) with global annual average uptake rate of 5.7, 3.3, and 2.64 kg CH_4/ha for temperate, tropical, and boreal forest biomes, respectively (Dutaur and Verchot, 2007). Grasslands, shrub lands, and steppe/savanna biomes have average annual uptake of 2.32, 2.25, and 1.49 kg CH_4/ha (Dutaur and Verchot, 2007). Cropland and desert have the lowest uptake rate with annual mean rate of 1.23 and 1.1 kg CH_4/ha, respectively (Dutaur and Verchot, 2007). Methane sink estimate by biome varies considerably depending on the estimation model (Saunois et al., 2019; Ito and Inatomi, 2012) but, owing to the combination of area and oxidation rate, forests represent the largest CH_4 soil sink followed by grazing lands (Murguia-Flores et al., 2018; Yu et al., 2017). Within grazing lands, the dry grazing lands in both temperate and tropical climates have about 2 to 3 times the uptake rate per hectare as moist grazing lands (Yu et al., 2017).

2.1.1 Factors affecting the soil methane sink capacity

The CH_4 oxidation potential and methanotrophic community size and structure can be affected by many environmental and anthropogenic factors (Boeckx et al., 1997, Dunfield, 2007). Environmental factors affecting the soil CH_4 sink can be divided into two types: those that have purely physical effects (primarily on diffusion), and those that influence methanotroph populations and activity. Water content has both physical and microbiological effects (Dunfield, 2007) as dry soil increases gas diffusion and increases CH_4 consumption but insufficient soil moisture reduces methanotroph activity. Changing climate and climatic factors, particularly seasonal precipitation variations in semiarid regions and drylands, also affect soil methane sink capacity, directly and indirectly, e.g., sensitive, vulnerable regions to changing climate and climatic factors (particularly to seasonal precipitation variations in semiarid regions and drylands). Soil OM also increases consumption through both pathways – pore space and pore size increase with increasing soil OM while soil OM carbon and nutrients increases methanotroph numbers (Gatica et al., 2020; Tang et al., 2019b). Physical factors are temperature (weak because of competing effects on methanotroph activity, soil water content, and gas diffusion rates), texture (uptake increases as sand increases), and bulk density (uptake increases as bulk density decreases) (Shukla et al., 2013). Hence land degradation that reduces soil OM and increases soil bulk density reduces the soil sink capacity while restoration increases the sink – however, the increase in sink capacity with restoration happens slower than the loss of sink capacity with degradation (Wu et al., 2020). Inorganic nitrogen addition depresses uptake because ammonia competes for CH_4 monooxygenase enzyme active sites and nitrite produced during nitrification and/or denitrification is toxic to methanotrophs (Dunfield, 2007). When nitrogen is added with organic amendments such as manure, the effect of nitrogen on CH_4 uptake is less. Pesticides and herbicides, metal pollution, and land use patterns can also have important effects over CH_4 oxidation and methanotrophic community (Boeckx et al, 1998; Priemé and Ekelund, 2001, Shukla et al., 2013).

2.1.2 Land management effects on soil methane sink

I. Pasture

A global meta-analysis showed that the addition of nitrogen to pastures reduced the soil sink capacity by 11.4 percent but application of phosphorus with the nitrogen roughly halved that reduction (Zhang et al., 2020). The livestock stocking rate has an important effect on CH_4 uptake. Globally, heavy grazing intensity reduces the sink capacity by 12 percent compared to moderate or light grazing due to the effect of heavy grazing pressure reducing plant productivity and soil OM while increasing soil bulk density from hoof action (Tang et al., 2019b). Adaptive multi-paddock grazing allows vegetation to recover without grazing pressure and this may explain why the potential CH_4 sink capacity of soil was found to be 50 percent higher under adaptive multi-paddock grazing than more continuous grazing (Shrestha et al., 2020). For low productivity grazing lands with low livestock stocking rates the soil sink can be an important part of the grazing system CH_4 budget. An empirical model for the steppes in China showed that the pasture CH_4 sink was equal to 50 percent of CH_4 from enteric fermentation and manure from grazing sheep at a stocking rate of 1 sheep/ha per year and 20 percent at a stocking rate of 4 sheep/ha per year (Tang et al., 2019a).

II. Forestry

Tree species composition in the system is a factor that affects the soil CH_4 sink (Dunfield, 2007) because soils under different forest compositions support different CH_4 uptake rates (Borken et al., 2003, Reay et al., 2005). Tree species effects are probably mediated through soil chemistry, moisture and microbiology, but the precise mechanisms are complex (Dunfield, 2007). Uptake rates are higher in primary forest than secondary forest or plantations and increase with time since land-use change (Gatica et al., 2020).

III. Cropland

Cropland typically has nitrogen addition so that reduces its CH_4 sink capacity. Otherwise, the CH_4 sink on upland cropland does not appear to be strongly affected by management as there is no consistent effect of tillage system (Venterea et al., 2005; Jacinthe and Lal, 2005; Kessavalou et al., 1998), biochar addition (Cong et al., 2018), or cover crops (Singh et al., 2020) on CH_4 uptake.

IV. Agroforestry

Because soil under trees typically has greater CH_4 uptake rate, the treed portion of the land has higher sink than the untreed cropland (Amadi et al., 2016). In an experiment in Colombia, an intensive silvopastoral system acted as a CH_4 sink with an accumulated flow of -1.01 mg/m^2 per hour compared with an improved pasture that had emissions equivalent to 46.7 mg/m^2 per hour during the same period (Rivera et al., 2019). In addition, the carbon sequestered in the shrubs and/or trees of silvopastoral systems provides an opportunity to offset some (Monjardino et al., 2010) or all (Torres et al., 2017) of the global warming effect of all livestock related CH_4 emissions.

Part 2. QUANTIFICATION OF METHANE EMISSIONS

3. Measurement

3.1 Animal based techniques

There are many different techniques and methodologies used to measure CH_4 emissions from ruminants (Hammond et al., 2016), including gas exchange measurements (e.g., respiration chambers, head or face masks, or spot sampling), tracer gas, and open-path laser technologies (Hill et al., 2016; Lassey, 2007; Storm et al., 2012). Table 1 lists critical aspects of different techniques. These techniques have specific requirements (i.e., methodologies) and assumptions that may limit their application outside of their intended purpose and may exacerbate CH_4 measurements if the conditions are not consistent with original assumptions. For instance, some techniques are more suitable for grazing animals (e.g., sulfur hexafluoride tracer gas, SF_6), while others can mainly be used for confined animals (e.g., open-path laser). Tracer release rate or air flow rate are most critical measurements to derive a CH_4 emissions rate.

Table 1. Characteristics of different techniques to measure methane[1].

Techniques	Cost	Level	Environment	Applications	Advantages	Disadvantages
Respiration and accumulation chambers	Generally high	Animal/ Manure		Research	Highly accurate, controlled environment; information about individual animals; include emissions from hindgut fermentation	Results are different from free-range animals; configurations still vary from one research group to another; an animal adaptation period is required; every 2–3 h accumulation chambers must release CO_2 that builds up. Needs calibration.
Hood and/or headbox systems	Moderate to high	Animal	Grazing/pasture, indoors free stall, or tie stall	Research	Portable and less expensive than a chamber; requires less space	Do not measure hindgut emissions; an animal adaptation period is required; some may be designed for grazing situations. Recovery test needed.
Tracers	Moderate	Animal		Research	Accurate; few interferences by other gases; the animal can free-range	Relies on SF_6, which is a greenhouse gas itself; does not completely capture all tracer and, therefore, relies on spot concentration measurements; high contact with animal, which can disrupt normal behavior; highly laborious
Gas sensor capsules	Low	Animal		Research	Compatible with new electronic technologies; relies on small, low-cost sensors; continuous measurements	Information about the relation between concentration and flux (emission); still under development
In vitro techniques	Low	In Vitro		Research and Commercial	High reproducibility but used to rank feeds for methanogenic potential and not for measurements of flux; allows different rumen microbial environments to be evaluated	Outcomes can be different from actual measurements; method relies on donor animals for rumen environment; standardization can be difficult.
Open-path laser	High	Pen/barn /building		Research	Information about many animals; data produced in natural grazing environment	Require expensive and accurate measurement approaches: data processing heavily influenced by microclimatic conditions; loss of data can be high
Unperson aerial/ground vehicles (UAV/UGV, drones)		Paddock/ pasture		Research		High variability and difficulty of air flow measurement
Satellite		Basin/Re gion		Research and Commercial		Only CH_4 concentration measurements
Computer models	Low	Diverse		Research and Commercial	Estimate the distribution of production; not limited to any configuration	It can be different from real scenarios; still rely on input data made from respiration and accumulation chambers measurements as well as tracer methods

LiDAR	Moderate	Pasture	Grazing	Research	Airborne; detect CO_2 and CH_4 concurrently

[1] Based on Hill et al. (2016).

3.1.1 Gas exchange technique

3.1.1.1 Respiration chambers

Respiration chambers have been the gold-standard technique to determine the energy expenditure of individual animals. The indirect calorimetry methodology relies on gas exchange of mainly O_2, CO_2, and CH_4 either using open-circuit chambers that analyze the composition of inflow and outflow air or closed-circuit chambers that analyze the composition of air accumulated over some time (Johnson and Johnson, 1995). A limitation of respiration chambers is that animals may not exhibit normal behaviors, e.g., might decrease their feed consumption; thus, resulting in an under-estimation of actual CH_4 emissions when compared to free-ranging animals under farm conditions (Huhtanen et al., 2019). A number of factors are essential when using this technology for controlled experiments, such as gas recovery, routine maintenance, chamber temperature (<27°C), relative humidity (<90 percent), CO_2 concentration (<0.5 percent), and ventilation rate (250–260 L/min) as suggested by Pinares-Patiño and Waghorn (2014). The utility of respiration chambers is also limited to quantifying gaseous emissions from relatively few animals (4 or less) mainly to account for the emissions from manure when they are accumulated in the barn with and without the animals (Mathot et al., 2016).

Respiration chambers are relatively expensive to build and maintain, but low-cost systems exist (Abdalla et al., 2012; Canul Solis et al., 2017; Hellwing et al., 2012). These systems use the same principles as for open-circuit indirect calorimetry, but use locally available materials for construction and air-conditioning systems are simpler than described for other open-circuit systems. The system is located in the daily environment of the cow (Canul Solis et al., 2017; Hellwing et al., 2012) or sheep (Abdalla et al., 2012). The system may consist of transparent polycarbonate chambers, thermic panels with acrylic windows, or sheep metabolism cages covered with 3-mm transparent polycarbonate walls, with a total volume of 9.97 m^3 and 17 m^3 for cows and 1.9 m^3 for sheep. Flow and gas concentrations may be measured continuously in the outlet or measured in air sampled from the outlet, using an infrared analyzer or gas chromatograph. Average recovery rates range from 99 ± 7 to 104 ± 9 percent.

An even simpler version of the respiratory chamber is the polytunnel that consists of one large inflatable or tent type tunnel made of heavy-duty polyethylene or PVC film in which individual or groups of cattle can be confined for selected periods of time during which the amount of CH_4 they produce is collected and then measured (Goopy et al., 2016). Polytunnels can be placed directly on pastures simulating semi-normal grazing conditions (Murray et al., 2001) or fixed close to the pastures where the daily offer and intake of forages can be measured (Gaviria-Uribe et al., 2020; Molina et al., 2016).

3.1.1.2 Spot sampling

Head-stall, also referred to as Automated Head-chamber system (AHCS) (Hristov et al., 2015) (e.g., GreenFeed Emission Monitoring™ system), and sniffer (e.g., GASMET 4030 system) apparatus are based on spot sampling of eructated and exhaled gases from the animals' mouth and nostrils. Sniffers measure concentration only. Head-stalls usually are programmed to deliver a small amount of feed to lure animals into inserting their head inside a chamber that will collect the gases using an active airflow. Methane emission determined with GreenFeed and empirical regressions developed from respiration chambers had a high correlation (r = 0.958) and low mean bias (12.9 percent of observed mean) for dairy cows (Huhtanen et al., 2019). The adequacy of head-stalls (e.g., GreenFeed) in measuring CH_4 is highly dependable on the daily frequency of the visits by the animal (within-day variation), animal behavior (within-animal variation), trial design and the number of days of data collection (Hammond et al., 2015; Thompson and Rowntree, 2020). Gunter and Bradford (2017) recommended at least 2.4 visits per day for 6.3 days. Hristov et al. (2015) recommended sampling eight times during a 24 hr feeding cycle, staggered in time over three days. Arbre et al. (2016) measured daily values and obtained a repeatability of 70 percent in 17 days and could increase it to 90 percent in 40 days. Coppa et al. (2021) reported a repeatability of 60 percent for a one-week measurement of daily CH_4 and increased it to 78 percent for an eight-week measurement period.

The GreenFeed Emission Monitoring System can be used for large-scale measurements and commercial production conditions of large and small ruminants (Zhao et al., 2020), but different units are needed dependent on animal size. The system is suitable for pasture (i.e., grazing conditions), indoors free-stalls or tie-stalls, but requires animal training. For maximum accuracy, one must perform CO_2 and CH_4 calibrations five times at the beginning and three times at the end of each gas measurement experiment. It is also necessary to perform the CO_2 recovery test at least once (3 releases is about one cylinder of CO_2) before each gas measurement experiment. For continuous applications, one must perform the recovery test once per month (Hristov et al., 2015). Benefits include its lower cost compared to the gas chambers, but similar to sniffers and gas tracers, it does not consider CH_4 emission from hindgut fermentation.

Sniffers are placed near the animal's muzzle at the feed or water troughs, and exhaled air is continuously sampled. Unfortunately, the precision of sniffers seems to be smaller (Bell et al., 2014) than the respiration chambers (Yan et al., 2010) due to high "between-animal variation," likely because it depends on the distance between the sniffers and animal's muzzle; ideally, it should be less than 30 cm (Huhtanen et al., 2015).

Portable accumulation chambers have been used frequently to determine short-term CH_4 emissions in grazing sheep (Goopy et al., 2011). These are bottomless boxes made with plexiglass on the sides and top that are lowered down on animals and sealed (Thompson and Rowntree, 2020). Three sampling ports are on the top of these boxes to follow gas accumulation in time. Comparisons with the respiration chamber measurements have indicated moderate correlations (up to 0.6) for up to 2-h sampling durations (Goopy et al., 2011; Goopy et al., 2015).

Chagunda (2013) evaluated the hand-held Laser Methane Detector (LMD) on-farm. The LMD is an infrared absorption spectroscopy that uses an excitation source and the second harmonic detection of wavelength modulation spectroscopy. This non-invasive and non-contact technique enables the measurement of CH_4 emission from the breath of ruminant animals. Some recovery tests may be needed. Methane has two strong groups of absorption lines, centered at 3.3 micrometers (v3 band) and 7.6 micrometers (v4 band). Most laser-based devices operate at near-infrared wavelengths, limited to below 2.2 micrometers. The most robust absorption band of CH_4 is located at 1.64 to 1.70 micrometers (2v3 band). This corresponds to the single-mode, single-frequency emission wavelength of indium gallium arsenide (InGaAs)-distributed feedback (DFB laser diode). Hand-held CH_4 detection systems are used in other industries and have been described by van Well et al. (2005). Because the instrument measures CH_4 at a range of several meters, it does not disturb animal behavior. The instrument accounts for the thickness of any CH_4 plumes, and the result is expressed as CH_4 concentration. Hence, the LMD enables real-time measurement with a fast response. One study reported the CH_4 measurement with the LMD had a strong agreement with measurements in respiration chambers (r = 0.8) (Chagunda and Yan, 2011). Furthermore, the LMD can segregate the CH_4 concentration from dairy cows performing different physiological activities (e.g., ruminating, feeding, sleeping). Although LMD uses spot sampling of the animal's breath, it is possible to calculate total emissions (g per day). One of the challenges of using LMD is related to the absence of gas sampling. It is necessary to separate the eructation episodes from the normal breath cycle of the animals (exhalation-inhalation cycles). To alleviate this challenge, a threshold value to separate the two events is being tested. Other challenges relate to applying this approach for grazing animals because wind speed and direction, relative air humidity, and atmospheric pressure can have a significant effect on the resultant concentration of CH_4. Wind speed negatively correlated with CH_4 concentration (r = -0.41). An additional limitation is the correct distance the device should be from the animal (Sorg, 2022) to avoid contamination from the neighboring animal. The LMD instrument is relatively novel in ruminant animals, and extensive studies will be required to determine the repeatability of the measurements (Chagunda, 2013) that could be used to develop standard protocols for data measurement and analysis (Sorg, 2022). However, such techniques may be helpful to improve the accuracy of current CH_4 inventories and monitor the efficacy of mitigation options (Chagunda, 2013).

3.1.2 Tracer technique

Methane emissions can also be determined by using a known quantity of tracer gas (e.g., SF_6) released in the rumen at the same rate as the CH_4 emission, keeping the dilution rates identical. The CH_4 emission rate is then computed by the known release rate of the tracer gas and the ratio of CH_4 and tracer gas concentrations (Johnson et al., 1994). Unfortunately, the difference in measurement between SF_6 method and respiration chambers can be greater than 10 percent (Storm et al., 2012; Ramírez-Restrepo et al., 2020), likely due to the inconsistent release of SF_6 from the permeation tubes deposited within the rumen, variations in animal's breath collection efficiency, interruption of normal behavior due to the sampling equipment harness, and inability to collect CH_4 emissions produced in the hindgut (Lassey, 2007). Modifications to the SF_6 method have been proposed to improve its predictability, such as continuous collection at a constant rate for 24 h and the incorporation of orifice plates rather than capillary tubes to restrict the rate of sample collection (Deighton et al., 2014). Arbre et al. (2016) suggested that a 3-day measurement period was needed to achieve a repeatability of 70 percent for CH_4 emissions per unit of feed intake (i.e., CH_4 yield), without any further increase in repeatability with more extended measurement periods. The SF_6 tracer gas technique is suitable for large and small ruminants, and it can potentially be used in outdoor (Ramírez-Restrepo et al., 2010) or indoor (Ramírez-Restrepo et al., 2016) systems. However, Hristov et al. (2016) suggested that SF_6 fits better in open spaces or well-ventilated buildings because, in poorly ventilated buildings, background CH_4 could affect the interpretation of results. Experiments carried out with this technique should be conducted away for non-experimental CH_4 (e.g., slurry, other animals, wet areas) and SF_6 sources (e.g., electricity transformers, industrial sites) (Jonker and Waghorn, 2020). The SF_6 technique is relatively inexpensive, but only one animal per unit can be measured. Adequate calibration of the release rate of the tracer gas from the permeation tube should be conducted in advance of placement in the rumen, with the experiment carried soon after this calibration due to a decrease in the permeation rate of the tube. For long-term trials, adjustments for the changing permeation rate should be performed (Jonker and Waghorn, 2020).

Madsen et al. (2010) suggested predicting CH_4 from CO_2 modeled from body weight, energy-corrected milk yield, and days of pregnancy, assuming that the energy utilization efficiency for maintenance and production is constant for dairy cows. Individual CH_4 concentration was recorded in an automatic milking system for three days, using a portable air sampler and analyzer unit, based on Fourier transform infrared detection and using CO_2 as a tracer gas (Lassen et al., 2012). Air was analyzed every 20 s when the animals were milked, and the ratio between CH_4 and CO_2 was used to measure CH_4 emission. The repeatability of the measurement (CH_4:CO_2 ratio) was 0.39 and 0.34 for Holstein and Jersey cows, respectively (Lassen et al., 2012). These results suggested that the CH_4:CO_2 ratio could be used for genetic evaluations of dairy cows (Lassen et al., 2012). Unfortunately, efficient cows (i.e., more milk per feed consumed) produce less heat and consequently CO_2 per unit of metabolic body weight and energy-corrected milk; thus, overestimating their CH_4 production. Hence, genetic selection for low CH_4 emitters using CO_2 production rate as a reference will favor inefficient dairy cows (Huhtanen et al., 2020). These methodological issues of the CH_4:CO_2 ratio technique should be taken into account.

3.1.3 Open-path laser technique

The open-path laser technique quantifies the dispersion of a specific gas from the source and uses the downwind concentration of the gas to establish the emission rate, using an 'inverse dispersion' approach (McGinn et al., 2006). The technique has been used for CH_4 (McGinn et al., 2006) and ammonia (NH_3) (McGinn et al., 2007) emissions. Validation assays have shown limitations of the technique regarding the time of data collection (McGinn et al., 2006, 2008). The open-path laser technique has been updated with different analyzers and atmospheric parameters integrated into a flying platform, showing more reliable and promising results (Hacker et al., 2016). These authors indicated that with the newer approach, CH_4 and NH_3 could be detected within a distance of at least 25 and 7 km, respectively, from the source.

Tomkins et al. (2011) compared open-path laser technique with an atmospheric dispersion model for grazing animals with respiration chamber animals consuming freshly cut *Chloris gayana*. Daily estimates were 136 and 114 g CH_4/d, respectively, and the authors suggested further comparisons using different forages and herds were needed. Subsequently, Tomkins and Charmley (2015) tested the open-path laser technique using the expected behavior of herding animals around water points during the day. The measurement was conducted on 4 to 16 days for 78 hours, with data collection every 10 min. Historical meteorological data for wind direction determined the physical arrangement of equipment at each site tested. The data needed to be filtered based on the environmental conditions which included light level, surface roughness, atmospheric stability, and variation of wind direction compared to historical data. Based on their results, the authors concluded that the open-path laser technique is a good option, when employed on aggregations of grazing cattle for 7 to 8 hours per day over 7 to 14 days and is an available option for directly measuring CH_4 emissions from cattle at the herd-scale in extensive grazing conditions.

3.1.4 In vitro techniques

The in vitro fermentation techniques have been used for several years to evaluate ruminal fermentation of feedstuffs and, more recently, to assess the effect of different nutritional strategies to mitigate CH_4 production under standardized conditions (Yáñez-Ruiz et al., 2016). Due to the aforementioned complexity and cost of the methodology for evaluating enteric CH_4 emissions directly on animals, the possibility of obtaining results through *in vitro* systems is an interesting alternative, especially at laboratory conditions for screening different alternatives for reducing methanogenesis such as tannins, plant secondary metabolites, essential oils (Tedeschi et al., 2021). Available *in vitro* techniques vary from batch culture systems (Mauricio et al., 1999; Pell and Schofield, 1993; Theodorou et al., 1994) to semi-continuous fermenters such as RUSITEC (Czerkawski and Breckenridge, 1977) or dual-flow continuous culture system (Hoover and Stokes, 1991). Most *in vitro* techniques are derived from Tilley and Terry's (1963) two-stage method, which consists of simulating rumen conditions (temperature, pH, anaerobiosis) using rumen inoculum (strained rumen fluid), buffer to avoid significant pH variation, medium to provide necessary minerals to the ruminal microbiota, and the substrate to be tested, which is the source of carbon, energy, true protein and other nutrients. The CH_4 production is usually expressed per unit of incubated or on a digested DM or OM basis.

3.2 Facility based techniques

3.2.1 Manure storages

Three different approaches for the quantification of manure CH_4 emissions from housing are commonly used: direct measurement methods, inverse modeling (manure and animals), and chamber technique (manure emissions) (Hassouna and Eglin, 2016). At the barn level, removal of cattle to estimate emissions from manure has been performed (Edouard et al., 2019; Mathot et al., 2012; Mathot et al., 2016). The measurement methods that exist today were developed for scientific purposes, which is why some methods can be implemented for measuring emissions from barn and manure storage at an experimental scale (Mathot et al., 2016). Their implementation in commercial farms is too expensive and time consuming. To date, there is no international standardization for these methods where it has been clearly demonstrated that the measurement of ventilation rate can have an impact on the result obtained (Qu et al., 2021). Moreover, one of the current challenges is the development of new methods, easier to implement and less expensive (Robin et al., 2010; Hassouna et al., 2010), adaptable to different contexts, to meet objectives such as the certification of emission reductions in real conditions or the quantification of emission factors taking into account intra-category variability.

3.2.1.1 Direct methods

Direct methods are the most widely used. An emission rate is calculated as the product of the housing ventilation rate (VR) and the in-house CH_4 concentration minus the background concentration (Hassouna et al., 2021). Methodology to quantify the uncertainty of aerial emissions for the direct methods has been outlined by Gates et al. (2009) and involves the statistical uncertainty of both the emissions concentration measurement and the ventilation rate measurement. Measurements associated with ventilation rate have been demonstrated to be the major contributor to the emissions rate uncertainty when utilizing the direct methods.

3.2.1.1.1 Ventilation rate

For the Ventilation rate (VR) quantification, three methods have been implemented mainly in studies and compared in literature: Internal gas and external tracer gas (indirect methods), and use of sensors (direct method).

i. Carbon dioxide balance

For this method (Barreto-Mendes et al., 2014; Liu et al., 2016), the main hypothesis is that VR determines the relationship between CO_2 production in the barn and the difference in CO_2 concentrations between the inside and outside of the barn (ΔCO_2) and CO_2 is used as the tracer gas. In the barn, CO_2 production comes from animals, deep litter, and gas or fuel heating systems if applicable in the barn. Pedersen et al. (2008) do not recommend using this method to calculate ventilation rate in the animal house with deep litter because of its high and variable CO_2 production. Animal CO_2 production can be estimated from animal heat production, CO_2 production per heat unit, and animal activity. In many studies, these parameters are calculated with models given by CIGR (2002). According to Zhang et al. (2010), associated errors ranging from 10 to 20 percent and more recent models that take into account the progress of animal genetics should be taken into consideration to improve the accuracy of the VR estimations. Concerning the accuracy of VR, Calvet et al. (2011) demonstrated that it is necessary to consider the daily variation of CO_2 production that depends on animal activity to have an accurate estimation of the daily variation of ventilation flow. This CO_2 balance method also requires ΔCO_2. Van Ouverkerk and Pedersen (1994) suggested that ΔCO_2 values should not be lower than 200 ppm in order for the method to yield reliable results.

ii. External tracer gas

The tracer gas method for measurement of the emissions at livestock buildings refers to a technique that relies on the release of a tracer gas that is not produced in the barn. This method is often used in naturally ventilated buildings (Ogink et al., 2013). The most widely used gas is SF_6 because it is easy to detect, chemically inert, and is not produced in the building. The barn ventilation rate is calculated using the tracer gas injection rate and the tracer concentration gradient, assuming perfect mixing of the air inside the barn, as well as steady-state conditions. Because of the high GWP of SF_6, low concentrations of SF_6 should be injected, and the concentration measurements have to be done with a sensor with a low detection limit. In livestock buildings, this method could be implemented using two different approaches: a constant injection of the tracer gas or with spot injections (concentration decay method). For the constant dosing method, the tracer gas is dosed into the barn or, more generally, close to an emitting area/point source. This tracer gas mimics the dynamic flow and the dilution of CH_4 or other target gas such as N_2O or NH_3 (Schrade et al., 2012). For the tracer decay method, a dose of tracer gas is injected and mixed into the housing until the desired threshold is achieved and uniform distribution of the tracer gas is reached. Then the injection is stopped, and the decrease of tracer gas concentration is monitored during a given period to calculate the barn VR (Mohn et al., 2018). This method requires a sensor or device to measure tracer concentration with a reasonably fast analysis frequency in highly ventilated barns like open barns and is not suitable for long-term airflow measurements (Ogink et al., 2013). Many studies have compared this method with the CO_2 method in different types of livestock buildings. Edouard et al. (2016) found that both methods gave similar results with the CO_2 mass balance method compared to the SF_6 tracer methods being 10 to 12 percent lower.

iii. Sensors

In mechanically ventilated houses, continuous monitoring of the static pressure differential and the operating status (on-off) of each fan can be used to estimate the fan's VR based on its theoretical or measured performance characteristics. Ideally, the *in-situ* performance of each fan is determined first, and the house VR can be estimated by summing all operating fan flow rates. Gates et al. (2004, 2005) developed and improved a fan assessment numeration system (FANS) to measure the *in-situ* performance curve of ventilation fans operating in a negative pressure mechanically ventilated animal house. This approach can provide ventilation estimates with uncertainties less than 10 percent in low airflow conditions and less than 25 percent in higher airflow conditions when regular *in situ* calibration is conducted (Gates et al., 2009). In naturally ventilated houses, Joo et al. (2014) proposed a method that relies on the implementation of a high number of ultrasonic anemometers at the openings of the barn. In the methods they developed, any positive velocities indicated air outflows while negative velocities denoted air flowing into the barns. The total air inflow rate was assumed as the sum of air inflows at the inlets, while the total air outflow rate was the sum of air outflow rates at the outlets.

3.2.1.1.2 Methane concentration measurements

For the quantification of the emission rate, CH_4 concentrations have to be measured inside and outside the barn. Most of the time, the same device is implemented for both measurements, implying that the device has to have the adapted detection range. Powers and Capelari (2016) listed many techniques that are commonly implemented for CH_4 concentration measurements, including gas chromatography, infrared spectroscopy, Fourier transform infrared spectroscopy technologies, photoacoustic spectroscopy, mass spectroscopy, tunable diode laser absorption spectroscopy technology, and solid-state electrochemical technology. These techniques are mainly spectroscopic and portable, but only techniques with a very selective detection system such as lasers are preferred for continuous measurements. Hassouna et al. (2013) highlighted interference problems with nonselective methods such as photoacoustic infrared spectroscopy (commonly used) that can lead to overestimation of CH_4 emissions. Gas chromatography can also be implemented, but the continuous measurement is more complicated in commercial farms because regular calibration is required. Nevertheless, not all sensors and gas analyzers on the market are suitable for detecting CH_4 in barns due to the adverse conditions found there (dust, moisture, NH_3, animals). The reliability of measurements over time is not always guaranteed. Testing new equipment can take a long period of time. Moreover, the available sensors and devices are typically costly.

3.2.2 Soil fluxes

For the collection of soil CH_4 fluxes *in situ*, the two possible approaches include chamber and micrometeorological methods that come in manifold designs and varying levels of complexity. The suitability of a given technique for determining CH_4 flux rates depends on multiple factors, including, but not limited to the purpose of the experimental study, the geographic scale, measurement frequency, replicability as well as available funds and labor. These techniques also rely on the deployment of different gas analyzers to quantify CH_4 fluxes with different levels of precision and temporal resolution.

3.2.2.1 Chamber techniques

Both closed and open chambers can be used for the collection of CH_4 fluxes from rice paddy soils and from various manure handling systems including liquid and solid storage systems (Husted 1993, Moller et al. 2004, Kreuzer and Hindrichsen 2006). The principles for collection and measurement via chambers apply to both soils and manure storage systems. A solid or clear open-bottomed chamber of a known volume is fitted onto a permanently installed ring or collar to enclose a given headspace. For closed or static chambers, the concentration of CH_4 builds up in the headspace of the chamber over time and air samples from inside the chamber are extracted over a time series (e.g at 0, 10, 20 and 30min). For non-CO_2 trace gases like N_2O and CH_4, longer time series are often required due to the low, negligible, or negative fluxes of these gases (Collier et al. 2014). Methane measurement in rice fields, however, have to enclose the plants in the chamber as the plant's aerenchyma is a conduit of methane. In turn, the enclosure interval is clearly limited during daytime to avoid stress for the plants caused by increasing temperature and CO_2 depletion. Alternatively, closed chambers may also be exposed during nighttime to limit these (Wassmann et al. 2019). Although emission rates are lower at night, the diurnal patterns may be taken into account for intercomparisons of varieties and treatments. A small fan is typically installed inside the chamber to thoroughly mix the atmospheric gases. Gas samples can be collected via syringe and transferred into vials for offsite analysis (Sass et al., 1990, Sass et al., 1991) or *in situ* analysis if using a dynamic system with automated sampling devices (Wassmann et al. 1993, 2000, Hall et al. 2014). The obvious advantages of dynamic systems are the high temporal resolutions and seamless observation periods, e.g., if the emission measurements encompass 2-h intervals over entire 24-cycles and stretch over the entire cropping season (Wassmann et al. 2000). These systems have proven very valuable in combination with modeling approaches, namely the validation of simulation models such as Landscape DNDC applied to the specific conditions of rice fields (Weller et al. 2016, Kraus et al. 2016, Janz et al. 2019).

In terms of applicability, the closed chamber systems with manual sampling procedures represent by far the most common approach used for rice fields and are now operated by many research groups. The growing number of these measurements can be illustrated by a literature search in Google Scholar (searching for the terms "rice" and "closed chamber") which has yielded 23 hits for the year 1991, 101 for 2001, 241 for 2011 and 632 for 2021. At this point, closed chamber measurements in rice fields have been conducted in almost all rice-producing countries of the world – in many cases as part of Tier 2 approaches of GHG inventories under the National Communications to the UNFCCC. The caveat of this wide-ranging applications is that the measurement results often remain as "grey literature" without peer-reviewed publication and are not always available to an international audience, e.g., the IPCC

Emission Factor Data Base only shows only 24 Emission Factors for methane in rice production (as of Jan. 2022).

Open chambers, i.e., dynamic or steady-state chambers, replace air inside the headspace with ambient air through an inlet port, and the CH_4 flux is estimated as the difference between the gas concentrations at the inlet and outlet ports (Pumpanen et al. 2004). Like closed chambers, gas analysis can occur *in situ* or through collection in glass vials for offsite analysis. Although these systems can in principle be used for emission measurements for all kinds of gases, their real advantage come into play for highly reactive gases such as the NO-NO_2-O_3 triad (Breuninger et al. 2011). Given the complexity of the gas sampling patterns, however, dynamic chamber systems are rarely used for non-reactive gases like methane, i.e., the current spike in available emission measurements in rice fields is exclusively based on closed chamber systems.

Gas chromatography (GC) is the conventional method used to analyze CH_4 concentrations in gas samples from soils and manure handling systems. As for methane analysis, the GC detectors of choice is flame ionization detector (FID) (Weiss 1981) whereas other detectors may be deployed for specific purposes such as mass spectrometry (Ekeberg et al. 2004) to determine isotopic composition or multiple gas analysis systems for parallel assessment of several GHGs (Hedley et al. 2006, Sitaula et al. 1992). Laser technologies, Fourier-transform infrared (FTIR), and other optical techniques continue to grow in popularity for analyzing CH_4 concentrations because of their low detection limits, higher degree of precision, and ability to measure multiple GHGs simultaneously at the sampling location (Brannon et al. 2016, Harvey et al. 2020). The available options include quantum cascade laser (QCL) (Cowan et al. 2014, Nelson et al. 2002), and other spectroscopic techniques with QCL like cavity ring down spectroscopy (Brannon et al. 2016, Christiansen et al. 2015), and off-axis integrated cavity output (Waldo et al. 2019, Brannon et al. 2016) (Harvey et al. 2020). Infrared adsorption measurement detectors are well suited for situations that require frequent, high precision measurements, e.g., to capture diel variation and short-term responses to experimental treatments (Ruan et al. 2014).

Other auxiliary measurements like soil and water temperature, air temperature inside and outside the chamber, and soil moisture should be collected at the time of collection (Pavelka et al. 2018) for use in seasonal and annual CH_4 flux calculations. Regardless of chamber type, care should be taken to ensure that the collection of gas samples does not introduce artificial environments or conditions that alter CH_4 flux. Collections rings or collars should be installed well in advance of sample collection, i.e., > 24 hours, to allow the diffusion of gas from the soil or litter layer to the atmosphere sufficient time to equilibrate after the disturbance event. More details about robust trace gas estimation with closed chambers and open chambers can be found in Pavelka et al. (2018), Collier et al. (2014), and Rochette and Hutchinson (2005).

Both open and closed chambers are widely accepted in the literature, but selecting between chamber types involves consideration of costs, labor availability, experimental design, and sampling conditions (e.g., site accessibility, climate, soil type, etc.). Closed chambers with manual sampling are advantageous because they need only low investment and are simple to deploy, but they require greater manual labor costs (Savage et al. 2014). Both non-through-flow and through-flow chambers can alter temperature, moisture, and gas diffusion dynamics during sample collection (Husted 1993) leading to errors in flux estimation (Pihlatie et al. 2013, Ueyama et al. 2015). Errors in flux estimation with closed chambers can be significantly reduced by increasing chamber size, i.e., height, area, and volume (Pihlatie et al. 2013).

The long duration times needed for measurement with closed chambers can also alter diffusion gradients (Davidson et al., 2002, Savage et al., 2014). Open chambers, particularly through-flow systems, allow for more frequent, and less time and labor-intensive measurements (Ueyama et al., 2015, Savage et al., 2014). Furthermore, open chambers may be more appropriate for manure handling systems given the differences in gas diffusion dynamics relative to soils (Husted 1993). However, these chambers require greater capital investments and maintenance, and may not be suitable in low infrastructure contexts (Collier et al., 2014).

3.2.2.2 Micrometeorological techniques

The main micrometeorological technique for measuring CH_4 fluxes from soils is by eddy covariance (EC). Eddy covariance relies on instantaneous covariance measurements of up and down drafts of air, i.e., "eddies," and the concentration of CH_4 or other GHGs within (Baldocchi 2014, Baldocchi 2003, Baldochi et al., 1988). Samples are taken rapidly, > 10 times per second, for long durations, > 30 minutes, to calculate GHG flux density between the soil and/or vegetation and atmosphere providing relevant spatio-temporal flux estimates for whole ecosystems (Baldocchi, 2014). One of the main advantages of micrometeorological techniques is that they allow for continuous gas sampling, and they can capture temporal variability in GHG fluxes, which is a major challenge with chamber techniques. They also offer low-to-no-disturbance and non-destructive ecological sampling (Eugster and Merbold, 2015, Baldocchi et al., 1988). However, EC is less suited to small-scale manipulation experiments, and exhibits some bias with spatially heterogeneous gases like CH_4 and N_2O (Baldocchi et al., 2012). Thus, EC may be more appropriate for ecosystem-level monitoring of CH_4 fluxes, and when applied in experimental contexts, it should be combined with chamber-based methods rather than a complete substitution for them (Eugster and Merbold 2015). Another aspect to consider is the large area ('fetch') required for EC measurements that represents major impediment for intercomparisons of different agronomic treatments. While the minimum fetch for EC measurements depends on the height where the sensors are placed, the typical set-up of a mast with 2 m height in a rice field translates into a 100 m radius and thus, a coherent experimental field of 4 ha (Alberto et al. 2009). Given that these measurement systems are relatively expensive, the practical solution can be a "roving tower" that is routinely shifted from one experimental field to the other (Alberto et al. 2012). While these flux records that integrate over larger areas avoid the artificial patchiness of chamber measurements, EC measurements have further constraints regarding a steady horizontal air flow within the fetch. This imperative requirement often leads to greater data gaps during night-time and effectively excludes EC measurements during weather periods with high turbulence which is often the case in tropical regions during the rainy season. EC measurements to determine methane emissions from rice fields have been applied in several countries, e.g. US (Reba et al. 2020), China (Ge et 2020), India (Swain et al. 2018) and the Philippines (Alberto et al. 2014). Additional research is needed to understand differences in seasonal flux estimates in rice paddies measured with chambers versus EC (Reba et al., 2020).

3.2.2.3 Remote sensing and satellite measurements

A new generation of remote sensing and satellite-based monitoring systems continue to support the quantification and monitoring of CH_4 fluxes from rice production. Satellite CH_4 emissions measurements provide better spatiotemporal coverage of emissions and emissions hotspots than more traditional *in situ* measurement techniques. Early satellite measurements of global CH_4 emissions were made with SCIAMACHY (ESA; Frankenburg et al., 2006), and later with GOSAT (JAXA; Kuze et al., 2016) (Houweling et al., 2014). The number of dedicated CH_4 focused missions have increased over the past several years and include GHGSat (GHGSat, Inc.; Varon et al., 2018), GOSAT-2 (JAXA, Glumb et al., 2014) geoCARB (NASA; Polonsky et al., 2014), and MethaneSAT (EDF; Staebell et al., 2021) (UNEP and CCAC 2021). Satellite-based measurements rely on inverse modeling to understand and quantify CH_4 emissions at regional and global scales (UNEP and CCAC 2021). Under inverse modeling, the atmospheric measurements made with satellites are used to back-calculate both the location of an emissions source and the rate of emission (Houweling et al., 2014, UNEP and CCAC 2021).

Zhang et al. (2020) used SCIAMACHY and GOSAT atmospheric CH_4 concentration measurements combined with MODIS time-series imagery of rice paddy production to better understand spatio-temporal dynamics of rice CH_4 emissions in continental monsoon Asia. They found strong associations between areas of rice production at the continental scale and atmospheric CH_4 concentration, and consistencies in seasonal rice growth and atmospheric CH_4 concentrations. The combination of geographic information with satellite measurements could help reduce the spatial uncertainties associated with rice CH_4 estimates in empirical and process-based models (Zhang et al., 2020). However, Zeng et al. (2021) reanalyzed the same atmospheric CH_4 concentration data with CH_4 simulations from a chemical transport model, and they found that there is insufficient evidence to support a correlation between spatial areas of rice production and atmospheric CH_4 concentrations. These authors caution against the use of correlation-based inference to estimate CH_4 emissions from rice production at regional and continental scales, and that more work is needed to combine satellite observations and model simulations to parse out different CH_4 emissions sources (Zheng et al., 2021).

Airborne and ground-based *in situ* measurements continue to be the main methods for measuring CH_4 concentrations despite their spatiotemporal limitations. Previous work from California in rice (Peischl et al., 2012) and dairy (Arndt et al., 2018) production systems demonstrates how remote sensing techniques can capture seasonal CH_4 emissions dynamics for regional production systems not accounted for in traditional bottom-up approaches. These measurement techniques are also sensitive to capturing CH_4 emissions dynamics under different types of management systems, i.e., residue burning vs. residue soil incorporation (Peischl et al., 2012), liquid slurry vs. dry manure storage (Arndt et al., 2018) with implications for GHG inventories and climate actions.

3.3 Large-scale techniques

3.3.1 Aircrafts

Airborne CH_4 measurements of dairy farms can be conducted using a series of concentric, closed flight paths, and the emission rates estimated with the application of Gauss's Theorem (Conley et al., 2017).

At the barn level, CH_4 mixing ratio, pressure, temperature, and horizontal wind are measured while an aircraft is flying a series of concentric close paths around the farm facilities to calculate the whole-facility CH_4 emissions. Aircraft measurements were compared with open-path measurements with inverse dispersion modeling, and vehicle measurements with tracer flux ratio method in California dairies and estimated CH_4 emission rates were compared on a whole-farm level and primary sources with a farm (e.g., animal housing and liquid manure lagoons) (Arndt et al., 2018; Daube et al., 2019).

3.3.2 Satellite and drone imagery

Precision imagery, such as drone or satellite imagery, can be utilized to determine and monitor soil and crop health and estimate the yield of crops given the good correlation between leaf area index and normalized difference vegetation index (Lamb et al., 2011; Nagy et al., 2018; Wahab et al., 2018). Drones could also be used to track and count animals (Laradji et al., 2020) and also shown to detect CH_4 leaks in natural gas pipelines (Barchyn et al., 2019; Tannant et al., 2018). There are potentials to adapt these technologies to assess and benchmark livestock-related CH_4 emissions on farms.

3.4 Uncertainties

Measurement error associated with the quantification of aerial pollutants, such as CH_4, comprises both systematic and random components. Uncertainty represents the quantification of the random component. Because uncertainty establishes the range of values that the true value of the measurement will be within, the uncertainty of emissions measurements must be known when using the measurements to develop emission inventories or emission factors or to certify emissions mitigation. Gates et al. (2009) reported how component error analysis could be used to quantify uncertainties such as air flow associated with direct measurement of aerial pollutant emissions such as CH_4. Hristov et al. (2018) examined the roots of uncertainties in predicting CH_4 for inventory purposes, and reported that, at the animal level, animal inventory, feed dry matter intake, chemical composition of the diets, and CH_4 emission factors, and predictions of enteric CH_4 emissions are the main culprit. Unfortunately up to now, uncertainty has not been evaluated for all published emissions values that makes difficult the comparison of the results between the different papers, the evaluation of the quality of the results and the certification of emission reductions. One future challenge will be to provide a standard methodology for uncertainty assessment associated with emission measurements. Hristov et al. (2018) concluded that quantitative attribution of changes in atmospheric CH_4 concentrations to CH_4 sources based on $\delta^{13}CH_4$ data (stable isotope signature, specifically $^{13}C/^{12}C$ used in top down methodology), is at least questionable.

4. Estimation

4.1 Bottom-up approaches

The so-called 'bottom-up' approaches sum up estimates of all identified source components of a given region or boundary to achieve an estimate of the global source of CH_4 emitters, including enteric, manure, and soil/crop emissions. According to Lassey (2008) many of these components are illquantified and lack agreement among distinct estimates. The 'bottom-up' approaches seem to follow a more mechanistic, conceptual, build-up approach rather than a reconciliatory approach (e.g., 'top-down') that may be ill-equipped if the actual sources are not known; thus, incorrectly assigning estimate shares to known sources. Vibart et al. (2021) provided an extensive discussion about mathematical models that can predict on-farm CH_4 and N_2O emissions.

4.1.1 Modeling to estimate enteric methane

There are many different types of mathematical modeling methods in agriculture; the most common ones can be classified as either empirical or mechanistic; stochastic or deterministic; and static or dynamic (France and Kebreab, 2008; Thornley and France, 2007). Some nutrition mathematical models may incorporate different (and sometimes complementary) methods for predictability purposes, often called levels or tiers of solutions (Tedeschi and Fox, 2020a). The simplicity of empirical models is often the dominant factor in the decision-making process when selecting models to predict CH_4 emissions. In part, the model simplicity is brought up by the inputs required for the execution of the model (essentially derived from statistical regression models and methods), which favors the selection of empirical models over more complex (and sometimes more complete) types of modeling such as mechanistic or even agent-based models. Empirical models do not take into account the underlying biological mechanisms behind a natural phenomenon, but they serve their intended purpose of deterministic predictions (Tedeschi and Fox, 2020a) if all inputs (e.g., variables) are available and within the range of the original dataset used to develop the statistical regression. Another factor that is rarely considered is that the new inputs must have similar correlations among themselves as the inputs of the original dataset; otherwise, the variable's coefficients might be incorrect, and the prediction will be biased. Therefore, cautionary notes should accompany model predictions because their limitations and intended use may not be the appropriate mathematical model for all types of production scenarios and specific conditions. Ideally, different alternatives for model predictability using contrasting modeling methods should be available and used. For instance, the Beef Cattle Nutrient Requirements Model (BCNRM) by the NASEM (2016) provided empirical and mechanistic options to predict CH_4 production in beef cattle. The BCNRM's empirical option was developed based on selected empirical equations for typical beef cattle production scenarios in North America (Escobar-Bahamondes et al., 2017). Whereas the BCNRM's mechanistic option was developed based on mechanistic and empirical approaches to model the rumen functions (Fox et al., 2004; NRC, 2000), often called functional models because they simultaneously have empirical and mechanistic elements in support of a specific predictive goal (Tedeschi and Fox, 2020a). Unfortunately, few mathematical nutrition models have explicitly modeled CH_4 emission from the hindgut of ruminants, in part because the rumen represents close to 90 percent of the CH_4 emission (Murray et al., 1976; Tedeschi and Fox, 2020a), and also because there is a lack of interest in predicting the fermentation dynamics in the hindgut because they contribute little, if any, to ruminant animal performance and production.

4.1.1.1 Empirical models

Bottom-up models to predict emissions have been used in lieu of actual measurement. These models use regional activity data to estimate emissions. The IPCC (2019) developed standard predictive bottom-up models. These models are generally stratified into tiers depending on the level of complexity. Tier 1 uses default emission factors based on general literature due to the paucity of data in a region. Tier 1 does not consider the characterization of livestock systems prevalent in a region, such as breed types, age of animals, physiological states, level of productivity (except for cattle and buffalo Tier 1a), and diet (intake and composition). Tier 2 is based on emission factors refined to consider feed and animal characterization. The emission factors for each livestock category are estimated based on the gross energy intake (GEI) and CH_4 conversion factor (Ym, expressed as percent of GEI converted to CH_4). Tier 3 is region-specific based on years of extensive research in the region. The IPCC models have been criticized because they assume ad libitum feed intake and that uncertainties accompanying the derived emission factors are ill-defined, which is often the case when prevailing conditions in a region are not considered (Goopy et al., 2018).

There are several empirical prediction models that have been developed in the last decade (e.g., Benaouda et al., 2019; Moraes et al., 2014; Niu et al., 2018; van Lingen et al., 2019). These models are based on dietary intake, proportions and compositions, and animal characteristics. There is a general agreement within the scientific community that DMI is crucial in predicting CH_4 production. For instance, Benaouda et al. (2019) reviewed 36 empirical models which involved 16 dietary and animal variables and found out that 56 percent of the models used DMI as the best predictor of enteric CH_4 production while 28 percent of the models selected GEI as the main predictor of CH_4 production. Niu et al. (2018) developed 42 empirical models and reported that increased complexity improved prediction. They also reported that models with DMI only had a good accuracy of prediction while other dietary variables improved the prediction of the models further. These findings are consistent with those discussed by Appuhamy et al. (2016), who reviewed 40 models involving 20 variables and found out that 43 percent of the models used DMI to predict CH_4 production.

Determination of DMI for stall-fed and confined animals is straightforward, but many livestock systems involve ruminants grazing on native pastures supplemented with crop residues and cultivated fodder/forage in mixed crop-livestock systems. The determination of dietary amounts and composition in these systems is complicated. In part, voluntary feed intake depends on the digestibility of the diet (or digestible energy), which, in turn, depends on the level of intake (Tedeschi et al., 2019). This complication becomes more convoluted because of the lack of proper characterization of the prevailing livestock systems (i.e., numbers, breeds, herd structures, body weight, physiological states, and level of productivity). General methods for estimating DMI include the use of empirical models such as those based on the net energy system (NASEM, 2016; NRC, 2001; NRC, 2007) and those utilizing animal characteristics, pasture conditions, and supplementation (CSIRO, 2007), use of internal and external markers and herbage disappearance (Macoon et al., 2003; Undi et al., 2008). These methods, being estimates, have inherent uncertainties that further compound and increase uncertainties in CH_4 predictive models. In such cases, it would be advisable to adapt DMI estimates to local conditions as much as possible. One such adaptation is the use of “feed basket,” a term referring to proportions of feeds on offer in a given season in a given region, making up the seasonal diet of livestock in that locality (Goopy et al., 2018; Marquardt et al., 2020).

Any predictive model is as good as the accompanying level of uncertainty. It is possible that the more region-specific the data and model, the lower the accompanying uncertainty. Predictive models are used to develop national emission inventories for monitoring, reporting, and verifying nationally determined contributions on mitigation of emissions (Bodansky et al., 2016).

Additional, targeted inputs might further improve the adequacy and predictability of empirical models. An example is the milk mid-infrared (MIR) spectra of milk components as a proxy to estimate individual CH_4 emissions when using chemometrics models. Indeed, common metabolic processes will affect both the amount of eructated CH_4 and the level of milk components (e.g., fatty acids). Milk mid-infrared spectra represent the chemical bonds from the components present in the milk. Moreover, milk MIR spectra can be obtained routinely at a reasonable cost (already collected for milk payment and/or milk recording). This proxy represents significant interest for large-scale studies (compare animals, herds, periods, geographical regions, genetic studies) (Vanlierde et al., 2020), but information about the limitation and applicability of milk MIR is lacking.

4.1.1.2 Mechanistic models

Mechanistic models represent the underlying processes that control emissions and their interactions. There are very few mechanistic models developed to predict CH_4 emissions. A dynamic mechanistic model designed to simulate digestion, absorption, and outflow of nutrients in the rumen was developed by Dijkstra et al. (1992). The model contains 19 state variables representing N, carbohydrate, lipid, and volatile fatty acid (VFA) pools. Enteric CH_4 production is estimated based on VFA stoichiometry developed by Bannink et al. (2006), which relates the VFA produced to the type of substrate fermented in the rumen. The assumption is that the hydrogen produced in the rumen from the fermentation of carbohydrate and protein is used: i.) to support rumen microbial growth, ii.) for biohydrogenation of unsaturated fatty acids, and iii.) for production of glucogenic VFA (i.e., propionate and valerate). The remaining hydrogen is used for the reduction of CO_2 to CH_4, and the prediction from rumen methanogenesis, and hindgut fermentation is described by Mills et al. (2001). The model has been used to estimate enteric CH_4 emissions mostly from dairy cattle (Alemu et al., 2011b; Kebreab et al., 2008; Morvay et al., 2011). A version with an updated VFA stoichiometry that includes the effect of rumen pH on the stoichiometry of VFA formed upon fermentation of soluble sugars and starch (Bannink et al., 2008) is used as a Tier 3 method for CH_4 inventory accounting in The Netherlands (Bannink et al., 2011). Ellis et al. (2010) introduced modifications to the model in order to be able to handle predictions for beef cattle better. MOLLY is another dynamic mechanistic model that simulates rumen digestion and whole-body metabolism in lactating dairy cows (Baldwin et al., 1987a, b, c; Baldwin, 1995). The model was constructed in a similar way as described above, but the VFA stoichiometry is based on the equations developed by Murphy et al. (1982), and later updated by Argyle and Baldwin (1988), which relate the amount of VFA produced to the type of substrate fermented in the rumen. In addition to the stoichiometric differences described above, the two mechanistic models differ in the number of microbial pools; MOLLY uses one microbial pool, whereas the model by Dijkstra et al. (1992) uses three pools (amylolytic, fibrolytic, and protozoa).

Several studies have evaluated the predictive potential of empirical and mechanistic models for enteric CH_4 production from cattle using independent data sources (Alemu et al., 2011b; Benchaar et al., 1998; Kebreab et al., 2006; Kebreab et al., 2008). Benchaar et al. (1998) compared the predictive capacity of two mechanistic and two linear models with a database constructed from literature. Predictions from linear equations were poor; the models explained between 42 and 57 percent of the variation. The mechanistic models, on the other hand, explained more than 70 percent of the variation. Alemu et al. (2011a) compared empirical models and the VFA stoichiometry used in mechanistic models to estimate and assess trends in enteric CH_4 emissions from western Canadian beef cattle. The authors concluded that a more robust approach might be to use mechanistic models to estimate regional Ym values, which are then used as input for IPCC models for inventory purposes.

Another mathematical model that can be used to forecast CH_4 emission was developed by Pitt et al. (1996) and Pitt and Pell (1997) to predict VFA and ruminal pH within the Cornell Net Carbohydrate and Protein System framework. The assumptions in developing the model were based on the mass balance approach and included i.) ruminal degradation of true protein yields negligible amounts of VFA and CH_4, ii.) CH_4 is the main sink of H_2, iii.) ruminal N balance is positive, and iv.) the end products of ruminal fermentation are essentially computed as one minus bacterial yield, multiplied by the amount of ruminally degraded carbohydrate corrected for bacterial ash, CP derived from NH_3-N, and the carbon skeletons of noncarbohydrate sources (Tedeschi and Fox, 2020a, b). Further additions to Pitt's model were discussed by Tedeschi and Fox (2020a, b) and incorporated into the NASEM (2016), including pectin impact on ruminal pH, adjustments for bacterial nitrogen, and optimization for ruminal pH given the rates of degradation and escape of carbohydrates, VFA, and lactate, and buffering capacity from saliva production and feed composition. Despite the limited evaluation of the VFA-pH- CH_4 model conducted by Pitt et al. (1996), the CH_4 emission has not been fully vetted.

The model developed by the French Institute for Agricultural Research (INRA, 2018) serves as the base of a Tier 3 method to estimate CH_4 emissions of indoor and grazing production systems, given available information on the type of animal, production level, and diet characteristics and consumption (Eugène et al., 2019).

4.1.2 Modeling to estimate manure methane

4.1.2.1 Empirical models

Similar to enteric CH_4, IPCC's (2019) guidelines for National Greenhouse Gas inventories indicate three tiers of complexity to estimate CH_4 produced during the storage and treatment of manure and from manure deposited on pasture. The Tier 1 approach is based on default emission factors per unit volatile solid (VS) by animal category and manure storage system. Tier 2 is based on country-specific estimates of VS and the impact of interactions between manure management systems and animal categories on total CH_4 emissions during excretion and storage, including manure treatments such as biogas production. Recent emission factor databases may help to refine the Tier 2 approach in line with the distribution of climate regions within a country (Beltran et al., 2021; Hassouna et al, 2019, Vigan et al., 2019; Van der Weerden et al., 2020). Finally, Tier 3 requires specific modeling approaches tailored to country-specific methodologies or measurement-based approaches to quantify emission factors. Likewise, several models have been used to estimate the CH_4 emissions from manure storage systems, however, they possess a higher degree of uncertainties. For example, using the IPCC Tier 2 method, for the management of liquid manure in anaerobic lagoons and slurry storage systems, the reported CH_4

emissions were in the range of 368 ± 193 and 101 ± 47 kg CH_4/head per year, respectively (Owen and Silver, 2015).

4.1.2.2 Mechanistic models

Mechanistic modeling of CH_4 emissions is challenging because of the complex data requirement and model parameterization (Li et al., 2012). Similar to enteric emissions, mechanistic models of manure emissions are scarce. One such model, Manure-DNDC (Li et al., 2012) is an extended version of DeNitrification-DeComposition (DNDC) model (Li et al., 1992). Manure-DNDC was developed to simulate biogeochemical cycles of C, N, and phosphorus (P) in livestock farms and can be applied to simulate GHG, ammonia, and nitric oxide emissions from major components of livestock production facilities. The model contains fundamental processes describing the turnover of manure's OM. A relatively complete suite of biogeochemical processes, including decomposition, urea hydrolysis, ammonia volatilization, fermentation, methanogenesis, nitrification, and denitrification, have been embedded in Manure-DNDC, which allows the model to compute the complex transfer and transformations of C, N, and P in livestock production systems. The model has been extensively calibrated for California cropping systems and has been used for developing California CH_4 emission inventory from rice paddies and N_2O emission inventory from synthetic fertilizers and crop residue (Deng et al., 2018a, b).

4.1.3 Soil/Crop modeling

4.1.3.1 Empirical models/IPCC methodology

The IPCC methodology for estimating methane emissions from rice cultivation was approved internationally as a part of the Revised IPCC Guidelines for National Greenhouse Gas Inventories in 1996 (IPCC 1996). The respective guidelines were updated in 2006 (IPCC 2006), followed by refinement in 2019 (IPCC 2019). The guidelines for rice cultivation comprise a fairly simple empirical model based on emission factors and scaling factors in combination with activity data on crop statistics and management information. It should be noted that these guidelines were developed for estimating emissions at national scale as required in the GHG inventories under the National Communications to be submitted to the UNFCCC. In the meantime, however, the methodology has been applied in a variety of contexts from local to global scale and thus, developed into kind or standard approach for calculating methane emissions from rice production.

$$CH_{4\,Rice} = \sum_{i,j,k} \left(EF_{i,j,k} \times t_{i,j,k} \times A_{i,j,k} \times 10^{-6} \right)$$

Where:

$CH_{4\,Rice}$ = annual methane emissions from rice cultivation, Gg CH_4 yr^{-1}

EF_{ijk} = a daily emission factor for conditions *i, j,* and *k*, kg CH_4 ha^{-1} day^{-1}

t_{ijk} = cultivation period of rice for conditions *i, j,* and *k*, day

A_{ijk} = annual harvested area of rice for conditions *i, j,* and *k*, ha yr^{-1}

i, j, and *k* represent different ecosystems, water regimes, type and amount of organic amendments, and other conditions under which methane emissions from rice may vary

The different conditions to be considered include 1) rice ecosystem type (irrigated, rainfed, deep water, and upland rice production), 2) flooding pattern before and during rice cultivation period, and 3) type and amount of organic amendments. Other conditions like soil type and rice cultivar can be considered for the detailed estimation if the specific information about the relationship between these conditions and methane emissions is available.

Three tiers can be used depending on the data availability. Tier 1 applies to countries where either methane emissions from rice production are not a key category or country-specific emission factors do not exist. In Tier 1, methane emissions are estimated based on the available data of the annual harvest area of rice after the disaggregation of the area according to the water regime: irrigated, rainfed, and upland. The calculations are done for each water regime and organic amendment separately. Tier 2 applies the same methodology as Tier 1, but country-specific emission factors and/or scaling factors should be used. Tier 3 comprises the application of simulation models (see below) that must be validated by independent observations from country or region-specific studies (IPCC 2006). Irrespective of the tier, IPCC recommends using activity data that is disaggregated at the sub-national level up to the best-possible resolution available for a respective country. Ideally, the activity data will routinely be updated through monitoring networks tailored to address the national circumstance of rice cultivation.

4.1.3.1.1 Daily emission factor and scaling factors

A global methane baseline emission factor proposed in the 2019 refinement is 1.19 kg CH_4 ha^{-1} d^{-1}, with confidence interval of 0.80-1.76. Regional methane baseline emission factors are also proposed, ranging from 0.65 to 1.32 kg CH_4 ha^{-1} d^{-1} (IPCC 2019), enabling the collection of more disaggregated activity data. The emission factor is adjusted with different scaling factors to account for the difference in water regime during and before the cultivation period and the type and amount of organic amendment applied (IPCC 2019). In Tier 2, scaling factors for soil type and rice cultivar can be included.

The scaling factor for water regime during the cultivation period relative to continuously flooded fields ranges from 0.06 for deepwater rice to 0.71 for the field with a single drainage period (IPCC 2019). The scaling factor for upland rice cultivation is zero. The scaling factor for water regimes before the rice cultivation period ranges from 0.59 for the fields without flooded preseason over one year to 2.41 with flooded preseason longer than 30 days.

The scaling factor for organic amendments is determined as a function of both, the application rate and the type of organic amendments. The latter comprises conversion factors ranging from 1 for fresh rice straw to the lowest value of 0.17 for compost (IPCC 2019).

4.1.3.1.2 Activity data

Estimation of methane emissions from rice cultivation by empirical models primarily is based on harvested area statistics, which should be available from a national statistics agency. In many rice growing countries, the duration of the cultivation period can also be obtained from statistics because this factor is closely related to the respective rice variety. In the refinement in 2019, the default cultivation period of rice is given for a global scale (113 days with an error range of 74—152 days) and also for a subcontinental scale (102—139 days)(IPCC 2019).The use of locally verified cultivation areas correlated with available data for emission factors would be most valuable. International data sources are also available for annual harvested area of rice although those do not distinguish between rice ecosystems (irrigated vs. rainfed rice) which is an important feature of the methodology. Data of rice area harvested can be obtained from the FAOSTAT on the website of FAO (www.fao.org/faostat). The Ricepedia online source provided by the International Rice Research Institute (IRRI, https://ricepedia.org/rice-around-the-world) includes harvest area of rice by ecosystems type for major rice-producing countries together with other useful information including a rice crop calendar for each country.

4.1.3.2 Mechanistic models

Among the soil biogeochemical process-based models, the Denitrification-Decomposition (DNDC) model is probably the most widely applied to evaluate GHG emissions from rice production (Gilhespy et al., 2014). However, other soil biogeochemical models like the Daily Century (DayCent; Parton et al., 1998, Del Grosso et al., 2001) and CH_4MOD (Huang et al., 2004) have also been used in for reporting national GHG emissions at the Tier 3 level in the United States, and Japan and China, respectively (IPCC 2019). The DayCent model has also been parameterized and validated for Chinese rice production systems (Cheng et al., 2013, Cheng et al., 2014). Through a specific methanogenesis submodel, DayCent integrates soil redox potential (Eh), soil temperature, and C substrate supply dynamics – via the soil organic matter (SOM) and plant production submodels – to simulate CH_4 production (Cheng et al., 2013).

The DNDC model is also well parameterized for estimating CH_4 emissions from major rice production regions (Giltrap et al., 2010), and it is used as a Tier 3 method by Japan for its national GHG inventory (IPCC, 2019, Katayanagi et al., 2017). The model was explicitly developed to represent carbon sequestration and trace gas emissions in agricultural production systems by modeling microbial activities in response to aerobic and anaerobic conditions, the latter being critical to the formation of CH_4 in soils (Li, 2007). For example, the use of DNDC simulations of CH_4 emissions was better able to represent the management factors that influence CH_4 production in rice systems in both Japan (Katayanagi et al., 2017) and India (Pathak et al., 2005) – i.e., organic matter inputs, total production area, drainage class types, and water management. In Japan, the DNDC model simulations were used to generate revised emissions factors (EF), which resulted in higher national CH_4 emissions than previously calculated but reduced uncertainty relative to Tier 1 estimates (Katayanagi et al., 2017).

While most soil biogeochemical process-based models simulate above and belowground plant C and N inputs, these models were not developed with the intent of rigorously modeling the impacts of varying cultivar types and certain environmental conditions (e.g., pest outbreaks) on crop yields, and, thus, the resulting variation in plant C and N inputs to soils. As important drivers of soil C sequestration rates and trace gas emissions, the over or underproduction of crop C and N inputs directly influences the GHG balance of the crop production system (Katayanagi et al., 2017). To overcome this challenge, Tian et al. (2021) combined the Decision Support System for Agrotechnology Transfer (DSSAT) (Jones et al., 2003, Sarka,r 2012, Tian et al., 2014) crop growth model, which incorporates rice genetic parameters, with DNDC to better represent crop yield, GHG emissions, and water use to identify best management practices to minimize food-water-GHG emissions tradeoffs in China. Future efforts to combine crop growth and production models with soil biogeochemical models can help improve GHG emissions estimates from rice paddy systems, but also help identify co-benefits and tradeoffs associated with management decisions.

Unlike other scientific fields, the use of ensemble modeling is still not common in soil science. An ensemble modeling approach combines multiple models or model versions to simulate GHG emissions. This approach helps address uncertainty in representing GHG emissions dynamics, which generally stem from differences in model structure and representation of different biogeochemical process (Parker 2013), but also the use of different model input datasets (Tian et al., 2019). While there is some work in this area for crop production and yields (Asseng et al., 2013), there is limited work on applying ensemble model simulations for soil N_2O (Ehrhardt et al., 2018, Tian et al., 2019) and soil C dynamics (Sándor et al., 2020). Ensemble model simulations of CH_4 emissions from rice production remains a major gap in the literature and a key area for future research.

4.2 Top-down approaches

Top-down approaches may provide the most accurate estimates of global CH_4 after mass balance is applied to global sources and sinks (Lassey, 2008). Measurements of CH_4 emissions are made along a spectrum of spatial and temporal scales ranging from instantaneous (e.g., individual sources) to global assessments of annual CH_4 emissions. Bottom-up approaches typically involve measuring at a scale of individual CH_4 emitters, such as livestock or manure storage facilities. It uses emissions factors developed based on data collected at individual, activity, and sometimes mechanistic models. Top-down approaches, in contrast, estimate emissions using observations of atmospheric CH_4 concentrations and models that account for atmospheric transport from an emitter to an observation location (NASEM, 2018). The isotopic characterization of CH_4 emission may provide powerful discrimination between sources (Nisbet et al., 2020). The proportion of biogenic emission (from wetlands, ruminants, or wastes) leads to a shift to negative values of $\delta^{13}C_{CH4}$ (atmospheric CH_4 changing carbon isotope ratio) (Nisbet et al., 2019). Various top-down techniques are used for measuring CH_4 emissions, including remote observations (e.g., atmospheric CH_4 by infrared spectrometry), towers, aircraft, and satellites. Many modeling approaches are suitable for spatial scales of 10 to 100 m (Lassey, 2007). However, such estimates still have a high uncertainty and also may be disputed (e.g., Hristov et al. (2013).

4.2.1 Comparison between bottom-up and top-down approaches

Comparing estimates produced from bottom-up and top-down techniques has helped identify information gaps and research needs. In some cases, top-down estimates of emissions and bottom-up inventories have significantly differed, leading to a reexamination of estimates from both approaches (NASEM, 2018). The challenge for top-down approaches is that estimates include emissions from all sources but may have difficulty attributing emissions to specific sources. Bottom-up approaches, on the other hand, provide estimates from specific sources. Miller et al. (2013) used atmospheric CH_4 observations, spatial datasets, and a high-resolution atmospheric transport model to estimate CH_4

sources in the United States. The authors concluded that emissions due to ruminants and manure are up to twice the magnitude of the bottom-up approaches used by the US Environmental Protection Agency (EPA). Hristov et al. (2013) challenged Miller's et al. (2013) top-down estimates and showed that the EPA estimates agree well with other more refined models used to quantify emissions at the individual scale. According to NASEM (2018), uncertainties in top-down CH_4 emission estimates arise due to uncertainties in atmospheric transport models. Further, NASEM (2018) reports that current global and regional atmospheric transport models are likely unable to accurately represent small-scale processes, making it difficult for them to simulate observed CH_4 at continental sites accurately. Contemporaneous top-down and bottom-up measurements were conducted by Arndt et al. (2018). The authors showed that whole-facility CH_4 emissions estimates were similar among open-path, vehicle, and aircraft measurements. Emissions from animal housing were similar to EPA estimates, but CH_4 emissions from liquid manure storage were 3 to 6 times greater during the summer than during the winter measurement periods. Therefore, short term measurements should not replace long term measurements. Top-down and bottom-up methods could be complementary in identifying gaps and may lead to better characterization of CH_4 emissions.

Part 3. Mitigation of methane emissions

5.1 Methane mitigation from enteric fermentation

In this section we provide a brief description of strategies with a potential to decrease enteric CH_4 emissions from ruminant production systems. The approaches have been broadly classified as: 1) Animal breeding and management, 2) Feed management, diet formulation and precision feeding, 3) Forages, and 4) Rumen manipulation. Some of the strategies are well researched and available for immediate adoption while others are considered experimental. In all cases the adoption potential of a given strategy is dependent upon the production system and the regional/local conditions; hence, the need for numerous approaches. Strategies that differ in mode of action may have potential additive effects when combined; however, there is still a need for research on the efficacy of combined mitigation approaches. Extensive production systems with grazing ruminants represent a unique challenge for mitigation because many of the dietary and rumen manipulation strategies (e.g., feed additive supplementation) may not always be applicable in those systems. For those systems, it will be necessary to evaluate the mitigation options and the limitations, if any.

Several metrics must be considered when addressing the efficacy of a particular enteric CH_4 mitigation strategy. Some strategies decrease absolute emissions (grams of CH_4 per animal per day), some decrease emissions yield (grams of CH_4 per kilogram of DMI), and others decrease emissions intensity (grams of CH_4 per kilogram of meat or milk produced). Methane mitigation can also be evaluated in terms of CH_4 energy loss as a proportion of ingested gross energy (GE, Ym), and as CH_4 produced per kilogram of digested OM. Methane yield, CH_4 produced per kilogram of digested OM, and Ym are variables important in research for helping to understand how emissions are mitigated by a certain strategy, and the potential consequences it may have on the animal's energy utilization efficiency. By adjusting for DMI, CH_4 yield assesses how efficacious a mitigation strategy may be independently of possible changes affecting feed intake -given that feed intake is the main factor affecting CH_4 production. Methane production per kilogram of digested OM further adjusts for the proportion of ingested feed that is actually digested. As a proxy of the feed fermented in the rumen available to produce CH_4, it can reflect changes in the rumen fermentation profile. In turn, Ym provides a metric of how much extra ingested energy is potentially available for an increase in animal production when CH_4 formation in the rumen is decreased. In this document, we have subjectively defined low efficacy as decreases in CH_4 emissions (any metric) lower than 15 percent, moderate efficacy as decreases between 15 and 25 percent, and high efficacy as decrease higher than 25 percent.

It is important to consider that mitigation of enteric CH_4 emissions from a farm, region, sector or country, or globally, does not depend solely on the effects of a mitigation strategy on absolute CH_4

emissions or on CH_4 emissions intensity. Most rumen manipulation strategies target ruminal methanogenesis and thus decrease absolute emissions, without affecting animal performance. Strategies that increase animal performance and efficiency of production tend to decrease CH_4 intensity because they dilute the feed energy associated with animal or herd maintenance. While decreased CH_4 emissions intensity represents a desirable improvement in GHG efficiency, absolute CH_4 emissions can actually increase if feed consumption and production increase proportionally more than the decrease in CH_4 emissions intensity. However, this is not commonly observed.

Respiratory CO_2 and CO_2 from rumen origin expelled by animals do not have greenhouse effects, because they result from the oxidation of organic carbon compounds ingested by the animals, which are in turn the result of plant biomass accretion from atmospheric CO_2 by photosynthesis; thus, CO_2 expelled by animals is a gross but not a net source of CO_2 to the atmosphere.

5.1.1 Animal breeding and management: Increased animal production

5.1.1.1 Description

Increasing beef and milk production through improvements in management, nutrition, disease prevention and treatment, and selective breeding or genetic improvement reduces CH_4 emission intensity but in most cases will increase absolute emissions on a daily basis. Various practices and technologies in animal feeding and husbandry can be used to increase animal production, such as improved diet formulation, reduced environmental stress, disease prevention, and selective breeding for greater weight gain or milk yield (Hristov et al., 2013; Knapp et al., 2014; Beauchemin et al., 2020).

5.1.1.2 Mode of action

Increased animal production reduces CH_4 emission intensity by dilution of maintenance (Capper and Bauman, 2013), as the proportion of ingested feed that supports animal maintenance functions is decreased, while increasing the proportion of feed that supports meat and/or milk production. However, increased animal production is generally associated with increased intake and absolute emissions, unless feed conversion efficiency is also improved so that the increase in production is obtained without an increase in feed consumption.

5.1.1.3 Efficacy

The magnitude of CH_4 intensity mitigation is variable ranging from high to low. Mitigation potential is larger in low-producing than in high-producing animal systems (Gerber et al., 2013). The mitigation potential is greatest for smallholders in low-income countries that typically rely on large numbers of low-producing animals to meet the demand for food production (Tricarico et al., 2020). For example, the reductions are largest in dairy systems that produce less than 2,000 kg of fat and protein-corrected milk per cow annually, with reductions in CH_4 intensity becoming smaller as production increases (Gerber et al., 2011). In all cases, the reduction in CH_4 emission intensity must be accompanied by a reduction in animal numbers to reduce absolute (daily) CH_4 emissions. This is because higher producing animals consume more feed to meet nutrient requirements for greater production thereby producing more enteric CH_4 and manure daily. Therefore, the increase in individual daily CH_4 emissions must be compensated by a proportionally greater decrease in the number of animals to decrease total emissions of the country or region.

Consideration deserves the possibility of replacing specialized beef herds with dairy herds producing beef. This could make possible maintaining or even increasing beef production with fewer animals, and thus absolute emissions and emissions intensity could decrease. This may not be applicable to all situations though, as in many countries or regions, beef calves are produced in lower quality soils with pastures that can only cover beef cows requirements for gestation and lactation, but not for fattening animals; semi intensive or intensive dairy production may not be possible under those conditions.

5.1.1.4 Potential to combine with other mitigation strategies

Increased animal production can be achieved using a combination of various practices and technologies in animal feeding, breeding and husbandry (Capper and Bauman, 2013). The potential to combine these practices and technologies with more focused CH_4 mitigation strategies, such as the use of feed additives or manure handling technology, is very high (Knapp et al., 2014).

5.1.1.5 Effects on other emissions

Increasing animal production may increase CH_4 and N_2O emissions from manure storage and land application, due to the increase in feed intake (Gerber et al., 2013). In addition, upstream CO_2 emissions may also increase as a result of greater energy use for crop cultivation and animal management associated with increased animal production. If grazing lands are abandoned as a consequence of increased animal production, wild herbivore populations may re-occupy livestock's ecological niches, causing a net increase in methane emissions (Manzano and White 2019).

5.1.1.6 Productivity and meat/milk/manure/crop/air quality

Animal production is increased along with manure production and crop cultivation due to increased feed intake by individual animals. However, resource use efficiency increases and emissions per unit of product decrease. This can increase farm profitability while reducing CH_4 emission intensity (Knapp et al., 2014). Increasing animal production can minimize the trade-offs between CH_4 mitigation, food security, and producer welfare particularly in low-producing systems.

5.1.1.7 Safety and health aspects

Most animal feeding and husbandry practices leading to greater animal production are safe for the animals as are the food products derived from them (FAO and IDF, 2011).

5.1.1.8 Adoption potential

The adoption potential for practices and technologies that increase animal production is high in all animal production systems, but especially those characterized by low productivity. Implementing these strategies requires education and knowledge transfer, availability of natural and technological resources, and positive return on investment for producers. Successful adoption requires the identification and breakdown of barriers for different livestock systems and regions as demonstrated by failures and successes in low-income countries in the adoption of recognized best practices to increase animal production (Owen et al., 2012).

5.1.1.9 Research required

Studies quantifying the effects of improved nutrition, health, reproduction, and genetics to increase animal production and decrease CH_4 emission intensity are required on a regional basis such that these measures are relevant and can be implemented. This information is needed to help farmers make management decisions based on economic and environmental outcomes. A key research question centers on the policies implemented to achieve lower global emissions from livestock production. If feed conversion efficiency is not improved or if animal numbers are not capped, then increased productivity increases CH_4 emissions. Reducing emission intensity becomes more important when expanding ruminant production to meet the demand for food from a growing population.

5.1.2 Animal breeding and management: Selection for low methane-producing animals

5.1.2.1 Description

Animal breeding that exploits natural animal variation in CH_4 emissions is an inexpensive, permanent and cumulative mitigation strategy (Hayes et al., 2013). At present there are only a few examples where CH_4 is included within breeding programs around the world, including a large-scale commercial trial with sheep farmers that is currently underway in New Zealand and a program in The Netherlands integrating CH_4 emissions into breeding dairy values (Rowe et al., 2019; De Haas et al., 2021).

5.1.2.2 Mode of action

Animal breeding exploits natural between-animal variation in CH_4 emissions (De Haas et al., 2017). Various possible modes of actions have been identified: lower feed requirement, increased feed efficiency, increased feed digestibility, decreased rumen size, increased rate of passage, improved health, and a different rumen fermentation profile, hydrogen dynamics and methanogen activity.

5.1.2.3 Efficacy

The magnitude of CH_4 mitigation possible to obtain is not fully understood. Earlier studies conducted have been relatively small-scale studies (Chagunda et al., 2009; Garnsworthy et al., 2012; Lassen and Lovendahl, 2016), and larger scale studies are needed to draw definitive conclusions on the potential for including CH_4 in breeding programs (De Haas et al., 2017). It has been estimated that decreases in CH_4 intensity in dairy production between 13 and 24 percent between 2018 and 2050 are possible, their magnitude depending on the economic weight on CH_4 production (de Haas et al., 2021).

5.1.2.4 Potential to combine with other mitigation strategies

Genetic selection is both complementary and additive with other mitigation strategies, therefore, genetic selection for CH_4 can be combined with other mitigation strategies. A challenge is that selection for a CH_4 trait takes selection pressure from other economically important traits of interest.

5.1.2.5 Effects on other emissions

Selection for decreased CH_4 may alter OM digestibility.

5.1.2.6 Productivity and meat/milk/manure/crop/air quality

Selecting for low total CH_4 production selects for lower DMI and can result in lower production (Lassen and Løvendahl, 2016; de Haas et al., 2017; Breider et al. 2019). Also, low-CH_4 producing animals should theoretically have a better conversion of digestible to metabolizable energy; however, their lower rumen retention times may result in lower digestibility (McDonell et al., 2016; Løvendahl et al., 2018). To include targeted selection for CH_4 production within a breeding program, the link between CH_4, animal productivity, and economics needs to be considered.

5.1.2.7 Safety and health aspects

No adverse issues of breeding have been reported in the literature.

5.1.2.8 Adoption potential

The adoption potential is high but requires considerable investment by industry to measure and identify low-CH_4 phenotypes. Assessing the CH_4 phenotype of an animal is difficult because CH_4 must be measured over an extended period (weeks) of time, and measurements of thousands of individuals are required to incorporate this trait into genetic selection programs. Proxies or indicators of CH_4 production are being explored as alternative means of phenotyping low-CH_4 animals. Once the trait is integrated within the breeding program there should be little impediment for adoption. It would be expected that there would be a considerable difference in the adoption potential in low-income and high-income countries. Investigation of genotype by environment interactions would determine whether optimum genetics identified in one country is suitable in another country or region. Interactions with diet type need to be explored.

5.1.2.9 Research required

Information is needed on low-CH_4 animal phenotypes, which requires measuring CH_4 production on a large cohort of animals (> 2000) (De Haas et al., 2017). Considerable analysis is required to determine the most appropriate traits for inclusion in a selection index; for example, CH_4 emission (g/day), CH_4 intensity (g/kg product), CH_4 yield (g/kg DMI), or other. Each trait will need to be evaluated to ensure that there are no negative consequences. Genetic breeding values will have to be developed and estimated with the CH_4 trait of relevance. The final step is to include the trait of interest in the respective selection index. This requires a linkage between the CH_4 trait of interest and economics, which could be done by placing a price on methane emissions.

5.1.3 Animal breeding and management: Improved feed efficiency

5.1.3.1 Description

Improving feed efficiency, defined as the ratio of animal product to feed intake (i.e., kg of meat or milk per kg DMI or kg of DMI per kg meat or milk), reduces CH_4 emissions intensity. Feed efficiency may be improved by increasing the nutrient density or digestibility of feed, altering rumen microbial composition, enhanced feed management practices (Knapp et al., 2014), and by selective breeding for animals with negative residual feed intake[3] (Løvendahl et al., 2018; Beauchemin et al., 2020) and smaller metabolic body weight (Vandehaar et al., 2016), or a combination of those.

5.1.3.2 Mode of action

Improved feed efficiency reduces the amount of feed animals consume to meet nutrient requirements to produce a unit of product (Løvendahl et al., 2018).

5.1.3.3 Efficacy

The potential for CH_4 mitigation through improved feed efficiency is low to modest in dairy cows (Knapp et al., 2014), but may be larger in beef cattle due to greater variability (Hristov et al., 2013).

5.1.3.4 Potential to combine with other mitigation strategies

Improving feed efficiency has the potential to be combined with other mitigation strategies.

5.1.3.5 Effects on other emissions

Improving feed efficiency will reduce absolute CH_4 emissions, CH_4 intensity as well as upstream emissions associated with feed production because less feed is required to produce a given quantity of animal product. In addition, CH_4 and N_2O emissions from manure storage and land application are also reduced because less manure is produced. A switch from fibre-rich forage to starch- and protein-rich cultivated fodders will result in increased fossil CO_2 emissions which. Depending on the magnitude of natural CH_4 emission baselines (Manzano & White 2019) this may not result in a net reduced warming effect.

5.1.3.6 Productivity and meat/milk/manure/crop/air quality

Improving feed efficiency increases animal productivity per unit of feed and may increase farm profitability depending on the cost of feed compared with revenues from meat and milk.

5.1.3.7 Safety and health aspects

Caution should be exercised in implementing certain animal nutrition practices that improve feed efficiency while increasing the risk of digestive upset such as greater inclusion of starch or fat in ruminant diets (Knapp et al., 2014). Caution should also be exercised when using unbalanced selection for negative residual feed intake because it could lead to undesirable effects due to negatively correlated traits (Hristov et al., 2013; Løvendahl et al., 2018).

5.1.3.8 Adoption potential

The adoption potential for improving feed efficiency is dependent upon the ability to safely increase the nutrient density or digestibility of feed, and development and incorporation of a complex feed efficiency trait in balanced selection indexes. Currently, genotyping an animal for feed efficiency is costly. The effects of improved feed efficiency on profitability will also need to be clearly defined.

[3] Residual feed intake is defined as the difference between an animal's actual feed intake and its expected feed intake based on its size and growth.

5.1.3.9 Research required

Studies are required to understand the interactions between feed efficiency and enteric CH_4 emissions, as there have been reports of negative correlations between these variables (Freetly and Brown-Brandl, 2013; Flay et al., 2019; Renand et al., 2019). Research is also needed to understand how the biological factors that influence feed efficiency and enteric CH_4 emissions interact (Cantalapiedra-Hijar et al., 2018; Løvendahl et al., 2018). Research is needed to study the effects on enteric CH_4 emissions (both intensity and absolute) as a result of improving feed efficiency under various genotype × environment × dietary conditions. The potential for cumulative or synergistic effects of improved feed efficiency and strategic dietary management and supplementation needs to be examined. Long-term bioeconomic evaluation of improving herd feed efficiency overtime is warranted. The potential for cumulative/synergistic effects of inherent feed efficiency potential and strategic dietary management/supplementation, as well as a holistic bioeconomic evaluation of improving herd feed efficiency overtime is warranted. Genetic selection for feed efficiency is not yet a breeding objective in most systems due to the lack of genomic tools to predict feed efficiency, thus further research is warranted.

5.1.4 Animal breeding and management: Improved animal health

5.1.4.1 Description

Improved animal health through breeding, disease prevention and treatment, enhanced nutrition, and/or husbandry to reduce CH_4 emission intensity.

5.1.4.2 Mode of action

Improved animal health typically increases animal production (Dürr et al., 2008; Hand et al., 2012) and feed efficiency (Potter et al., 2018). It decreases the feed energy and nutrients used by the immune system in response to disease and for maintaining the animal. For example, when mastitis occurs, an immune response is elicited, and depending on the pathogen, a series of local and systemic effects may occur, including a decline in DMI (Ballou, 2012) increasing emission intensity. Rather than mobilizing tissue reserves to compensate for this loss of dietary energy, nutrient partitioning changes and animal production declines (Ballou, 2012).

5.1.4.3 Efficacy

It depends on whether the disease in particular negatively affects feed intake, digestibility and animal productivity. Improved health is likely to increase absolute enteric CH_4 emissions, but decrease CH_4 emission intensity (Potter et al., 2018). A review that modelled the increased intake and production as well as animal longevity due to improved animal health showed reduced emission intensity (von Soosten et al., 2020). Other studies have suggested that there would be no effect or reduction of daily enteric CH_4 emissions and low to high reductions in emission intensity (Hristov et al., 2015; Houdijk et al., 2018; Özkan et al., 2018; Potter et al., 2018; von Soosten et al., 2020). Therefore, the overall effect of improving animal health on CH_4 emissions will depend on whether animal performance is negatively affected by disease, and whether improved health increases productivity.

5.1.4.4 Potential to combine with other mitigation strategies

Mitigation effects of improved health are assumed to be additive to other CH_4 mitigation strategies.

5.1.4.5 Effects on other emissions

Improved animal health is likely to increase upstream emissions associated with crop production if feed intake and animal performance increase. Nitrous oxide emissions from manure might decrease if animals produce more product as more dietary nitrogen would be retained in meat and milk (Arndt et al., 2015). However, if there is an increase in feed intake there may also be increased N_2O from manure as a result of increased nitrogen excretion.

5.1.4.6 Productivity and meat/milk/manure/crop/air quality

Animal production losses and costs of improving animal health can vary depending on many factors, such as animal age and previous infections. For example, losses from mastitis vary depending on stage of lactation at time of infection, previous infections (Cha et al., 2013), parity (Bartlett et al., 1991), and the causative pathogen (Cha et al., 2011). Milk production losses have been shown to be as little as 0.35 kg/day (Halasa et al., 2009) to as much as 4.18 kg/day (Wilson et al., 2004). Cha et al. (2011) reported that, on average, a single case of mastitis cost farmers between $95.31 and $211.03 US for treatment, discarded milk, labor, and culturing tests. Similarly, gastrointestinal parasitism in ewes has been shown to increase enteric and manure CH_4 intensity, and manure N_2O intensity, by 11, 32 and 30 percent, respectively (Houdijk et al., 2017). In general, decreasing mortality of young animals will decrease GHG emissions, as fewer non-productive animals will have to be maintained in the herd. Improved animal health also diminishes adult animal culling and the needs for growing replacements (Hristov et al., 2013b).

5.1.4.7 Safety and health aspects

No adverse effect has been reported in the literature.

5.1.4.8 Adoption potential

The adoption potential for existing strategies to improve animal health is greater in high income countries. However, in low- and middle-income countries the adoption potential of animal health improvement strategies is low to medium, because of costs of treatments and preventive care and of access to treatments.

5.1.4.9 Research required

Most of the research on the effect of animal health on CH_4 production is based on modelling (Hristov et al., 2015; Özkan et al., 2018; von Soosten et al., 2020) and only few studies have directly measured the effect of health on enteric CH_4 emissions (Arndt et al., 2015; Houdijk et al., 2018). In general, it is possible to calculate the impacts of decreased mortality of young and adult animals on the number of replacements and herd emissions of enteric CH_4. More research is needed to better understand how improvements in health affect enteric CH_4 emission of individual animals through affecting DMI including digestive and metabolism aspects.

5.1.5 Animal breeding and management: Improved animal reproduction

5.1.5.1 Description

Increasing the reproductive performance of suckler ruminants through management, nutrition and breeding results in fewer non-productive replacement animals required within a herd. In dairy production, improved reproduction increases the proportion of lactating animals. Improved reproductive performance can occur through reproductive management and genetic selection for herd fertility. These approaches shorten the calving interval and age of first calving and increase longevity of animals in a herd.

5.1.5.2 Mode of action

Increased fertility reduces CH_4 emission intensity of meat and milk production by reducing the number of replacement animals in the herd. However, the age profile of the herd increases, which increases total daily emissions from the herd.

5.1.5.3 Efficacy

The magnitude of CH_4 mitigation depends on the reproductive status of the herd. Research conducted generally involves modelling at herd level instead of systems level taking into account the growing and non-productive animals required per productive animal in the herd (Lovett et al., 2006; Lovett et al., 2008; O'Brien et al., 2010; Lahart et al., 2021).

5.1.5.4 Potential to combine with other mitigation strategies

The potential to combine increased reproductive performance with other mitigation strategies is very high (Knapp et al., 2014).

5.1.5.5 Effects on other emissions

Improved reproductive performance allows producing the same amount of milk or beef with fewer animals. Fewer animals decrease manure output and associated emissions of CH_4 and N_2O. In pasture-based systems where the grazing season length is linked to calving date, N_2O emissions may change substantially with changes in grazing season length and required feed production.

5.1.5.6 Productivity and meat/milk/manure/crop/air quality

Increasing reproductive performance may increase animal production if the proportion of multiparous cows in the herd increases, because they have greater milk yields than primiparous cows (Hutchinson et al., 2013). Increasing animal fertility should increase farm profitability (Shalloo et al., 2014) because fewer replacement animals would be required to maintain the herd. If excess replacement heifers are used to produce beef, total animal numbers will increase along with associated emissions. However, greater beef production from excess dairy calves could potentially offset beef production and emissions elsewhere.

5.1.5.7 Safety and health aspects

No adverse effect has been reported in the literature.

5.1.5.8 Adoption potential

A balanced approach is needed for genetic selection programs to incorporate reproductive traits in addition to other economically important traits. Solely selecting on the basis of improved animal production has been associated with reductions in herd fertility. The adoption potential for practices and technologies that increase fertility is high. However, successful adoption requires education, knowledge transfer, availability of and access to resources, and positive return on investment. Implementation will also depend on the availability of genetic selection programs that include fertility in the breeding objectives. In low-income countries there may be limitations due to many of these components.

5.1.5.9 Research required

Studies quantifying the effects of improved reproduction on CH_4 emissions are required. This information will be valuable for farmers to make management decisions based on both economic and environmental outcomes. The impact of using sexed semen and embryo transfer to increase the beef merit of animals from the dairy herd needs to be quantified for its impact on CH_4 emissions. Use of sexed semen along with good herd fertility could allow targeted breeding to maximize genetic gain while at the same time maximizing beef merit thereby limiting herd expansion.

5.1.6 Feed management, diet formulation and precision feeding: Increased feeding level

5.1.6.1 Description

Increasing the feed intake of animals. In this section, we discuss the isolated effects of increasing feeding level without altering diet composition. In practice, however, it should be considered that there may be few production situations in which animals can be fed extra feed without altering diet composition. For example supplementing grazing animals with a concentrate will decrease the forage to concentrate ratio. A taller pasture with greater grass availability will likely be less digestible. In turn, altering feed intake and diet composition affect animal production.

5.1.6.2 Mode of action

Increasing the feed intake of ruminants decreases retention time of feed in the rumen due to higher passage rates. Shorter retention time limits microbial access to OM, thus reducing the extent of ruminal fermentation (Galyean and Owens, 1991), leading to a decline in CH_4 losses per unit of DMI or as a percentage of gross energy intake (GEI). Also, a rapid passage rate increases growth rate of methanogens and H_2 concentration, inhibiting acetate, H_2, and CH_4 production and favoring propionate production, which is a competitive pathway for the use of H_2 (Boadi et al., 2004; Janssen, 2010). Importantly, increased feed intake decreases the proportion of ingested and absorbed nutrients and energy associated with animal maintenance. As a result, increased feed intake dilutes CH_4 production associated with maintenance, and a greater proportion of CH_4 emitted is associated with animal production (Capper et al., 2009). The result is that total CH_4 production increases because there is more feed to ferment, but CH_4 as a proportion of DMI or GEI, and CH_4 per unit of animal product, usually decrease at higher intakes.

5.1.6.3 Efficacy

Increasing feed intake increases total CH_4 emissions but reduces CH_4 emission rate (percent of GEI or Ym) and yield (CH_4/kg DMI) (Blaxter and Clapperton, 1965; Yan et al., 2010). For example, Beauchemin and McGinn (2006) reported that Ym declined by 0.77 percentage units per unit increase in level of intake above maintenance while Hammond et al. (2013) observed a decline in CH_4 yield of up to 11 percent per unit of DMI with a twofold increase in DMI. Johnson and Johnson (1995) reported an average 1.6 percentage unit decrease in Ym per increased level of feed intake above maintenance. Moreover, CH_4 intensity (per unit of product) decreases with increasing intake, as increased intake is positively related to increased productivity. Knapp et al. (2014) reported 2 to 6 percent decrease of CH_4/energy-corrected CH_4 milk for each kilogram increase in DMI.

Empirical prediction models for CH_4 production show greatest accuracy when DMI is included as a variable (Appuhamy et al., 2016; Hristov et al., 2017; Niu et al, 2018), demonstrating the high impact of DMI on CH_4 production. In these models, the positive linear relationship between DMI and predicted CH_4 yield showed variability across models (11.3 to 15.3 g CH_4/kg DMI), and was mainly attributed to different chemical composition and digestibility of diets within the datasets used to develop the different models (Niu et al., 2018), although the measurement technique employed could also have affected the estimations (Hristov et al., 2018).

5.1.6.4 Potential to combine with other mitigation strategies

Easy to combine in practice. However, the effect of feed intake can interact with other strategies (e.g. diet quality and composition). Also, other CH_4 mitigation strategies may depress feed intake, such as inclusion of tannins (Jayanegara et al., 2012) or coconut oil (Hollmann and Beede, 2012) and other lipids.

5.1.6.5 Effects on other emissions

As is the case for increased absolute CH_4 emissions with increased feed intake, total CO_2 and N_2O emissions may also increase due to the additional feed required, although CO_2eq emissions per unit of product is decreased (Capper et al., 2009).

5.1.6.6 Productivity and meat/milk/manure/crop/air quality

Increasing feeding level can increase productivity depending on the animal category. For example, suckler beef cows and sheep in the first two thirds of gestation may not benefit from increased or ad libitum feed intake due to their relatively low energy requirement. Also, greater intake increases the excretion of feces and urine, potentially affecting manure composition and emissions (Hristov et al., 2013) but perhaps not per unit of animal product.

5.1.6.7 Safety and health aspects

This is a safe mitigation strategy for the animal, the environment, and consumers, which has been applied by producers and does not require government regulations. However, increased intake of high grain diets can increase the risk of rumen- and systemic acidosis and should be carefully managed and monitored under such feeding conditions.

5.1.6.8 Adoption potential

This mitigation strategy is easily adoptable in production systems where increasing feed offered is possible. However, in extensive grazing production systems the possibilities of increasing feed intake can be limited or may require considerable additional expense. In all cases, decisions about supplementing extra feed will depend on the economic return.

5.1.6.9 Research required

The general principles of the effects of increasing feed intake on digestion, fermentation, and CH_4 production are well established. However, refinement of extant prediction models that estimate CH_4 production in response to DMI, as well as development of new models for particular regions or diets, is important. It is recommended that this mitigation strategy is accompanied by a broader evaluation of diet characteristics that could impact efficacy. Studies related to increasing feed intake should also consider its effect on the emissions of other GHG.

5.1.7 Feed management, diet formulation and precision feeding: Decreased forage to concentrate ratio

5.1.7.1 Description

Decreasing the forage to concentrate ratio of the diet to increase the energy density of the diet.

5.1.7.2 Mode of action

Forages are composed of mainly structural carbohydrates while concentrates are high in sugars, starch and highly fermentable fiber. The composition of the carbohydrates consumed affects the VFA profile and CH_4 production (Johnson and Johnson, 1995). With high forage diets, acetic acid production is promoted resulting in greater CH_4 production per unit of feed (Hegarty and Gerdes, 1998; Janssen, 2010). Higher proportion of concentrates in the diet decreases structural carbohydrate proportion, and increases rumen outflow rate. Higher rates of growth of methanogens causes H_2 to accumulate, which inhibits acetate and CH_4 production and favors propionic production as an alternative sink of metabolic hydrogen (Hegarty and Gerdes, 1998; Benchaar et al., 2001; Janssen, 2010). Moreover, rapid fermentation rate of grains lowers ruminal pH, which inhibits the growth of methanogens and protozoa (Van Kessel and Russel, 1996; Hegarty, 1999; Janssen, 2010), thereby decreasing CH_4 production per unit of feed fermented.

5.1.7.3 Efficacy

There is general agreement that feeding concentrates to ruminants reduces CH_4 emissions, expressed relative to GEI, DMI and product, although the reported magnitude varies. Johnson and Johnson (1995) reported a 2-3 percentage unit decrease in GE losses as CH_4 in feedlots using high concentrate diets (i.e. more than 90 percent concentrate). McAllister et al. (1996) reported up to a 3.9 percentage unit reduction in the percentage of GEI lost as CH_4 with increasing concentrate intake between 40 and 68 g DM/kg$^{0.75}$ per day. Beauchemin and McGinn (2005) reported 1.5 percent percentage unit less CH_4 (4.5 percent versus 6.0 percent of GEI) from beef cattle fed primarily grain versus forage diets. Knapp et al. (2014) reported a 2 percent decrease in the CH_4 to energy-corrected milk ratio for each 1 percent increase of non-fiber carbohydrates in the diet, up to a maximum of 15 percent decrease. Sauvant and

Nozière (2016) quantified the effects of percentage of concentrate on CH_4/OMD from results of calorimetric measurements gathered in the Rumener database, concluding that energy losses as CH_4 are minimised with high percetage of concentrate fed at high intake levels. The difference in magnitude of effect of concentrates on CH_4 especially in mixed diets depends on the proportion of concentrate in the diet, the type of concentrate, and the fermentation characteristics (Moss et al., 1994).

Some experiments evaluating concentrate supplementation of grazing animals have shown a decrease of CH_4 per DMI and energy-corrected milk (Jiao et al., 2014) while others reported no change (Muñoz et al., 2015, Lovett et al., 2005, Young and Ferris, 2011). The discrepancies for pasture studies may be attributed to the substitution rate (concentrate versus pasture), pasture characteristics or differences in methodology used to estimate DMI.

Although increasing concentrate supplementation decreases CH_4 production per kilogram of DMI, OM digested, and animal product, it can lead to an increase in absolute emissions of CH_4. This is because concentrate supplementation can increase DMI and digestibility (especially in low quality forage systems), resulting in more OM fermented in the rumen.

5.1.7.4 Potential to combine with other mitigation strategies

This strategy can be easily combined with other mitigation strategies. Several studies have shown additive effects of concentrate and oil inclusion on mitigating total CH_4 emissions and emission intensity (Lovett et al., 2003; Bayat et al., 2017). Methanogenesis inhibitors such as 3-NOP show synergy with concentrates whereby the mitigation potential of inhibitors is increased in high concentrate diets (Schilde et al., 2021). Yeast showed an additive relationship with increased concentrate proportion in an in vitro study (Phesatcha et al., 2020); those results would need to be confirmed in vivo.

5.1.7.5 Effects on other emissions

Increased use of grain to decrease CH_4 output per product will, however, be accompanied by increased emissions of CO_2 and N_2O from the use of fossil fuels and nitrogen fertilizer to produce the grain (Boadi et al., 2004, Beauchemin et al., 2009). Conversion of pasture land to cropland results in the loss of soil carbon. Some studies have shown a reduction of total CO_2eq per unit of product with increasing concentrate (Johnson et al., 2002b, Lovett et al., 2006). This emphasizes the need to evaluate total CO_2eq emissions using a LCA for individual farms and geographical regions (Beauchemin et al., 2008). Changes in soil carbon need to be incorporated into the LCA.

5.1.7.6 Productivity and meat/milk/manure/crop/air quality

Concentrates are highly digestible and thus feeding concentrates in general allows for higher levels of animal productivity. Milk and meat from animals fed concentrates has more saturated fat and less polyunsaturated and rumenic and vaccenic acids compared to animals fed conserved forages, especially for grazing animals. If increasing concentrate percentage in the diet increases intake, the amount of manure may also be increased depending upon digestibility.

5.1.7.7 Safety and health aspects

Increasing the percentage of concentrates in the diet is considered safe, and it does not require regulatory approval. However, increasing concentrate percentage in ruminant diets can cause clinical and subclinical acidosis and therefore should be implemented and monitored carefully.

5.1.7.8 Adoption potential

Cereal grains can be consumed by humans and non-ruminant animals, whereas ruminants can convert fibrous feeds that are unsuitable for human consumption to high-quality protein sources (i.e., milk and meat). In this regard, feeding to ruminants concentrates that are edible by humans implies a feed vs. food competition, and is regarded as undesirable. In addition to forages, non-human edible crop coproducts are consumed in abundance by ruminant livestock. This niche role of ruminants should therefore be balanced against the decrease in CH_4 emissions yield and intensity (Boadi et al., 2004), considering also that an increase in absolute CH_4 emissions might occur. This strategy is easily adoptable in production systems in which intensification is possible. Substantial increases in cereal grain use would be difficult, or even impossible, to implement in many areas of the world where cereal crops cannot be grown or are too expensive (Beauchemin et al., 2009). However, ruminants consume considerable quantities of food waste and co-products, converting these low-value materials into high-quality products. There is opportunity to increase the use of these materials from grains and oilseeds not suitable for human consumption (e.g., frozen, off-grade, distillers grains, etc.) (Ominski et al., 2021). Adoption will depend on availability and the cost:benefit ratio of concentrate supplementation. It should also be considered that some consumers prefer animal products from grazing animals.

5.1.7.9 Research required

As the general scientific concepts are well established, further research should be focused on regional scale adoption potential using LCA approach. A quantification of natural baseline CH_4 emissions in natural or rewilded grazing ecosystems is needed to assess the effectiveness of increased concentrate ratio to mitigate global warming.

5.1.8 Feed management, diet formulation and precision feeding: Starch concentrate sources and processing

5.1.8.1 Description

Processing of grains and feeding specific sources of concentrates to promote starch fermentation in the rumen and/or shift the site of starch digestion from the rumen to the intestines.

5.1.8.2 Mode of action

Promoting starch fermentation in the rumen increases propionate production, which serves as an alternative sink of metabolic hydrogen to methanogenesis (McAllister and Newbold, 2008; Ungerfeld, 2015). Moreover, increasing starch fermentation decreases ruminal pH and inhibits the proliferation of methanogenic archaea (Van Kessel and Russell, 1996) and decreases the abundance of rumen protozoa (Franzolin and Dehority, 2010). The inhibitory effect on protozoa limits the symbiotic role of protozoa in protecting methanogens from oxygen toxicity and decreases the generation of H_2 as substrate to methanogens for CH_4 formation (Newbold et al., 2015). Additionally, processing method and source of grain can affect DM and starch degradability in the rumen. Slower rate of ruminal OM degradability will allow a greater proportion of OM digestion to occur in the intestines, which decreases the availability of substrate for CH_4 production in the rumen.

5.1.8.3 Efficacy

The anti-methanogenic effect of grain-based diets depends on the type of grain and the processing method (Johnson and Johnson, 1995). The magnitude of CH_4 abatement from grain sources appears to follow the order: wheat > corn > barley (Beauchemin and McGinn, 2005; Moate et al., 2017, 2019). Feeding a wheat-based diet to dairy cows reduced CH_4 emissions, yield and intensity by an average of 30 percent, 48 percent and 41 percent, respectively, compared to corn-based and barley-based diets. Similarly, Ramin et al. (2021) reported that an oat-based diet reduced CH_4 emissions in dairy cows by 5 percent compared to a barley-based diet. It has also been shown in finishing feedlot cattle that feeding a corn-based diet decreased CH_4 yield by 30 percent compared to a barley-based diet (Beauchemin and McGinn, 2005), possibly attributed to decreased ruminal starch digestibility (Yang, et al., 1997). Furthermore, grain processing method (application of various combinations of heat, moisture, time and mechanical actions) can modify ruminal digestion of starch (Theurer, 1986), which could influence the amount of CH_4 produced. Compared to a dry-rolled corn-based diet, feeding a steam-flaked corn-based diet to steers reduced CH_4 yield by 17 percent (Hales et al., 2012). However, the anti-methanogenic effect of grain processing is variable across studies, and is greatest for animals fed high-concentrate diets. Methane emission did not differ between single-rolled or double-rolled barley-based diets fed to dairy cows (Moate, et al., 2017) nor for ground versus pressure-cooked corn-based diets fed to calves (Pattanaik et al., 2003).

5.1.8.4 Potential to combine with other mitigation strategies

There is limited information on the synergistic effect of combining this strategy with other mitigation strategies. However, it appears feasible to combine this strategy with other CH_4 mitigation strategies, particularly the use of methanogenesis inhibitors. In vitro experiments have shown that the CH_4 mitigating effect of wheat was greater when combined with methanogenesis inhibitors (nitrate, fat or 3-NOP) compared to the individual effect of wheat (Alvarez-Hess et al., 2019).

5.1.8.5 Effects on other emissions

Feeding grain-based diets may increase the GHG emissions associated with feed production especially if the grain processing method involves the use of fossil fuel for thermal treatment. Digestibility of nutrients might differ depending on grain source and processing method, which might increase the excretion of nutrients such as fermentable OM and nitrogen (Beauchemin and McGinn, 2005; Hales, et al., 2012), and the amount of CH_4, ammonia and N_2O emissions from manure (Gerber et al., 2013).

5.1.8.6 Productivity and meat/milk/manure/crop/air quality

This strategy is expected to maintain or improve animal performance (milk yield or weight gain) if the ration formulation is well balanced to supply the nutrient requirements of the animals. However, milk protein and fat concentrations might decrease when feeding wheat or oat-based diets compared to corn- or barley-based diets (Moate et al., 2019; Moate et al., 2017; Ramin, et al., 2021) if rumen pH declines. The decrease in milk components could reduce the profitability of dairy producers.

5.1.8.7 Safety and health aspects

Grains have been routinely fed to high-producing ruminants and do not pose safety issues. However, feeding high-concentrate diets containing grains such as wheat and barley can lower rumen pH and increase the risk of sub-acute acidosis and other metabolic incidences such as laminitis and liver abscesses, which could impair animal health.

5.1.8.8 Adoption potential

This CH_4 mitigation strategy is readily available and can be easily implemented in intensive/confined feeding systems but has limited potential for application in grazing systems. Processing and feeding various grain sources is easily implemented by farmers and does not require government approval. Formulating diets with grain sources requires some technical expertise to ensure that the nutrient requirements of the animals are met. The success of this strategy will depend on the type of grain available, price volatility of grains and the cost of processing grains. The combination of these factors could increase feed costs and limit the adoption of this mitigation strategy. Moreover, this strategy, as well as modifying the forage to concentrate ratio (see **Section 5.1.7**), can increase food-feed competition and may contrast with the positive image of ruminants in utilizing human-inedible feed resources.

5.1.8.9 Research required

While a considerable amount of research has been conducted in dairy cows, more research is required to characterize how grain source and processing method could influence enteric CH_4 emission in beef cattle and small ruminants. The effect of grain processing method and degree of processing on rate and extent of starch digestion needs to be clarified regarding the impact on metabolic disorders such as acidosis. Although the magnitude of CH_4 abatement of wheat-based diets is attractive compared to other grains, the wide adoption of this feeding strategy might be limited due to the negative effect on milk fat production and profitability. Thus, further research is required to identify the appropriate ration formulation balance with wheat-based diets that would counteract the negative effect on milk fat while retaining its CH_4 mitigation potential. Finally, the impact of this CH_4 mitigation strategy on feed emissions and nutrient excretion should be considered when accounting for the net reduction effect on the emission intensity of meat or milk.

5.1.9 Feed management, diet formulation and precision feeding: Supplementation of lipids

5.1.9.1 Description

Dietary supplementation of lipids

5.1.9.2 Mode of action

Dietary lipids elicit their CH_4 mitigating effect through various mechanisms that modify the rumen ecosystem and fermentation. These mechanisms include toxicity against methanogens and protozoa; biohydrogenation of unsaturated fatty acids serving as a minor alternative H_2 sink; shifting ruminal fermentation process to promote the production of propionate resulting in lower CH_4 production, and by decreasing feed fermentability in the rumen (Newbold et al., 2015; Honan et al., 2021). Lipids can encapsulate feed particles, which reduces rumen fermentation differing digestion to the small intestine. In addition, as lipids are largely unfermentable (except for the glycerol moiety), the replacement of carbohydrates with lipids reduces fermentable OM contributing to a decrease in enteric CH_4 emissions.

5.1.9.3 Efficacy

Supplementation of dietary lipids is an effective CH_4 mitigation strategy although efficacy depends on the form (refined oil vs. oilseeds), source and amount of supplemental fat, degree of saturation and number of carbons of the fatty acids in the supplemental fat, and nutrient and fatty acid composition of the basal diet (Grainger and Beauchemin, 2011; Patra, 2013). Various meta-analysis studies have been conducted to elucidate the CH_4 mitigating effect of dietary lipids in ruminants (Beauchemin et al., 2008; Eugène et al., 2008; Grainger and Beauchemin, 2011; Patra, 2013; Patra, 2014; Arndt et al., 2021). These studies show that the anti-methanogenic effects of dietary lipids vary considerably over a broad range of conditions. Beauchemin et al. (2008) reported that adding fat to the diets of sheep, beef and dairy cattle reduced CH_4 yield (g/kg DMI) by 5.6 percent per 10 g/kg DM inclusion of supplemental fat. In other meta-analysis studies, CH_4 yield decreased by 3.77 percent in cattle (Patra, 2013) and 4.30 percent in sheep (Patra, 2014) for every 10 g fat/kg DM added to the diet. Patra (2014) indicated that the anti-methanogenic effect of dietary lipids is greater in sheep than cattle due to the comparatively lower depression of DM digestion and consequent lower decrease of CH_4 production. Medium-chain fatty acids (MCFA; lauric, myristic and capric and caprylic acids) and polyunsaturated fatty acids (PUFA) are the most effective fatty acids for reducing CH_4 emissions. Feeding refined oils rich in MCFA (e.g., coconut oil and palm kernel oil) or purified forms of MCFA such as myristic acid (Machmüller, 2006; Odongo et al., 2007; Hollmann et al., 2012) have been shown to reduce CH_4 emissions. Similarly, feeding oils or oilseeds rich in PUFA sources (e.g., fish oil, sunflower, canola, linseed, cottonseed, camelina, soybean, rapeseed) have proven effective in reducing CH_4 emissions (Fievez et al., 2003; Jordan et al., 2006a; Martin et al., 2008; Beauchemin et al., 2009; Grainger et al., 2010; Bayat et al., 2015; Ramin, et al., 2021).

Most oilseeds need to be processed prior to feeding to ensure availability of the lipids in the rumen. Oils are typically more effective than crushed oilseeds (Beauchemin et al., 2008), although the comparison depends on the extent of processing of the oilseeds. In a meta-analysis, Arndt et al. (2021) showed that feeding oils/fats and oilseeds had comparable mitigation effects on daily CH_4 production (-19 percent and -20 percent), CH_4 yield (-15 percent and -14 percent) and CH_4 intensity for milk (-12 percent and -12 percent). However, feeding oilseeds had no effect on CH_4 intensity for weight gain whereas supplemental oils and fats reduced CH_4 intensity of weight gain by 22 percent (Arndt et al., 2021). Few studies have examined the long-term effects of dietary lipids on CH_4 emission; whilst some results indicate that lipid supplementation has persistent anti-methanogenic effects (Jordan et al., 2006b; Grainger et al., 2010), a recent study under grazing conditions showed otherwise (Muñoz et al., 2021). Extrusion of linseed but not of rapeseed was effective for decreasing CH_4 yield and intensity in dairy (Martin et al., 2011). The inhibitory effect of dietary lipids on CH_4 emission is greater in concentrate-based diets compared to forage-based diets (Patra, 2013), possibly due to lower rumen pH associated with concentrate-based diets, which enhances the inhibition of fatty acids on methanogens (Zhou et al., 2015).

5.1.9.4 Potential to combine with other mitigation strategies

The synergistic effect of combining dietary lipids with other mitigation strategies has been investigated in only a few studies. An additive effect of dietary lipids on CH_4 abatement was confirmed when canola oil was combined with 3-NOP (Zhang et al., 2021) and when linseed oil was combined with nitrate (Guyader et al., 2015). However, there was no additive effect when soybean oil was combined with an extract rich in tannins (Lima et al., 2019) or saponins (Mao et al., 2010).

5.1.9.5 Effects on other emissions

Feeding fats can create emission trade-offs from feed and manure. Supplementing fats can lead to an increase in feed emissions associated with the cultivation, processing and transportation of refined oils or processed oilseeds. The effect on feed emissions can be larger in the case of soybean and palm kernel oil sourced from some parts of Latin America and Asia due to the higher global warming potential associated with substantial land-use changes. Feeding a high concentration of fats can decrease feed digestibility (Patra, 2013; Patra, 2014), which might increase the excretion of OM and CH_4 losses from manure (Møller et al., 2014; Hassanat and Benchaar, 2019). However, feeding supplemental fats at levels that do not affect feed digestibility might not affect emissions from manure (Hristov et al., 2009).

5.1.9.6 Productivity and meat/milk/manure/crop/air quality

Supplementing fats at up to 4 to 6 percent of the dietary DM (total dietary fat of 6 to 8 percent maximum) can improve milk production while reducing CH_4 emissions (-15 percent) in cattle (Patra, 2013). However, feeding higher concentrations of fats can exert detrimental effects on rumen fermentation, feed digestion and animal performance (Grainger and Beauchemin, 2011; Patra, 2013; Patra, 2014). The meta-analysis conducted by Arndt et al. (2021) quantitatively showed that feeding oils and fats decreased DMI (-6 percent) and digestibility (-4 percent) but had no effect on milk production or weight gain. However, feeding oilseeds did not affect DMI but decreased digestibility (-8 percent) and weight gain (-13 percent), with no effect on milk production (Arndt et al., 2021). Supplementing dietary lipids rich in long-chain unsaturated fatty acids can improve the nutritional quality of meat or milk by increasing the content of healthful fatty acids including PUFA, conjugated linoleic acids and vaccenic acid (Flowers et al., 2008; Bayat et al., 2015).

5.1.9.7 Safety and health aspects

This strategy is not known to pose a risk to the safety of animals and humans and is not subject to regulatory approval processes.

5.1.9.8 Adoption potential

This CH_4 mitigation strategy is readily available and can be easily implemented in intensive/confined feeding systems. This strategy does not pose a safety risk and is accepted by farmers and regulatory agencies. Ration formulation requires some technical expertise considering that supplemental fats also supply digestible energy and care must be taken to ensure that dietary fat levels do not exceed the threshold of 6 to 8 percent of the diet DM. Feeding refined oils can be costly, with limited potential for commercial application. Alternatively, processed oilseeds can be less expensive and might stimulate the adoption of supplementing dietary lipids. Although limited options exist to apply this strategy in grazing systems, there have been promising efforts to breed grasses with high levels of fats rich in PUFA (Winichayakul et al., 2008) or providing supplemental fat through drinking water (Osborne et al., 2008).

5.1.9.9 Research required

To stimulate uptake, further research is needed to identify cost-effective fat sources and their respective supplemental level that would reduce CH_4 emissions without impairing feed digestibility and animal production. The interaction of fats and fatty acids with other dietary factors (such as NDF and non-fibre carbohydrate) needs to be understood concerning the CH_4 inhibitory effect of dietary lipids. There is a need to ensure that CH_4 inhibition due to lipid supplementation of diets is not due to a decrease in fiber digestibility. Studies are also required to ascertain the long-term effect of supplemental fats in suppressing CH_4 emission. Considering the potential impact on feed emissions and nutrient excretion, the effectiveness of this mitigation strategy needs to be addressed using LCA.

5.1.10 Forages: Forage storage and processing

5.1.10.1 Description

Forage management at or after harvesting, such as form of preservation or alteration of particle size, to modify its physicochemical characteristics.

5.1.10.2 Mode of action

More than one mode of action may be involved. Ensiling of forage can decrease CH_4 production compared to preservation as hay because of fermentation of the soluble carbohydrate fraction during silage making, thereby reducing fermentation in the rumen (McDonald et al., 1991). Processing strategies such as pelleting increase rumen outflow rate. Greater passage rate decreases OM degradation in the rumen (Thomson 1972; Huhtanen and Jaakkola, 1993; Hironaka et al., 1996; LeLiboux and Peyraud, 1999) which results in less CH_4 production. Also, increased passage rate increases methanogens growth rate, and consequently H_2 concentration increases according to the Monod function. Greater H_2 concentration thermodynamically inhibits H2 production, and as a result, acetate production, which releases H_2, is also inhibited. Less H_2 being produced means less H_2 being incorporated into CH_4 production. Fermentation is shifted towards propionate production (Janssen, 2010).

5.1.10.3 Efficacy

Johnson et al. (1996) reported a decrease of CH_4 yield between 20 to 40 percent when forage was ground or pelleted compared with feeding long forage. Benchaar et al. (2001) reported similar findings in a simulation study, with approximately 20 percent reduction of CH_4 production (g/day and percentage of GEI) for pelleted compared with long alfalfa hay. The efficacy of forage processing for decreasing CH_4 production is greatest when animals are fed ad libitum rather than restrictively (Johnson and Johnson, 1995; Le-Liboux and Peyraud, 1999). Pelleting also promotes increased DMI when intake is limited by rumen fill (Vermorel et al., 1974), with the efficacy of pelleting being more pronounced for low-quality forages (Hironaka et al., 1996). Relatively few studies have examined the effect of forage preservation method on CH_4 production (Boadi et al, 2004; Knapp et al., 2014). Bechaar et al. (2001) simulated 33 percent less CH_4 (g/day, percent of GEI) for alfalfa silage compared with alfalfa hay, due to lower ruminal degradation of OM as carbohydrates are partly fermented in silage making (McDonald et al., 1991). However, a decrease in CH_4 production due to decreased ruminal digestion of OM when ensiling or pelleting feed may not decrease CH_4 per unit of meat and milk produced, unless DMI and animal production increase. The effects of preservation method will depend upon forage species and stage of maturity of the forage harvested (Evans, 2018).

5.1.10.4 Potential to combine with other mitigation strategies

Forage processing and storage methods are easily combined with other CH_4 mitigation strategies, but whether the interactions are positive or negative, or the effects are additive, will have to be evaluated in each case.

5.1.10.5 Effects on other emissions

Ensiling and processing increases the use of fuel and results in additional CO_2 emissions as compared to grazing fresh herbage. Moreover, reduced NDF digestibility due to processing can lead to increased manure emissions of CH_4 (Knapp et al., 2014). Therefore, there is a need for whole-farm LCA analysis (Beauchemin et al., 2008).

5.1.10.6 Productivity and meat/milk/manure/crop/air quality

No major concerns because decreased digestibility is generally more than compensated by increased intake, resulting in increased intake of digestible nutrients. Decreased NDF digestibility could decrease milk fat production (Boadi et al., 2004).

5.1.10.7 Safety and health aspects

Fine grinding can increase the risk of ruminal acidosis (Boadi et al., 2004). It would have to be carefully managed by adapting the animals gradually and monitoring intake of individual animals.

5.1.10.8 Adoption potential

Easy to adopt in non-grazing systems. Forage preservation methods that optimize feed nutritional quality, and hence animal performance, are recommended. These strategies (especially ensiling) are already adopted in many parts of the world. However, the need for machinery or contracting services is increased, which leads to additional costs.

5.1.10.9 Research required

While storage and processing of forage has been shown to decrease CH_4 yield, it is not clear if CH_4 per unit of animal product is also decreased as there is limited published literature (Martin et al., 2009). Studies need to consider whole farm CO_2eq emissions as a decrease in enteric CH_4 production may increase emissions elsewhere in the farming system. This can vary widely among systems and regions and therefore studies are necessary to parameterize local production systems to develop predictive models.

5.1.11 Forages: Increased forage digestibility

5.1.11.1 Description

Increasing forage digestibility leads to improved animal performance, decreasing the emissions of CH_4 per unit of product.

5.1.11.2 Mode of action

Forages are more digestible when in a vegetative phenological stage of maturity. In pastoral systems, forage digestibility can be increased through optimizing grazing management so that pre-grazing herbal mass and height is not excessive. Digestibility of OM is often higher for low than high herbal mass swards. Digestibility of forages conserved as hay or silage can be maximized by cutting and preserving at a vegetative phenological stage. Treatments with alkalis, urea, fibrolytic enzymes and lignolytic fungi have been also investigated for increasing digestibility of mature forages (Adesogan et al., 2019). Increasing forage digestibility increases animal productivity and forage intake and digestion. Responses in absolute CH_4 production to increased forage digestibility can be variable, but absolute CH_4 production usually increases due to greater DMI and increased OM fermentation in the rumen. On the other hand, emissions intensity consistently decreases, as CH_4 produced is diluted by greater amounts of animal product (Hristov et al., 2013; Beauchemin et al., 2009, 2020).

5.1.11.3 Efficacy

Greater forage digestibility can increase absolute CH_4 emissions, but low to moderate decreases in CH_4 emissions intensity generally result (Hristov et al., 2013; Beauchemin et al., 2020). Dairy cows grazing swards differing in pre-grazing herbal mass produced similar amounts of total daily enteric CH_4 per cow, but the increase in milk production with low herbage mass resulted in 10 percent lower enteric CH_4 intensity (Muñoz et al., 2016). Cows fed fresh herbage grass cut after a shorter regrowth period produced more fat- and protein corrected milk and the same total amount of CH_4, but CH_4 intensity were 12 percent less with the shorter grass regrowth period (Warner et al., 2015). Warner et al. (2016) compared grass ensiled at three stages of maturity, and reported that ensiling less mature grass resulted in greater DMI and digestibility, and milk production. Absolute CH_4 production was 6 percent greater with the earliest cut grass, but CH_4 intensity was 24 percent less. Macome et al. (2018) evaluated grass ensiled at four different stages of maturity, reporting that CH_4 yield, CH_4 production per kilogram of ingested digestible OM, and CH_4 intensity of dairy cows were 16, 24 and 21 percent less, respectively, for the youngest compared to oldest cut grass.

5.1.11.4 Potential to combine with other mitigation strategies

From a practical point of view, improved forage digestibility is easy to combine with other CH_4 mitigation strategies at the farm level. Whether biological responses are additive, or whether positive or negative interactions exist, remains to be investigated when combining increased forage digestibility with other CH_4 mitigation strategies.

5.1.11.5 Effects on other emissions

Emissions of GHG other than enteric CH_4 will be altered by grazing (e.g., changes in stocking rates) or cutting management changes that affect the digestibility of conserved forages. Earlier cutting of herbage for ensilage or hay making will result in lower grass biomass available, thus affecting the emissions of fossil fuels CO_2 per kilogram of DM conserved, although less fossil fuel per hectare may be needed to harvest and ensile/bale the forage. Downstream emissions will also be affected, as greater digestibility would decrease manure output and change its composition, perhaps decreasing the emissions of CH_4 from manure. Nitrogen excretion in urine and feces may also be affected, as forages contain more N at vegetative stages. Best practices in grazing and better movement of animals on pasture can reduce the heterogeneous distribution of manure thus reducing N_2O emissions. A LCA will be needed, and local research is recommended for generating reliable practices for each region.

5.1.11.6 Productivity and meat/milk/manure/crop/air quality

Animal productivity is expected to increase as forage quality increases, while manure output is expected to decrease. Changes in manure composition and degradation characteristics, and its methanogenic capacity need to be investigated. Higher stocking rates can result in higher ammonia emissions from manure deposited to soils, which causes air quality degradation.

5.1.11.7 Safety and health aspects

There are no safety concerns for animals, humans, food or the environment. No approvals from government agencies are required.

5.1.11.8 Adoption potential

Increasing forage quality with resulting increases in animal productivity is favorably regarded by producers (Knapp et al., 2014). However, cutting less biomass of forage for making hay or ensiling will increase costs and can be unattractive to some producers, unless overall benefits in production and profitability can be demonstrated. Local research to determine optimal cutting stages is recommended. Demonstration systems at model commercial farms may be needed for widespread adoption of economically beneficial systems.

5.1.11.9 Research required

The required knowledge in terms of biological responses, and the necessary technologies, exist and are available. More research is needed on how forge characteristics affect CH_4 emissions. Local research is recommended for establishing optimal cutting times for forages that maximize animal production and farm profitability. Life cycle assessments conducted at the regional level are needed. Such research will also help to develop emission factors specific to each type of grassland and pasture.

5.1.12 Forages: Perennial legumes

5.1.12.1 Description

Increasing the proportion of legume forages (i.e., alfalfa) in ruminant diets.

5.1.12.2 Mode of action

The highly variable nutritive profile of forages affects enteric CH_4 production. At the same physiological stage of maturity, legume forages contain less neutral detergent fiber (NDF) than grasses, and although the fiber in legumes is more lignified, the decline in fiber digestibility with advancing maturity is much greater for grasses than for legumes. Fiber that is more digestible results in a ruminal fermentation that decreases the acetate:propionate ratio and methanogenesis. In addition, legumes contain secondary compounds that decrease CH_4 production (i.e., condensed tannins and saponins; see Sections 5.1.25. and 5.1.26.;), although concentrations of these compounds is highly variable (MacAdam and Villalba, 2015; Aboagye and Beauchemin 2019; Kozłowska et al., 2020). There is interest in tannin-containing tropical legumes such as *Leucaena leucocephala* and *Desmanthus* spp. (Suybeng et al., 2019). Rate of passage from the rumen and consequently DMI can be greater for legumes than for grasses, which decreases CH_4 yield (g/kg DMI and g/kg OM digested). Lastly, animal performance is often increased with inclusion of legumes in ruminant diets, which decreases CH_4 intensity.

5.1.12.3 Efficacy

It is difficult to quantify the reduction in CH_4 production due to dietary inclusion of legumes because it depends on the quality of the forages being compared, as differences in feed intake and digestibility confound the results. For temperate forages, a meta-analysis (n = 112 treatment means) by Archimède et al. (2011) reported no difference in CH_4 between legumes and C3 grasses. In other studies comparing temperate forages, reductions in CH_4 production due to feeding legumes rather than grasses have been non-existent or inconsistent (Chaves et al., 2006; Dini et al., 2012; Hassanat et al., 2013, 2014; Arndt et al., 2015). For forages grown in warmer environments, Archimède et al. (2011) reported that legumes produced less CH_4 per kilogram of intake (DM, -19 percent; OM, -24 percent; digestible OM, -26 percent) than C3 or C4 grasses. However, those results were not substantiated by Kennedy and Charmley (2012) who reported that CH_4 for cattle fed tropical grasses was 5.4–7.2 percent of GEI (10.9–13.4 percent of digestible energy intake), whereas for tropical grass–legume mixtures, the values were 5.4–6.5 percent of GEI (8.6–13.0 percent of digestible energy intake). The notable exception was the legume *L.leucocephala*, which decreased CH_4 yield by 11 percent when its inclusion rate was doubled, without similar effects for other legume species (Kennedy and Charmley, 2012). Thus, the use of legumes may be a CH_4 mitigation strategy in areas with warmer climates where the digestibility of grasses declines rapidly with increasing maturity, with the mitigation effect being highly dependent upon forage species and quality. When the nutritive value of the diet (digestibility, CP) increases with incorporation of legumes, animal performance would be expected to increase, thereby decreasing CH_4 intensity.

5.1.12.4 Potential to combine with other mitigation strategies

It can be easily combined with other strategies, especially those with different modes of action.

5.1.12.5 Effects on other emissions

Perennial legume forages biologically fix N, which reduces the amount of N fertilizer used and consequently the CO_2 emissions from manufacturing N-containing fertilizers (Rochon et al., 2004; Lüscher et al., 2014). Biological fixation of N also increases available N for associate and subsequent crops (Schultze-Kraft et al., 2018). The N fixed by legume forages is still subject to losses, and thus contributes to N_2O emissions when their residues decay (Guyader et al., 2016), although emissions of N_2O by legumes are lower compared to grass swards (Lüscher et al., 2014). Perennial forages can increase soil carbon storage (Little et al., 2017) helping to rehabilitate degraded soils especially in tropical areas (Schultze-Kraft et al., 2018). Changes in dietary forage source can affect the physicochemical characteristics of manure. For example, CH_4 emissions from manure slurry were less when feeding dairy cows alfalfa compared with maize silage (Massé et al., 2016). Fossil fuel CO_2 emissions from the use of farm equipment are also less for perennial compared with annual forages, such as maize silage (Hawkins et al., 2015). Forage legumes generally have high nutritive value (digestible energy and crude protein), which can decrease the use of purchased supplements and associated costs and emissions (Schultze-Kraft et al., 2018).

5.1.12.6 Productivity and meat/milk/manure/crop/air quality

Effects of increasing the proportion of forage legumes in the diet on animal productivity is highly dependent on the production system and specific forages, and thus cannot be broadly quantified. Rochon et al. (2004) estimated positive economic benefits from legume and legume-grass silages compared with grass silage for the UK and E.U. Johansen et al. (2018) conducted a meta-analysis of temperate forages in dairy cows diets and concluded that overall, legume-based diets resulted in higher dry matter intake and milk yield than grass-based diets, but there was no difference in feed conversion efficiency. Milk fat and milk protein concentrations were lower on legume-based diets compared with grass-based diets. However, there were differences in DMI and energy corrected milk among the legumes, thus not all legumes are equally effective. Both animal and forage productivity need to be considered when conducting a system analysis.

5.1.12.7 Safety and health aspects

No major concerns. Grazed clovers and alfalfa can cause bloat (timpanism), but this aspect is known and generally can be well managed by farmers. It should be considered that some legumes contain tannins, which if consumed in excess can depress digestibility. Accepted by regulatory officials.

5.1.12.8 Adoption potential

Adoption potential is high, but highly dependent upon climate, soil, and growing environment. Needs a regional approach using life cycle assessment prior to recommending. May have greater CH_4 mitigation potential in tropical areas, where digestibility of grasses declines rapidly with increasing maturity, and where concentrations of secondary plant compounds are relatively high. Legumes fix nitrogen and decrease the needs of nitrogenous fertilizers, although they can increase the need of applying phosphorus.

5.1.12.9 Research required

Life cycle assessment studies that consider climate, soil type, land use, and production system are needed to determine optimum use of legumes in different locations. Assessments need to compare animal productivity under different forage management systems to identify optimum legume inclusion that minimizes emission intensity. Forage productivity and persistence must also be considered. Controlled animal research studies are needed that account for differences in intake, digestibility and plant secondary compounds to examine the true CH_4 mitigation potential of tropical and temperate legumes. Need to quantify the plant secondary compounds in different legume species, accounting for effects of stage of maturity and storage.

5.1.13 Forages: High starch forages

5.1.13.1 Description

Use of forages with high-starch concentration (i.e., whole plant cereals, sorghum and maize).

5.1.13.2 Mode of action

With high starch forage there is an increase in starch and decrease in fiber concentration of the diet, resulting in a rumen fermentation that promotes propionate production (Arndt et al., 2015), which competes with methanogenesis for metabolic hydrogen. It may also decrease rumen pH (Hassanat et al., 2013), which inhibits methanogens. These forages can enhance animal performance due to high total digestible nutrient content and greater DMI (Benchaar et al., 2014; Gislon et al., 2020).

5.1.13.3 Efficacy

Methane production may increase, remain stable or decrease, depending upon changes in DMI (increasing the energy density of the diet can increase intake when diets are limited by rumen fill). Up to 15 percent less CH_4 (percent GEI) for diets containing maize silage compared with some other forages has been reported (Hassanat et al., 2013; Benchaar et al., 2014; Gislon et al., 2020). However, effects on CH_4 per unit of animal productivity are highly variable, and may differ with nutritional values of harvested forages (Arndt et al., 2015). Efficacy of using high-starch forages to reduce CH_4 depends on the stage of maturity of the various forages (i.e., time of harvest), and the relative differences in starch concentration.

5.1.13.4 Potential to combine with other mitigation strategies

It can be easily combined with other strategies, especially those with different modes of action.

5.1.13.5 Effects on other emissions

A change in forage source will impact other emissions, therefore, promoting high-starch forages as a CH_4 mitigation strategy needs to be assessed at the farm scale using LCA. Forage production systems are highly variable and dependent upon farm site conditions (e.g., soil type and fertility, water, climate) and management practices. These factors affect forage yield and nutritive value, field emissions, animal performance and manure characteristics and emissions. Rotz et al. (2010) reported that increasing the ratio of maize silage to alfalfa silage in dairy cow diets resulted in N being used more efficiently, which resulted in a small decrease in excreted manure N and in turn decreased the emission of N_2O from cropland. Maize silage production led to less CO_2 emissions from machinery and fuel compared with alfalfa, with a net result of 13 percent less CO_2eq emissions per kilogram of milk. In contrast, Uddin et al. (2021) reported only a 2.5 percent decrease in the CO_2eq per kilogram of milk for maize silage compared with alfalfa silage in the diet of lactating dairy cows. However, carbon storage in soils was not considered by Rotz et al. (2010) or Uddin et al. (2021). Little et al. (2016) showed that although replacing alfalfa silage with maize silage in the diet of lactating dairy cows lowered Ym by 10 percent, differences between the two forage systems on CO_2eq emissions per kilogram of milk were minimal. Furthermore, the perennial forage had greater potential to store soil carbon than the maize silage rotation, illustrating the importance of considering all emission sources and soil carbon changes prior to recommending high-starch forages to decrease enteric CH_4 production.

5.1.13.6 Productivity and meat/milk/manure/crop/air quality

Effects of high-starch forages on animal productivity depends on forage nutritive value. Chemical composition and digestibility of maize silage hybrids is highly variable (Ferraretto and Shaver, 2015; Zardin et al., 2017), as is the case with most forages. A meta-analysis of 547 treatment means for maize silage diets indicated that milk yield per tonne of DM was highly positively correlated with starch content (r = 0.65) and NDF digestibility (r = 0.49) and negatively with NDF content (r = -0.72) (García-Chávez et al., 2020). Use of maize silage can decrease the N content of diets, and has been associated with greater N use efficiency by animals, decreased manure N excretion, and decreased ammonia-N and N_2O emissions from manure (Ardnt et al., 2015). Effects on manure CH_4 emissions are not well known.

5.1.13.7 Safety and health aspects

No major concerns and accepted by regulatory officials.

5.1.13.8 Adoption potential

Maize silage is already widely used in the diets of beef and dairy cattle worldwide where growing conditions are favourable. Maize is a warm season crop, and thus it is not agronomically suitable in many locations worldwide. Other high-starch forages such as small-grain cereals (barley, oat, triticale, and wheat) are widely grown in temperate locations with sorghum more suitable for semi-arid, warmer climates.

5.1.13.9 Research required

Feeding high-starch forages to reduce enteric CH_4 emissions is not recommended unless accompanied by LCA indicating the net emissions of meat and milk production are also decreased. Greatest potential of high-starch forages to reduce total CO_2eq emissions may be when used to replace another annual forage crop. As forage quality directly affects animal productivity, further research should examine the potential to use locally adapted high-starch forages to increase animal productivity and lower the CO_2eq emission intensity of animal products. The research needs to consider local agronomical and animal production conditions. Comparison of yields of forages and their feeding value produced per ha is needed.

5.1.14 Forages: High sugar grasses

5.1.14.1 Description

Use of high-sugar grasses (mainly cultivars of perennial ryegrass, *Lolium perenne* L.), with elevated water soluble carbohydrate (WSC) concentration. The WSC contents are typically increased in high sugar grasses to 250 g/kg of DM, but can be as high as 350 g/kg of DM (Lovett et al., 2006; Rivero et al., 2020). The WSC concentration is mainly increased at the expense of CP, and in some cases, NDF content. The concentration of WSC varies with cultivar, stage of maturity and forage management (Lovett et al., 2006; Rivero et al., 2020).

5.1.14.2 Mode of action

The greater concentration of readily available carbohydrates decreases acetate:propionate ratio in rumen fermentation and consequently CH_4 production is decreased (Rivero et al., 2020). High-WSC grasses also improve rate of fermentation, rumen microbial protein synthesis, with less ammonia-N absorbed and excreted as urea in the urine. The balance of carbon and N in the rumen is improved, leading to enhanced N utilization by the microorganisms.

5.1.14.3 Efficacy

In vitro studies generally report less CH_4 for high- versus low-sugar grasses (Lovett et al., 2006; Wang et al., 2020), but in vivo results are inconsistent. Ellis et al. (2012) estimated that an increase in WSC concentration of 40 g/kg of DM or more may be required to alter in vivo CH_4 production. Mitigation potential also depends on the concomitant changes in CP and NDF concentration and digestibility. Using a modeling approach with high-sugar grasses incorporated into dairy cow diets, Ellis et al. (2012) concluded that CH_4 (g/day and percentage of GEI) actually increased, especially when WSC increased at the expense of CP. However, the simulated CH_4 intensity decreased by up to 17 percent when DMI increased due to feeding high-sugar grasses. Zhao et al. (2016) fed fresh perennial ryegrass to sheep and reported moderate correlations (r = 0.44 to 0.54) between WSC concentration and various expressions of CH_4 production. For fresh forages, CH_4 per kilogram of dry matter intake was 9 percent less for sheep fed high-sugar versus control ryegrass (259 versus 221 g WSC/kg of DM; Jonker et al., 2014). However, with dried forages, there was no difference in CH_4 production, yield or intensity for dairy cows fed high- versus low-sugar grasses (193 versus 103 g WSC/kg DM; Staerfl et al., 2012b). It appears that the CH_4 mitigation potential of high-sugar grasses may be reduced when conserved as hay.

5.1.14.4 Potential to combine with other mitigation strategies

Can be easily combined with other strategies, especially those with different modes of action. The type of interaction (negative, positive, or additive effects) will need to be examined in each case.

5.1.14.5 Effects on other emissions

High-sugar grass cultivars have been shown to decrease total N excretion, and particularly the proportion of N excreted in urine (Staerfl et al., 2012b; Foskolos and Moorby, 2017). Consequently, ammonia and N_2O emissions are reduced. A life cycle assessment of milk production indicated that total CO_2eq per kilogram of milk was 3 percent less when dairy cows were fed high-sugar compared with conventional ryegrass pastures (Soteriades et al., 2018).

5.1.14.6 Productivity and meat/milk/manure/crop/air quality

In theory, increased supply of readily fermentable carbohydrates should increase animal productivity in a manner similar to supplementation with concentrates. Increased (+9 percent) DMI due to increased digestibility was reported for dairy cows in early lactation fed fresh high-sugar grass (243 versus 161 g WSC/kg of DM; Moorby et al., 2006). A meta-analysis by Ellis et al. (2012) reported a 3.3 percent average increase in DMI with increased WSC concentration (+39 g/kg of DM) of grass leading to increased milk yield. However, a more recent meta-analysis indicated that, on average, feeding dairy cattle high-sugar grasses did not increase milk production although urinary N excretion was decreased by 26 percent (Foskolos and Moorby, 2017). Additionally, the lower CP concentration of high-sugar grasses may negatively affect productivity of high producing ruminants if protein requirements are not met. For example, milk production was 18 percent less when dairy cows were fed dried high-sugar compared with control ryegrass (193 vs. 103 g WSC/kg DM), possibly because the diets were not isonitrogenous (158 vs. 254 g crude protein/kg of DM, respectively; Staerfl et al., 2012b). Also, productivity and other agronomic characteristics of high-sugar grasses will have to be considered, as they may impact the area of grassland necessary to sustain a certain level of production. Any differences in persistency could affect how soon a pasture needs to be resown, which will impact the emissions of CO_2 and N_2O associated with the use of fossil fuels and fertilizers.

5.1.14.7 Safety and health aspects

No major concerns and accepted by regulatory officials.

5.1.14.8 Adoption potential

Perennial ryegrass is easy to establish and manage in agronomically suitable areas. It grows well in a wide range of soil fertility conditions, with high forage yields and digestibility. However, its productivity and nutritional components are greatly affected by season, fertilization rate and cultivar (Rivero et al., 2019). Adoption potential of high-sugar ryegrass cultivars is widespread in temperate areas where ryegrass is commonly grown. Prior to recommending high-sugar grasses for CH_4 mitigation, the climate, soil, growing environment and yield potential need to be considered. Perennial high-sugar grasses are currently not available for tropical or subtropical areas.

5.1.14.9 Research required

Most of the research to date on high-sugar grass cultivars has been limited to the UK, NL, and NZ, and thus, an expanded geographical analysis is required. Further in vivo studies are needed to quantify the effects of high-sugar grasses on CH_4 production, yield and animal performance for various production systems. Whether CH_4 mitigation effects differ for pasture versus conserved high-sugar grass needs to be examined. A more in-depth understanding of the chemical composition and digestibility of high-sugar grasses is also needed. Finally, LCA studies that consider climate, soil type, land use, and production system are needed to determine optimum use of high-sugar grasses in different geographical locations.

5.1.15 Forages: Pastures and grazing management

5.1.15.1 Description

Grasslands are important feed sources for ruminants and provide secure livelihoods and economic opportunities for rural communities (Chará et al., 2017; Mottet et al., 2018). Grazing systems vary with climate, plant species, soil types and livestock and include season-long continuous grazing, rest-rotation grazing, deferred rotation grazing, and intensively managed grazing. These systems manage pastures to provide forage resources for animals by balancing livestock demand (both quantity and quality) with forage availability while promoting rapid pasture re-growth during the grazing season as well as long-term pasture persistence. Adequate grazing management can improve herbage quantity and quality leading to increased animal production per hectare (Alcok and Hegarty, 2006; Congio et al., 2018; Savian et al., 2018), with increased soil carbon stocks and decreased CH_4 intensity (Guyader et al., 2016; Silva et al., 2016; Makkar, 2018; Savian et al., 2018). The use of pastures for sustainable production and production of animal protein sources contributes to the FAO's Sustainable Development Goals.

In addition to traditional pasture-based systems, silvopastoral systems (SPS) that incorporate trees and shrubs in pastures increase the amount of biomass per unit of area and provide other ecosystem and biological services (Murgueitio et al., 2011). SPS promote sustainable intensification of land without using fossil fuels, while increasing biodiversity, water use efficiency and biomass production, and respecting animal welfare (Mauricio et al., 2019). Use of SPS can be a viable option especially in Latin America. Vandermeulen et al. (2018a) showed that SPS with multipurpose shrubs and trees delivered ecosystem services while the woody fodder improved ruminal protein digestion, reduced parasitic infestation and decreased CH_4 emissions, but limitations such as toxins can restrict use. Mauricio et al. (2019) showed that SPS based on different forage species, shrubs, and trees, enhanced the capacity to produce meat and milk without the use of grain.

5.1.15.2 Mode of action

This strategy is based on intensification of grazing systems. The intent is to improve herbage quality and quantity through grazing management systems that promote rapid regrowth. These systems consider pre-grazing and post-grazing sward height and maximize herbage nutritional quality and increase digestible OM intake by grazing ruminants and improve land-use (Muñoz et al., 2016; Gregorini et al., 2017; Congio et al., 2018; Savian et al., 2018).

5.1.15.3 Efficacy

Grazing management can lower enteric CH_4 yield and intensity, but CH_4 production is not expected to change, although it may increase if DMI is increased, and if increased forage production allows greater stocking rates. The extent to which grazing management lowers CH_4 intensity is extremely variable depending upon the production system and local conditions. For example, rotational grazing based on sward pre- and post-grazing heights increased digestible OM intake of sheep grazing Italian ryegrass (*Lolium multiflorum*), reducing CH_4 intensity by 17 percent, although absolute CH_4 production was not affected (Savian et al., 2018). For dairy cattle, managing the sward height of tropical, non-irrigated elephant grass (*Pennisetum purpureum* Schum. cv. Cameroon), decreased CH_4 intensity by 21 percent due to increased milk production, although absolute CH_4 production was not affected. For beef cattle, CH_4 intensity (g/kg carcass) was 10 percent less for heavy versus light continuous grazing although soil carbon sequestration was less for heavy grazing (Alemu et al., 2017).

Pasture species may also contain phytocompounds such as condensed and hydrolysable tannins that reduce enteric CH_4 production (Durmic et al., 2017; Vandermeulen et al., 2018b; Stewart et al., 2019; Ku-Vera et al., 2020). The presence of shrubs or legume (e.g., *Macrotyloma axillare*) forages in pasture lands can improve the nutritional quality of the diet while reducing CH_4 emissions due to the presence of tannins (Lima et al., 2020). Therefore, the inclusion of diversified forage species in pastures can increase the quantity of biomass for animals while decreasing enteric CH_4 emissions.

5.1.15.4 Potential to combine with other mitigation strategies

It is expected that grazing management and use of SPS would have additive effects with other CH_4 mitigation strategies.

5.1.15.5 Effects on other emissions

Grazing management affects CO_2eq intensity of beef production by influencing diet quality, animal performance and soil carbon reserves. The tree species in the SPS can affect soil CH_4 sinks (Borken et al., 2003) due to complex mechanisms involving soil chemical composition, moisture and microbiology (Dunfield, 2007). Increased feed intake may increase manure emissions unless forage digestibility is also increased. However, intensification of animal production would be expected to decrease total CO_2eq emissions per unit of livestock product (Capper et al., 2009); therefore, LCA is needed when determining efficacy.

5.1.15.6 Productivity and meat/milk/manure/crop/air quality

In most studies, improved grazing management has improved animal production, due to increased DMI and improved forage quality. For example, optimizing grazing efficiency and herbage quality in the study by Congio et al. (2018) improved milk production efficiency by 51 percent, while decreasing CH_4 emission intensity by 20 percent and CH_4 yield by 18 percent. Greater milk production efficiency increased CH_4 emissions per hectare by 29 percent. The authors considered that strategic grazing was cost effective.

5.1.15.7 Safety and health aspects

No major concerns and accepted by regulatory officials.

5.1.15.8 Adoption potential

Improved management of pastures is possible to immediately implement in extensive and intensive livestock systems. Selection of forages, shrubs and fodder species to be used needs to be tailored according to region and grazing management system. Limitations to adoption are the high cost of implementing rotational systems (fences, water troughs). Nevertheless, pasture management can be implemented at the farm-level, is suitable for all grazing ruminant categories, and has a high acceptability by farmers and consumers. There are limitations that need to be overcome such as the need for external inputs (e.g. fertilizer), potential decreases in biodiversity in some cases, and negative impacts on animal welfare (e.g., heat stress). Implementation of SPS is an objective of the Global Agenda for Sustainable Livestock (GASL, 2021).

5.1.15.9 Research required

A holistic approach involving multidisciplinary research teams and involvement of stakeholders (rural extension services, associations, cooperatives and farmers) is needed to improve pastures that promote carbon sequestration and CH_4 sinks in soils, reduce external inputs (energy, fertilizers) and improve animal welfare. Life cycle assessment of pasture-based systems needs to encompass soil carbon in addition to accurate estimates of enteric CH_4 emissions and excreta, and other aspects of the landscape and environment. Long-term regionally focused research is needed. Extension services supported by public policies (i.e., payment for environment services) may be needed to encourage adoption.

5.1.16 Rumen manipulation: Ionophores

5.1.16.1 Description

Dietary supplementation with ionophores to improve feed efficiency and decrease the acetate/propionate ratio in the rumen, thereby mitigating enteric CH_4 emissions.

5.1.16.2 Mode of action

Ionophores are carboxylic polyether substances interfering with ion transport across cell membranes of gram-positive bacteria and protozoa. Ionophores modify the flux of ion transport across cell membranes of microorganisms, increasing the concentration of protons (H^+) in the cytoplasm (Duffield and Bagg, 2000; Duffield et al., 2008a and 2008b; Hersom and Thrift, 2012; Azzaz et al., 2015). For maintaining cell equilibrium, the bacterial cells use energy to extrude H^+ out, resulting in reduced growth and death of the cells (Duffield and Bagg, 2000). Due to the structure of the cell membranes, ionophores are mainly active against gram positive bacteria and protozoa (Beauchemin et al., 2009; Hersom and Thrift, 2012; Azzaz et al., 2015), but they do not target methanogens directly (Mathison et al., 1998; Beauchemin et al., 2009). By shifting the bacterial population of the rumen, ionophores modify VFA production from acetate (H_2 source) towards propionate (H_2 sink), thus leading to reduced methanogenesis (Mathison et al, 1998; Duffield and Bagg, 2000; Duffield et al., 2008a; Beauchemin et al., 2009; Hristov et al, 2013; Azzaz et al., 2015). Also, increased feed efficiency (Beauchemin et al., 2009; Hersom and Thrift, 2012; Hristov et al., 2013) reduces CH_4 emissions intensity. The potential for adaptation of the rumen microbes to ionophores is not clear, with some (Mathison et al., 1998; Beauchemin et al., 2009), but not all (Appuhamy et al., 2013) reports indicating time limited effects.

5.1.16.3 Efficacy

In their meta-analysis of 22 published studies, Appuhamy et al. (2013) demonstrated that monensin reduced the CH_4 conversion factor Ym by 0.5 percent unit (5.97 vs 5.43 for control and test groups, respectively) in beef cattle, with diets high in NDF concentration showing the greatest effects. However, there was no effect of monensin on the Ym value for dairy cows. Different doses were tested in the beef and dairy studies. When adjusting for monensin dose, the CH_4 mitigation effects were similar for dairy cows and beef steers (-12 $\pm$ 6 g/d and -14 $\pm$ 6 g/d, respectively). When adjusted for DMI differences, monensin reduced Y_m in dairy cows and beef steers by 0.23 $\pm$ 014 and 0.33 $\pm$ 0.16, respectively. The duration of the treatment period did not significantly modify the monensin effect.

5.1.16.4 Potential to combine with other mitigation strategies

Using combinations of ionophores or rotational feeding of ionophores may help avoid microbial adaptation (Mathison et al., 1998). Good potential to combine with other strategies with different modes of action. No interaction effects were observed when monensin was combined with 3-nitrooxypropanol in the diets fed to beef cattle (Vyas et al., 2018).

5.1.16.5 Effects on other emissions

With improvements in feed conversion efficiency due to monensin, the quantity of OM excreted in the manure might also be reduced, further reducing CH_4 emission from the farm. Improved N metabolism due to ionophores reduces urinary N excretion and associated potential emissions of NH_3 and N_2O. Given that monensin dose in the diet is small, the increase in CO_2 emissions from the use of fossil fuels in monensin manufacture and transport is thought to be rather small.

5.1.16.6 Productivity and meat/milk/manure/crop/air quality

Ionophores are used to improve feed efficiency and productivity of beef cattle and dairy cows (Hersom and Thrift, 2012). Ionophores also improve feed efficiency (reducing feed intake, by about 0.3 kg/day and increasing milk yield by 0.7 kg/day in the case of monensin fed to dairy cows; monensin being the most extensively studied ionophore (Duffield et al., 2008b), leading to greater production for the same amount of feed consumed (Mathison et al., 1998; Duffield and Bagg, 2000; Duffield et al., 2008b; Beauchemin et al., 2009; Hersom and Thrift, 2012; Hristov et al., 2013). The use of ionophores may affect the milk fatty acid profile, reducing the short chain fatty acids and stearic acid, while increasing conjugated linoleic acid (Duffield et al., 2008b). In addition to productivity benefits, ionophores may also improve ruminant health, particularly reducing the risk of sub-clinical ketosis (Duffield and Bagg, 2000; Duffield et al., 2008a), subacute acidosis (Appuhamy, 2013) and bloat (Duffield and Bagg, 2000; Duffield et al., 2008a, b, Appuhamy, 2013).

5.1.16.7 Safety and health aspects

The concentration of ionophores in the diet should be limited to avoid toxicity (Novilla, 1992; Hall, 2000). Use of monensin is subject to approval by regulatory agencies, and it is banned in some countries, including the EU. It has been questioned whether the widespread use of ionophores might contribute to cross-resistance to other antibiotics (Wong, 2019).

5.1.16.8 Adoption potential

The use of ionophores in feed for ruminants is regulated and, in some jurisdictions (e.g., European Union), their use is prohibited. Where the use of ionophores is authorized, the adoption can be high in production systems where dairy cows and beef cattle are supplemented with minerals or compound feeds. Ionophores are supplemented via feed (Hersom and Thrift, 2012) and therefore do not necessitate specific investments on the farm. Improvements in animal performance provide economic benefits that generally offset the cost of the ionophore. Ionophores can also be provided in the form of slow-release capsules, which can be useful for more extensive pasture-based systems.

5.1.16.9 Research required

The use of ionophores in beef and dairy cattle feed is well known and its commercial application is widespread. Numerous studies and meta-analyses have demonstrated the benefits; however, it is recommended that meta-analyses be updated to include more recently published information.

5.1.17 Rumen manipulation: Chemical inhibitors of methane production

5.1.17.1 Description

Several chemical compounds investigated since the 1960s inhibit the formation of CH_4 in rumen fermentation when present in the diet in small concentrations. The investigational compound 3-nitrooxypropanol (3-NOP), which is commercially available in some countries, is discussed separately below (5.1.18). Studies investigating the use of chemical inhibitors of methanogenesis in preruminant animals are discussed in the Early Life Intervention section.

5.1.17.2 Mode of action

Chemical inhibitors target methanogens but not all of them through inhibiting methanogenesis directly. Methane halogenated analogues chloroform, bromoform, iodoform, bromochloromethane (BCM), carbon tetrachloride, and others (Bauchop, 1967; Trei et al., 1971; Lanigan, 1972) inhibit the last step of methanogenesis by reacting with vitamin B_{12} to block cobamide-dependent methyl transfer (Wood et al., 1968). Coenzyme M analogues bromoethanesulfonate (BES; Gunsalus et al., 1978) and 3-NOP (Duin et al., 2016) also inhibit the last step of methanogenesis by blocking methylcoenzyme M reductase. Hydroxymethylglutaryl-SCoA inhibitors mevastatin and lovastatin inhibit the synthesis of membrane lipids in Archaea (Miller and Wolin, 2001). It was speculated that 9, 10-antraquinone may disrupt electron transfer and hinder ATP generation in methanogens (García-López et al., 1996). Direct inhibition of methanogens by other chemicals such as pyromellitic diimide (Martin and Macy, 1985), and halogenated compounds 2, 2, 2-trichloroacetamide (Trei et al., 1971) and hemiacetal of chloral and starch (Trei et al., 1972), among others, is evidenced by accumulation of H_2, but their exact mechanisms of action in the methanogen cell has not been demonstrated.

5.1.17.3 Efficacy

In two recent meta-analyses of in vivo studies (Veneman et al., 2016; Arndt et al., 2021), chemical inhibitors of methanogens were found to cause the strongest decrease in absolute CH_4 production among various antimethanogenic strategies. In some in vivo studies, absolute CH_4 production was inhibited more than 90 percent compared with control treatments (Mathers and Miller, 1982; McCrabb et al., 1997; Mitsumori et al, 2012). The highly specific methanogenesis inhibitor BES is very potent in vitro, but its effects lasted for only 3 d in vivo (Immig et al., 1996). Conversely, long term inhibition of methanogenesis in vivo by different chemical compounds has been observed in other studies (e.g., Trei et al., 1971, 1972; Clapperton et al., 1974, 1977; Davies et al., 1982; Tomkins et al., 2009).

5.1.17.4 Potential to combine with other mitigation strategies

High specificity of these compounds makes it possible to find additive effects when two or more compounds with different mechanisms of action are combined, and likewise, when combined with other antimethanogenic strategies with different mechanisms of action. Different methanogens are inhibited by chemical compounds to different extents (Ungerfeld et al., 2004; Duin et al., 2016), and thus combinations or rotations of different inhibitors of methanogenesis is a research direction of interest. In vitro experiments as proof of concept are recommended (e.g., Zhang and Yang, 2012; Patra and Yu, 2013).

5.1.17.5 Effects on other emissions

Manufacturing and transporting these chemical compounds result in emissions of fossil fuels-generated CO_2. However, because their dietary concentration is very low, the significance of these emissions on CO_2eq daily production or intensity basis is very low.

5.1.17.6 Productivity and meat/milk/manure/crop/air quality

In general, inhibition of methanogenesis with chemical compounds does not affect animal productivity (Ungerfeld, 2018; Arndt et al., 2021). Digestibility is not affected but DMI mostly decreases (Ungerfeld, 2018). Amount and chemical composition of manure is probably little affected, but passage of chemical inhibitors to manure has not been evaluated (except for 3-NOP).

5.1.17.7 Safety and health aspects

Compounds such as CH_4 halogenated analogues can be toxic to the animal and are volatile and can deplete the ozone layer of the atmosphere. Concentration of BCM in muscle, fat and offal off steers fed BCM was within maximum limits in Australia, although potential losses due to volatility were not considered (Tomkins et al., 2009). Toxicity, passage of chemical inhibitors to animal products and voiding to the environment must be carefully examined before these compounds can be recommended and marketed.

5.1.17.8 Adoption potential

Chemical inhibitors can allow a strong and consistent decrease in enteric CH_4 emissions with minimal effects on other GHG emissions but, so far research did not demonstrate this clearly. Inclusion of an inhibitor in a diet will increase costs, and will be unattractive to producers if not accompanied by a higher price of a product associated with lower carbon footprint. It may also be possible to take advantage of greater energy retention and changes in rumen and animal metabolism to increase animal productivity, but more research is needed to explore those possibilities. Chemical inhibitors may not be suitable for grazing systems in which animals are not supplemented, unless slow-release forms adequate for those systems can be formulated or a lower efficacy accepted. Approval by government agencies can be lengthy and expensive. Consumers may be reluctant to accept them, but we are unaware of published consumer surveys on this matter.

5.1.17.9 Research required

This research area is of much interest due to the highest efficacy observed. Methanogen enzymes are being screened for the development of new chemical inhibitors (Carbone et al., 2018), and new inhibitors are being investigated (e.g., Zhang et al., 2019, 2020). At the same time, there are older reports in which some compounds were shown to have long lasting effects on methanogenesis (see above for some examples); we are not aware of further investigation about toxicity, passage to animal products or damage to the environment for those compounds, and probably those lines of research were abandoned because of the limitations put ot handling these compounds in the feed/food chain. Also, research to understand changes in rumen microbial and whole animal metabolism induced by methanogenesis inhibition is recommended in order to optimize the outcome of the interventions (Ungerfeld, 2018; Ungerfeld and Hackmann, 2020).

5.1.18 Rumen manipulation: 3-Nitrooxypropanol (3-NOP)

5.1.18.1 Description

3-Nitrooxypropanol (3-NOP) is a CH_4 inhibitor developed and commercialized by DSM Nutritional Products (Basel, Switzerland). It is the mononitrate ester of 1,3-propanediol with the formula $HOCH_2CH_2CH_2ONO_2$ (Duval and Kindermann, 2012; Yu et al., 2021).

5.1.18.2 Mode of action

3-NOP is a small molecule with a shape similar to that of methyl-coenzyme M (methyl-CoM; Duin et al., 2016). Methyl-CoM is a substrate of coenzyme M reductase (MCR) in the last step of methanogenesis. As an analogue of methyl-CoM, 3-NOP selectively binds into the active site of MCR in a position similar to natural ligand methyl-CoM and inactivates MCR by oxidizing the active site nickel +1 in co-factor F_{430}. Additionally, the nitrate group of 3-NOP is reduced to nitrite in the process, which also inactivates MCR (Duin et al., 2016). As a result, CH_4 production is inhibited and the flow of metabolic hydrogen in rumen fermentation shifts from acetate and CH_4 towards propionate, butyrate and valerate (Romero-Perez et al., 2014; Schilde et al., 2021).

5.1.18.3 Efficacy

There is a growing number of scientific publications (> 50) describing its efficacy in dairy and beef cattle in a range of different diets and management systems, with several reviews and meta-analysis (Dijkstra et al., 2018, Jayenagara et al, 2018, Kim et al., 2020; Arndt et al., 2021, Yu et al, 2021). This extensive body of published data (in vitro, short term and long-term studies) in conjunction with studies run under the specific guidelines and requirements needed for registration of 3-NOP in Europe, allowed the evaluating panel to assess and conclude that 3-NOP has the potential to be efficacious in all ruminant species (Bampidis et al., 2021). The meta-analysis by Dijkstra et al. (2018), was recently updated to include 22 additional recent studies (E. Kebreab, pers. comm.), and resulted in an average decrease of 30.6 ± 2.91 percent and 27.1 ± 2.89 percent (mean ± SEM) for CH_4 production and yield, respectively, for the mean 3-NOP dose of 118 mg/kg of DM in the ration of dairy and beef cattle. Efficacy was positively related to 3-NOP dose (0.22 percent decrease in CH_4 yield per mg 3-NOP/kg of DM). Efficacy was also negatively affected by NDF concentration of the diet and positively by starch concentration. Mean responses were greater in dairy (38.2 ± 3.33 percent and 34.9 ± 3.43 percent, for CH_4 production and yield respectively) compared with beef cattle (26.1 ± 2.76 percent and 21.1 ± 2.99 percent, for CH_4 production and yield respectively). Although most long-term studies have shown that 3-NOP effectiveness remained constant, a couple of studies report that 3-NOP effectiveness declined slightly over time, which might be related to the low dose used (Yu et al., 2021).

5.1.18.4 Potential to combine with other mitigation strategies

Good potential to combine with other strategies with different modes of action. Incremental mitigation effects on CH4 yield were reported for 3-NOP when combined with unsaturated lipids (Zhang et al., 2021), higher concentrate proportion (Schilde et al., 2021) and monensin ionophore (Vyas et al., 2018).

5.1.18.5 Effects on other emissions

The emissions from producing 3-NOP in small scale conditions have been reported as 48 to 52 kg CO_2eq/kg 3-NOP (Alvarez-Hess et al., 2019; Feng and Kebreab, 2021). Based on recent industrial optimization, the footprint of 3-NOP is estimated to be substantially reduced at approximately 25 kg CO_2eq/kg (DSM personal communication). Using the published figure of 50 kg CO_2eq, a mean addition of 3-NOP of 118 mg/kg diet DM (E. Kebreab, pers. comm.) would represent approximately 6 g CO_2eq/kg diet DM. For example, for a dairy cow consuming 25 kg DM/d and emitting 274 g/d of CH_4 (~100 kg/yr), the increase in CO_2eq emissions due to feeding 3-NOP would represent about 2 percent of the basal CO_2eq emissions from enteric CH_4 (calculations not shown), without considering emissions of CO_2eq from manure, N_2O, and fossil fuels. Nkemka et al. (2019) showed no residual effects of feeding 3-NOP to beef cattle on manure CH_4 emissions when used in an anaerobic digester. Owens et al. (2021) also established in field conditions that manure from cattle fed 3-NOP had unchanged emissions patterns. To further study emissions upon manure spreading, Weber et al. (2021) conducted a lab scale study using soils amended with manure from cattle fed 3-NOP and concluded that GHG emissions were dependent on soil texture. For coarse-textured soil (Black Chernozemic), GHG emissions were greater when amended with manure from cattle fed 3-NOP compared with control manure (mainly due to increased N_2O emissions), but this effect was not observed for other soil types or when the manure was first composted. This aspect thus needs further study.

5.1.18.6 Productivity and meat/milk/manure/crop/air quality

Studies report no negative effects of 3-NOP on digestibility, with possible small increases in digestibility in some cases (Zhang et al., 2012; van Gastelen et al., 2020). In most dairy studies, supplementing diets with 3-NOP has not led to improved animal performance (Arndt et al., 2021; Jayanegara et al., 2018). Supplementing dairy diets with 3-NOP (40 to 80 mg/kg DM) did not affect DMI, milk yield, milk component yield, or feed efficiency in most dairy studies although slightly increased body weight gain (Haisan et al., 2014; Hristov et al., 2015; van Gastelen et al., 2020) and small changes in milk components (Jayanegara et al., 2018; Schilde et al., 2021) were reported in some studies. Depending upon the diet and dose of 3-NOP (100 to 200 mg/kg DM), most beef cattle studies report a decrease in DMI of 2 to 6.5 percent (Alemu et al., 2020, 2021), with no negative effects on animal performance (Alemu et al., 2020, 2021; Vyas et al., 2016, 2018), except when feeding a high grain diet with high 3-NOP dose (200 mg/kg DM). Enhanced gain:feed ratio (by 2.5 percent to 5 percent) has been reported in some (Alemu et al., 2021; Vyas et al., 2016, 2018), but not all beef cattle studies (K. Beauchemin, pers. comm.). Impact of 3-NOP on rumen fermentation has been assessed in both beef and dairy cattle. A distinct shift towards greater propionate and butyrate concentration, and a reduction in the acetate:propionate concentration ratio upon supplementation with 3-NOP has been observed (Jayenegara et al., 2018). It has been hypothesized that this shift might lead to higher energy and glucose availability for the animal (Ungerfeld, 2013, 2018; Ungerfeld et al., 2022). Ruminal pH has also been shown to be higher with 3-NOP supplementation indicating less risk of rumen acidosis (Jayenegara et al., 2018).

5.1.18.7 Safety and health aspects

Safety of 3-NOP for animal use and humans consuming meat and milk from animals fed 3-NOP has been assessed on the basis of an extensive set of studies, by regulatory officials in the E.U. (Bampidis et al., 2021), and will need to be assessed by regulatory officials in other jurisdictions. The E.U. market authorization process for feed additives concluded the product to be safe for dairy cows, the consumer and the environment when given up to the maximum recommended dose of 88 mg 3-NOP/kg complete feed (with a DM content of 88 percent) (Bampidis et al., 2021). 3-NOP is rapidly hydrolyzed in the rumen post-dosing (2 to 3 h; Thiel et al., 2019a) to 1,3 propanediol and nitrate, which are low-toxicity compounds naturally occurring in the rumen of non 3NOP supplemented cows. 1,3 propanediol is further hydrolyzed and used in energy metabolism with 3-NOP carbon incorporated into carbohydrates, amino acids and fatty acids (Thiel et al., 2019a). In lactating goats, 3-NOP was shown to be extensively metabolized to CO_2, with less than 5 percent of ^{14}C-labeled 3-NOP excreted via urine and feces with minute quantities in milk lactose (Thiel et al., 2019a). 3-NOP and its metabolites are not expected in milk fat or protein because of their high water solubilities. Residues in beef meat have been shown to be minute or non-existent (Thiel et al., 2019a). Thiel et al. (2019b) reported that in rats 3-NOP and its metabolites pose no mutagenic and genotoxic potential.

5.1.18.8 Adoption potential

3-NOP is already approved for use in Brazil, Chile, and the E.U., and the authorization process is ongoing for other markets. It is commercially available in some markets and has clear potential for adoption by confinement systems using total or partial mixed rations. In its current form, 3-NOP may not be suitable for grazing ruminants because it is most effective when mixed into the ration such that it is consumed throughout the day (unless a lower efficacy or higher dosing is accepted) thus matching the fermentation of feed and production of methane. Preliminary studies using a prototype slow-release form of 3-NOP have proven successful (Muetzel et al., 2019) and will require further testing in larger scale studies. 3-NOP requires approval by regulatory officials. A survey regarding farmers/sector experience with 3-NOP in dairy diets will start in the Netherlands in 2022. In its current form, 3-NOP may not be suitable for grazing ruminants because it is most effective when mixed into the ration such that it is consumed throughout the day. Some advantages of using 3-NOP in rations are its low effective dose (1-2 g/d), high specificity towards methanogens, relatively sustained decrease in CH_4 over long periods of time (i.e., months), and safety. Including 3-NOP in animal diets will result in increased feed costs, as will be the case for many other CH_4 inhibitors, and unless there is an increment in the price of animal products produced with a lower carbon footprint, or a consistent improvement in animal performance, producers may not readily adopt the inclusion of 3-NOP in diets. We are not aware of the existence of surveys regarding consumer acceptance of 3-NOP.

5.1.18.9 Research required

Research is needed to develop a stable form of 3-NOP for grazing animals or a slow-release form that could be fed less frequently. The optimum dose in diets that vary in chemical composition needs refinement. Efficacy of using 3-NOP in long-term beef and dairy cow studies under various conditions needs further validation. Studies that combine 3-NOP with other mitigation strategies are needed. Further evaluation of the GHG emissions from manure of animals fed 3-NOP are needed although no negative impacts on digestibility have been observed. Research is also needed to achieve a more complete understanding of the changes in rumen and whole animal metabolism that result from inhibiting rumen methanogenesis, which may potentially be translatable into improved animal productivity.

5.1.19 Rumen manipulation: Immunization against methanogens

5.1.19.1 Description

Vaccination against rumen methanogens.

5.1.19.2 Mode of action

Stimulation of the ruminant´s immune system to produce antibodies against methanogens. Antibodies are delivered to the rumen via saliva.

5.1.19.3 Efficacy

Effects on CH_4 yield have been mild or non-existent in sheep (Wright et al., 2004; Leslie et al., 2008; Williams et al., 2009) and goats (Zhang et al., 2015). Antibodies against methanogens decreased CH_4 production in mixed rumen cultures in a non-peer reviewed study (Baker and Perth, 2000), and effects were variable in another mixed rumen cultures study (Cook et al., 2008). Growth and CH_4 production of a pure culture of *Methanobrevibacter ruminantium* were inhibited by antibodies against methanogens (Wedlock et al., 2010). In vivo studies with sheep have shown that vaccination with a model methanogen antigen increased antibodies concentrations in saliva, estimated to result with up to 10^4 molecules of antigen-specific IgG per methanogen cell in the rumen (Subharat et al., 2016).

5.1.19.4 Potential to combine with other mitigation strategies

Appears feasible, but experiments have not been conducted to investigate synergisms among mitigation strategies. If vaccines can be efficacious, use of other additives that directly target methanogens could represent duplicative mitigation.

5.1.19.5 Effects on other emissions

It seems likely that fossil fuels CO_2 emissions from manufacturing, packaging, transporting and storing vaccines would be minimal. It is assumed that digestibility and nutrient excretion will be unaltered by vaccination.

5.1.19.6 Productivity and meat/milk/manure/crop/air quality

There have been no effects of methanogens vaccines on DM intake and body mass gain (Wright et al., 2004; Williams et al., 2009). A non-peer review publication claimed greater DMI and wool growth in sheep vaccinated against methanogens (Baker and Perth, 1999). Effects on animal performance and product quality would need to be thoroughly evaluated if an efficacious vaccine is developed that clearly demonstrates a reduction in CH_4 emissions as other CH_4 mitigating measures normally do not show such beneficial effects.

5.1.19.7 Safety and health aspects

Unknown at present, but can be speculated to be low risk, as antibodies naturally exist in animal tissues eaten by humans. Developed vaccines would need to go through appropriate regulatory approval processes.

5.1.19.8 Adoption potential

This enteric CH_4 mitigation strategy is attractive for extensive grazing production systems with infrequent or no supplementation and limited potential for intensification. It is also interesting in that it is unlikely to significantly affect the emissions of other GHG, would be easy and safe to apply, would not require specialized technical skills, and would likely be acceptable to government agencies and consumers. If proven effective, development of vaccines against methanogens is perhaps the most desirable approach for controlling CH_4 emissions from extensive ruminant production systems.

5.1.19.9 Research required

This CH_4 mitigation strategy is at present at a proof-of-concept stage, as shown by responses elicited by vaccination on antibodies in serum, saliva and rumen fluid (Wright et al., 2004; Zhang et al., 2015; Subharat et al., 2015, 2016). Identification of membrane-associated and surface-exposed proteins present in a broad range of rumen methanogens that can serve as antigens is necessary to develop a successful vaccine. Sequencing of genomes of rumen methanogens has been useful to identify potential antigens (Leahy et al., 2013; Wedlock et al., 2013). Vaccination against methanogens has been shown to induce the production of antibodies in saliva and their delivery to the rumen (Subharat et al., 2015, 2016). Antibodies against methanogens have been shown to have some stability in rumen fluid (Subharat et al., 2015) and to agglutinate methanogens in vitro (Wedlock et al., 2010). However, even though individual steps in the development of vaccines against methanogens seem to have been successful, in vivo effects on CH_4 production have so far been small or non-existent (Baca-González et al., 2020). It is somewhat surprising that there are more peer-reviewed studies in which CH_4 production has been examined in vivo (Wright et al., 2004; Leslie et al., 2008; Williams et al., 2009; Zhang et al., 2015) than in mixed cultures (Cook et al., 2008). Vaccination did not affect methanogens abundance but increased methanogens diversity, suggesting that lack of effects on CH_4 production might be due to lack of broad spectrum of the vaccine on the rumen methanogenic community (Williams et al., 2009). More work is required to select appropriate antigens present across diverse rumen methanogens clades, assess their efficacy against cultivable rumen methanogens and in vitro mixed batch and continuous cultures, develop adequate adjuvants, and assess the persistence of immune responses across ruminant populations.

5.1.20 Rumen manipulation: Bromoform-containing seaweeds (Asparagopsis sp.)

5.1.20.1 Description

Some red seaweeds (macroalgae) inhibit methanogenesis due to their capacity to synthesize and accumulate halogenated compounds, such as bromoform and dibromochloromethane (Machado et al., 2016). Two red seaweeds, *Asparagopsis taxiformis* and *A. armata*, have shown high inhibitory effects on CH_4 production *in vitro* and *in vivo* (Kinley et al., 2016; Li et al., 2016; Roque et al., 2019, 2021; Stefenoni et al., 2021).

5.1.20.2 Mode of action

The anti-methanogenic property of *Asparagopsis* is due to its content of halogenated compounds, of which bromoform is the most abundant (Machado et al., 2016). Halogenated CH_4 analogues react with vitamin B_{12} to block cobamide-dependent methyl transfer (Wood et al., 1968) into mercaptoethanesulfonate (coenzyme M) to produce methyl-coenzyme M, which is in turn the methyl donor in the last step of methanogenesis (Harms and Thauer, 1996).

5.1.20.3 Efficacy

In vivo studies with sheep, steers and dairy cows reported dose-dependent decreases between 9 to 98 percent of CH_4 production by supplementing *Asparagopsis* to the diet (Li et al., 2016; Kinley et al., 2020; Roque et al., 2019, 2021; Stefenoni et al., 2021). Severe inhibition of methanogenesis (>50 percent) was observed with 1 percent or less *Asparagopsis* in the diet DM (Li et al., 2016; Kinley et al., 2020; Roque et al., 2019, 2021). Efficacy of *Asparagopsis* on CH_4 mitigation depends on its concentration of bromoform, which ranged from 3.28 to 39 x 10^{-3} µg/kg DMI in different studies (Kinley et al., 2020; Roque et al., 2019, 2021). Additionally, *Asparagopsis* may be more effective at decreasing CH_4 production in high concentrate than in high forage diets (Roque et al., 2021). Efficacy of *Asparagopsis* was lost over time (Stefenoni et al., 2021), and it was concluded that this was probably due to instability and loss of bromoform from the product with time, rather than an adaptation of the rumen microbes, although this should be investigated further. However, over a 5 month period, Roque et al. (2021) did not report loss of efficacy from the product.

5.1.20.4 Potential to combine with other mitigation strategies

The combination with other mitigation strategies was not experimentally examined but is expected to have potential when different bioactive components or modes of action are involved. Combination with other CH_4 mitigation strategies may allow decreasing the concentration of bromoform in the diet to alleviate potential detrimental effects on DMI, health and safety (see Safety and Health aspects below).

5.1.20.5 Effects on other emissions

The CO_2eq emissions of growing, harvesting, processing (drying), storing, and transporting *Asparagopsis* at a large scale need to be considered in a life cycle assessment to determine the net impact on GHG intensity of meat and milk production. Assessment of ozone related environmental impacts probably deserves considerations as bromoform is documented as an ozone-depleting substance (Papanastasiou et al., 2014); in a preprint currently under review, the potential global depletion of stratospheric ozone was estimated to be relatively small for Asparagopsis growth conditions in Australia (Jia et al., 2021).

5.1.20.6 Productivity and meat/milk/manure/crop/air quality

Dietary supplementation with *Asparagopsis* reduced feed intake in most (Roque et al., 2019, 2021; Stefenoni et al., 2021; Muizelaar et al., 2021) but not all (Kinley et al., 2020), experiments. A feed supplement containing *Asparagopsis* was rejected by some individual sheep (Li et al., 2016) and dairy cows (Muizelaar et al., 2021) fed high levels of seaweed. *Asparagopsis* increased (Kinley et al., 2020) or did not affect (Roque et al., 2021) body mass gain of steers, although in both studies feed efficiency was improved due to reduction of feed intake. There were no effects of *Asparagopsis* inclusion in the diet on carcass or meat quality (Kinley et al., 2020; Roque et al., 2021). Milk production was decreased by supplementation with 1 percent DM *Asparagopsis* because of DMI reduction (Roque et al., 2019; Stefenoni et al., 2021). Effect of *Asparagopsis* on manure GHG emission is unknown.

5.1.20.7 Safety and health aspects

Long-term oral exposure of animals to bromoform can cause liver and intestinal tumors. Therefore, it is classified in the U.S. as a Group B2, probable human carcinogen (EPA, 2000). Bromoform residues were not detected in meat, fat, organs or feces from sheep and beef fed *Asparagopsis* (Li et al., 2016; Kinley, et al., 2020; Roque et al., 2021), but accumulation of iodine in meat was reported (Roque et al., 2021). Roque et al. (2019) and Stefenoni et al. (2021) did not find passage of bromoform to milk from dairy cows fed *Asparagopsis*, although Muizelaar et al. (2021) detected bromoform in milk of cows offered *Asparagopsis* in samples from some experimental days. Iodine and bromide accumulated in milk from cows fed *Asparagopsis* (Stefenoni et al., 2021). The ruminal mucosa of animals that consumed *Asparagopsis* showed pathological signs in sheep (Li et al., 2016) and cows (Muizelaar et al., 2021).

5.1.20.8 Adoption potential

Bromoform-containing seaweed will not be ready for adoption until various challenges are successfully addressed, notably potential safety risks for animals and humans. So far, in vivo studies have used wild-harvested *Asparagopsis*, with variable bromoform content (Vijn et al., 2020). Adoption will require consistent growth and processing of seaweed species to accumulate halogenated compounds and maintain their concentrations throughout transporting, handling, and animal feeding. Paradoxically, on the other hand, bromoform and other halogenated alkanes are a concern for animals, food, and environmental safety, which will need to be resolved for adoption to occur. Passage of bromoform to milk in animals fed *Asparagopsis* is inconsistent (Roque et al., 2019; Stefenoni et al., 2021; Muizelaar et al., 2021). Animal and food safety concerns relative to transfer and accumulation in milk and meat of iodine and bromide will also need to be examined and successfully addressed for adoption of seaweed-based mitigants. If inclusion of bromoform-containing seaweeds at low levels deemed to be safe could be regarded as acceptable, administration of pure bromoform in other forms (e.g. slow release) could equally be considered. This criterion might be extended to other haloalkanes inhibitory of methanogenesis, such as chloroform and bromochloromethane, although an encapsulated form of bromochloromethane was regarded as unlikely to be allowed for commercial use in Australia (Tomkins et al, 2009). Finally, inclusion of *Asparagopsis* in animal feed, adoption as with any other additive, will imply an extra cost, therefore, cost effectiveness must be considered.

5.1.20.9 Research required

More *in vivo* research is needed to determine CH_4 mitigation and productivity changes under different diet and management conditions. Effective methods for growing, processing, and storing *Asparagopsis*, how to improve its palatability and the best delivery methods will need to be established. Growth conditions that promote bromoform are key to *Asparagopsis* efficacy, but in contradiction, potential hazards of bromoform to the animals, farmers, consumers and the environment are a concern for its use. The metabolic fate of bromoform ingested by the animal and the partition of ingested bromide among the different excreta (via feces, urine, milk and exhalation) needs to be established. The bromide-containing compounds present in milk (Stefenoni et al., 2021) need to be identified to determine possible risks to consumers. Safety issues associated with iodine and heavy metals also need to be addressed. It is recommended to study the combination of *Asparagopsis* with other CH_4 mitigation strategies. Bromoform volatilization from production sites should be prevented.

5.1.21 Rumen manipulation: Other seaweeds

5.1.21.1 Description

Seaweeds (macroalgae) other than *Asparagopsis* may inhibit methanogenesis due to the presence of specific bioactive components, but the research on these alternative seaweeds is limited mainly to *in vitro* studies (as reviewed by Abbott et al., 2021).

5.1.21.2 Mode of action

Seaweeds have highly variable chemical composition, depending upon the species, time of collection, and growth environment. The anti-methanogenic property of these alternative seaweeds may be due to low concentrations of bromoform and numerous other bioactive constituents, including polysaccharides, proteins, peptides, bacteriocins, lipids, phlorotannins (i.e., similar to condensed tannins found in terrestrial plants; only found in brown seaweeds), saponins, and alkaloids (Morais et al., 2020; Abbott et al., 2020). These compounds are known to decrease CH_4 production by suppressing archaea and protozoa, resulting in a shift in rumen fermentation pathways, and in some cases an undesirable decrease in substrate degradability. Some of these seaweeds produce bromoform, but store less in biomass than *Asparagopsis* (Carpenter and Liss, 2000): *Laminaria digitata* (brown), *Macrocystis pyrifera* (brown), *Pterocladia capillacea* (red), *Rhodymenia californica* (red), *Ulva intestinalis* (green), and *Ulva* spp. (green).

5.1.21.3 Efficacy

Several seaweeds have been identified as having high *in vitro* CH_4 mitigation potential (>50 percent decrease): *Cladophora patentiramea* (green), *Cystoseira trinodis* (brown), *Dictyota bartayresii* (brown), *Gigartina* spp. (red), *Padina australis* (brown) and *Ulva* spp. (green) (Dubois et al., 2013; Machado et al., 2014; Maia et al., 2016). Red and brown seaweeds seem to have greater effects on CH_4 production than green seaweeds (McCauley et al., 2020). *In vivo* efficacy is unknown and needs to be investigated.

5.1.21.4 Potential to combine with other mitigation strategies

Good potential to combine with other strategies with different bioactive components or modes of action. Negative interactions might occur if combined with compounds with similar modes of action.

5.1.21.5 Effects on other emissions

The CO_2eq emissions of growing, harvesting, processing (drying), storing, and transporting seaweed at a large scale needs to be considered in LCA to determine the net impact on GHG intensity of meat and milk production (McCauley et al., 2020). The importance of upstream emissions of CO_2 would depend on the percentage of seaweed inclusion in the diet. There is also potential to purify or extract seaweed bioactives, which would decrease emissions related to drying and transportation. The potential fixation of CO_2 through photosynthesis has been regarded as a benefit towards mitigation in the emission of GHG (McCauley et al., 2020); however, this is likely a minor benefit as most of the CO_2 would be released to the atmosphere by the animal or humans consuming animal products, as is the case with other feedstuffs.

5.1.21.6 Productivity and meat/milk/manure/crop/air quality

The nutritive value of seaweed varies considerably depending on their composition and animal adaptation; thus, would need to be evaluated *in vivo* for any seaweed found to have antimethanogenic potential. Low doses (<2 percent of DM) may perhaps not affect ration intake, digestibility or amount of manure excreted; however, phlorotannin-containing seaweeds may shift nitrogen excretion from urine to feces (Antaya et al., 2019). Protein digestibility was lower in a brown than for a red seaweed (Abbott et al., 2020). High mineral concentration limits digestible OM concentration in many seaweeds. Beneficial effects such as improvement of immune and antioxidative status and inhibition of pathogens has been reported (Makkar et al., 2016), but this is probably highly species-dependent. Quality of animal products may be positively affected by seaweed through increasing the content of beneficial fatty acids (McCauley et al., 2020).

5.1.21.7 Safety and health aspects

Seaweeds tend to concentrate minerals, specifically heavy metals such as arsenic and copper, as well as iodine and nitrate, therefore the safety and health impacts need to be determined for each seaweed (Makkar et al., 2016; Abbott et al., 2020; McCauley et al., 2020; Morais et al., 2020). High iodine concentration has been found in milk of cows fed the brown seaweed *Ascophyllum nodosum* (Antaya et al., 2015), a finding that limits adoption potential for dairy cows. Health problems have been reported in sheep naturally adapted to consuming large amounts of seaweeds in coastal areas (Makkar et al., 2016). Potential toxicity and residues in meat and milk will depend on the content of toxic minerals and the level of inclusion of seaweeds in the diet.

5.1.21.8 Adoption potential

Immediate adoption potential is low, but there is good future potential especially in coastal areas with native seaweeds where they may be consumed wet. Otherwise seaweed needs to be rapidly dried before becoming mouldy. Low temperature drying reduces inactivation of biochemical compounds (Makkar et al., 2016). Sustainable production of seaweeds will be needed to meet the demand (Abbott et al., 2020). Palatability due to high salt content and toxicity may be limitations particularly when offered free choice or to grazing ruminants. It might be most effective to incorporate seaweeds into a total mixed ration or extract the bioactives such that they can be used as a feed additive. Adoption will depend on cost: benefit analysis and regional availability. Approval by government agencies will depend on the content of potentially toxic minerals, which may have to be analyzed from batch to batch unless consistent composition can be shown. Inclusion of seaweeds in ruminant diets may be acceptable to consumers if there is no risk of toxicity and no off-flavors in meat or milk.

5.1.21.9 Research required

Several aspects need to be considered for the use of seaweed to reduce enteric CH_4 emission (Vijn et al., 2020). Substantial *in vivo* research is needed to determine CH_4 mitigation potential and environmental impacts of seaweed farming. Bioactive compounds and the growth conditions that promote these bioactives is key. Palatability, best delivery method, quality control and potential to extract bioactive compounds will need to be established. Safety issues associated with high concentrations of certain bioactives, iodine, and heavy metals need to be addressed. A comparison with synthetically derived, identical bioactive compounds needs investigation.

5.1.22 Rumen manipulation: Defaunation

5.1.22.1 Description

Some rumen methanogens are ecto- (Vogels et al., 1980) or endosymbionts (Finlay et al., 1994) of protozoa, which supply them with H_2 and formate. It has been proposed that elimination of protozoa would cause the loss of their methanogenic symbionts, resulting in a decrease in CH_4 production in the rumen. Protozoa can be eliminated from the rumen using chemicals, lipids, by freezing rumen contents, or by isolating newborn animals (Newbold et al., 2015). In this section, we discuss defaunation targeting the elimination of rumen protozoa, rather than the decrease in protozoal numbers through the addition of phytochemicals such as saponins and tannins, or ionophores such as monensin. Those rumen manipulation strategies are discussed in other sections.

5.1.22.2 Mode of action

Protozoa do not dispose of metabolic hydrogen in propionate production (Goopy, 2019), and the removal by symbiotic methanogens of the H_2 and formate that they produce favors carbohydrate fermentation. It has been estimated that protozoa-associated methanogens contribute between 9 and 37 percent of CH_4 produced in rumen fermentation (Newbold et al., 1995, 2015). The presence of protozoa is not strictly necessary to rumen function and animal survival (Morgavi, et al., 2010; Newbold et al., 2015), therefore, their elimination has been proposed as a means of decreasing enteric CH_4 production through the simultaneous removal of symbiotic methanogens. Defaunation does not have a clear effect on the abundance of total methanogens (Huws et al., 2020), but protozoa-associated methanogens seem to be more active CH_4 producers than free-living methanogens (Jami and Mizrahi, 2020). Protozoa may also favor methanogens by protecting them from oxygen toxicity (Morgavi et al., 2010).

5.1.22.3 Efficacy

From a summary of *in vivo* and *in vitro* experiments, Hegarty (1999) concluded that eliminating protozoa resulted in an average decrease of 13 percent in CH_4 production, which was not solely due to removal of protozoa-associated methanogens. Meta-analyses of *in vivo* experiments with cattle, sheep and goats by Morgavi et al. (2010), Newbold et al. (2015) and Li et al. (2018) found that defaunation caused decreases in CH_4 production of 10 to 11 percent, although this was highly variable. The meta-analysis by Veneman et al. (2016) reported an average 17 percent (4 to 29 percent CI_{95}) decrease in CH_4 yield. The recent meta-analysis by Arndt et al. (2021) reported that defaunation resulted in decreases of 10 and 20 percent in absolute CH_4 production and yield, respectively. Linear relationships between protozoal numbers and CH_4 yield have been reported (Morgavi et al., 2010; Guyader et al., 2014). The meta-analysis by Li et al. (2018) suggested long-term adaptation of CH_4 production to defaunation. In agreement, Morgavi et al. (2012) showed a numerical decrease in CH_4 production of short term-defaunated wethers, and a numerical increase in CH_4 production in wethers that had been defaunated for more than two years. In contrast, previous work had not found long term adaptation to defaunation (Morgavi et al., 2008).

5.1.22.4 Potential to combine with other mitigation strategies

Not much is known about the interactions of defaunation with other CH_4 mitigation strategies. Defaunation interacted with nitrate supplementation, with nitrate decreasing CH_4 yield in faunated sheep, but numerically increasing it in defaunated animals (Nguyen et al., 2016). With regards to chemical inhibitors of methanogenesis, it was speculated that free-living rumen methanogens may be more resistant to 2-bromoethanesulfonate than protozoal symbionts, conferring defaunated rumen fluid resistance to this inhibitor of methanogenesis (Ungerfeld et al., 2004).

5.1.22.5 Effects on other emissions

Because defaunation can improve the efficiency of N utilization and decrease N elimination in urine (Eugene et al., 2004; Newbold et al., 2015), defaunation may decrease the emissions of N_2O from N voided to the environment in animal urine. Fiber excreted in manure may increase, as defaunation has been shown to decrease fiber digestibility (Eugene et al., 2004; Newbold et al., 2015; Li et al., 2018).

5.1.22.6 Productivity and meat/milk/manure/crop/air quality

The meta-analyses by Eugene et al. (2004) and Newbold et al. (2015) reported beneficial effects of defaunation on weight gain, feed conversion efficiency, and wool production, with no effects (Eugene et al., 2004) or decreased DMI (Newbold et al., 2015). The meta-analysis by Arndt et al. (2021) reported no effects of defaunation on DMI or weight gain. There were consistent decreases in rumen and overall tract OM and NDF digestibility and rumen VFA and ammonia concentration, and increased microbial nitrogen production, and a shift in nitrogen excretion from urine to feces (Eugene et al., 2004; Newbold et al., 2015; Li et al., 2018). Decreases in fiber digestibility can partially account for the decrease in CH_4 production caused by defaunation (Firkins et al., 2020). Beneficial effects on animal performance were more important with high-forage, low quality diets (Eugene et al., 2004). Protozoal numbers have been found to associate positively with DMI and NDF digestibility (Guyader et al., 2014). Defaunation decreased biohydrogenation of polyunsaturated fatty acids (Newbold et al., 2015).

5.1.22.7 Safety and health aspects

Through engulfing starch grains and metabolizing lactate, protozoa can help maintain a more stable rumen pH when feeding highly fermentable diets thereby preventing acidosis (Eugene et al., 2004; Newbold et al., 2015). There is no evidence to suggest that defaunation could harm the animal´s health, the environment, or pose a risk to the consumption of animal products.

5.1.22.8 Adoption potential

Defaunation results in mild decreases in CH_4 emissions. In addition, it is difficult to defaunate and maintain defaunated animals in production settings. Hence, defaunation has not been recommended as a CH_4 mitigation strategy for practical application (Hristov et al., 2013; Newbold et al., 2015; Huws et al., 2020).

5.1.22.9 Research required

There are differences among protozoa regarding their associated methanogens and contribution to CH_4 production, as well as in their cellulolytic capacity (Morgavi et al., 2010; Firkins et al., 2020) and bacterial predatory activity (Newbold et al., 2015). Targeting specifically the order Vestibuliferida has been suggested as a research direction because of the high CH_4 producing and low fiber degrading activity of this order (Huws et al., 2020), but this type of "fine tuning" protozoal manipulation strategies are not available at present. There is a need for further refinement in the understanding of how different protozoal taxa affect methanogenesis, intra-ruminal nitrogen recycling, fiber digestion, utilization of soluble carbohydrates, oxygen scavenging, as well as their rumen sequestration and passage (Firkins and Mackie, 2020).

5.1.23 Rumen manipulation: Alternative electron acceptors

5.1.23.1 Description

Dietary supplementation with organic and inorganic compounds that draw electrons away from methanogenesis towards alternative hydrogenotrophic pathways in rumen fermentation.

5.1.23.2 Mode of action

Organic alternative electron acceptors are carboxylic acid intermediates of rumen fermentation pathways, which, either incorporate metabolic hydrogen themselves (fumarate, which is reduced to succinate in the propionate randomizing pathway), or can be metabolized to compounds which incorporate metabolic hydrogen (malate, which is dehydrated to fumarate; acrylate, which can be esterified to acrylyl-CoA and incorporated into the propionate non-randomizing pathway; crotonate, which can be esterified to crotonyl-CoA and incorporated into butyrate formation; Russell, 2002; Carro and Ungerfeld, 2015; Ungerfeld and Hackmann, 2020). Importantly, the resulting electron sinks (propionate and butyrate) are absorbed through the rumen wall and have a nutritional value for ruminants.

Inorganic alternative electron acceptors are strong anions, which, when added as salts to the diet, dissociate and when they are reduced draw electrons away from CH_4 formation. Nitrate complete reduction yields primarily ammonium, which can be incorporated into microbial N or absorbed through the rumen wall. Nitrate reduction intermediate nitrite also exerts direct inhibition of methanogens (Hulshof et al., 2012; Latham et al., 2016; Yang et al., 2016). Reduction of sulfate yields hydrogen sulfide, which can be expelled as a gas (dissimilatory reduction) or be incorporated into microbial amino acids and co-factors (assimilatory reduction; Drewnowski et al., 2014).

For added alternative electron acceptors to draw metabolic hydrogen away from CH_4 formation, their reduction has to be thermodynamically more favorable than methanogenesis at the in vivo rumen concentration of all metabolites involved (Cord-Ruwisch et al., 1988; Ungerfeld and Kohn, 2006).

5.1.23.3 Efficacy

The mode of action of alternative electron acceptors imposes a theoretical limitation to their efficacy resulting from the stoichiometry of incorporation of metabolic hydrogen in their reduction. For example, the reduction of 1 mole of fumarate to 1 mole of succinate incorporates 1 mole of reducing equivalents ([2H]), which theoretically would suppress the formation of 0.25 mole of CH_4 through hydrogenotrophic methanogenesis ($CO_2 + 4\ H_2 \rightarrow CH_4 + 2\ H_2O$) (Carro and Ungerfeld, 2015). For example, a decrease of only 10 percent in CH_4 production of a cow producing 328 g/d (~500 L/d) of CH_4 would require the animal to ingest 1.4 kg/d of sodium fumarate, i.e., a considerable part of its diet (Newbold et al., 2005). Furthermore, meta-analyses of *in vitro* experiments have shown that the decreases in CH_4 production are below the theoretical expectation for fumarate and malate because fumarate and malate were apparently partially converted to acetate rather than to propionate, thus releasing, instead of incorporating, [2H] (Ungerfeld et al., 2007; Ungerfeld and Forster, 2011). *In vivo* results of fumarate and malate supplementation have produced variable results ranging from no effects in some studies to mild and moderate decreases in CH_4 production (i.e., 10 to 23 percent) in others (Carro and Ungerfeld, 2015). Wood et al. (2009) reported pronounced decreases in CH_4 production beyond what the stoichiometrical reduction of fumarate to succinate would result. It is possible that the inclusion of an elevated level (10 percent as fed) of highly fermentable fumaric acid in the diet further decreased CH_4 production and shifted fermentation to propionate (Janssen, 2010) beyond what fumarate reduction would allow.

Stoichiometrically, 4 moles of hydrogen are redirected toward reduction of 1 mole of nitrate, equivalent to 258.7 g CH_4 per kilogram of nitrate. The consumption of 173 g/d of sodium nitrate fully reduced to ammonium would decrease 10 percent of CH_4 emitted from a cow producing 328 g/d of CH_4 (~500 L/d; calculations not shown). This ideal stoichiometry is complicated by the incomplete reduction of nitrate, which would result in lower CH_4 decrease, and the direct toxicity of nitrate reduction intermediate nitrite on methanogens, which would increase the mitigation of CH_4 production. Nitrate supplementation consistently decreases CH_4 production *in vivo* in long term experiments (Lee and Beauchemin, 2014), including experiments as long as 407 consecutive days (Granja-Salcedo et al., 2019). From their meta-analysis, Lee and Beauchemin (2014) reported a linear decrease of 8.3 g of CH_4 per kilogram of DM intake per gram of nitrate ingested per kilogram of body mass and per day. In a later meta-analysis, Feng et al. (2020) reported that the mean dose of nitrate supplementation of 16.7 g/kg DM decreased CH_4 production by 13.9 percent on average, although this depended on nitrate dose, type of animal (greater efficacy in dairy than in beef cattle) and DMI. Efficacy diminished with increasing DMI. On average, their finding results in 364 g of sodium nitrate decreasing CH_4 production by 10 percent in a cow consuming 24 kg DM/d and producing 328 g/d of CH_4, i.e., about 50 percent of theoretical mitigation efficiency (calculations not shown). Mitigation efficiency can be greater in individual studies, e.g., Hulshof et al. (2012) achieved 87 percent efficiency.

5.1.23.4 Potential to combine with other mitigation strategies

The addition of fumarate (Tatsuoka et al., 2008; Ebrahimi et al., 2011) or malate (Mohammed et al., 2004) to *in vitro* incubations in which methanogenesis was inhibited helped redirect accumulated dihydrogen towards propionate formation; in contrast, the addition of butyrate precursors as electron acceptors did not relieve accumulation of dihydrogen through enhancing butyrate formation (Ungerfeld et al., 2006). The addition of fumarate to the diet of goats did not interact with the forage to concentrate ratio with respect to CH_4 production (Yang et al., 2012).

Inorganic electron acceptors nitrate and sulfate had additive effects on CH_4 decrease (van Zijderveld et al., 2010). Nitrate addition tended to negatively interact with linseed oil (Guyader et al., 2015) but interacted synergistically with canola oil (Villar et al., 2020). Nitrate interacted negatively with defaunation on CH_4 production (Nguyen et al., 2015). Addition of nitrite reducing bacterium *Propionibacterium acidipropionici* did not interact with nitrate addition (or had an effect) on CH_4 emissions from sheep (de Raphélis-Soissan, 2014). To compensate for the small reduction in feed consumption, diets with nitrate could be associated with supplementation with oils and fats, increasing both the energy density and the mitigation potential of the diet.

5.1.23.5 Effects on other emissions

Emissions of CO_2 from fossil fuels associated with manufacturing, or extracting and isolating fumarate and malate from natural sources, may not be negligible because of the relatively important dietary concentrations of these compounds that are needed to exert an effect on CH_4 emissions. Malate is naturally present in some forages at vegetative stages (Callaway et al., 1997); selecting varieties with high and sustained malate content and desirable agronomic traits could avoid additional CO_2 emissions.

Apart from emissions associated to the manufacture of nitrate salts, nitrate can be partially reduced to N_2O in the rumen (de Raphélis-Soissan et al., 2014; Petersen et al., 2015). Unless nitrate is supplemented to a N-deficient diet, nitrate should isonitrogenously replace another N source in order of not to increase N voided to the environment, which can potentially increase N_2O emissions (Beauchemin et al., 2020).

5.1.23.6 Productivity and meat/milk/manure/crop/air quality

Malate can help prevent acute acidosis by stimulating lactate utilization by *Selenomonas ruminantium*, as well as ameliorate subclinical acidosis. Malate and fumarate decreased biohydrogenation of linoleic and linolenic acids in vitro, and increased production of rumenic acid, which may potentially improve nutritional qualities of animal products. Most studies have not shown effects of moderate inclusion of malate on DMI, whilst fumarate effects have been more inconsistent, with decreased DMI in some studies and lack of effect in others. Malate supplementation did not affect weight gain or milk production in some studies and improved them in others, whilst fumarate supplementation has not affected milk production (Carro and Ungerfeld, 2015).

Overall, benefits of nitrate supplementation on animal productivity have not been demonstrated (Yang et al., 2016), unless nitrate is added to nitrogen deficient diets (Nguyen et al., 2015). Wang et al. (2018) found that replacing urea by nitrate on an isonitrogenous basis in a low protein content diet increased microbial N production and milk yield, which may be related to additional microbial ATP generation resulting from nitrate reduction (Yang et al., 2016).

5.1.23.7 Safety and health aspects

Fumarate and malate are natural intermediates of rumen fermentation. They are regarded as safe, and registered as animal feed ingredients in the E.U. and U.S. (Carro and Ungerfeld, 2015). Nitrate´s fermentation intermediate nitrite is absorbed through the rumen wall and enters blood circulation, reacting with hemoglobin to produce methaemoglobin, which cannot carry oxygen. Nitrate poisoning can be fatal, but it is possible to adapt the rumen gradually to increase the rate of reduction of nitrite to ammonium (Lee and Beauchemin, 2014; Yang et al., 2016). Traces of nitrate have been found in tissues (Doreau et al., 2018) and milk (Guyader et al., 2016) of animals fed nitrate but have not been deemed dangerous to consumers (Beauchemin et al., 2016). The inclusion of nitrate in animal feeds is not approved in the U.S. and Canada (Beauchemin et al., 2020). In Australia, carbon credits can be obtained by feeding nitrate to beef cattle (https://www.legislation.gov.au/Details/F2015C00580). High dietary sulfate results in hydrogen sulfide production, which can cause polioencephalomalacia (Drewnoski et al., 2014).

5.1.23.8 Adoption potential

Feeding fumarate and malate to ruminants is largely limited by cost because of the level of inclusion in the diet needed to obtain an effect on CH_4 mitigation, and their inconsistent effects on animal performance. Nitrate supplementation requires gradual adaptation of animals and can only be recommended for farms in which feed intake is carefully managed. Also nitrate content of herbage and forages needs to be taken into account to prevent too high levels. The potential increase in N_2O emissions as a consequence of feeding increased N levels should be carefully assessed. It was estimated that supplementing nitrate instead of urea as a non-protein N source would be more than twice as expensive (Callaghan et al., 2014).

5.1.23.9 Research required

In vivo experiments with combinations of inhibitors of methanogenesis and fumarate or malate to examine the incorporation of accumulated dihydrogen into propionate production would be of interest. Selection of grasses with malate content that stays high through maturity can be a possible route of supplementation. Efforts to decrease nitrite accumulation by adding nitrite-reducing bacteria have been successful in *in vitro* experiments (Sar et al., 2005a, b) but only numerically lowered nitrite and methaemoglobin concentration in plasma *in vivo* (de Raphelis-Soissan et al., 2014). More efforts to examine different doses and frequencies of administration of nitrite-reducing bacteria are recommended, as well as isolation of new nitrite reducers from the rumen environment.

5.1.24 Rumen manipulation: Essential oils

5.1.24.1 Description

Essential oils are complex mixtures of volatile lipophilic secondary metabolites. Traditionally extracted from plants by boiling water and steam distillation, methods also include solvent extraction, supercritical CO_2 extraction, and expression extraction. They are plant specific and are responsible for a plant's characteristic flavour and fragrance (Benchaar and Greathead, 2011). Essential oils can be extracted from many parts of a plant, including the leaves, flowers, stem, seeds, roots and bark

(Benchaar et al., 2008). When extracted and concentrated, essential oils may exert antimicrobial activities against a wide variety of microorganisms including bacteria, protozoa, and fungi (Dean and Ritchie, 1987; Sivropoulou et al., 1996; Chao et al., 2000). In addition to plant sources, essential oils can be chemically synthesized for commercial use. Chemically, essential oils are variable mixtures of principally terpenoids, mainly monoterpenes and sesquiterpenes, although diterpenes may also be present, and a variety of low molecular weight aliphatic hydrocarbons, acids, alcohols, aldehydes, acyclic esters or lactones and exceptionally N- and S-containing compounds, coumarins and homologues of phenylpropanoids (Dorman and Deans, 2000).

5.1.24.2 Mode of action

Most essential oils exert their antimicrobial activities by interacting with processes associated with the bacterial cell membrane, including electron transport, ion gradients, protein translocation, phosphorylation, and other enzyme-dependent reactions (Ultee et al., 1999; Dorman and Deans, 2000). Gram-positive bacteria appear to be more susceptible to the antibacterial properties of essential oils than gram-negative bacteria. The resistance of gram-negative bacteria to the antimicrobial activity of essential oils is due to an outer layer surrounding their cell wall that acts as a permeability barrier, limiting the access of hydrophobic compounds of essential oils (Burt, 2004). However, phenolic compounds (e.g., thymol and caravacrol contained in some essential oils (e.g., thyme and oregano)) can inhibit the growth of gram-negative bacteria by disrupting the outer cell membrane (Helander et al., 1998). It seems that the small molecular weight of essential oils allows them to penetrate the inner membrane of gram-negative bacteria (Nikaido, 1994; Dorman and Deans, 2000). Ruminal gram-positive bacteria are involved in fermentation processes that produce, among other end products, acetate, butyrate, formate, lactate, hydrogen, and ammonia (Russell and Strobel, 1989). Most of these fermentation processes are coupled with the production of CH_4, which is a reductive step required for disposing of reducing equivalents largely produced by this group of bacteria (Owens and Goetsch, 1988). On the other hand, gram-negative bacteria are involved in fermentation pathways associated with the production of propionate and succinate (Russell and Strobel, 1989; Russell, 1996). When this group of bacteria predominates in the rumen, rumen fermentation pattern shifts towards more propionate (H_2-using pathway) and less acetate (H_2-producing pathway) production, which reduces the availability of hydrogen for ruminal methanogenesis (Bergen and Bates, 1984). Neither methanogens nor protozoa (i.e., symbiotic relationship with methanogens) appear to be sensitive to the activity of essential oils (Benchaar and Greathead, 2011).

5.1.24.3 Efficacy

A number of essential oils (e.g., oregano, thyme), garlic oil and its derivatives have been shown to decrease CH_4 production *in vitro* (Cobellis et al., 2016) but results from *in vivo* studies have been less conclusive (Benchaar and Greathead, 2011). Essential oils with high content of phenolic compounds (e.g., thymol, carvacrol), garlic, and its active compounds (alliin, diallylsulphides and allicin) appear to be effective for CH_4 reduction in vitro, although their efficacy was not confirmed or was less pronounced in vivo (Klevenhusen et al., 2011; Zijderveld et al., 2011; Benchaar, 2020, 2021). Commercial products containing various essential oils have been shown in a very limited number of studies a potential to decrease CH_4 production. For instance, a commercial product of oregano oil (Orego Stim®, Anpario plc,

, Nottinghamshire, UK) fed to lactating dairy cows was reported to reduce CH_4 yield by 22 percent (Kolling et al., 2018). Feeding 15 g/day of a commercial product containing citrus extract and allicin from garlic (Mootral GmbH, Switzerland) to feedlot steers reduced enteric CH_4 yield by 23 percent, but only at the final week (week 12) of the study (Roque et al., 2019). A 10 percent decrease in CH_4 yield was reported for a commercial mixture of coriander, eugenol, geranyl acetate and geraniol (Agolin® Ruminant; Agolin SA, Bière, Switzerland) when fed to dairy cows at the rate of 1 g/day (Belanche et al., 2020). Based on literature data available to date, it appears that essential oils and their compounds may offer promise for CH_4 mitigation, but further animal feeding studies, especially long-term studies, are required to determine their efficacy.

5.1.24.4 Potential to combine with other mitigation strategies

Opportunities exist to combine with other mitigation strategies with different or similar mechanisms of action. For instance, given the lack of effects of essential oils on protozoa, combining these substances with other phytocompounds known for their antiprotozoal activity (e.g., saponins) may increase their mitigating effect. Monensin is known for its inhibitory effect of ruminal methanogenesis by inhibiting gram-positive bacteria (i.e., increase of propionate production at the expense of that of acetate), and thus, its combination with essential oils, which also inhibit the same group of bacteria may enhance the reducing effect on CH_4 production. Given that most essential oils do not act directly on methanogens, their combination with other direct inhibitors (e.g., chemical inhibitors) could contribute to mitigating effects.

5.1.24.5 Effects on other emissions

Some essential oils and their compounds have been reported to reduce dietary protein degradation in vitro but in vivo studies have been inconsistent (Cobellis et al., 2016). If this decrease is accompanied with a reduction in urinary N excretion, potential reduction in N_2O and ammonia may occur.

5.1.24.6 Productivity and meat/milk/manure/crop/air quality

In general feeding essential oils to ruminants does not affect animal productivity or product (milk, meat) quality (Benchaar et al., 2009). Adverse effects of essential oils on feed digestion were reported (Benchaar et al., 2009; Cobellis et al., 2016) and if such effects occur in animals, it would negatively affect productivity. There is potential for transfer of essential oils to animal products (milk, meat) exists (e.g., terpenes, garlic) which could affect the organoleptic properties of meat and milk. The amount and chemical composition of manure are likely not affected but if feed digestion in the rumen is depressed, the amount of manure excreted and associated emissions could increase.

5.1.24.7 Safety and health aspects

Little is known on the safety of use of essential oils in ruminant nutrition. At doses generally recommended by the feed industry, the probability of essential oils being toxic to animals is low. However, caution should be taken especially if essential oils are fed at high doses. For example, a number of essential oil components (e.g., carvacrol, cinnamaldehyde, eugenol, thymol) have been registered by the European Commission for use as flavourings in foodstuffs. However, essential oil compounds such as estragole and methyl eugenol were deleted from the list in 2001 due to genotoxic properties (Burt, 2004). The S-containing compounds in garlic have been shown to be responsible for haemotoxic effects in beef cattle (Rae, 1999) and horses (Pearson et al., 2005). The cytotoxicity of organo-sulphur compounds from garlic has been demonstrated as cell damage (Amagase, 2006). Use of essential oils as feed additives in livestock production must also be safe for the feed manufacturing personnel and farm workers. These substances have been reported to be potentially irritating and may cause allergic dermatitis (Burt, 2004), suggesting that caution should be taken by users in handling such feed additives.

5.1.24.8 Adoption potential

Because they are plant derived products, essential oils are considered safer than antibiotics or chemical additives. Essential oils have a wide spectrum of antimicrobial activity which makes it difficult to target specific microbial groups and can adversely affect feed digestion in the rumen. In addition, it has been reported that microbial populations are able to degrade and/or adapt to essential oils over time. The challenge remains to identify essential oils that selectively inhibit rumen methanogenesis, with lasting effects and without depressing feed digestion and animal productivity. Because essential oils are highly volatile, most commercial products are coated and formulated to control the release of the active ingredient once added to the animal's diet. However, long-term stability of products and need for controlled storage conditions can be a limiting factor. Finally, unless there are productivity benefits, adoption of essential oils may be discouraged because of additional cost.

5.1.24.9 Research required

The potential of essential oils to mitigate enteric CH_4 emission has been mostly examined in vitro and there is need to conduct more in vivo studies to determine the efficacy of essential oils. The range of essential oils available is extensive (>3000) and more work is needed to identify the most effective ones to reduce enteric CH_4 production. Many of the concentrations that have shown effects in vitro are too high for in vivo applications and thus, more research is warranted at optimal doses, under various dietary conditions allowing CH_4 mitigation without negatively affecting animal productivity. Furthermore, the favorable effects obtained in vitro can be because of microbial adaptation in vivo. Consequently, further long-term animal studies are needed to investigate the extent to which microbes adapt to these substances. Also, more work is required to assess the transfer of essential oils into animal products and the potential impact on quality of animal products.

5.1.25 Rumen manipulation: Tannin extracts

5.1.25.1 Description

Dietary supplement of tannin-rich extracts.

5.1.25.2 Mode of action

Tannins exert their anti-methanogenic effects by modifying the rumen microbial community and its function. As highlighted by Aboagye and Beauchemin (2019), several mechanisms have been proposed for the anti-methanogenic activity of tannins including direct inhibition of methanogens and the protozoal population associated with methanogens; decreasing hydrogen production through inhibition of fibrolytic bacteria and fibre digestibility, and acting as an alternative hydrogen sink to methanogenesis.

5.1.25.3 Efficacy

Tannins derived from vegetable sources can be classified into condensed (CT) and hydrolysable (HT) tannins. When tannins are extracted, both tannin types can be present at different concentrations depending on the plant part from which the extract was obtained, the plant maturity stage, and growing conditions. The anti-methanogenic effect of tannin-containing feeds is variable depending on factors such as the plant source, structural complexity (CT and HT have high and low molecular weights, respectively), dose, type of basal diets and ruminant species (Mueller-Harvey, 2006; Jayanegara, et al., 2012; Aboagye and Beauchemin, 2019). Feeding purified tannin-rich extracts compared to non-extracted tannins (i.e., tannins present in whole plants or agro-industrial by-products) could limit how other compounds confound with the anti-methanogenic activity of tannins. A meta-analysis of *in vitro* and *in vivo* studies showed decreased CH_4 production with increasing dietary tannin level, with a more consistent discernible effect observed when tannin inclusion was >20 g/kg dietary DM (Jayanegara, et al., 2012). Studies conducted in cattle, sheep and goats have shown effective anti-methanogenic activity when supplementing HT-rich extracts from *Acacia mearnsii* (Carulla, et al., 2005; Staerfl et al., 2012a; Alves et al., 2017; Denninger et al., 2020), CT from *Sericea lespedeza* plus quebracho extract (Liu et al., 2019) or a combination of HT and CT extracts from chestnut and quebracho (Duval et al., 2016; Aboagye et al., 2018). In these *in vivo* studies, the decrease in CH_4 emission ranged from 6 percent to 45 percent and the CH_4 mitigation effects were observed in both forage-based and concentrate-based diets. However, several studies have reported no effects on CH_4 emissions when supplementing CT extracts from quebracho and *Mimosa tenuiflora* (Beauchemin et al., 2007; Ebert et al., 2017; Lima et al., 2019) or HT extracts from chestnut and valonea (Śliwiński et al., 2002; Wischer et al., 2014). Nonetheless, supplementing tannin-rich extracts is a promising CH_4 mitigation strategy and there is evidence indicating that feeding tannins could exhibit long-term CH_4 mitigating effects (Staerfl et al., 2012a; Duval et al., 2016; Aboagye et al., 2018).

5.1.25.4 Potential to combine with other mitigation strategies

It appears feasible to combine tannin extracts with other CH_4 inhibitors but inconsistent additive effects on CH_4 reduction have been reported in some studies. The additive effects on CH_4 mitigation have been demonstrated when tannin extract from *Swietenia mahogani* was combined with *Sapindus* saponin extract *in vitro* (Jayanegara et al., 2020) and for supplementation of tannin extract from *Acacia mearnsii* with cottonseed oil in dairy cows (Williams et al., 2020). However, studies conducted in sheep and goats have reported no additive anti-methanogenic effect when tannin extract from *Acacia mearnsii* was combined with nitrate (Adejoro et al., 2020), *Mimosa tenuiflora* extract was combined with soybean oil (Lima et al., 2019) and when tannins from *Sericea lespedeza* plus quebracho extract were combined with monensin, soybean oil or coconut oil (Liu et al., 2019).

5.1.25.5 Effects on other emissions

If tannin supplementation decreases fibre digestibility, excretion of fermentable OM would be expected to increase, which might increase CH_4 losses from manure (Gerber et al., 2013). However, Staerfl et al.

(2012a) showed that feeding acacia tannin extract reduced fibre digestibility without affecting CH_4 emission from manure. Tannins have been demonstrated to inhibit manure CH_4 emission when ingested or added directly to manure (Whitehead et al., 2013; Pham et al., 2017). Therefore, the anti-methanogenic effect of ingested tannins may persist in manure. Additionally, numerous studies (especially those involving high-protein diets) have shown that tannins bind and interact with dietary proteins in the GIT, which improves N utilization and decreases urinary N losses (Mueller-Harvey, 2006; Waghorn, 2008; Aboagye and Beauchemin, 2019). Consequently, manure ammonia and N_2O emissions are decreased (Powell et al., 2011; Duval et al., 2016).

5.1.25.6 Productivity and meat/milk/manure/crop/air quality

Tannin-containing feeds can be less palatable because tannins bind to salivary glycoproteins resulting in an astringent taste (Mueller-Harvey, 2006). Moreover, feeding high concentrations of tannins (i.e., >50 g/kg DM) can elicit anti-nutritive properties that decrease intake, fibre and protein digestibility, and animal performance (Aboagye and Beauchemin, 2019). Supplementing purified tannin extract rather than non-extracted tannins can limit the interaction between tannin characteristics and the nutritional composition of the diet, thereby reducing the confounding effect on animal performance (Aboagye and Beauchemin, 2019). To avoid the negative effects of tannins, it has been recommended feeding a low to moderate dosage threshold (i.e., <30 to 50 g/kg DM diet), which can improve animal performance (weight gain and milk yield), prevent bloat, enhance N utilization, control intestinal parasites and mitigate enteric CH_4 emissions (Mueller-Harvey, 2006; Waghorn, 2008; Patra and Saxena, 2011). Moreover, dietary supplementation of tannins can improve the fatty acid composition and oxidative stability and sensory qualities of meat and milk (Salami et al., 2019; Frutos et al., 2020).

5.1.25.7 Safety and health aspects

Compared to CTs, HTs are more susceptible to microbial hydrolysis in the gut, producing metabolites that may elicit potential toxic effects to the animal post-absorption (Reed, 1995; McSweeney et al., 2001). Feeding high concentration (i.e., >50 g/kg DM diet) of HTs may cause toxic effects including liver necrosis, kidney damage, hemorrhagic gastroenteritis, and even mortality (Reed, 1995). Feeding a high concentration of CT may also affect the intestinal mucosa thereby decreasing the absorption of essential nutrients such as amino acids, which in turn could increase the risk of toxicity to plant compounds such as cyanogenic glycosides (Reed, 1995). The negative effect of tannins, particularly HT, can be prevented by gradual adaptation and continuous feeding or feeding lower concentrations (i.e., <50 g/kg DM diet) (Aboagye and Beauchemin, 2019). Tannins have not been shown to pose a risk to the safety of animal products for human consumption.

5.1.25.8 Adoption potential

Tannins are secondary metabolites naturally present in plants. Production of tannin extracts is scalable and some tannin extracts (e.g., extracts from tara, mimosa, quebracho, gambier, pine and chestnut plants) are currently produced on a commercial scale for different applications in the wood, dyeing, leather, and wine industries (Fraga-Corral et al., 2020). Tannin extracts can be easily incorporated into the diets of animals in intensive/confined feeding systems. Tannins are safe to apply and do not require specialized technical skills to implement; care should be taken not to apply excessive doses that could compromise digestibility and nutrient utilization. Because they are plant-based, in most jurisdictions tannin extracts are subject to a less lengthy regulatory approval process compared with chemical feed additives, even though risks on negative side effects are present.

5.1.25.9 Research required

More research is required to elucidate how the structural complexity of HT and CT extracts influence their anti-methanogenic activity, and to identify the optimum concentration of specific sources of tannin extracts for reducing CH_4 emission without negative impacts on animal performance. Moreover, further studies should focus on developing an effective combination of tannin extracts with other CH_4 inhibitors that could exhibit additive and long-term enteric CH_4 mitigating effects. The effect of supplemental tannins on manure CH_4 emission needs to be clarified for different types of basal diets and the mechanism of such anti-methanogenic needs to be understood. The ability of tannins to reduce N losses and N_2O emissions indicates that a LCA approach needs to be used when implementing this CH_4 mitigation strategy.

5.1.26 Rumen manipulation: Saponins

5.1.26.1 Description

Dietary supplementation of saponin-containing plants or saponin-rich extracts.

5.1.26.2 Mode of action

The anti-methanogenic effect of saponins is mainly related to their ability to inhibit the rumen protozoa population (which indirectly decreases the protozoa-associated methanogens), altering ruminal fermentation by promoting the production of propionate and reducing the availability of hydrogen for methanogenesis (Jayanegara et al., 2014; Patra and Saxena, 2009a). Additionally, the anti-methanogenic activity of saponins could be directly related to a decrease in the activity and number of methanogens (Patra and Saxena, 2009a).

5.1.26.3 Efficacy

The CH_4 mitigating effect of saponins is highly variable depending on the source, chemical structure and dosage of saponins, diet composition, and adaptation of the rumen microbes to saponins (Goel and Makkar, 2012; Patra and Saxena, 2009b). Most *in vitro* and *in vivo* studies have shown that *Sapindus* saponins, tea saponins, *Quillaja* saponins, *Yucca* saponins, lucerne saponins, and *Sesbania* saponins decreased CH_4 production although some studies have reported no effects (Patra and Saxena, 2009a; Goel and Makkar, 2012; Ramírez-Restrepo et al., 2017; Jafari, et al., 2019). A meta-analysis of *in vitro* studies found that CH_4 production decreased with increasing levels of saponins and the anti-methanogenic effectiveness of saponin sources numerically followed the order: *Yucca* > tea > *Quillaja*, although no statistical difference was observed between them (Jayanegara et al., 2014). The inconsistent anti-methanogenic effect of saponins may be partly associated with their transient anti-protozoal effect (Wina, et al., 2005) due to inactivation of saponins that occur through deglycosylation to sapogenins by rumen microbes (Newbold et al., 1997; Teferedegne et al., 1999). Thus, maintaining the anti-protozoal activity of saponins in the rumen would be a strategy to improve the consistency of their anti-methanogenic effects. The maintenance of anti-protozoal activity could be achieved by combining saponins with glycosidase inhibitors to avoid deglycosylation (Ramos-Morales et al., 2017b) or through modification of the chemical structure of saponins to prevent enzymatic cleavage for microbial degradation (Ramos-Morales et al., 2017a).

5.1.26.4 Potential to combine with other mitigation strategies

Saponins may be combined with other CH_4 inhibitors that have complementary mechanisms of action on methanogenesis. However, some studies suggest that this synergistic anti-methanogenic effect may depend on the saponin source. *In vitro* studies have shown that supplementing a low dose of *Quillaja* saponins in forage-based and concentrate-based diets exhibited an additive CH_4 mitigating effect when combined with garlic oil, nitrate or both without adverse effect on feed digestion and rumen fermentation (Patra and Yu, 2013; Patra and Yu, 2014; Patra and Yu, 2015a,b). Additionally, additive anti-methanogenic effects were observed *in vitro* for *Quillaja* saponin combined with nitrate and sulfate (Patra and Yu, 2014) and *Sapindus* saponin combined with tannin extract of *Swietenia mahogani* (Jayanegara et al., 2020). However, no additive CH_4 mitigating effect was found when tea saponin was combined with soybean oil (Mao et al., 2010) or fumarate (Yuan, et al., 2007) in sheep diets.

5.1.26.5 Effects on other emissions

Saponins could reduce rumen NH_3 concentration and improve N utilization efficiency possibly due to their NH_3-adsorption property and anti-protozoal activity which reduces proteolysis and deamination of dietary proteins in the rumen (Wina, et al., 2005; Patra and Saxena, 2009a). Consequently, feeding saponins, particularly *Yucca* saponins, has the potentials to reduce NH_3 emissions from manure although this effect has been inconsistent in some studies (Li and Powers, 2012; Sun et al., 2017; Adegbeye et al., 2019). Moreover, the positive effect of saponins at improving N use efficiency could reduce manure N losses and N_2O emission (Yurtseven et al., 2018).

5.1.26.6 Productivity and meat/milk/manure/crop/air quality

The inclusion of saponins at an appropriate level in the diet might not have negative effects on animal performance. A meta-analysis of *in vitro* studies found that dietary inclusion of higher saponin levels did not have negative effects on feed digestion and rumen fermentation (Jayanegara et al., 2014). Although the benefits of saponins on animal productivity is variable, their anti-protozoal effect could increase the efficiency of microbial protein synthesis, and enhance the supply of metabolizable protein and improve animal performance especially for roughage-based diets (Wina et al., 2005; Patra and Saxena, 2009a). Moreover, there are indications that dietary saponins could exhibit functional effects such as antioxidant and anti-inflammatory activities, that could reduce oxidative stress, improve immunity and animal health (Zhou, et al., 2012; Wang et al., 2017) and hence indirectly contribute to lower emissions. Additionally, supplementing dietary saponins could potentially improve the fatty acid profile and oxidative stability of ruminant meat, although limited improvements have been observed in milk quality (Vasta and Luciano, 2011; Szczechowiak et al., 2016; Toral et al., 2018).

5.1.26.7 Safety and health aspects

Saponins have not been shown to pose a risk to the safety of animal products for human consumption. However, saponins (mostly steroidal saponins) from some plants can be toxic to animals, causing photosensitization followed by liver and kidney degeneration and intestinal problems such as gastroenteritis and diarrhoea (Wina et al., 2005). An overview of toxic saponin-containing plants has been provided by Wina et al. (2005). Nonetheless, saponins might be subjected to less stringent regulatory approval than chemical inhibitors because they are plant derived.

5.1.26.8 Adoption potential

Supplementation ruminant diets with saponin-containing plants or extracts is readily available for adoption. Production of saponin extracts is scalable and some saponin extracts (e.g., Yucca and Quillaja bark saponins) have been commercially produced for applications in the pharmaceutical, food and cosmetic industries (Güçlü-Üstündağ and Mazza, 2007). At least one patent exists involving the use of saponins in ruminant feeding (Aoun et al., 2003). Saponins are safe to apply and do not require specialized technical skills for formulation of diets.

5.1.26.9 Research required

Yucca, tea and *Quillaja* saponins have shown potential to reduce CH_4 emissions but more studies are required to identify the optimum dosage and interaction with basal diets that will enhance understanding of their anti-methanogenic effects in the long-term. The combination of *Quillaja* saponin with other methanogenesis inhibitors (particularly nitrate) appears promising to achieve a greater anti-methanogenic effect but *in vivo* studies are required to confirm their synergistic CH_4 mitigating effects in ruminants. The potential of saponins to reduce N losses from animals, and manure NH_3 and N_2O emissions from manure requires further investigation. Potential interaction of saponins with other emissions (NH_3 and N_2O) apart from enteric CH_4 suggests that this CH_4 mitigating strategy should be examined using LCA.

5.1.27 Rumen manipulation: Biochar

5.1.27.1 Description

Dietary supplementation with biochar. Biochar is formed as a result of the pyrolysis (350–600 °C with limited oxygen) of various biomass sources.

5.1.27.2 Mode of action

Biochar has been proposed to enhance biofilm formation (Leng, 2014) and hydrogen transfer among members within microbial communities (Chen et al., 2014). Transfer of dihydrogen to acceptors other than CO_2 could result in a reduction in enteric CH_4 emissions.

5.1.27.3 Efficacy

Addition of biochar at 2 percent of dietary DM suggested that it could lower CH_4 emissions from an artificial rumen system (Saleem et al., 2018), but subsequent studies using other sources of biomass failed to detect any impact of biochar on CH_4 emissions in rumen continuous culture systems (Tamayao et al., 2021a, b). Subsequent studies using beef cattle also failed to detect any impact of biochar on CH_4 emissions in finishing cattle (Terry et al., 2019; Sperber et al., 2021). Efficacy of biochar may be influenced by biomass source as well as pyrolysis conditions and by secondary treatment of the biochar with acidic or alkali solutions. As biochar appears to be largely indigestible by mixed rumen cultures (Tamayao et al., 2022) reductions in CH_4 emissions could be associated with a depression in digestibility if biochar comprises a significant proportion of the diet.

5.1.27.4 Potential to combine with other mitigation strategies

Synergistic responses of combining biochar with biofat, an industrial byproduct of cashew nuts shell, have been reported to reduce CH_4 emissions in vitro (Saenab et al., 2020), but synergism with other mitigation strategies have not been reported in vivo.

5.1.27.5 Effects on other emissions

Depending on pyrolysis conditions and emissions capture, formation of biochar can release variable amounts of CO_2, CH_4 and N_2O (Sparrevik et al., 2015). Addition of biochar to ruminant diets may increase the level of recalcitrant carbon in manure and increase stable carbon levels (Romero et al,. 2021) and reduce N_2O emissions from soils (Shakoor et al., 2021). In contrast, direct addition of biochar to stored liquid manure was found to increase GHG emissions (Liu et al., 2021).

5.1.27.6 Productivity and meat/milk/manure/crop/air quality

Biochar has been shown to improve feed efficiency in lambs (Mirheidari et al., 2020) and carcass quality in beef cattle (Terry et al., 2020), but these responses do not appear to be accompanied by reductions in enteric CH_4 emissions.

5.1.27.7 Safety and health aspects

Biochar has been used as a feed coloring agent and as a chelator of toxins within the digestive tract of livestock. Biomass sources should be assessed for the presence of heavy metals, PCB's, dioxins or other potential toxicants before being used as feedstock for the production of biochar that could be fed to livestock.

5.1.27.8 Adoption potential

Biochar is available on the market and produced on an industrial scale as a soil amendment for use on farms and urban gardens. At this point it does not appear to possess enteric CH_4 mitigation properties, but is available in the marketplace should it reduce GHG emissions within the overall livestock production cycle. Ease of handling would improve if it was administered in a pelleted form and care must be taken owing to the explosive potential of biochar dust within confined spaces.

5.1.27.9 Research required

Biochar appears to have limited potential to lower enteric CH_4 emissions. Alternative biomass sources for pyrolysis and secondary chemical treatment of biochar could still be explored for their potential to reduce ruminal CH_4 emissions. Additional work should focus on the role that biochar can play in altering the chemical composition of manure including increasing the level of stable carbon that it can contribute to promote OM accumulation and the retention of manure nutrients within the plant root profile.

Exploration of the use of biochar to lower GHG emissions from livestock should be conducted from an LCA perspective with consideration of all emissions and sinks throughout the production chain. Long term, basic research could address understanding how the channeling of dihydrogen to different hydrogenotrophic microbial groups is controlled.

5.1.28 Rumen manipulation: Direct fed microbials

5.1.28.1 Description

Direct fed microbials, or live microbial additives, are viable microorganisms (e.g., fungi, yeasts, bacteria) which when ingested by a ruminant can modify rumen fermentation. For the present analysis, we will focus exclusively on the decrease of CH_4 production, although other objectives also exist, such as stabilizing rumen pH or improving lactate utilization or fiber digestion.

5.1.28.2 Mode of action

There can be various modes of action. Generally, the addition of a live microbial additive intends to redirect metabolic hydrogen away from CH_4 production and towards an alternative product of fermentation nutritionally useful to the host ruminant animal. This may be achieved through the incorporation of dihydrogen into pathways other than methanogenesis, or through the stimulation of pathways not producing dihydrogen, or through anaerobic CH_4 oxidation (Jeyanathan et al., 2013). For a live microbial additive to be successful, the pathway carried out by the added microorganism has to be thermodynamically feasible and the affinity for the reaction substrates has to be high; the supply of additional enzyme activity as a live microorganism to a thermodynamically non-spontaneous process will be ineffective. For example, a hydrogenotroph should have a low dihydrogen threshold and high affinity for dihydrogen to compete with hydrogenotrophic methanogens (Ungerfeld, 2020). Another possibility is adding live microbial additives producing bacteriocins capable of directly inhibiting methanogens (Gilbert et al., 2010; Jeyanathan et al., 2013).

5.1.28.3 Efficacy

Efficacy has been variable. Effects of yeasts, *Aspergillus oryzae*, and lactic acid bacteria on rumen fermentation and CH_4 production have been inconsistent, as they have not been selected to decrease CH_4 production (Jeyanathan et al., 2013; Weimer, 2015). A strategy investigated is the stimulation of propionate production as a pathway incorporating metabolic hydrogen (Jeyanathan et al., 2013, Elghandour et al., 2015). Some strains of propionibacteria have caused mild decreases in CH_4 production in in vitro batch cultures (Alazzeh et al., 2012). Mamuad et al. (2014) and Kim et al. (2016) observed stronger decreases in CH_4 production in in vitro batch cultures with the addition of fumarate reducers. In vivo experiments with propionibacteria found a numerical decrease in CH_4 production with a high forage (Vyas et al., 2014a) but not with a mixed (Vyas et al., 2015) or a high concentrate diet (Vyas et al., 2014b). A patent claims that a combination of a *Propionibacterium* and *Lactobacillus rhamnosus* strain 32 caused a 25 percent decrease in CH_4 production of lactating Holstein cows fed a mixed diet, with no effect in those fed a diet higher in starch (Berger et al., 2014). Addition of nitrate and sulfate-reducing bacteria have also decreased CH_4 production in vitro (Jeyanathan et al., 2013). The addition of reductive acetogens, which have the ability to reduce CO_2 with dihydrogen to produce acetate, to in vitro rumen fermentation, had minimal or no effects on CH_4 production if not accompanied with a chemical inhibitor of methanogenesis (Nollet et al., 1997; le Van et al., 1998; Lopez et al., 1999). The magnitude of CH_4 oxidation in the rumen was estimated as minimal (Jeyanathan et al., 2013).

5.1.28.4 Potential to combine with other mitigation strategies

The physicochemical mode of action of live microbial additives allows for a rational design of combinations with other CH_4 mitigation strategies. Live microbial additives can enhance the flow of metabolic hydrogen through desirable, thermodynamically feasible metabolic pathways, whose rate is constrained by enzyme kinetics (Ungerfeld, 2020). For example, inhibiting methanogenesis with chemical compounds in batch cultures allowed reductive acetogenesis conducted by added reductive acetogens to be functional (Nollet et al., 1997; le Van et al., 1998; Lopez et al., 1999). Nitrite- and nitrate-reducing bacteria have been successfully tested in various studies with in vitro cultures with added nitrate to decrease CH_4 production and enhance the rate of nitrite reduction to ammonium, and avoid the accumulation of nitrite (Jeyanathan et al., 2014), as the reductions of nitrate and sulfate are thermodynamically more favorable than methanogenesis in the rumen (Ungerfeld and Kohn, 2006). In an experiment with sheep fed nitrate as a CH_4 mitigation strategy, the addition of nitrite reducing bacterium *Propionibacterium acidipropionici* decreased plasma nitrite concentration only numerically (de Raphélis-Soissan, 2014). The addition of live succinate or propionate producers could also improve the conversion of added fumarate or malate to propionate.

5.1.28.5 Effects on other emissions

Growing, storing, and transporting live microbial additives would imply some emission of fossil fuels CO_2. Effects on the animal´s efficiency of N utilization would have to be evaluated. Overall, additional emissions of CO_2eq are presumably low.

5.1.28.6 Productivity and meat/milk/manure/crop/air quality

In vivo results of CH_4 mitigation experiments with direct fed (live) microbial additives are scarce. Vyas et al. (2014a, b, 2015) did not find an effect of the addition of propionibacteria to different diets on DMI or weight gain. Berger et al. (2014) did not find an effect of adding a *Propionibacterium* alone or in combination with one of two lactobacilli on DMI or production of milk or milk components of dairy cows.

5.1.28.7 Safety and health aspects

Approval by regulatory agencies is usually needed and necessitates the microorganism in question to be completely characterized and described, and the potential for pathogenicity discarded. Live microbial additives have been studied to prevent disorders such as acidosis and decrease the load of pathogens in cattle (Jeyanathan et al., 2013; Elghandour et al., 2015). The commercial availability and use of probiotics in human and domestic animal nutrition and health is widespread.

5.1.28.8 Adoption potential

The potential is good if consistent in vivo results can be obtained. In particular, live microbial additives have a very good potential as a companion manipulation of chemical inhibitors of methanogenesis, as they may be able to improve productivity in certain animal categories and diets through directing metabolic hydrogen accumulated as dihydrogen towards desirable products. Because their adoption would imply an additional feeding cost, it would be important to take advantage of possible changes in the absorption of metabolites to improve animal performance. Preparations of live microbial additives should maintain viability for prolonged periods of time, and be easy to use, store, and transport. With few exceptions, live microbial additives do not persist in the rumen and need to be dosed frequently to exert effects on digestion and fermentation (Weimer, 2015). In that regard, live microbial additives may not always be applicable to extensive beef production systems in which animals have only sporadic contact with their human keepers. For those systems, it will be necessary to evaluate the mitigation option to consider the limitations, if any.

5.1.28.9 Research required

Considerable in vitro and in vivo research on the optimization of rumen fermentation with live microbial additives is still needed, and in particular when combining them with other CH_4 mitigation strategies such as chemical inhibitors of methanogenesis and alternative electron acceptors. A rational approach that takes into consideration the possible physicochemical limitations of fermentation pathways is recommended. If live microbial additives can be demonstrated to be consistently effective, applied research to optimize the frequency, dose and mode of administration will be needed. An understanding of physiological and metabolic changes that can occur in the animal will be needed to optimize the production and absorption of metabolites to improve animal productivity.

5.1.29 Rumen manipulation: Early life interventions

5.1.29.1 Description

The use of interventions in preruminant animals during the establishment of the rumen microbiome aimed at decreasing enteric CH_4 emissions later in the lifetime of the animal.

5.1.29.2 Mode of action

Adult microbiota is resilient in that it recovers from perturbations after those perturbations cease (Weimer, 2015). In contrast, the newborn undergoes various stages of microbial colonization, and interventions at early life stages may modify and program the post-weaning and adult microbiota in favorable directions. Early life events can influence microbial composition post-weaning through rumen development, microbial establishment, and host immunity (Abecia et al., 2014, 2018; Yañez-Ruiz et al., 2015; Furman et al., 2020). The concept of early life electron redirection was illustrated by Fonty et al. (2007) with gnotobiotic lambs inoculated with reductive acetogens after birth, in which reductive acetogenesis remained to be the main hydrogenotrophic pathway up to 12 months of age.

5.1.29.3 Efficacy

Abecia et al. (2013) supplemented does after they gave birth to twin goat kids with the methanogenesis inhibitor BCM for two months. One kid per doe received BCM for three months after birth. Three months after the administration of BCM was discontinued, kids that had previously received BCM still produced 20 percent less CH_4 per kilogram of DMI than those who had not, although the decrease in CH_4 production was lesser than when the BCM treatment was stopped. The greatest efficacy occured when both kids and their mothers were supplemented with BCM. Meale et al. (2021) administered 3-NOP to calves until 14 weeks of age. At weaning at 11 weeks, CH_4 production was 10.4 percent lower in heifers receiving 3-NOP, and at one year of age, a 17.5 percent CH_4 decrease was still observed in those calves which had received 3-NOP early in life.

On the other hand, Debruyne et al. (2018) did not find long-lasting effects of coconut oil supplementation to goat kids up to 11 weeks of age on CH_4 production in incubations with rumen inoculum from control and treated animals, conducted with rumen inoculum taken from the lambs when they were 28 weeks old. Saro et al. (2018) did not find effects of administering a linseed and garlic oil mixture to lambs during their first 10 weeks of age on their CH_4 production at 20 weeks of age, although those lambs that received a second treatment with the linseed and garlic oil mixture decreased their CH_4 production.

5.1.29.4 Potential to combine with other mitigation strategies

There may be negative interactions between the same anti-methanogenic treatments administered early and again later in life: rumen inoculum from 6- and 12-months old calves which had been supplemented with extruded linseed from birth until four months of age responded less to the addition of linseed oil in vitro addition as a CH_4 mitigation additive compared to control rumen inoculum from calves which had not been supplemented extruded linseed (Ruiz-González et al., 2017). On the other hand, Saro et al. (2018) did not find interactions on CH_4 production between supplementing a linseed and garlic oil mixture at two stages in early life.

5.1.29.5 Effects on other emissions

Other CO_2eq emissions will likely be affected in the same direction that each particular intervention may affect other CO_2eq emissions later in life. However, the magnitude by which other emissions might be affected is expected to be considerably smaller, given that the treatment would be short lived and is conducted in young animals with a small body size.

5.1.29.6 Productivity and meat/milk/manure/crop/air quality

Effects on productivity are probably largely dependent on the intervention used. Abecia et al. (2013) reported greater weight gains and a tendency for decreased concentrate intake in goat kids supplemented with BCM; performance of the animals later in life was not reported. Supplementation of goat kids with coconut oil decreased body weight at 28 weeks of age (Debruyne et al., 2018). Saro et al. (2018) did not observe effects of supplementing lambs with a linseed-garlic oil combination during 10 weeks on weight gain at 10 and 20 weeks of age. Meale et al. (2021) did not observe effects of supplementing 3-NOP to calves during their first 14 weeks of life on weight gain between birth and 23 weeks or weeks 57 to 60, although there were numerical differences in favor of control animals, the same as with preweaning concentrate intake. It is possible that, given that early life treatments are applied for a relatively short period of time, negative effects on animal performance, if they occur, might be offset by compensatory growth.

5.1.29.7 Safety and health aspects

Potential consequences on safety and health effects of early life interventions will depend on the strategy used. However, there will be wash-out periods of several months during the growing phase of the animals before they produce milk or meat. Furthermore, doses of any additives would be much diminished in comparison with an adult animal with a much greater body size, which also diminishes potential negative environmental effects. Therefore, it is likely that additives that could impose unacceptable levels of risk to the environment or to consumers when fed to adult animals could be acceptable when administered to newborn animals, provided that they do not harm the young animal. Nonetheless, the safety of each early life intervention will have to be approved by regulatory agencies. Long-term efficacy of early life interventions in adult animals may also need to be demonstrated for their usage to be approved by regulatory agencies as a mitigating measure.

5.1.29.8 Adoption potential

The concept of early life interventions is very attractive in that the cost of applying long-lasting manipulations for a short period of time to animals with a small body size would be greatly diminished compared to adult animals, in which most interventions would have to be applied continuously. In addition, it may be safer to consumers and the environment to use smaller doses for shorter periods of time followed by long wash-out periods. Furthermore, this strategy may be advantageous for grazing ruminants where supplementation of feed additives is not possible. Research on early life interventions is at an early stage. There are few and contradictory results about the efficacy of early life interventions to decrease CH_4 production later in life and the persistency of the effects observed, although some recent results are encouraging (Meale et al., 2021). Efficacy likely depends on the additive or dietary modification used, the dose, mode and duration of the administration, and the animal species, among other factors.

5.1.29.9 Research required

The persistence in CH_4 decrease after one year in animals treated until 11 weeks of age (Meale et al., 2021) is of great interest, but these results need to be confirmed in further experiments. There is a need for more research to establish the most effective interventions and their optimal doses, modes and frequencies of administration, the minimal duration and the endpoint of each intervention, as well as the expected period of persistence of the effects on CH_4 production. It will also be very important to study the effects of each early life intervention on future animal performance and health. It will also be important to identify and understand the mechanisms involved, such as the permanent change in the establishment of the composition of the rumen microbiota, anatomical and functional changes in GIT development, and possible changes in the immune system (Yàñez-Ruiz et al., 2015).

5.1.30 Rumen manipulation: Phage and lytic enzymes with activity against methanogens

5.1.30.1 Description

Phage and the lytic enzymes they produce are being investigated for their activity against rumen methanogens as an enteric CH_4 mitigation strategy.

5.1.30.2 Mode of action

Archaeal phage produces lytic enzymes that breakdown pseudomurein, the principal cell wall component of rumen methanogens. This disruptive activity could reduce the production of CH_4 in the rumen.

5.1.30.3 Efficacy

A novel archaeal lytic enzyme (PeiR) displayed on bionanoparticles was shown to reduce CH_4 production in specific pure methanogen cultures by up to 97 percent over a period of 5 days (Altermann et al., 2018). Efficacy of the lytic enzyme decreased against methanogens that were more phylogenetically distant from *Methanobrevibacter ruminantium* M1, the original host of the provirus. No *in vivo* or mixed culture studies have been undertaken to investigate the ability of phage or their lytic enzymes to reduce ruminal CH_4 emissions.

5.1.30.4 Potential to combine with other mitigation strategies

Appears feasible, but experiments have not been conducted to investigate synergisms with other mitigation strategies. Synergism may be most likely with other mitigation strategies that specifically target those methanogens more distantly related *Methanobrevibacter ruminantium* M1 that lack sensitivity to administered phage or lytic enzymes.

5.1.30.5 Effects on other emissions

The infrastructure that would be required for phage or enzyme production would rely on the establishment of manufacturing facilities that would likely require the use of fossil fuels. Scale-up and production of phage or lytic enzymes to a commercial scale could prove challenging. It is assumed that phage would not alter N_2O emissions or the efficiency of milk or meat production, but such relationships have not been investigated as the technology has not been assessed outside of the laboratory.

5.1.30.6 Productivity and meat/milk/manure/crop/air quality

No studies have been undertaken to investigate the impact of phage or lytic enzymes on productivity parameters.

5.1.30.7 Safety and health aspects

Presumed to be low risk as phage are already used in therapeutic applications in medicine and food safety and none of the 65 known archaeal viruses have been linked to animal pathogenesis (Wirth and Young, 2020).

5.1.30.8 Adoption potential

This enteric CH_4 mitigation strategy would require administration of a phage or lytic enzymes on a continuous basis, making the technology more suitable for use with total mixed diets and less suitable for extensive grazing systems. The technology would be more desirable if lytic as opposed to temperate archaeal phages could be isolated, possibly enabling self-propagating biocontrol of ruminal methanogens. However, to date lytic phages with activity against rumen methanogens have not been identified and current candidate lytic enzymes have been identified as a result of the sequencing of prophage within the methanogen genome (Leahy et al., 2010).

5.1.30.9 Research required

This CH_4 mitigation strategy is not yet at the proof- of-concept stage, as the technology has not been investigated beyond its impact on pure cultures of rumen methanogens. Although it is well known that the rumen harbors a rich and diverse virome (Gilbert et al., 2020), there is only a single preliminary report of the isolation of an intact phage with potential activity against methanogens (Baresi and Bertani, 1984). Only three pseudomurein endoisopeptidases have been characterized for their potential activity against methanogens (Schofield et al., 2015; Altermann et al., 2018). Intact lytic phage are known to play a major role in the ecology of methanogens within other anaerobic habitats (Danovaro et al., 2016) and it is almost certain that they play a significant role in the ecology of rumen methanogens. More work is required to define the diversity of archaeal viruses (Coutinho et al., 2019) as they are likely underrepresented in genomic approaches that characterize the rumen virome. Efforts to identify lytic phage with activity against methanogens could be the next step in advancing this mitigation strategy, although it is likely that a cocktail of phages will be required to cover the complete complement of methanogens that reside in the rumen.

5.1.31 Summary tables

In the following summary tables (Table 2 to Table 4) we delineate the possibilities and hurdles for application of the various enteric CH_4 mitigation emissions for ruminants in three main production systems:

i) Confinement systems that include feedlots and dairies in which animals are penned or housed in drylots or buildings. In these non-grazing systems, all the feed ingested by the animals is provided by human operators. Feed ingredients can be many, including cereal grains, oilseeds and meals, conserved forages, byproducts, and premixes containing minerals, vitamins and additives. The feeding frequency and management (i.e., total mixed ration or feed components offered separately) is decided by the farm operator.

ii) Grazing with no supplementation. In these systems, animals ingest exclusively plants by grazing pastures. Extensive beef and sheep ranching systems are an example, although other dairy, beef and small ruminants production systems based on grazing pastures without supplementation are also considered within this category.

iii) Mixed grazing systems, in which grazing animals are supplemented concentrates and/or conserved forages. Typically, the proportion of total DMI by the animal by grazing pastures versus the proportion supplemented varies throughout the year with the pasture growth curve. In mixed dairy systems lactating cows are typically supplemented twice daily during milking. In other mixed grazing systems supplementation may take place once daily, although this can vary.

It is acknowledged that within each system there is ample variation depending on animal species and category, climate (tropical and subtropical, temperate), eco-zone, and so forth. The application of each enteric CH_4 mitigation strategy for each of the three production systems as shown in the tables is based on a qualitative assessment, as follows:

1. Available knowledge generated by applied research, indicating the number of existing peer-reviewed in vivo published studies in which the effects of the mitigation strategy on enteric CH_4 production have been reported (Column 1).

2. The magnitude of the change in CH_4 production, both on an absolute (per animal and per day) and intensity (per unit of animal product) basis (Column 2).

3. Average measured or likely effects of the application of an enteric CH_4 mitigation strategy on the emissions of other GHGs at other points of the production chain. Upstream changes may include the direct and indirect release of CO_2 and N_2O in the growth and manufacture of feeds, feed additives, or other products. Downstream changes may also occur in the emissions of CH_4 and N_2O from manure. Changes in crop production and grazing management can affect carbon sequestration in soils. In some cases changes in other GHGs have been found to be minimal whilst in others a life cycle assessment is recommended for a defined production unit such as a farm, region or country (Column 3).

4. Effects of the application of the enteric CH_4 mitigation strategy on animal productivity. Only those studies in which the effects of the mitigation strategy on enteric CH_4 and animal productivity were simultaneously measured and reported are considered (Column 4).

5. Present stage of technical development of an enteric CH_4 mitigation strategy. A mitigation strategy may be considered to be fully developed and available for adoption at the farm level, although further research to optimize its application may still be needed. Government approval and manufacture scaling and distribution may still be pending but those aspects are considered in the last column (as Government and Accessibility, respectively). Secondly, a mitigation strategy may be in its last stages of technical development and close to its practical application. Finally, some mitigation strategies are at an early stage of research and their application may potentially take place in the long term with a considerable degree of uncertainty depending on the outcome of future basic and applied research (Column 5).

6. Existing concerns with regards to potential toxicity to animals, human operators, residues in animal products, and the environment (Column 6).

7. Various aspects representing potential barriers to adoption of a mitigation strategy within a particular production system (Column 7). Those can be highly variable among countries, regions and farms.

Table 2. Summary of enteric methane mitigation strategies for confined ruminant (dairy/beef/other) systems (greater detail and references are provided in the text).

Strategy	In vivo research conducted on CH_4 mitigation F=few (< 5); S=some (5-10), M=many (>10)	Expected CH_4 decrease range H=≥25 percent, M=15-24 percent, L=≤15 percent, I=increase may be observed, U=unknown (not examined) ; V=variable		Effects on other GHG emissions U = upstream; M = Manure; Mi=minimal; Ma=major changes expected, needs LCA; Un=unknown; V=variable.	Animal productivity (meat & milk production, feed efficiency) I=increase D=decrease Nc=no change, U=unknown, V=variable	Technical availability R = available now, C = close to being available, U = long term or uncertain availability	Risk management D = max dose[1]; safety for A= animals, H = humans, F=food, E= environment, N = none, U=Unknown	Main barriers to adoption on-farm F, resistance to change[2, 3]; C, increased cost/lack of financial incentives; M, animals are managed sparingly A, accessibility; T, technical support[2]; G, government approval; CA, consumer acceptance; S, safety
		g/d	g/kg meat or milk					
Animal breeding and management								
Increased animal production	M	I	L[4]	Ma	I	R	N	C, T
Selection for low-methane producing animals	S	L	L	Mi	Nc	U	N	C, A, T
Increased feed efficiency	M	V	L	Ma	I	R	N	C, T
Improved animal health	F	I	L	Mi	V	R	N	C, T
Improved animal reproduction	F	I	L	Mi	I	R	N	F, C, T
Feed management, diet formulation and precision feeding								
Increased feeding level	M	I	L	Ma	I	R	N	C, T

Decreased forage to concentrate ratio	M	L	L	Ma	I	R	A	C, A, T
Concentrates sources and processing	M	L	L	Ma	V	R	A	C, A
Supplementation of lipids	M	M	M	Ma	I	R	N	C, A, T
Forages								
Forage storage and processing	S	I	L	Ma	I	R	N	C, T, A
Increased forage digestibility	M	I	L	Ma	I	R	N	C, T
Perennial legumes	F	L	L	Ma	V	R[5]	N	C, A, T
High starch forages	S	L	L	Ma	V/I	R	N	C, A, T
High sugar grasses	F	L	L	Ma	V	R[5]	N	C, A, T
Pastures and grazing management	Not applicable	-	-	-	-	-	-	-
Species (use of forbs, diverse mixtures)	Not applicable	-	-	-	-	-	-	-
Tannin-containing forages	S	L	L	Ma	V	R	D	C, A, T
Rumen manipulation								
Ionophores	M	L	L	Mi	I	R	D	C, G, CA
Chemical inhibitors of methane production	M[6]	H	H	Mi	Nc/V	U	D, A, H, F, E[7]	C, G, CA, S
3-Nitrooxypropanol (3-NOP)	M	H	H	Mi	Nc/V	C, R	D	C, G, CA

Immunization against methanogens	F	L	L	Mi	Nc	U	N	C, G, CA
Bromoform-containing seaweeds (*Asparagopsis* sp.)	S	H	H	Ma/U	V	R, C	D, A, H, F, E	C, A, G, CA, S
Other seaweeds	F	U/L	U	Ma/U	U	U	D, A, F, E, H	C, A, G, S, CA
Defaunation	M	L	L	Mi	I or Nc meat production and feed efficiency	U	N	C, A, T, G, CA
Alternative electron acceptors. I. Carboxylic acids	M	L	L	Ma	I or Nc meat and milk production	R, U	D	C, A, G, S, CA
Alternative electron acceptors. II. Inorganic electron acceptors	M	L to M	L to M	Ma	Nc	R, U	D, A, F, E	C, A, T, G, S, CA
Essential oils[8]	F	L	L	Mi	U/Nc	R[5]	D	C, A, T, G
Tannin extracts	F	L	L	M	V	R[5]	D	C, A, T, G
Saponins	F	L	L	Mi	U	U	U	C, A, T, G
Biochar	F	None to L	None to L	Ma	Nc	R	D	C, A, G
Direct-fed microbials	F	L	L	Mi	Nc	U[5]	N	C, A, T, G
Early life interventions	F	U	U	Mi	V/U	U	D, A	T,G,CA,S
Phage and, lytic enzymes with activity against methanogens	F	U	U	Mi	U	U	U	C, G, T, CA

[1]A maximum dose exists, although it may be unknown; [2]It is acknowledged that Resistance to change (F) and the need for Technical support (T) are highly subjective evaluations and will vary considerably among particular producers, but both aspects should be considered in decision making; [3]Resistance to change because of aversion to financial risk is considered under Cost (C); only aversion to technical change is considered under Resistance to change (F); [4]Low in the short term but can be high in the long term; [5]Some are

currently available in various markets, but few in vivo studies have shown consistent methane decrease. [6]Many in total, but some if only the most investigated compounds are considered; [7]Will depend on the chemical nature of the compound; [8]Highly variable chemical nature; need individual evaluation.

Table 3. Summary of enteric methane mitigation strategies for extensive pastoral/ranching systems (beef/dairy/other) based on grazing without supplementation (Greater detail and references are given in the text).

Mitigation strategy	**In vivo research conducted on CH_4 mitigation** F=few (< 5); S=some (5-10), M=many (>10)	**Expected CH_4 decrease range H=≥25 percent, M=15-24 percent, L=≤15 percent, I=increase may be observed, U=unknown (not examined) ; V=variable**		**Effects on other GHG emissions** U = upstream; M = Manure; Mi=minimal; Ma=major changes expected, needs LCA; Un=unknown; V=variable.	**Animal productivity (meat&milk production, feed efficiency)** I=increase; D=decrease; Nc=no change, U=unknown, V=variable	**Technical availability** R = available now, C = close to being available, U = long term or uncertain availability	**Risk management** D = max dose[1]; safety for A= animals, H = humans, F=food, E= environment, N = none, U=Unknown	**Main barriers to adoption on-farm** F= resistance to change[2, 3]; C= increased cost/lack of financial incentives; M, animals are managed sparingly, A= accessibility; T= technical support[2]; G= government approval; CA= consumer acceptance; S= safety
		g/d	**g/kg meat or milk**					
Animal breeding and management								
Increased animal production	F	I	L	Ma	I	R	N	C, T
Selection for low-methane producing animals	F	L	L	Mi	Nc	U	N	C, M, A, T
Increased feed efficiency	F	V	L	Ma	I	R	N	C, T
Improved animal health	F	I	L	Mi	I	R	N	C, M, T
Improved animal reproduction	F	I	L	Mi	Nc	R	N	F, C, M, T
Feed management, diet formulation and precision feeding								
Increased feeding level	F	I	L	Ma	I	R	N	C, T
Decreased forage to concentrate ratio	Not applicable	-	-	-	-	-	-	-

Concentrates sources and processing	Not applicable	-	-	-	-	-	-	-
Supplementation of lipids	Not applicable	-	-	-	-	-	-	-
Forages								
Forage storage and processing	Not applicable	-	-	-	-	-	-	-
Increased forage digestibility	F	I	L	Ma	I	R	N	C, T
Perennial legumes	F	I	L	Ma	I	R[4]	N	C, A, T
High starch forages	Not applicable	-	-	-	-	-	-	-
High sugar grasses	F	L	L	Ma	V	R[4]	N	C, A, T
Pastures and grazing management	F	I	L	Mi	I	R	N	F, C, M, T
Species (use of forbs, diverse mixtures)	F	V	L	Ma	I	R	N	C, A, T
Tannin-containing species	S	L	L	Ma	V	R	N	C, A, T
Rumen manipulation								
Ionophores	F	U	U	Mi	I	R	D	M, CA
Chemical inhibitors of methane production	F	U	U	Mi	U	U	D, A, H, F, E[5]	C, M, A, G, CA, S
3-Nitrooxypropanol (3-NOP)	F	U	U	Mi	U	U	D	C, M, G, CA
Immunization against methanogens	F	U	U	Mi	U	U	N	C, G

Bromoform-containing seaweeds (*Asparagopsis* sp.)	F	U	U	Ma	U	U	D, A, F, H, E	C, M, A, G, CA, S
Other seaweeds	F	U	U	Ma	U	U	D, A, F, E	C, M, A, G, S
Defaunation	F	U	U	Mi	U	U	N	C, M, A, T
Alternative electron acceptors. I. Carboxylic acids	F	L	L	Ma	Nc milk production	R	D	C, M, A, G
Alternative electron acceptors. II. Inorganic electron acceptors	S	L to M	L to M	Ma	Nc or I meat production	R	D, A, F, E	C, A, T, G, S
Essential oils[6]	F	L	L	Mi	U	R[4]	D	C, M, A, T, G
Tannin extracts	M	L	L	M	V	R	D	C, M, A, T, G
Saponins	F	L	L	Mi	U	U	U	C, M, A, T, G
Biochar	F	U	U	Ma	U	R	D, A	C, M, T, G
Direct-fed microbials	F	U	U	Mi	U	U[4]	N	C, M, A, T, G
Early life interventions	F	U	U	Mi	U	U	D, A	M, T
Phage and, lytic enzymes with activity against methanogens	F	U	U	Mi	U	U	U	C, M, T, G

[1]A maximum dose exists, although it may be unknown; [2]It is acknowledged that Resistance to change (F) and the need for Technical support (T) are highly subjective evaluations and will vary considerably among particular producers, but t is advised to consider both aspects for decision making; [3]Resistance to change because of aversion to financial risk is considered under Cost (C); only aversion to technical change is considered under Resistance to change (F); [4]Some are currently available in various markets, but few in vivo studies have shown consistent methane decrease.[5]Will depend on the chemical nature of the compound; [6]Highly variable chemical nature; need individual evaluation.

Table 4. Summary of enteric methane mitigation strategies for mixed grazing with supplementation of concentrates, by-products and conserved forages (greater detail and references are provided in the text).

Mitigation strategy	**In vivo research conducted on CH_4 mitigation F=few (< 5), S=some (5-10), M=many (>10)**	**Expected CH_4 decrease range H=≥25 percent, M=15-24 percent L=≤15 percent I=increase may be observed, U=unknown (not examined) ; V=variable**		**Effects on other GHG emissions U = upstream; M = Manure; Mi=minimal; Ma=major changes expected, needs LCA; Un=unknown; V=variable**	**Animal productivity (meat&milk production, feed efficiency) I=increase; D=decrease; Nc=no change, U=unknown; V=variable**	**Technical availability R = available now, C = close to being available, U = long term or uncertain availability**	**Risk management D = max dose[1]; safety for A= animals, H = humans, F=food, E= environment, N = none, U=Unknown**	**Main barriers to adoption on-farm F= resistance to change[2, 3]; C= increased cost/lack of financial incentives; M, animals are managed sparingly, A= accessibility;T=, technical support[3]; G= government approval; CA=consumer acceptance; S= safety**
		g/d	**g/kg meat or milk**					
Animal breeding and management								
Increased animal production	S	I	M[4]	Ma	I	R	N	C, T
Selection for low-methane producing animals	S	L	L	Mi	Nc	U	N	C, A, T
Increased feed efficiency	F	V	L	Ma	I	R	N	C, T
Improved animal health	F	V	L	Mi	I	R	N	C, T
Improved animal reproduction	F	I	L	Mi	Nc	R	N	F, C, T
Feed management, diet formulation and precision feeding								
Increased feeding level	S	I	M	Ma	I	R	N	C, T
Decreased forage to concentrate ratio	M	L	L	Ma	I	R	A	C, A, T
Concentrates sources and processing	F	V	L	Ma	V	C	A	C, A, T
Supplementation of lipids	F	L	L	Ma	Nc	C	N	C, A, T

Forages								
Forage storage and processing	F	I	L	Ma	I	R	N	C, A, T
Increased forage digestibility	M	I	L	Ma	I	R	N	C, T
Perennial legumes	F	I	L	Ma	U	R[5]	N	C, A, T
High starch forages	S	L	L	Ma	V	R	N	C, A, T
High sugar grasses	F	L	L	Ma	V	R[5]	N	C, A, T
Pastures and grazing management	S	I	L	Mi	I	R	N	F, C, T
Species (use of forbs, diverse mixtures)	F	L	L	Ma	U	R	N	C, A, T
Tannin-containing species	S	L	L	Ma	V	R	D	C, A, T
Rumen manipulation								
Ionophores	M	L	L	Mi	I	R	D	C, G, CA
Chemical inhibitors of methane production	F	U	U	Mi	U	U	D, A, H, F, E[6]	C, A, G, CA, S
3-Nitrooxypropanol (3-NOP)	F	H	H	Mi	Nc	C	D	C, G, CA
Immunization against methanogens	F	U	U	Mi	U	U	N	C, G
Bromoform-containing seaweeds (*Asparagopsis* sp.)	F	U	U	Ma	U	R	D, A, F, H, E	C, A, G, CA, S
Other seaweeds	F	U	U	Ma	U	U	D, A, F, E	C, A, G, S
Defaunation	F	U	U	Mi	U	U	N	C, A, T

Alternative electron acceptors. I. Carboxylic acids	F	U	U	Ma	U	R	D	C, A, G
Alternative electron acceptors. II. Inorganic electron acceptors	F	L to M	L to M	Ma	Nc	R	D, A, F, E	C, A, T, G, S
Essential oils[7]	F	L	L	Mi	U	R[5]	D	C, A, T, G
Tannin extracts	F	L	L	M	U	R	D	C, A, T, G
Saponins	F	L	L	Mi	U	U	N	C, A, T, G
Biochar	F	U	U	Ma	U	R	D, A	C, G
Direct-fed microbials	F	U	U	Mi	U	U[5]	N	A, C, T, G
Early life interventions	F	U	U	Mi	U	U	D, A	A, T, G
Phage and, lytic enzymes with activity against methanogens	F	U	U	Mi	U	U	U	C, G, T

[1]A maximum dose exists, although it may be unknown; [2]It is acknowledged that Resistance to change (F) and the need for Techical support (T) are highly subjective evaluations and will vary considerably among particular producers, but t is advised to consider both aspects for decision making; [3]Resistance to change because of aversion to financial risk is considered under Cost (C); only aversion to technical change is considered under Resistance to change (F); [4]Medium in the short term but can be high in the long term; [5]Some are currently available in various markets, but few in vivo studies have shown consistent methane decrease. [6]Will depend on the chemical nature of the compound; [7]Highly variable chemical nature; need individual evaluation.

5.2 Methane mitigation strategies from housing, manure management and land application

This section provides descriptions of current strategies to mitigate CH_4 emissions during the collection, storage, and utilization of animal manures. Because manure is often stored within livestock and poultry housing systems, some of these strategies address CH_4 mitigation from animal housing systems as well. Numerous strategies have been suggested to mitigate CH_4 emissions from manure. These strategies include the collection and capture of biogas (Clemens and Ahlgrimm, 2001), utilization of anaerobic digestion systems to maximize CH_4 production for collection and use as fuel (Clemens et al., 2006, Montes et al., 2013), frequent manure removal from animal housing or storage (Andersen et al., 2015), manure cooling (Ni et al., 2008), manure acidification (Petersen et al., 2012), addition of amendments that inhibit CH_4 production (Andersen et al., 2018), solids separation, use of biofilters and scrubbers, manure management systems that promote aerobic conditions (Montes et al., 2013), as well as land application and land management strategies. Environmental conditions such as temperature, pH, retention time, and favorable anaerobic conditions for methanogenic bacteria activity result in increased CH_4 production, while environments or the presence of inhibitory compounds that inhibit the growth of CH_4 producing bacteria, can reduce CH_4 production (Andersen, 2018).

Anaerobic digestion followed by biogas collection and utilization is one of the most effective means of reducing manure CH_4 emissions, if fugitive emissions are well controlled. Anaerobic digestion reduces the carbon (C) content in the manure (Parajuli R et al., 2018). Lowering the C content in the manure decreases readily available C and provides less energy to support the denitrifying bacteria, reducing the N_2O formation potential of digested manures applied to the soil (Montes, et. al., 2013). While not manure management strategies per se, CH_4 reduction strategies involving animal nutrition and grazing systems have been included in this section because they reduce the amount of manure produced, and hence the resulting emissions.

Table 5 provides a brief qualitative assessment of each listed strategy including mode of action, efficacy potential, current adoption potential and antagonistic effects on N_2O production. Potential efficacy ratings of Low, Medium and High are provided in Table 5, where Low represents a reported efficacy of up to 33 percent CH_4 mitigation, Medium is listed for > 33 percent and < 66 percent reported mitigation control and High is listed when a CH_4 mitigation efficacy of greater than 66 percent has been reported. This classification follows the system outlined in Maurer et al. (2016). In cases where differing mitigation efficacies have been reported, the range of potential efficacies has been listed, i.e., “Low to Medium” or “Medium to High”. A more detailed description for each strategy is included below Table 5. These descriptions provide more information about each strategy, including quantitative information, potential changes in ammonia (NH_3) emissions (increase or decrease), and reference publications for further study. While adoption potential ratings of Low to High have been included in Table 5, it is important to note that the adoption potential for a strategy within a specific county or region may be higher or lower due to local regulations or availability or cost of the technology compared to other area. Where this is the case, it is discussed in more depth in the section regarding the mitigation strategy in question.

The listing of available mitigation strategies is not a listing of Best Management Practices per se. A specific strategy may work well in one situation and be a poor choice in another. It should also be noted that while the focus of this report is on CH_4, some strategies that mitigate CH_4 result in the formation of other GHG emissions, i.e., N_2O, as well as increased NH_3 emissions. When a CH_4 mitigation strategy may result in the antagonistic production of N_2O emissions it has been noted in Table 5 as well as the strategy description. Complete assessment of CH_4 emissions for the whole manure management chain (from animal housing to land application) should be considered. It should be noted that some mitigation techniques may be combined for increased efficacy as well either on the same manure management step or at farm level. For example, anaerobic digestion could be combined with sub-surface injection manure land application to provide greater overall efficacy.

Table 5. Methane Mitigation Strategies from Housing, Manure Storage and Land Application.

Strategy	Mode of Action	Efficacy Potential	Current Adoption Potential	Antagonistic GHG Emission Effects
Biogas Collection and Utilization	Engineered system to collect and use biogas.	High if fugitive emissions are controlled.	High	No
Decrease manure storage temperature	Reduction in growth rate of methanogenic bacteria.	Low to Medium. 5 percent CH_4 reduction per 1°C drop in temperature below 20°C reported.	Low to Medium	No
Manure acidification	Reduction in growth rate of methanogenic bacteria	High if pH is reduced to below 6 and more.	High	No
Addition of methane inhibitors to manure (narasin, etc.)	Compounds cause changes to the microbial community that can inhibit CH_4 production.	Medium to High. Efficacy increases with increasing dosage.	High	May increase CH_4 production for 1st week following addition to stored manure.
Decreased storage interval	Shortened manure storage reduces CH_4 formation in storage.	Medium	Medium	Yes. Cumulative N_2O emissions may increase with an increased number of land application events.
Solids separation	Removal of carbon through volatile solids removal.	Low to High	High	No
Composting and Aeration	Aerobic process creates adverse conditions for CH_4 formation.	High	High	Yes. Composting process may create N_2O emissions.
Biofilter and Scrubbers	Methanotrophic bacteria oxidize CH_4.	Low	Medium	Yes. N_2O may be produced in biofilter.
Manure Incorporation and Injection	Soil serves as a CH_4 sink.	Negative to High depending on soil conditions.	High	Yes. N_2O emissions may increase under some soils conditions.

Manure Application Timing	Soil temperature and moisture content impact methanogenic bacteria activity.	Low	Medium	Yes. N_2O emissions may increase under some soils conditions, but they may be decreased under others
Nutritional Strategies	Reduction of the quantify of manure through improved feed conversion rate, linked to increased feed digestibility.	Medium	Medium	No

5.2.1 Biogas collection and utilization

5.2.1.1 Description

The enhanced production and engineered collection of CH_4 via biogas from manure can be used to reduce CH_4 emissions from animal manure storage. Biogas collection can be used on traditional manure storages or purpose-built anaerobic digestion systems can be used to increase CH_4 production for collection and use as an energy source.

5.2.1.2 Mode of action

Collection and utilization (flaring, engine combustion or injection into pipeline for distributed use) of CH_4 replaces the direct release of CH_4 into the atmosphere.

5.2.1.3 Efficacy

It is important to note that engineered manure anaerobic digestion systems can be expected to produce up to two orders of magnitude more CH_4 than traditional manure storage systems (Hilhorst et al., 2002). As such, fugitive emissions must be well controlled in order to achieve a reduction in CH_4 emissions through the use of anaerobic digesters. If manure is stored in a gas-tight structure, all CH_4 emissions from stored manure can be eliminated through the use of anaerobic digester systems (Clemmons et al., 2006). Similarly Maurer et al. (2016) reports CH_4 mitigation from anaerobic digestion to be “High”, meaning greater than 67 percent.

5.2.1.4 Potential to combine with other mitigation strategies

The use of some CH_4 mitigation strategies, such as manure acidification and addition of CH_4 inhibitors will reduce the conversion of carbon to CH_4, through manure anaerobic digestion. While a reduction in CH_4 production will not reduce the efficacy of biogas collection, strategies implemented upstream in of the anaerobic digestion that are antagonistic to CH_4 production should be avoided when anaerobic digestion will be utilized. Otherwise, this technology can be used in combination with most other mitigation strategies. The utilization of anaerobic digestion prior to manure land application is reported to reduce N_2O emissions following land application in some circumstances (Chadwick, et. al., 2011).

5.2.1.5 Effects on other emissions

The production of CO_2 is also increased during the anaerobic digestion process, but it is collected and utilized as a component of the biogas (Li at al., 2017). Tapping of CH_4 (as fuel) or converting (upgraded CH_4) can mitigate the GHG emissions. Digestate can also contribute to indirect GHG credits with respect to the equivalent chemical fertilizers it substitutes.

5.2.1.6 Productivity and meat/milk/manure/crop/air quality

No impact on meat or milk production. While the anaerobic digestion of manure does not remove nutrients, it will result in transposing manure nutrients from inorganic to more readily plant available organic forms. Sulfur in manure will result in hydrogen sulfide formation in the biogas and must be considered during biogas utilization.

5.2.1.7 Safety and health aspects

The CH_4 contained in biogas is flammable and safety considerations when dealing with a flammable gas must be followed. Methane is explosive when mixed with air at concentrations of 5 percent to 15 percent CH_4 in air.

5.2.1.8 Adoption potential

Manure anaerobic digestion technology is well developed and ready for use. It is readily adopted to liquid manure slurries and has a long usage history with both cattle and swine manures. The primary obstacle to the adoption of manure anaerobic digestion has been cost recovery given the cost of biogas production compared to other available energy sources (Beddoes et al., 2007; Torrijos, 2016).

5.2.1.9 Research required

No major research gaps exist.

5.2.2 Decreased manure storage temperature

5.2.2.1 Description

Active cooling of slurry areas can significantly reduce CH_4 emissions.

5.2.2.2 Mode of action

Temperature affects methanogenesis, thus lower temperatures decrease the activity of methanogens during manure storage.

5.2.2.3 Efficacy

Reducing manure storage temperature reduces methanogenic bacteria activity in stored manure and results in decreased CH_4 emissions (Montes, 2013). Reducing temperature in pig slurry storage tanks has been shown to reduce GHG emissions by 21 percent compared with uncontrolled manure storage (Sommer et. al., 2004). Hilhorst et. al. (2002) reported that reducing the manure storage temperature from 17°C to 10.2°C between resulted in a CH_4 emission reduction of 66 percent from swine manure slurry. For cattle slurry a reduction of 1-2°C resulting in a CH_4 emissions reduction of 5-10 percent.

5.2.2.4 Potential to combine with other mitigation strategies

Manure cooling can be additive with other mitigation strategies.

5.2.2.5 Effects on other emissions

Manure cooling can also mitigate NH_3 emissions (a precursor of N_2O emissions) from in-house manure storage.

5.2.2.6 Productivity and meat/milk/manure/crop/air quality

No impact on meat or milk production. Manure cooling can assist with mitigation of NH_3 emissions.

5.2.2.7 Safety and health aspects

No safety or health concerns.

5.2.2.8 Adoption potential

Controlling manure storage temperature is a technically feasible, albeit expensive (depending on climate) control strategy. It may be a cost-effective option if the exchanged heat can be used. Decreasing manure temperature to <10°C by removing the manure from the building and storing it outside in cold climates can reduce CH_4 emissions (Hilhorst et. al., 2002).

5.2.2.9 Research required

Most of the research has been done in the context of reducing NH_3 emission in house and the measured impact on CH_4 emissions has been limited. Additional demonstration of efficacy through the evaluation of CH_4 emission might be necessary at this specific scale. Thus, further development of cooling systems that could be easily implemented in different types of houses is still required.

5.2.3 Manure acidification through dietary measures

5.2.3.1 Description

Incorporating benzoic acid in the diet of pigs to decrease the pH of manure for NH_3 and CH_4 emission mitigation (pig slurry).

5.2.3.2 Mode of action

Benzoic acid used in the diets of piglets, pigs and sows is metabolized in the liver and excreted after conversion into hippuric acid by metabolic conjugation with the amino acid glycine (Bühler et al., 2006, Halas et al., 2010; Galassi et al., 2011). Hippuric acid has a low pH and the increased concentration in urine leads to a reduction of its pH.

5.2.3.3 Efficacy

The supplementation of diets fed to pigs for fattening with 0.7 percent of benzoic acid during the starter phase and 1.7 percent during the growing/finishing phase reduced urine pH by 1.81 and 2.46 points in the starting and growing/finishing phase, respectively (7.50 vs 5.69 and 7.48 vs 5.02, respectively). Consequently, the slurry pH was reduced by 0.48 and 0.78 points for each of the phases, respectively (den Brok, 1999). The urinary pH was reduced significantly with the incorporation of benzoic acid at a dose of 1 percent in pigs for fattening diet (6.4 $\pm$ 0.6 vs 7.3$\pm$ 0.2 for the test and control animals, respectively), while the reduction was not significant at an incorporation rate of 0.5 percent (Guingand et al., 2005). The addition of 1 percent benzoic in the diet of pigs for fattening reduced urinary pH by one pH unit, independent on the level of protein (7.93 vs 7.09 (low protein diets), 7.77 vs 6.76 (high protein diets), for the control and the test groups, respectively), through the increased concentration of hippuric acid in the urine (Bühler et al., 2006). Halas et al. (2010) showed a significant decrease of pH in both the urine (6.1 vs 7.0, for the test and control groups, respectively) and the feces (6.7 vs 7.2 for the test and the control groups respectively), when incorporating benzoic acid at 0.5 percent in their diet. Similarly, the pH of the slurry was reduced by 0.46 pH-points (8.43 vs 8.89) when adding 1 percent of benzoic acid in the diet of Italian heavy pigs (Galassi et al., 2011). While feeding benzoic acid to pigs is clearly effective in decreasing pH, its efficacy for reducing CH_4 emissions from manure has not been established. However, benzoic acid shows potential as a mitigation strategy given that direct acidification of manure slurry using sulfuric acid has been shown to substantially reduce CH_4 emissions.

5.2.3.4 Potential to combine with other mitigation strategies

Due to its unique mode of action, benzoic acid can be used with other mitigation techniques leading to the reduction of OM excretion. It can also be combined with other manure management strategies reducing CH_4, which are not dependent on the manure pH. The use of benzoic acid in feed may negatively influence anaerobic digestion.

5.2.3.5 Effects on other emissions

The reduction of urinary pH was systematically accompanied by a reduction of NH_3 emission, either in the house ambiance or in the exhaust air.

5.2.3.6 Productivity and meat/milk/manure/crop/air quality

In addition to the pH reduction, benzoic acid also improved weight gain and feed conversion rate.

5.2.3.7 Safety and health aspects

The use of benzoic acid is safe under the proposed conditions of use and has been registered in various countries. The reduction of NH_3 emission in animal housing provides additional safety and welfare benefits for the animals and farmers.

5.2.3.8 Adoption potential

As benzoic acid can be easily incorporated in pig feed, it can be adopted, when farmers are using compound feed or when producing feed on farm. Its positive impacts on animal productivity and welfare compensates usually for the cost of incorporation. The adoption of this strategy may be limited by the registration status of benzoic acid in different jurisdictions, as well as in certain livestock production systems, such as organic farming.

5.2.3.9 Research required

Most of the research has been done in the context of reducing NH_3 emission from the farm and impact on CH_4 emission has not been measured. Demonstration of efficacy through the evaluation of CH_4 emission might be necessary.

5.2.4 Manure acidification through direct amendment

5.2.4.1 Description

The reduction of manure pH by the direct addition of acids to manures slurries or stockpiles.

5.2.4.2 Mode of action

Methanogenic bacteria are inhibited as pH decreases.

5.2.4.3 Efficacy

Manure slurry acidification to a pH of 5.5 has been reported to reduce CH_4 production by 67 to 87 percent in cattle manure slurries (Petersen et al., 2013) and Sokolov et al. (2020) reported CH_4 reductions of 77 percent in dairy cattle manure.

5.2.4.4 Potential to combine with other mitigation strategies

The acidification of manure is not compatible with anaerobic digestion. It may be combined with other mitigation strategies.

5.2.4.5 Effects on other emissions

Manure acidification will reduce NH_3 emissions. Acidification of liquid manures may increase hydrogen sulfide emissions.

5.2.4.6 Productivity and meat/milk/manure/crop/air quality

No impact on meat, milk, or manure quality. Acidification of manure to pH in the 5.5 range typically does not pose problems for crop production. Acidification does reduce the loss of N as NH_3 which results in increased N available to crops while reducing NH_3 emissions during and after the land application of manure and surface application of acidified slurry is a good alternative to slurry injection (Fangeuiro at al., 2017).

5.2.4.7 Safety and health aspects

The storage and handling of acidic compounds requires appropriate safety measures.

5.2.4.8 Adoption potential

This technology is well developed because it is listed as a Best Available Technology (BAT) for NH_3 mitigations. Nevertheless, technical barriers (risks related to storage and handling of acid, corrosion of materials) and psychological barriers (consumers consideration) have been observed in some countries.

5.2.4.9 Research required

Research that better quantifies N_2O emissions from acidified manures following their incorporation and injection into the soil is needed. Long-term effect on soil properties should also be study in different pedoclimatic conditions.

5.2.5 Methane inhibitors

5.2.5.1 Description

The addition of amendments such as tannins (Whitehead et al., 2013), monensin (Clanton et al., 2012) and narasin (Andersen et al., 2018) added directly to manure storage have been demonstrated to limit the formation of CH_4 in stored manures.

5.2.5.2 Mode of action

Amendments such as monensin and narasin are ionophores, which are lipid-soluble molecules. These molecules transport ions across cell membranes and cause changes to the microbial community that can inhibit CH_4 production. Tannins are polyphenolic compounds found in some plant species that have an inhibitory effect on methanogenic microbes.

5.2.5.3 Efficacy

Narasin has been shown to strongly inhibit CH_4 production for up to 25 days following its addition to swine manure at 3.0 mg narasin kg^{-1} manure. Andersen et al. (2018) reported CH_4 production rates were reduced by 9 percent for each mg of narasin added per kg of manure, and this reduction was effective for up to 25 days. Some level of Inhibition was noted for up to 120 days. Quebracho condensed tannins added at 0.5 percent weight per volume to manure slurries has been demonstrated to reduce CH_4 production by over 85 percent for up to 28 days.

5.2.5.4 Potential to combine with other mitigation strategies

The use of CH_4 inhibitors in feed or manure can reduce the efficacy of anaerobic digesters that utilize these manures as feed stocks. Otherwise, this technology can be used in combination with most other mitigation strategies.

5.2.5.5 Effects on other emissions

For the first week following the initial application of these inhibitors CH_4 production may increase and then will be inhibited.

5.2.5.6 Productivity and meat/milk/manure/crop/air quality

When added directly to manure no impact on meat or milk occurs.

5.2.5.7 Safety and health aspects

None.

5.2.5.8 Adoption potential

This technology is well developed and ready for use. However, the adoption of such mitigation strategy will depend on the registration status of the substances used for that purpose in the different jurisdictions. The primary obstacle to adoption is the additional cost of purchasing these compounds for direct addition to stored manure with no associated increase in production.

5.2.5.9 Research required

No major research gaps exist.

5.2.6 Decreased manure storage interval

5.2.6.1 Description

A reduction in CH_4 emissions from stored manure can be achieved by reducing the manure storage interval in-house (manure frequent removal) and during the outdoor storage**.**

5.2.6.2 Mode of action

Reducing the length of time manure is stored reduces the amount of CH_4 that can be generated during storage (Andersen et al., 2015).

5.2.6.3 Efficacy

The highest efficacy will be realized for animal production systems that have the greatest CH_4 production from stored manure, such as deep-pit swine production systems (Park et al., 2006). Petersen et al. (2013) reported 40 percent to 50 percent CH_4 emission reduction due to the frequent manure removal for pigs. For animal production systems where the majority of the CH_4 emissions are not generated during manure storage this approach will be limited in effectiveness.

5.2.6.4 Potential to combine with other mitigation strategies

This mitigation strategy may be combined with any other mitigation strategies.

5.2.6.5 Effects on other emissions

If the removed manure is being land applied on a more frequent basis this strategy could result in increased N_2O and CO_2 emissions. Nevertheless, this technique leads to lower NH_3 emissions and odors in-house (Santonja et al., 2017) and during storage.

5.2.6.6 Productivity and meat/milk/manure/crop/air quality

None.

5.2.6.7 Safety and health aspects

None.

5.2.6.8 Adoption potential

This strategy can be adopted by producers who have viable utilization options for manure that is available more frequently. Producers who do not have manure land application or other utilization opportunities will not be able to employ this strategy. The in-house implementation of this technique can easily be considered in new houses. In existing houses, costly modification of the manure management system could be required.

5.2.6.9 Research required

Additional research on the potential increase in N_2O and CO_2 emissions from more frequent land application is needed.

5.2.7 Solid-liquid separation

5.2.7.1 Description

Solid-liquid separation has become a complementary management option for manure management systems, particularly for anaerobic systems. The separation process can help to divert solids with high phosphorous-to-nitrogen ratio to nutrient deficit areas. This can help to reduce GHG emissions that would occur during manure storage and during manure application. Methane emission reduction is possible because volatile solids are separated along with the solid stream. Solid separation also reduces crust formation, which is useful to limit the anaerobic conditions during manure storage.

5.2.7.2 Mode of action

Removing part of the OM (i.e., volatile solids) prior to delivering manure to storage structures and land application.

5.2.7.3 Efficacy

The CH_4 reduction ranges from 7.0 percent-49.0 percent depending on several factors such as system design (e.g., screen size), concentration of solids in processed manure, manure flow rate, and type and configuration of manure processing pit prior mechanical separator (Zhang et al., 2019).

5.2.7.4 Potential to combine with other mitigation strategies

This mitigation strategy may be combined with other mitigation strategies.

5.2.7.5 Effects on other emissions

Emissions of N_2O and NH_3 from land application of solid separated (Aguirre-Villegas et al., 2019).

5.2.7.6 Productivity and meat/milk/manure/crop/air quality

None.

5.2.7.7 Safety and health aspects

Generally safe with potential hazards of moving parts.

5.2.7.8 Adoption potential

There are several designs that could be applied on different farm sizes. They can also be used with current manure management systems with little modification. The adoption will depend on the cost of retrofitting the existing management systems.

5.2.7.9 Research required

Measurements of emissions of different gases after land application over different seasons.

5.2.8 Manure composting/aeration

5.2.8.1 Description

Manure composting is the biological oxidation of manures in conjunction with an additional organic carbon source typically at thermophilic temperatures produced by microbial heat production. Manure can either be left undisturbed during the composting process (passive composting), mechanically turned (extensive composting) or actively aerated (intensive composting).

5.2.8.2 Mode of action

Composting is an aerobic process that reduces or prevents the release of CH_4 during OM breakdown. If the process is fully aerobic then composting does not produce CH_4 because CH_4 producing microbes are not active in the presence of oxygen. In practice composting systems may not achieve completely aerobic conditions and both aerobic and anaerobic conditions may exist within the compost pile or windrow.

5.2.8.3 Efficacy

Maurer et al. (2016) reports composting to be 70 percent effective at all scales for dairy manure but reports a reduction by 34 percent of CH_4 emissions) for swine manure composting at all scales. This large range in emissions from composting processes reflects the difference in CH_4 emissions from composting systems with differing aerobic / anaerobic conditions.

5.2.8.4 Potential to combine with other mitigation strategies

Composting can be combined with other CH_4 mitigation strategies. Composted is often used following manure separation to prepare the separated solids for use as bedding material in dairy cattle systems.

5.2.8.5 Effects on other emissions

Composting is an aerobic process that produces both CO_2 and N_2O. In addition, nitrogen losses from composting systems in the form of NH_3 emissions can be significant. Maurer et al. (2016) reports N_2O emission controls of -685 percent and -388 percent (where - indicates an increase in N_2O emissions) for swine and dairy manure composting systems respectively.

5.2.8.6 Productivity and meat/milk/manure/crop/air quality

No impact on meat or milk production. Losses of N during composting can be high, especially via NH_3 emissions but also N_2O (depending on the composting process) and are increased by frequent turning and mixing of the manure during the composting process.

5.2.8.7 Safety and health aspects

Composting can generate NH_3 emissions. Safety precautions should be taken with windrow turning and management equipment.

5.2.8.8 Adoption potential

Composting and aeration technologies are well developed and ready for use. Composting is readily adopted to solid manures and can also be implemented for slurry with the addition of a carbon source.

5.2.8.9 Research required

No additional research is required to implement this strategy.

5.2.9 Biofilters and scrubbers

5.2.9.1 Description

Biofilters and biofilter/scrubber combinations have been shown to be effective in reducing CH_4 emissions from both animal housing (mechanically ventilated) and manure storage through the action of methanotrophic bacteria (Hilhorst et al., 2002).

5.2.9.2 Mode of action

Methanotrophic bacteria grown in the biofilter oxidize thereby reducing or eliminating the emissions.

5.2.9.3 Efficacy

Maurer et al. (2016) reports 17 percent to 24 percent CH_4 mitigation effect across all species at all scales in their summary performance data for technologies to control gaseous emissions livestock operations.

5.2.9.4 Potential to combine with other mitigation strategies

This mitigation strategy may be combined with any other mitigation strategies.

5.2.9.5 Effects on other emissions

Biofilters and scrubbers are utilized to control NH_3 emissions. While they are very effective at reducing NH_3 emissions, N_2O is typically formed in the biofilter as well.

5.2.9.6 Productivity and meat/milk/manure/crop/air quality

None.

5.2.9.7 Safety and health aspects

None

5.2.9.8 Adoption potential

Biofilters and scrubbers require the replacement of ventilation fans with units correctly sized to work against the pressure drop developed in the biofilter. This retrofit can be cost-prohibitive for many operations.

5.2.9.9 Research required

Additional research on limiting N_2O production in biofilters is needed.

5.2.10 Manure incorporation and injection

5.2.10.1 Description

The incorporation of manure following land application either through cultivation practices or direct injection of manure 15 to 20 cm below the soil surface.

5.2.10.2 Mode of action

Soils can serve as either a source or a sink for CH_4, depending on the conditions and if methanogenic or methanotrophic bacteria are active (Topp and Pattey, 1997). When soils serve as a sink, methanotrophic bacteria can oxidize CH_4 following the incorporation or injection of manure below the soil surface. If soil conditions are favorable for methanogenic bacteria activity, CH_4 emissions can increase following the incorporation or injection of manure.

5.2.10.3 Efficacy

High mitigation efficacy when soil conditions favor methanotrophic bacteria growth. When soil conditions favor methanogenic bacteria growth, soils can become a CH_4 source. Methane emissions from the soil have been shown to spike immediately following manure application but fall to very low levels quickly following incorporation or injection (Montes et al., 2013). Lovanh et al. (2008) reported that the injection of swine manure resulted in an order of magnitude reduction of CH_4 when compared to surface applied swine manure. Reports of increased CH_4 emissions following manure injection compared to commercial fertilizer controls can also be found in the literature. Sistani et al. (2010) reports that CH_4 emissions from cropland fertilized with injected swine was significantly higher than the control which used commercial fertilizer.

5.2.10.4 Potential to combine with other mitigation strategies

The potential to combine other strategies such as anaerobic digestion or solids separation with incorporation or injection is excellent. The anaerobic digestion of manure or separation of solids prior to incorporation or injection reduces the C available to be converted to CH_4 and further enhances the CH_4 mitigation potential of this strategy.

5.2.10.5 Effects on other emissions

The incorporation and especially injection of manure below the soil surface can lead to increased N_2O emissions. It should be noted however, that conflicting results are reported regarding N_2O emissions following the land application of manure via injection. Vallejo et al. (2005) reported no significant difference in N_2O emissions between surface application and injection of swine manure. The inconsistency in reported N_2O emissions measured following the land application of manure is likely due to the diversity of soil conditions that the emissions were measured under.

5.2.10.6 Productivity and meat/milk/manure/crop/air quality

No impact on meat, milk, or manure quality. Incorporation or injection of manure has been demonstrated to conserve nutrients for plant use and thereby increase plant nutrients available for crop uptake. Incorporation and injection reduce NH_3 emissions to the atmosphere but can increase N_2O emissions.

5.2.10.7 Safety and health aspects

None

5.2.10.8 Adoption potential

This technology is well developed and ready for use. It should be noted that in order to utilize this technology producers will have to purchase the purpose specific equipment required to pump, transfer and sub-surface inject manure. The initial cost of this equipment may prove a barrier to adoption for some farmers.

5.2.10.9 Research required

Research that better quantifies CH_4 and N_2O emissions following the incorporation and injection of manure into the soil.

5.2.11 Manure application timing

5.2.11.1 Description

Application of manure, at different times of the day and of seasons, with different methods that are currently employed for incorporation and surface applications.

5.2.11.2 Mode of action

Effect of soil temperature and moisture content affect methanogenic bacteria activity.

5.2.11.3 Efficacy

Montes et al. (2013) lists application timing as having an efficacy of ≤10 percent in their table of possible mitigation strategies.

5.2.11.4 Potential to combine with other mitigation strategies

Applicable with other manure treatment technologies that could provide flexibility of application timing such as manure storage and production of stable manure products such as manure compost.

5.2.11.5 Effects on other emissions

It may affect the emissions of N_2O depending on weather conditions and soil conditions (i.e., temperature, soil freeze-thaw cycles), and manure type and treatment (He et al., 2020). Soils with high moisture content may promote the emissions of N_2O (Montes et al., 2013). Ammonia emissions increased in the first 10 hours after manure application (Gordon et al., 2001). In addition, when the available pool of N and C in the soil is greater, denitrification rates can increase resulting in greater N_2O emissions. As such timing manure applications such that actively growing crops are present can reduce N_2O emissions compared to field applications during fallow times when greater pools of N would remain available (Chadwick, et al., 2011). Thorman et al. (2007) reports direct N_2O emissions from fall/winter manure slurry applications were 64 percent greater than spring applications when emissions are considered as a percentage of the total N applied.

5.2.11.6 Productivity and meat/milk/manure/crop/air quality

None.

5.2.11.7 Safety and health aspects

Safety precautions should be taken when using equipment for manure application.

5.2.11.8 Adoption potential

Achieved in practice when storage volume and weather conditions allow.

5.2.11.9 Research required

Measurements of the emissions of N_2O and NH_3 under different weather conditions and cropping systems.

5.2.12 Nutritional strategies

5.2.12.1 Description

Reducing of the amount of OM excreted has a positive impact on the emission of CH_4 from manure.

5.2.12.2 Mode of action

Nutritional mitigation options that improve feed conversion rate of animals, through improved diet digestibility (e.g., feed formulation, feed processing, forage management, enzymes, direct fed microorganisms, botanical extracts, and so forth), decrease the amount of OM excreted. Furthermore, preparation of feed in the form of pellets may also reduce feed losses in pig farms.

5.2.12.3 Efficacy

The efficacy depends on the different mitigation options and on the status of the farm. Improvement of feed conversion ratio between 2 and 5 percent can be achieved under typical farm conditions.

5.2.12.4 Potential to combine with other mitigation strategies

Nutritional options can be combined with specific approaches of manure management (e.g., acidification). They may negatively impact the operation of anaerobic digesters.

5.2.12.5 Effects on other emissions

Usually, improved feed conversion rate also reduces nitrogen excretion, leading to reduced NH_3 and N_2O emissions. Enteric CH_4 production is also decreased in ruminants.

5.2.12.6 Productivity and meat/milk/manure/crop/air quality

Feed conversion ratio is an important productivity parameter for farmers. Effects on animal productivity are covered elsewhere in this document.

5.2.12.7 Safety and health aspects

The nutritional mitigation solutions used to improve feed conversion ratio of animals are typically considered as Generally Recognized as Safe or evaluated for safety by regulatory authorities.

5.2.12.8 Adoption potential

Nutritional solutions are easily adopted by farms where compound feeds or mixed rations are used. In this case, adoption potential is high. In general, the cost of the nutritional mitigation solution is compensated by the improvement in feed conversion rate. However, the implementation of these strategies will depend on the regulatory environment (e.g., authorisation of feed ingredients) and may not be allowed in certain livestock systems, such as organic farming.

5.2.12.9 Research required

Research on the efficacy of new nutritional solutions that improve feed conversion efficiency needs to include measurements of reduced OM excretion and associated emissions.

5.2.13 Grazing practices – production system

Grazing systems encompass the entire production system change rather than applying a single mitigation method for stored or land applied manure. As such, grazing systems have not been included in Table 5. Modifying the grazing system to decrease CH_4 emissions can affect the amount and composition of manure excreted by animals. Methane emissions from urine and dung droppings for grazing animals is minimal compared to that emitted in manure storage systems via animal confinement (Pellerin at al., 2017). The reduction in CH_4 emissions from animal excreta is large, especially under warm climates. Modifying the grazing system involves a whole change in system functioning.

Generally, grazing animals produce greater enteric CH_4 emissions because of the high fiber concentration of the herbage compared with animals in confinement systems fed mixed diets. Grazing systems, when intensively managed, have larger N_2O emissions at the field level. On the other hand, NH_3 emissions (a precursor of N_2O) is generally reduced in grazing vs. confined systems. Grazing systems reduce the amount of manure produced in the farm because bedding is not used, and excreta is delivered directly to the pasture. Within grazing systems, there can be differences that affect the potential for soil organic sequestration.

5.3 Mitigation of methane emission from rice paddy

Methane is emitted from rice paddies due to anaerobic decomposition of organic matter, such as soil organic matter, plant residue, and rice roots, under highly reduced conditions when the land is flooded. Methane produced in anoxic rice soil is oxidized in the oxic rhizosphere and surface soil in part. Thus, the balance of methane production and oxidation controls methane emission (Fig. 1.3). Several management practices that induce increased redox potential of soil suppress methane production and thus the emission from rice fields.

5.3.1 Water management

Modifications of the water management have a proven track record to reduce methane emissions from rice fields and are deemed the most promising way to mitigate methane emissions from rice paddies (Wassmann 2019). Drainage of rice fields increases the redox potential, which strongly suppresses the microbial processes of methane production and concomitantly stimulates methane oxidation. However, the draw-back of floodwater generates short-term spikes of gaseous methane which has been entrapped in the flooded soils (Wassmann et al. 1994). Nevertheless, the overall amount of methane emitted from the soil through the course of the cropping season is significantly reduced as demonstrated in numerous field measurements (Sander et al. 2014). Either single or multiple drainage approaches like alternative wet and dry (AWD) management has shown consistently significant mitigation potential – although the magnitude of reduction is given in a wide range by different studies (Yagi et al., 2020).

While the baseline is defined as continuous flooding, the scaling factors for other water regimes (SF_w) in the IPCC guideline range from 0.41 to 0.94 with an extensive error range due to the difference in the extent of drainage in terms of duration and frequency (IPCC et al., 2019). According to a recent meta-analysis based on 201 paired observations, non-continuous flooding practices reduced CH_4 emissions by 53 percent as compared to continuous flooding (Jiang et al. 2019). In terms of GWP, the reduction effect has a slightly lower percentage, namely 44 percent, which is attributed to trade-offs through higher N_2O emissions. Increments in N_2O emissions under unstable water regimes are well documented, but – with the exception of individual records of excessively high N_2O emissions (Kritee et al. 2020) – do not reverse the trend of GHG savings through AWD (Majumdar 2003, Yagi et al., 2020). Athough the global meta-analysis by Jiang et al. (2019) also revealed a slight yield reduction through AWD, the economic feasibility of this water management practice will largely depend on local circumstances AWD, namely the potential savings in pumping costs. In the Vietnamese Mekong Delta, the application of AWD improved farm profitability by up to 13 percent corresponding to about $100 per hectare (Frith et al, 2021).

Water management before the cultivation period also affects methane emissions during rice cultivation. A prolonged non-flooded preseason over one year had a significantly lower methane emission scaling factor (0.41-0.84), while flooded preseason over 30 days before cropping had a more than doubled scaling factor (2.13-2.73) (IPCC et al., 2019).

Such mitigation practices are only feasible wherever complete control of water supply and drainage is possible. In the tropics, water management will be less effective in mitigating methane emissions during rainy seasons (Yagi et al., 2020). This impact of precipitation is also taken into account in a newly developed method for GIS-mapping of AWD suitability (Nelson et al. 2017). However, if available, appropriate water management promotes rice production while effectively mitigating methane emissions (Yagi et al., 2020). Land leveling allows the spatially homogeneous water management, and would contribute to effective methane migration. Better water management in rice paddies to mitigate methane emission could also contribute to sustainable water, an important goal for agriculture (FAO, 2020). On the other hand, the prolonged aerated condition could cause an enhanced decomposition of soil organic matter, lowering the carbon storage and fertility of rice field soils in the long term.

5.3.2 Organic amendments

More methane is emitted from soils amended by organic compounds of easily decomposable carbon. Methane emission also increases as a function of the amount of organic amendments that are applied to th soils. If rice straw is incorporated into the soil after harvest, the timing of rice straw application significantly affects methane emission. A long interval between straw incorporation and flooding lowers methane emission during the rice-growing season as compared to incorporating rice straw just before flooding (IPCC et al., 2019). Either removing or burning rice straw drastically reduces methane emission, but implies adverse effects on the local air quality (in case of burning) and may decrease in soil organic carbon and soil fertility in the long term (Yagi et al., 2020). On the other hand, long-term experiments with flooded rice fields showed high stability of soil organic matter even if the straw has been routinely been removed for more than a decade of double cropping rice (Pampolino et al. 2008). Given the overall objective of resource recycling, the application of composted rice straw to the soil is another option to reduce methane emission from rice fields as compared to a baseline of incorporating fresh straw (Buendia et al., 2019; Yagi et al., 2020). The N-content of rice straw, however, will not suffice for reasonable yield levels on its own, so that additional organic amendments, e.g. animal manure, will be required. Moreover methane production during the composting process should be taken into account (Nguyen-Van- Hung et al. 2020). Farmyard manure and green manure also have a lower scaling factors than the incorporation of fresh rice straw (Buendia et al., 2019), thus giving another option of applying organic amendments to sustain the fertility and carbon storage in soil.

Biochar has been considered as an option to reduce GHG emissions from rice cultivation. Although the long-term effect remains unclear, it has often been demonstrated that biochar application is an effective way to reduce methane emission from flooded rice fields (Jeffery et al., 2016; Mohammadi et al., 2020; Yagi et al., 2020). Environmental life cycle assessment studies showed that the carbon footprint of rice produced in biochar-treated soil was estimated to range from −1.43 to 2.79 kg CO_2eq per kg rice grain, implying a significant reduction relative to rice produced without a biochar soil amendment (Mohammadi et al., 2020). At this point, however, the application of biochar in rice production remains at the scale of pilot studies as the practicability and environmental impacts of available stoves is still unclear. Combination of AWD water management with biochar application may further reduce methane emissions (Sriphirom et al., 2020).

5.3.3 Fertilizer and other amendments

Application of sulfate-containing fertilizer, such as ammonium sulfate and phosphogypsum, reduces methane emission (Yagi et al., 2020; Kumar et al., 2020) as sulfate ion supports sulfate reduction in flooded rice field soils that outcompetes methane production (Achtnich et al., 1995).

Bio-fertilizers, e.g., *Azolla* (aquatic pteridophyte with symbiotic cyanobacteria) and blue-green algae (cyanobacteria), are widely used to increase soil fertility and rice yields with their nitrogen fixation activity. They can mitigate methane emission by oxygenating the rice soil through photosynthetic activities (Maylan et al., 2016).

It is reported that nitrification inhibitors, which slow down the conversion of ammonia into nitrate, can reduce not only nitrous oxide but also methane emission from rice fields (Malyan et al., 2016). Nitrification inhibitors promote rice plant growth through increased nutrient uptake, and increase the redox potential in the rhizosphere, which reduces methane emission (Boeckx et al., 2005).

Reduction of ferric iron also competes with methanogenesis (Achtnich et al., 1995). Addition of steel slag can mitigate methane emission from paddy fields (Kumar et al., 2020). Silica oxide in steel slag also can mitigate methane emission from rice by promoting the development of aerenchyma of rice roots, which increases oxygen transportation from the atmosphere to the root region and enhances rhizospheric methane oxidation (Kumar et al., 2020).

5.3.4 Planting methods and crop management packages

Direct seeding has been reported to reduce methane emissions (per m^2 and day) as compared to the traditional transplanting of rice seedlings (Yagi, 2020, Malyan, 2016). Although the yield of direct-seeded rice could be lower than transplanted rice (Yagi et al., 2020), this practice is getting increasingly popular due to labor savings and could also be optimized in view of mitigation potential in many rice-growing areas.

Regarding pre-season conditions, methane emissions are reduced by prolonged period without flooding (Yagi et al., 2020), e.g. caused by a long fallow season or a crop rotation with an upland crop. This effect is considered in the IPCC guidelines through a pre-season scaling factor, namely SF_{pre} = 0.59 for “non flooded preseason >365 d” in contrast to a baseline (SF_{pre} = 1) for “non flooded preseason >180 d”.

The system of rice intensification (SRI) is a farming methodology characterized by a low-water, labor-intensive management, implies features of low-emission management (Malyan et al., 2016; Yagi et al., 2020). The term “SRI”, however, has been used in the literature for a very wide range of crop management practices, in particular regarding the application of organic manure (Ly et al, 2013). The original SRI concept encompasses high amount of organic inputs that result in insofar

a high background level of methane emissions. As SRI also prescribes intermittent flooding, the actual increment in emissions will be lower than under continuous flooding, so that suppresses methanogenesis. In turn, SRI has been considered as a mitigation strategy which can be justified if compared to continuous flooding (Ly et al. 2013) or as long as organic amendments are omitted (Jain et al. 2014). Insofar,the calculated mitigation effect by SRI will ultimately depend on the definition of the baseline management as well as the sub-type of SRI used for the comparison.

5.3.5 Selecting/breeding rice varieties

The difference in methane emissions for different rice varieties has been documented in several case studies. The underlying mechanisms to reduce methane through variety selection remain still unclear – except for the straight-forward approach of replacing long-duration with short-duration varieties which was proposed back in 2000 (Setyanto et al. 2000). All other possible changes in plant morphology and physiology showed inconsistent results in different studies due to complex G × E × M (genetics, environment, management) interactions that directly or indirectly alter the methane budget (Wassmann et al. 2000). Derived from plant morphology, low permeability of the aerenchyma constrains the methane transfer from the soil to the atmosphere (Butterbach-Bahl et al. 1997, Aulakh et al. 2002). Since this trait will also limit transfer of oxygen into the root system which has the opposite effect on methane fluxes, so that the net impact on methane emissions varies according to specific circumstances. From a physiological perspective, root exudation determines the amount of methanogenic material and thus, shows a strong correlation to methane emission (Lu et al. 1999). The actual amount of root exudation, however, is largely affected by the nutrient status of the rice plant (Lu et al. 2000), so that its impact on methane emissions may be concealed by other factors. High efficiency of the physiological carbon sink, i.e. the allocation of metabolites in the grain, has been shown as favorable for low-emission plants in greenhouse experiments (Denier van der Gon et al. 2002) as well as through genetically modified organisms (Su et al. 2015). Broadly speaking, rice cultivars with few unproductive tillers, small root system, high root oxidative activity, high harvest index, low root exudation and which are early maturing we proposed for mitigating methane emission in rice fields (Malyan et al., 2016). However, mechanistic understanding is still needed to select and breed varieties that emit less methane (Balakrishnan et al., 2018; Yagi et al., 2020).

5.3.6 Reducing methane from straw burning

Although methane emission from rice production is generally equated with biogenic emissions from flooded fields, the common farming practice in many Asian countries also generates sizable amounts of pyrogenic methane. Open field burning entails incomplete combustion of rice straw and this generates methane as well to a lesser extent nitrous oxide (Romasanta et al. 2017). In spite of intensive efforts to eliminate this practice, straw burning is still rampant in many parts of Asia causing enormous problems in terms of local air pollution (Gadde et al. 2009). While rice straw is typically kept in piles on the fields after harvest, the proportion of incomplete combustion is a function of moisture content in these piles and thus, of local rainfall events (Romasanta et al. 2017). One alternative management practice to straw burning is soil incorporation, but straw amendments increase the subsequent methane emissions once the field gets flooded. Based on the IPCC 2019 guidelines, this increment in emissions could be curtailed through proper timing of the soil incorporation, i.e. the conversion factor of "straw incorporated long (>30 days) before cultivation" is 0.19 as opposed to the baseline of "straw incorporated shortly (<30 days) before cultivation" (conversion factor = 1).

The options for external straw use rely on its removal from the field which represents a relatively laborous activity under the typically low levels of mechanization in most rice producing regions at this point. Straw could be used to produce compost and then be returned to the field. While the conversion factor of compost is fairly low (0.17) as compared to fresh straw, the low N-content of straw will require some additional organic material for compost production such as animal manure (Nguyen-Van-Hung 2020a). Straw could also be fed to cattle, but its low nutritional value will cause sizable methane emissions from the animals (see above). In principle, straw represents a valuable feedstock for bioenergy as shown for wheat straw in many industrialized countries. Rice straw, however, has a high silica content that typically causes technical problems ("slagging") in combustion devices (Chieng and Seng 2020). Moreover, this type of commercial use will require its availability in a compact form facilitating easy transport and storage. To this end, the ongoing mechanization trends in rice production, namely in form of new baling machines, may transform the availability of straw that can be traded as a commodity (Nguyen-Van-Hung et al. 2020a).

5.3.7 Choice of options

The appropriate water management, including midseason drainage, AWD, and SRI, is the most promising option to mitigate methane emission from flooded rice fields and thus would be the first choice if practicable. To avoid the introduction of fresh organic matter like rice straw into the soil is a feasible way to prevent excessive methane production and thus high emission in rice fields.

The other options in fertilization may to a certain extent be considered for the mitigation of methane – either as an additional option or as long as water management options are not possible. Sulfate-containing fertilizers may be helpful to reduce methane emission, but it is not suitable for the soil with less amount of reducible iron that forms insoluble FeS because otherwise reduced sulfur (S^{2-}) would damage the rice roots. Biofertilizers (*Azolla* and blue-green algae) can oxygenate the surface soil, reducing methane production and promoting methane oxidation, but assuming a discernable impact under field conditions is speculative at this point. The application of iron and silica-containing materials that can keep higher redox conditions in soil and rhizosphere is also helpful to reduce methane emissions. The meta-analysis demonstrated the mitigation potential of biochar, but as for the other options there is no evidence on its applicability at larger scale.

Many of the above-mentioned methods could contribute to better plant growth and higher yields, which reduces GHG emission per yield, i.e., "GHG intensity" expressed as tonnes of CO_2eq per ha, as well as carbon footprints defined as emissions per product amount expressed as kg CO_2eq per kg of rice product. As for the private sector, the latter is of much more relevant than the area-based emissions, so that future efforts to reduce methane emissions from rice may be driven by user-friendly calculation tools and product labelling of carbon footprint (Wassmann et al. 2022). This will then encompass a wide range of production enhancing approaches, e.g., rice hybrid technologies. Such efficiency gains in food production have routinely been considered as mitigation options in animal systems (see above) but have hardly been mentioned in the context of rice production.

5.3.7 Newly emerging technologies

In addition to the currently available methods for reducing methane emission from rice, several new technologies show high potential as mitigation options derived from ongoing investigations. For example, it is suggested that plant growth-promoting rhizobacteria (PGPR) such as diazotrophs could increase root mass, thereby promoting O_2 exchange to the soil (Singh and Strong 2016). The potential of transgenics to decrease methane emission is demonstrated by using a barley transcription gene (Su et al., 2015). Microbial fuel cells (MFCs) generate electricity in rice field soil, which compete with methane production and thus can mitigate methane emission from the rhizosphere (Kouzuma et al., 2014). These new technologies are still in infancy stage and need further investigations and verification for application at a field scale (Pratt and Tate, 2018).

5.4 Cross-cutting methane mitigation

5.4.1 General guidance for taking an integrated approach to methane mitigation strategies

To reliably assess the potential for CH_4 emission reduction and ensure that recommended mitigation strategies are appropriate and minimize potential trade-offs, the wider agricultural and systemic context and implications must be considered. In this section we give a brief overview of why these broader considerations are necessary, discuss the tools to ensure holistic appraisals, and provide some illustrative examples of CH_4 reduction strategies being considered in a wider context.

Agricultural production involves complex interactions between biological systems, time- and location-specific environmental conditions, and management practices. These factors result in considerable uncertainty and variation in agricultural emissions (Dudley et al., 2014). Interventions targeting one concern (i.e., CH_4 emissions) can entail multifaceted interactions with other components of the system. These interactions can result in wider co-benefits; for example, general increases in production efficiency may reduce emissions of other GHG including N_2O and CO_2 in addition to CH_4, alongside reductions in wider resource use and environmental impacts (Capper, 2011). In other cases, there may be trade-offs. For example, efforts to reduce CH_4 emissions may increase other GHG (Cardoso et al., 2016) or raise concerns related to animal welfare (Llonch et al., 2017).

Comprehensive assessments that cover multiple impact categories are typically provided by LCA, with attributional LCA being the most common. Attributional LCA tracks energy and material use and pollutant releases occurring along the supply chain and production process to report the total inventory or 'footprint' attributable to a given output or functional unit (ISO 14044). The functional unit may be a product or commodity of certain quality (e.g., milk, for dairy production), or a more specific aspect of the outputs (e.g., protein or calorie content). The choice of functional unit depends on the assessment being made and intended usage of life-cycle information.

Life-cycle system boundaries are extended as far as possible and relevant for a given question, ideally starting from the point of production of all inputs ('cradle'), capturing impacts occurring before the agricultural production phase, such as energy-use in manufacturing fertilizer. In many agricultural and food-related LCAs, the production process is tracked until the end of the agricultural production phase (leaving the 'farm-gate'), as this is where most impacts are accrued and where changing agricultural practices has the greatest ability to reduce impacts, but the chain can also be followed through to processing, consumption, and disposal (for complete 'cradle-to-grave' LCA). Disposal from production such as infrastructure or manure are part of the production system.

In this way, LCA provides a useful methodology to explore CH_4 reduction strategies in a wider context. By taking a life-cycle perspective, we look beyond just the reduction in CH_4 emissions that might be achieved through different measures and consider wider associated impacts – positive or negative. This could entail, for example, considering the impacts associated with manufacturing of CH_4-reducing feed additives. It also provides the whole production-system perspective necessary to identify wider co-benefits or potential trade-offs, as noted above. LCA generally captures emissions per functional unit, which is often products. However, from a global perspective, it is absolute emissions that matter for the ultimate climate consequences. Some mitigation strategies may reduce emission intensity by increasing efficiency, which may then facilitate greater production, resulting in increased absolute emissions. It depends on the wider policy and development objectives whether emissions intensity or total emissions are the most relevant aspects to characterize mitigation outcomes.

In addition to its role in setting a comprehensive framework to compile a life-cycle inventory, LCA is also commonly used to assess the impacts resulting from this inventory. This is done through the translation of inventory data to potential impacts of interest through standardized reporting indicators. The climate impact assessment component of an LCA (often referred to as the 'carbon footprint') takes the inventory data for individual GHG emissions and combines them into a single climate impact indicator. More discussion on various metrics that can be considered for LCA is given in the Chapter 6.

It must be noted that GWP100 is just one potential climate impact indicator. It is a 'midpoint' indicator that is only part of the way along the chain of translating GHG emissions into eventual contribution to climate change and resulting damage. Depending on, for example, the timeframe or aspect of climate change of interest, other indicators may be equally justified, yet give a different answer as to whether a specific intervention has overall positive or negative impacts. Recent guidance recommends considering multiple metric choices in Life Cycle Impact Assessments (Levasseur et al., 2016), including also GTP. As a short-lived GHG, the relative valuation of CH_4 is particularly sensitive to metric-choice and time-horizon. The section of this report on 'metric guidance' contains extended discussion on the usage of different GHG metrics and wider discussion on interpreting contributions to climate change. The metrics section of this document further discusses what other ways the impact of CH_4 mitigation can be quantified.

Climate impact is only one component of a total impact assessment in LCA, and other common impacts include, for example, water scarcity, land use, biodiversity loss, air and water pollution. Interventions to reduce CH_4 emissions can then be compared against these other outcomes, similar to exploring the influence they may have on other GHG emissions, as described above.

These wider impact categories also have standard, simplifying indicators to report results and provide a simple appraisal of relative performance. As with the assessment of climate change impacts, there may be different indicators and modelling approaches suitable for different purposes, with guidance to explore sensitivity to different metrics and ensure the method used can be sensibly applied to the question posed (Frischknecht et al., 2016). While we recommend as comprehensive an assessment as possible, whether to explore other categories and which ones to choose in addition to GHG emissions is ultimately at the discretion of the user/investigator. Separate impact categories can also be weighted and combined into aggregated indicators. Examples are 'disability-adjusted life years' that estimates total burden on human health, financial valuation to provide a common currency for all impacts and outputs, or abstract scores to act as a simple communication device for total impacts. There is, however, no universally agreed upon method of indicator weighting or aggregation, and doing so can obscure individual results, so it is standard practice to retain separate reporting categories in addition to fully aggregated indicator results.

It is widely recognized that these challenges result in limitations and potential subjectivities in agricultural LCAs. For example, van der Werf et al. (2020) argued that LCA is currently ill-equipped to reliably assess the impacts of organic or lower-intensity agriculture because some impact indicators remain weak with the focus on product-level assessment being too narrow in the context of wider aims and impacts. In the context of assessing CH_4 reductions, therefore, it is important to note that LCA may give us some insight into and quantification of wider benefits and/or trade-offs, but its results depend on methodological choices leading to high uncertainty of the results. There may be other considerations that determine how policymakers or society-at-large view certain system transitions as positive.

Some of these broader issues may be addressed through consequential LCA – a method that links LCA data and methodologies to consequential (largely economic) models of what might happen in response to changes (e.g., changes in production method or type or quantity of functional unit produced), rather than just comparing individual system impacts. Where attributional LCA allocates elementary flows to individual products, which may then be compared, consequential LCA estimates the deviations in the elementary flows resulting from a system change (Rebitzer et al., 2004; Ekval and Weidema, 2004).

Consequential LCA may be particularly relevant where proposed CH_4-reduction measures entail major, systemic impacts, such a global shift towards more intensive ruminant production, or reductions in total ruminant production. Consequential LCA also has challenges and limitations (Yang and Heijungs, 2018), and a full review is beyond the scope of this report. However, we suggest it may provide another useful approach which has arguably been underutilized in assessment of specific agricultural interventions.

Concerns around larger scale assessment must also be kept in mind when considering the scalability of CH_4 mitigation methods. Some potential methods that are applicable to intensive systems, such as feed additives or regular CH_4-inhibitory vaccination, may not be possible or appropriate for more extensive systems. This will limit the total mitigation potential associated with a given technology or management. In summary, the complex and interlinked nature of agricultural production means we must consider CH_4 reductions in a wider context, as explored further in the examples presented below. LCA remains a valuable method to ensure comprehensiveness and help compile inventories of activities that may be associated with climate and environmental (or other) impacts. LCA can also help guide and provide useful frameworks for impact assessment. However, exhaustive analysis of the impacts of agricultural production systems, the extent to which they may be deemed 'sustainable', and the full considerations required for decision-making, may need a deeper appraisal and interpretation/prioritization than LCA impact indicators can provide alone. This may include an assessment of whether reductions in emissions intensity or absolute emissions are the relevant measure of success. There is a growing body of literature that continues to develop Life-Cycle Impact Assessment methodologies and suggests refinements to how they are applied. Given the focus of this report on CH_4 reduction, we provide extended discussion of how CH_4 emissions, in particular, are reported and potential implications for interpretation and decision-making elsewhere, but the wider context outlined here remains important.

5.4.2 LCA scenario analysis for intensive systems

Livestock farming systems contribute to GHG emissions arising directly from enteric and manure CH_4, manure N_2O emissions, and indirectly from crop production, soil emissions, and use of fossil fuel for machinery use and manufacturing of inputs (fertilizer, imported feeds). Some mitigation options, particularly feed and manure additives have associated CO_2 and N_2O emissions during their production and transportation. Therefore, it is important to consider the net reductions in total CO_2eq emissions when promoting a CH_4 mitigation strategy.

Methane-reducing dietary formulations, feed additives and supplements can be effective in reducing enteric CH_4 emissions in beef feedlots and dairies (Nguyen et al., 2012; Beauchemin et al., 2020); however, the net benefits/burdens on CO_2eq emissions should be quantified by including the related life cycle impact of producing such diets, feed additives, and supplements. In the California intensive dairy system, Feng and Kebreab (2020) evaluated the net mitigating effect of two feed additives, 3-NOP and nitrate. In the case of 3-NOP the diet was not changed so only the additional emission in producing 3-NOP was considered in the calculations. The authors reported that the emissions associated with 3-NOP production were 35 to 52 kg CO_2eq/kg 3-NOP produced depending on how and where the additive was produced. Additionally, the transportation of additives to the farm was also included. For nitrate, the emissions associated with the production of the additive as well as the impact of changing the diet composition were considered, as nitrate supplementation replaces other nitrogen sources in the diet. In a meta-analysis, Dijkstra et al. (2018) reported that 3-NOP on average reduced CH_4 production and CH_4 yield by 32.5 percent and 29.3 percent, respectively. Another recent meta-analysis by Feng et al. (2020) indicated that nitrate reduced CH_4 production and CH_4 yield by 14.4 percent and 11.4 percent, respectively, in a dose-response manner. In the final analysis, Feng and Kebreab (2020) using a cradle to farm gate system boundary (Fig. 5.1) reported that the average net reduction rates with supplementation of 3-NOP and nitrate in the California dairy farming system were 11.7 percent and 3.95 percent, respectively, when upstream and downstream emissions were included in the LCA. Animal production was assumed to be not affected by feed additive inclusion.

Implementation of mitigation options reviewed in the previous sections have associated effects on the system including changes in diet composition, transportation, manure composition and manure application to soil. While Owens et al. (2020) reported that supplementing beef cattle with 3-NOP did not significantly affect manure CH_4 emissions during storage, other mitigation options, particularly those that change the chemical composition of diet should be analyzed for downstream emission effects. To analyze the effect of more complex mitigation strategies, e.g., combining mitigation measures or measures that have effects at different levels of the farm (e.g., animal vs. manure management), an LCA approach that uses fixed emission factors may not have sufficient capacity to capture the interactions within farm components and therefore would be unable to evaluate potential trade-offs of GHG mitigation. For these situations, frameworks that capture internal feedbacks and loops between farm components are required (Del Prado et al., 2013; Rawnsley et al., 2016). Integrating whole-farm modelling with LCA, for example, can be used as a framework to study climate change mitigation and adaptation in ruminant-based farming systems (Del Prado et al., 2013). In particular, this type of framework has been shown to identify how effective GHG mitigation methods may, in some cases, alter emissions of other forms of pollution and have very different impacts on broader aspects of sustainability, including profitability (Del Prado et al., 2010). The downside of this type of approach is the lack of availability beyond academia and level of complexity which is greater than that of emission factor-based frameworks.

For manure management, the manure N applied to soil influences feed production and composition, and therefore affects animal productivity. A high-fat diet for dairy cattle, for example, can reduce enteric CH_4 emissions but it may also increase the CH_4 production potential of the slurry (if OM digestibility is decreased due to fat supplementation) and thus lead to greater CH_4 emissions from manure during storage (Petersen et al., 2013). Hence, unless anaerobic digestion is used to capture this additional CH_4 from slurry, fat-rich diets could result in a negative interaction with respect to GHG mitigation. Moreover, farm models have been used to identify potentially non-additive effects of combined mitigation measures, i.e., effectiveness of the combined mitigation methods may not be equal to the sum of the individual methods when applied singularly (Del Prado et al., 2010).

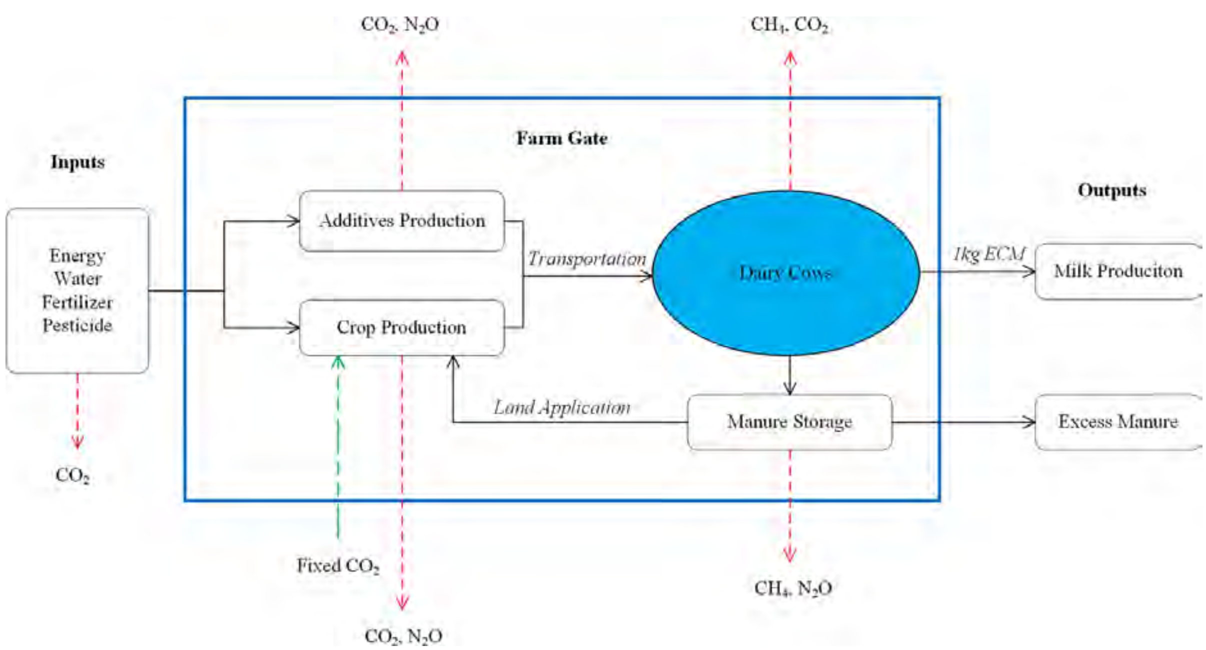

Figure 4. System boundary of the life cycle assessment for California milk production (from Feng and Kebreab, 2020).

ECM = energy corrected milk. It should be noted that CO_2 from animal origin is considered a net zero with plant sequestration (see Section 5.1).

5.4.3 LCA scenario analysis for less intensive systems

Less intensive livestock production systems tend to have a greater share of their total carbon footprint as CH_4, especially from enteric fermentation. In contrast to intensive ruminant production systems where enteric CH_4 typically comprises less than 40 percent of the total CO_2eq/kg product based on a GWP100 (e.g., cattle: 24 percent, del Prado et al., 2013; sheep: 25 percent, Batalla et al., 2015; and goats, 39 percent, Pardo et al., 2016), extensive systems have greater proportions of enteric CH_4 due to the use of forages and limited use of concentrates, combined with less emissions from use of fossil fuel. In some extensive systems, enteric CH_4 can comprise >70 percent of the carbon footprint of meat and milk due to the use of less digestible, fibrous feed and reduced level of animal productivity (Flysjö et al., 2011; Chobtang et al., 2016; Gonzalez-Quintero et al., 2021; Zubieta et al., 2021).

Typically, for pasture-based systems the most desirable production system is one that efficiently utilizes high levels of grazed pasture in the animals' feed budget, exploits existing facilities on the farm and returns the greatest profit (Crosson et al., 2011). One enteric CH_4 mitigation measure for grassland-based livestock systems is pasture quality improvement. Better pasture renewal practices or diet improvements have been identified as promising measures to reduce enteric CH_4 from low input systems (Goopy, 2020). Increasing the digestibility of forage has been identified as a strategy to decrease enteric CH_4 emissions intensity. However, the mitigation effect needs to be analyzed case by case. For example, a shift from feeding less grass to more whole plant silage maize was shown to reduce N excretion and enteric CH_4 intensity by 6 percent and 14 percent, respectively, as simulated by a farm model (Del Prado et al., 2011) although some systems may not have silage as an option. However, such a change in feeding strategy required land use change from pasture to maize (which is not possible for marginal lands), leading to soil C and N losses that can be much greater than animal level emission reductions (Vellinga and Hoving, 2011). Yan et al. (2013) conducted LCA to assess GHG emissions from pasture-based milk production relying mainly on (i) fertilizer N, or (ii) white clover, and the results indicated that the carbon footprint for white clover was 11 to 23 percent less (per kg of energy corrected milk) than that of fertilizer N, suggesting clover could be used to reduce the carbon footprint of milk of grazing dairy cows. Similarly, Schils et al. (2005) found that GHG intensity from a grass-clover system were 10 percent lower than those from a grass-fertilizer N system.

Lahart et al. (2021) compared the effect of dairy genetic merit in Holstein-Friesian cows across three contrasting feeding pasture-based production systems (extensive to intensive). The authors reported improved genetic merit as well as reducing concentrate supplementation led to a general improvement in GHG intensity as well as improved N use efficiency within the context of pasture-based dairy production systems. In agreement, van der Weerden et al. (2018) compared ‘improved’ dairy production systems designed to reduced N leaching with existing pasture based dairy production systems in New Zealand and reported that lower feed supplies and associated lower stocking rates of the 'improved' systems were the key drivers of lower total GHG emissions.

Research also showed that high concentrate diets leading to increased average daily gain and shorter finishing periods reduced CH_4 emissions per unit of product (Lovett et al., 2005). Both Pelletier et al. (2010) and Murphy et al. (2017) reported that GHG emission intensities were greater for beef finished at pasture than on a high concentrate diet. Although the proportions of enteric fermentation were similar for pasture based and high concentrate-based finishing systems, the quantities were significantly greater for pasture-based finishing systems as emissions were accumulated over a longer production system compared to the shorter concentrate-intensive production system.

A number of studies have shown that slaughtering animals at a younger age reduces GHG emissions per animal finished and per kg of carcass. However, Taylor et al. (2020) showed that earlier age at slaughter did not necessarily lead to the greatest profitability due to the lower gross output value achieved. In improved pasture-based systems, the ability to slaughter animals at a younger age often results in a greater stocking density, thus resulting in increased GHG emissions per hectare compared with more extensive production systems, although emissions per kg of beef would be less. Crosson et al. (2011) and Murphy et al. (2018) reported that increasing output per hectare is often consistent with lower GHG emission intensity. More generally, strategies that increase dry matter production per hectare tend to reduce the emissions intensity of food production but increase total emissions per hectare. Whether reducing emissions intensity or reducing absolute emissions per hectare is the relevant measure of success depends on overall mitigation objectives that may differ between countries and even within countries depending on domestic policy frameworks. Higher intensity can lead to land sparing with constant output or enhanced output with constant land use and therefore the system needs to analyze these consequential land uses too.

Grasslands can be a carbon source or sink depending on climate, site characteristics including soil type, and management practices such as grazing management, level of fertilizer and lime application, inclusion of legumes and historical land use (Bellarby et al., 2013). Inclusion of carbon sequestration from permanent grassland would significantly improve the relative performance of pasture compared with grain-based production systems from net GHG emission perspective (Soussana et al., 2010). However, due to the temporal and spatial uncertainties in calculating the potential of soil carbon sequestration, carbon sequestration is often omitted from modelling studies of pasture-based ruminant systems (Crosson et al., 2011). The same applies to GHG emissions due to land use change that might result from lower production, depending on the economy and policies.

Part 4. Metrics for quantifying impact of methane emissions

6.1 Introduction

Different GHGs have distinct chemical and physical properties, and these determine the effects of these emissions on the climate, in terms of both the strength and duration of any climate impacts. The fundamental properties of different GHGs, and their ultimate effects on global warming, are generally well-understood and scientifically uncontested. For some purposes, including most climate science, we can work directly from our physical understanding of individual gases, using climate models of varying complexity to explore the contribution of different GHGs to warming or other climate impacts, or quantify the benefits of potential emission reductions.

Emission metrics can provide a means of comparing different greenhouse gas emissions by putting them on to one scale, typically by quantifying a specified climate impact of a non-CO_2 gas relative to that of a CO_2 emission, reported as 'CO_2-equivalents'.

Emission metrics are used for a variety of purposes, in particular for reporting and monitoring emissions at global, national, regional or institutional levels, trading emissions of different GHG against each other, and aiding mitigation decision making especially in trade off situations when reducing one gas is very costly but reducing another gas is much less costly; or when decreasing the emissions of one GHG contributes to increasing the emissions of another GHG.

In principle, emission metrics can also be used to compare the effect of non-gaseous climate forcers (e.g., aerosol or albedo change; Collins et al., 2013; Bright and Lund, 2021) with the effect of greenhouse gas emissions. However, there are also important differences in that the climate impact of aerosol emissions depends strongly on the location of emissions and can have variable impacts on precipitation. In this report, our focus is on metrics for greenhouse gas emissions only, primarily CH_4, and to a lesser extent N_2O. We will, therefore, use the 'GHG emissions metrics' terminology.

The following definition is from the glossary of the IPCC's Sixth Assessment Report (IPCC 2021):

> **Greenhouse gas emission metric:** A simplified relationship used to quantify the effect of emitting a unit mass of a given greenhouse gas on a specified key measure of climate change. A relative GHG emission metric expresses the effect from one gas relative to the effect of emitting a unit mass of a reference GHG on the same measure of climate change. There are multiple emission metrics and the most appropriate metric depends on the application. GHG emission metrics may differ with respect to (i) the key measure of climate change they consider, (ii) whether they consider climate outcomes for a specified point in time or integrated over a specified time horizon, (iii) the time horizon over which

the metric is applied, (iv) whether they apply to a single emission pulse, emissions sustained over a period of time, or a combination of both, and (v) whether they consider the climate effect from an emission compared to the absence of that emission, or compared to a reference emissions level or climate state.

Notes: Most relative GHG emission metrics (such as the Global Warming Potential (GWP), Global Temperature change Potential (GTP), Global Damage Potential, and GWP*), use CO_2 as the reference gas. Emissions of non-CO_2 gases, when expressed using such metrics, are often referred to as "CO_2 equivalent" emissions. A metric that establishes equivalence regarding one key measure of the climate system response to emissions does not imply equivalence regarding other key measures. The choice of a metric, including its time horizon, should reflect the policy objectives for which the metric is applied.

A wide range of emission metrics have been proposed. As different greenhouse gases are not direct analogues of each other, with differences in how each emission affects the climate over time, any definition of 'equivalence' relies on a judgment about what aspect is being compared. Consequently, different emission metrics sometimes provide strikingly divergent results, despite being based upon the same physical understanding of the effects of GHG emissions on the climate. The differences between metrics rests on which aspects of the physical response are used as proxies to represent climate change and over what time horizon. For short-lived species such as CH_4 the CO_2-equivalence can differ substantially between different metrics, whereas for longer-lived species such as N_2O, the values provided are relatively consistent across different metrics on timescales up to a century.

A fundamental conclusion from the scientific literature on metrics is that the most appropriate metric depends on the objective (i.e., on the specific environmental or climatic information being sought, or the policy-question to be addressed, and over which time horizon). For some applications, there may be external requirements to use a specific emission metric. For example, the Paris Agreement rulebook states that countries must report their emissions using the 100-year Global Warming Potential (GWP_{100}), and the GWP_{100} is the *de facto* standard metric for a range of other purposes. This is despite the cautious note added by the IPCC when it first introduced the GWP in its First Assessment Report in 1990. Specifically, the authors noted: "It must be stressed that there is no universally accepted methodology for combining all the relevant factors into a single [metric] . . . A simple approach [i.e., the GWP] has been adopted here to illustrate the difficulties inherent in the concept" (IPCC, 1990; brackets added by Shine, 2009).

Apart from conceptual consistency between metrics and policy objectives, relevant considerations can also include the scientific uncertainty of metric values, the ease of communication and tangible relevance of a metric for a variety of stakeholders and uses (e.g. the link between physics-based metrics and their interpretation in an economic or broader policy context), and the consistency or compatibility of any given metric with existing climate change targets and obligations (e.g. Balcombe et al, 2018). Given this wide set of criteria, most metrics are reasonably well suited for some applications and less well suited for others. For some applications, emission metrics may not be necessary at all. The ultimate choice must balance the need to use a large number of different metrics for scientific and policy completeness and the need to use a small set of metrics that may be imperfect but could be considered good enough for a range of applications and/or a pragmatic policy choice.

In this chapter, we expand on these points, describing and explaining some of the key emission metrics, and discuss how they might relate to different scientific or policy concerns. We will guide the reader through the meanings and implications of some key metrics, with simplified illustrations of their uses. This description is primarily aimed at aiding those involved in baseline assessments, and in making greenhouse gas mitigation choices within agricultural supply chains.

6.2 Context and definitions

6.2.1 Key Principles of GHG emissions metrics

The primary role of greenhouse gas emission metrics is to help provide information on how different greenhouse gas emissions (or activities emitting them) contribute to climate change and its resulting impacts (e.g., Fuglestvedt et al., 2010), or conversely, on the benefit that avoiding any given emission(s) would provide in terms of avoided climate change and impacts. This may take the form of describing how different activities or sectors contribute to overall climate change or climate change impacts, the assessment of priorities and tradeoffs of emitting and mitigating different GHGs, or it may aid decision making and identify the most critical and/or efficient routes to meeting overarching climate targets. Emission metrics then provide a shortcut in the cause-effect chain and translate emissions to impacts, as shown in the figure below (reproduced from Myhre et al., 2013).

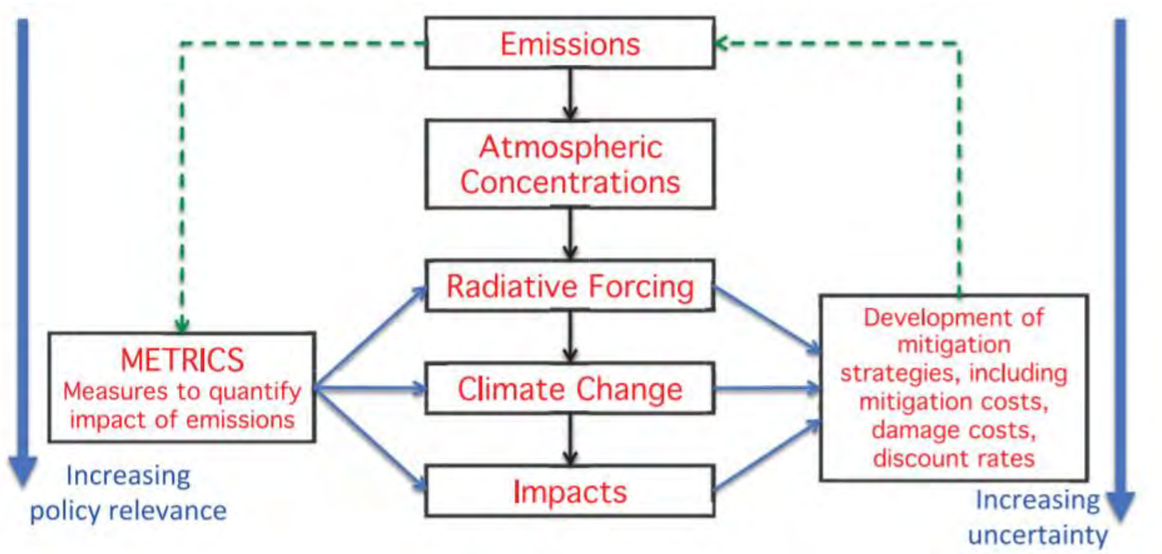

Figure 5. The cause–effect chain from emissions to climate change impacts.
It illustrates the role of metrics in defining the estimated responses to emissions (left) and for the development of multi-component mitigation strategies (right). The relevance of the various effects increases from emissions to impacts but the uncertainty increases as well. The dotted line on the left shows that effects and impacts can be estimated directly from emissions, while the dotted line on the right side indicates the use of these estimates in the development of strategies to reduce emissions. (Adapted from Fuglestvedt et al., 2003, and Plattner et al., 2009.). Figure reproduced from Myhre et al., 2013, Chapter 8 IPCC AR5.

To provide context for the further description and elaboration of different metrics, here we briefly describe the steps in the cause-effect chain (Figure 5) from emissions to climate change impacts.

Emitting a greenhouse gas increases the atmospheric concentration of that gas for a characteristic length of time, depending on how long it takes for that gas to break down or dissipate in the atmosphere.[4] One-off (pulse) emissions of short-lived gases such as CH_4 (with an average atmospheric lifetime of around a decade) will raise atmospheric concentrations for a couple of decades, while emissions of long-lived gases such as N_2O (with an average atmospheric lifetime of around a century) will result in more prolonged concentration increases. CO_2 has a complex atmospheric lifetime as it is removed from the atmosphere by different processes with different rates, but can largely be considered an extremely long-lived gas, with a significant fraction of emissions remaining in the atmosphere for millenia (Archer et al., 2009; Joos et al, 2013). Figure 6 shows the different effects on radiative forcing and temperature change for one gigatonne (Gt) pulse emissions of CO_2, CH_4, and N_2O. The remainder of this section will explain the key principles which underlie GHG emission metrics.

[4] The lifetime of a greenhouse gas is the time scale for the increased concentration arising from an instantaneous pulse emission to decay in the atmosphere. For gases following an exponential decay, the lifetime is characterised by its exponential decay constant.

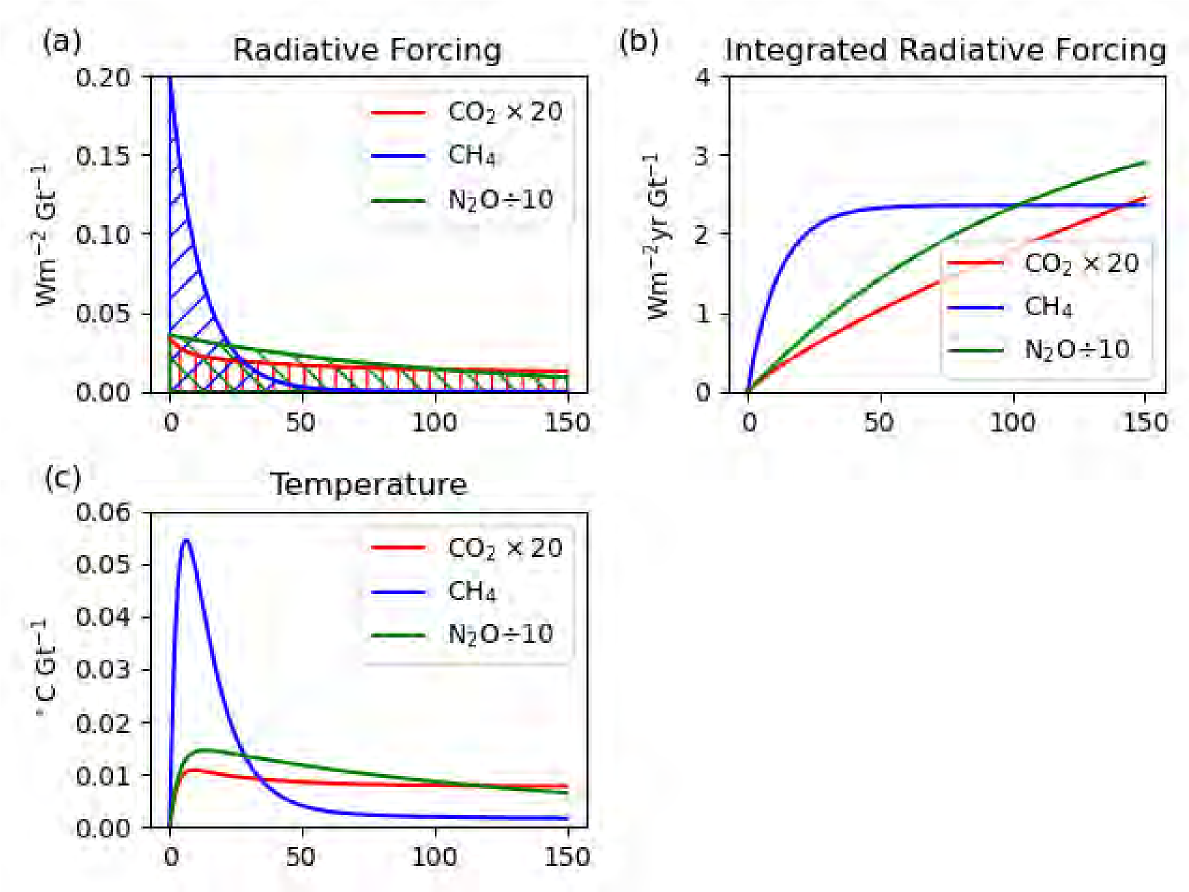

Figure 6. Different effects on radiative forcing and temperature change for one gigatonne (Gt) pulse emissions of CO_2, CH_4, and N_2O.

(a) solid lines are the global mean radiative forcing change following a pulse emission of each gas. The absolute GWP for each gas is defined as the area under each curve (hatched) up to the chosen time horizon. (b) lines represent the areas under the curves in the left-hand panel. The absolute GWP is the value of the curve at the chosen time horizon. (c) lines represent the global mean temperature change following a pulse emission of each gas. The absolute GTP for each gas is defined as the value of the curve at a chosen time horizon. Contributions from each gas have been scaled by different multipliers to make it easier to compare different gases on the same graph.

Changes in greenhouse gas concentrations impact the climate by changing the atmospheric energy balance (radiative forcing). The extent to which a given change in concentration of a gas leads to radiative forcing is known as its 'radiative efficiency', and can be thought of as a measure of the 'greenhouse strength' of different gases (Forster et al., 2021).

Any gas that warms the surface perturbs the terrestrial and oceanic carbon fluxes (Arora et al., 2020), typically causing a net flux of CO_2 into the atmosphere and hence further warming. This aspect is already included in the carbon cycle models that are used to generate the climate effects of a pulse of CO_2 (Joos et al., 2013), so for consistency this also needs to be included for non-CO_2 gases (Gillett and Matthews, 2010; Gasser et al., 2017). The metric values provided by IPCC AR6 (Forster et al., 2021) therefore now include the carbon cycle response by default.

For emissions of chemically-reactive gases, their impacts on other greenhouse gases also need to be accounted for. For example, as CH_4 breaks down in the atmosphere, it leads to the formation of tropospheric (lower atmosphere) ozone and stratospheric (the layer of the atmosphere above the troposphere) water vapour. Increased concentrations of tropospheric ozone and stratospheric water vapour also result in radiative forcing, and CH_4 emission metrics generally also include these indirect effects in their assessment of the effect of CH_4 emissions on the climate (see Forster et al., 2021).

As the physical driver by which climate is affected, radiative forcing presents a potential proxy measure of 'climatic impacts' to compare emissions of different gases and is used as the point of comparison in the most common GHG emission metric, the Global Warming Potential (GWP, see below for further details). It is also possible to continue along the cause-effect chain (see Figure 5), and base comparisons on the expected climate change (e.g., increase in global temperature) that will result from this radiative forcing. Another relatively common emission metric, the Global Temperature change Potential (GTP, also discussed below), takes this approach, comparing emissions on their relative contribution to global temperature change at a specific point in time following the emission.

Metrics can progress still further to quantify impacts as the damages resulting from climate change, for example economic damages (Hammitt et al., 1996), individual environmental impacts such as precipitation or sea-level rise (Shine et al., 2015; Sterner et al., 2014; Kirschbaum, 2014). As highlighted in Myhre et al. (2013), using a point of comparison further along the cause-effect chain can provide more direct information for communicating impacts and informing decision-making, but it also adds to greater uncertainty as more processes must be modelled at each step along the cause-effect chain. Some of the relatively simple physical metrics such as GWP and GTP can also be linked to cost-benefit and cost-effectiveness approaches to climate policy in specific contexts (see Sections 6.3.2 and 6.3.3 below for details).

6.2.2 Pulse-emission metrics

Most greenhouse gas emission metrics are based on the comparison of a pulse of emissions of 1 kg of one gas to another and provide a relative valuation or 'exchange rate' to compare the impacts of those emissions. This valuation is typically made in relative terms, with CO_2 taken as the reference gas to provide a single weighting factor to convert emissions of non-CO_2 gases to a 'CO_2-equivalent' (CO_2e) quantity, i.e. the values in tables 6.1 and 6.2 show how many kg of CO_2 a 1 kg emission of CH_4 is equivalent to. Different gases differ both in their climatic impacts and atmospheric lifespan. Quantification and comparison between different gases, therefore, requires a prior definition of the assessed climate impact and relevant time-horizon. Even though GWP is a relatively simple metric based on physical science only, it can serve as a proxy for metrics that evaluate the damage resulting from emissions based on an economic perspective (Tol *et al*., 2012). Global damage potentials are discussed in section 6.3.3.

6.2.2.1 GWP

The most common GHG emission metric, the Global Warming Potential (GWP), compares the radiative forcing accumulated over a user-defined time-horizon resulting from a pulse-emission of a specific GHG compared to a pulse-emission of equal mass of CO_2. The most frequently used, and effective 'standard', version of this metric is the 100-year Global Warming Potential (GWP_{100}). It is defined as the total radiative forcing occurring over the subsequent 100-year period after a GHG emission, relative to that of a pulse emission of CO_2 of equal mass. Myhre et al. (2013) described it as: "A direct interpretation is that the GWP is an index of the total energy added to the climate system by a component in question relative to that added by CO_2."

For short-lived greenhouse gases, such as CH_4, GWP values vary significantly depending on the time-horizon used. With increasing time-horizons, the relative valuation of short-lived vs long-lived gases declines, as there is an extended period over which the long-lived gas continues to exert a radiative forcing effect on the climate while the short-lived gas is no longer in the atmosphere and can no longer exert a direct radiative effect. This is shown in Table 6 below (GWP values from the IPCC's AR6, Forster et al., 2021), where the 20-year GWP for CH_4 is much greater than its 100-year GWP. N_2O has a lifetime of over a century, so its GWP values are less sensitive to the choice of time-horizon (up to 100-years, at least) than for CH_4 (Table 6). There are large uncertainties in all metrics (30 - 40 percent) due to uncertainties in the radiative efficiency of different gases as well as indirect effects, and uncertainty about the atmospheric longevity of both CO_2 and any gases that CO_2 is compared with. The categorization of CH_4 into "Fossil" and "non-Fossil" reflects whether the carbon introduced into the atmosphere is considered new or not (or already included in budgets) (see section 6.3.7).

Table 6. GWP values from the IPCC's Sixth Assessment Report (AR6), Forster et al., 2021.

	GWP_{20}	**GWP_{100}**
Fossil CH_4	82.5 +/- 25.8	29.8 +/- 11
Non-fossil CH_4	79.7 +/- 25.8	27.0 +/- 11
N_2O	273 +/- 118	273 +/-130

6.2.2.2 GTP

Another metric that is relatively common is the Global Temperature change Potential (GTP). It compares the temperature increase resulting from a pulse-emission of a specific GHG compared to the effect of a pulse-emission of CO_2 of equal mass, at a specific user-defined point in time after the emission (Shine et al., 2005). So, for example, the 20-year GTP of CH_4 represents the increase in global average temperature resulting from a pulse CH_4 emission compared to that of a pulse CO_2 emission of the same mass 20 years after these emissions. The 100-year GTP provides the same comparison 100 years after the emission (i.e., for emissions occurring in the year 2021, it compares gases based on the temperature increase resulting from these emissions in the year 2121). As shown in Table 7, the GTP for short-lived gases is highly sensitive to the choice of time-horizon.

The GTP is more sensitive to the choice of time-horizon than the GWP because it is an end-point metric that compares impacts only at the end-point of the specified time-horizon whereas the GWP integrates impacts over all individual years within the time-horizon. As an integrated metric, the GWP provides insights into total impacts (with radiative forcing as the proxy impact measure) that result from a given emission over the whole time-horizon. This can be appropriate for trying to reduce the overall potential damages when the effect depends on how long the change occurs for, not just how large the change is at a single future point in time. In contrast, the GTP, as an end-point metric, provides information about impacts (with temperature change as the proxy impact measure) only at the individual year specified. A key application of GTP would be for the quantification of the contributions of different gases to the goal of not exceeding any set temperature target at a specific future point in time.

GTP can also be applied to a sustained constant change in emissions (i.e., an emission of 1kg of gas per year, instead of a single emission) and is then known as the sustained GTP or GTP_s (Shine et al., 2005). Another related metric is the integrated GTP, e.g. $iGTP_{100}$ integrates GTP over 100 years, and has values which are similar to GWP_{100} (Peters et al., 2011). We have not shown values for the sustained or integrated GTP here, as they do not appear in AR6.

Table 7. GTP values based on formulae from the IPCC's Sixth Assessment Report (AR6), Forster et al., 2021.

	GTP_{20}	GTP_{100}
Fossil CH_4	54 +/- 21	7.5 +/- 2.9
Non-fossil CH_4	52 +/- 21	4.7 +/- 2.9
N_2O	297 +/- 134	233 +/- 110

6.2.3 'Step/Pulse' Metrics

Due to the strong influence of the chosen time horizon on the pulse-emission metrics for shorter-lived species described above, alternatives for calculating climate equivalence have been developed. 'Step/pulse' equivalence has been proposed as an alternative means of comparing the emissions of long and short-lived greenhouse gases. This type of 'equivalence' is possible because a single pulse emission of CO_2 and a sustained step-change increase in CH_4 emissions have similar impacts on global mean temperature increases (Allen et al., 2022a). This approach can be thought of as defining 'equivalence' by working backwards from the respective temperature outcomes. If an individual CO_2 emission has a certain impact on global temperature, is it possible to define 'equivalent' CH_4 emissions that would result in approximately the same temperature impact? A number of papers published over the past decade (e.g. Smith et al., 2012; Lauder et al., 2013; Allen et al., 2016; Collins et al., 2020) have suggested that this can be achieved by equating a permanent step-change in the rate of CH_4 emissions to an individual pulse of CO_2 emissions, as both would result in a similar incremental increase in long-term global mean temperature. An alternative perspective resulting from this type of equivalence is that the global mean temperature effect over time of an individual CH_4 emission is more akin to a large CO_2

release followed by a subsequent removal of a slightly smaller amount of CO_2, rather than to a single individual pulse of CO_2 emissions (Allen et al., 2021).

As this type of equivalence is based on matching eventual warming outcomes of the emissions being described, it has been suggested that step/pulse metrics can report 'CO_2-warming-equivalence' (CO_2-we), in contrast to CO_2-e from pulse-emission comparisons (Cain et al., 2019). There were earlier attempts to match warming or forcing outcomes under a scenario based on pulse- or step-based metrics (Wigley, 1998; Tanaka et al., 2009, 2013).

Under step/pulse metrics, introducing a new sustained CH_4 emission from a source (i.e. a step-change from no emission to a constant emission) could be considered as equivalent to a large one-off pulse of CO_2 emission, both adding significant additional warming. For this new sustained CH_4 source, most of the resultant warming will be realised within the first few decades, as ongoing emissions will be balanced by the chemical reactions which destroy atmospheric methane after a few decades of stable CH_4 emission rates. That will result in stable atmospheric concentrations of CH_4 and a stable contribution to radiative forcing. Additional warming will continue at a much lower rate for several centuries, as the climate fully adjusts to the elevated radiative forcing (Cain et al., 2019; Smith et al., 2021). This scenario is shown by the middle column of panels in Figure 7, which illustrates constant emissions of CO_2 and CH_4 and the resultant level of warming they each generate.

If these sustained CH_4 emissions are reduced at any point, CH_4 concentrations will decline as natural removals continue without the removed CH_4 being replaced. This will then lead to lower temperatures (right column in Figure 7). To similarly reduce the level of warming from an earlier CO_2 emission, it would have to be actively removed from the atmosphere. Definitions and assumptions about past emissions and the climatic impacts they may still be exerting, and how to define existing or new sources, therefore have a large impact on the calculated equivalent CO_2 emissions that would result in the same temperature change.

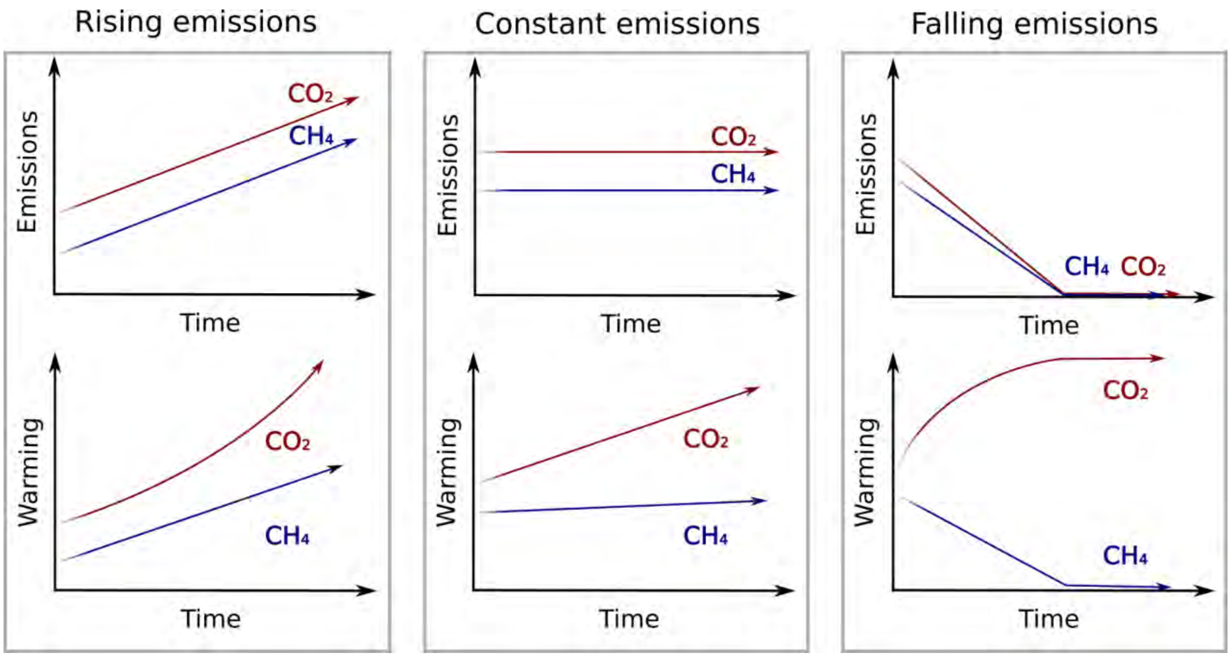

Figure 7. An illustration of how rising (left), constant (middle) and falling (right) emissions of CO_2 (red) and CH_4 (blue) affect levels of global warming.

For both CO_2 and CH_4, rising emissions drive temperatures up. For constant emissions, CO_2 drives temperatures up at a slower rate than for rising emissions, but for CH_4 the level of warming is only very slightly rising. For falling emissions, CO_2 continues to drive temperatures up until emissions are eliminated. For CH_4, falling emissions lead to falling temperatures. This fundamental difference between CO_2 and CH_4 is why pulse emission metrics do not reflect temperature changes arising from short lived pollutants accurately, and why step/pulse metrics were developed to assess temperature outcomes. Figure reproduced from Allen et al., (2022b).

'Step/pulse' equivalence has been defined via a small number of approaches. One approach (referred to as GWP*, denoting a modified GWP approach) estimates the equivalence in terms of global mean surface temperature increase between a sustained flow of CH_4 emissions and an individual pulse-emission of CO_2 (Allen et al., 2016). This approach has been updated to improve the accuracy of the relationship between the CO_2-warming-equivalent emissions calculated using GWP* and modelled temperature (Cain et al., 2019, Smith et al., 2021). Lynch et al. (2020) demonstrated GWP* in a wider range of scenarios exploring its use to estimate temperature responses to non-global emission trajectories and Cain et al., (2021) used GWP* to evaluate scenarios which aim to achieve the Paris Agreement temperature goals.

The equation to convert a methane emission ($CH_4(t)$) to a CO_2-warming-equivalent ($CO_2\text{-we}(t)$) emission, using GWP*, is:

$$CO_2\text{-we}(t) = GWP_{100} \times (4.53 \times CH_4(t) - 4.25 \times CH_4(t-20))$$

which simplifies to:

$$CO_2\text{-we}(t) = 8 \times CH_4(t) + 120 \times \Delta CH_4(t)$$

where GWP_{100} here is the normal GWP for pulse emissions of CH_4 and CO_2 from AR5 (following Smith et al., 2021 and Forster et al, 2021); $CH_4(t)$ and $CH_4(t-20)$ are the current CH_4 emission rates and those 20 years earlier; and $\Delta CH_4(t) = CH_4(t) - CH_4(t-20)$, i.e., the difference in methane emission rate between time t and 20 years prior (Smith et al. (2021)).

This formula for GWP* allows calculation of CO_2-warming-equivalent emissions for any time series of CH_4 emissions, i.e., not just a single and permanent step-change. The resulting CO_2-we emissions will then result in approximately the same change in temperature as the time series of CH_4 emissions. This is shown for two future scenarios in Figure 8. Figure 8a shows a lower ambition scenario for methane emissions, and panel b a higher ambition scenario. The modelled warming from the emissions is shown by the heavy black line. Cumulative CO_2-we emissions calculated using GWP* are shown in green, and they are a good approximation of the modelled warming for both scenarios. The GWP* is a 2-term approximation to find the CO_2-equivalent emissions that would generate the same radiative forcing time series as is generated by the CH_4 emissions (Allen et al., 2021).

A second approach, developed by Collins et al. (2020), provides an alternative method where the forcing or temperature of a pulse of CO_2 emissions is compared with a step change in the rate of emissions of short-lived gases over a specified period to report the 'combined global warming potential' (CGWP) and 'combined global temperature change potential' (CGTP), respectively. The CGTP metric is similar to GWP* in that it compares the warming resulting from a change in the rate of CH_4 emissions with the warming that results from a pulse emission of CO_2. The approximation made in CGTP is that the time evolution of the CH_4 emissions is unimportant, and that only the difference between the initial and final emission rates are relevant (provided that most of the change in emission rates is achieved a few decades before the end of time horizon of interest). This makes it useful for addressing the effects of permanent changes in CH_4 emission rates on long-term warming, though is less accurate when the CH_4 emission rates vary close to the time frame of interest. Cumulative CO_2-we emissions calculated using CGTP100 are shown for the two scenarios in Figure 8 in orange, and also show good agreement with the modelled warming for both. The two step-pulse metrics (CGTP100 and GWP*) are able to capture the reduction in warming resulting from methane emissions cuts, which cannot be captured with GWP100 (dark blue) or GWP20 (light blue), if it is assumed that the warming from past CH_4 emissions would persist in the same way as as it does for CO_2. GWP* also represents the historical period more closely. Further discussion of both can be found in Forster et al. (2021).

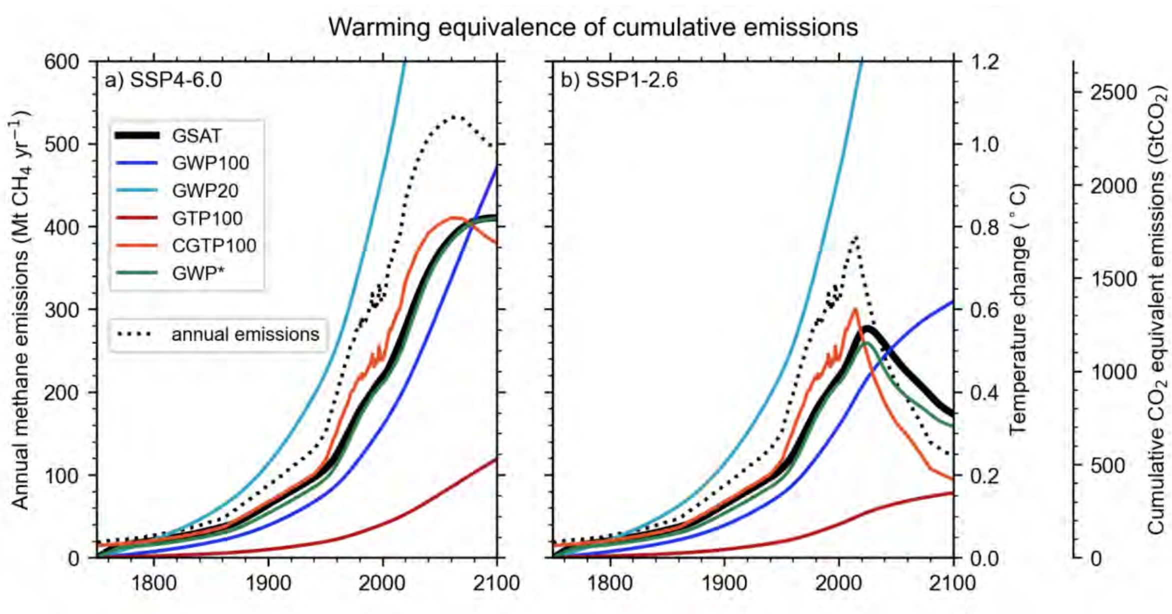

Figure 8. Cumulative CO_2 equivalent emissions of methane are shown calculated using different metrics, for two mitigation scenarios named SSP4-6.0 (panel a) and SSP1-2.6 (panel b).

Temperature response from these emissions calculated using an emulator is shown in the black line (labelled GSAT for global surface air temperature). Figure reproduced from Forster et al., (2021), Chapter 7 of AR6.

6.2.4 Key differences between 'step/pulse' and pulse-metrics

As highlighted above, there is a fundamental distinction between pulse-emission metrics and step/pulse metrics. One way to consider the different metric concepts is to explore how they might be used, e.g. to assess marginal impacts (i.e. the impact of emitting vs not emitting an individual emission) or to assess the additional warming an emission would cause (i.e. the impact of changes in emissions of short-lived gases from their current values). In this section, we use the term 'marginal' to refer to the effect of future emissions compared to those future emissions not occurring. Marginal emissions capture the effect from those emissions and therefore the benefit of avoiding those emissions, which is relevant for choices about the effort and costs that might be justified (from a cost-benefit or cost-effectiveness perspective) to mitigate future emissions (Dhakal et al., 2022, supplementary material). We use the term 'additional' warming to mean the effect on temperature of emissions after a specific year, relative to the level of warming in that specific year. The marginal warming from future CH_4 emissions is always positive and can be compared to the marginal warming from CO_2 (see Figure 9). The additional warming from future CH_4 emissions can be negative if they are reduced year on year.

Climate change impacts could be assessed by using modelling of radiative forcing or temperature change as proxies, or by going into greater detail in describing the connection between temperature changes and resultant impacts (Kirschbaum, 2014, 2017). Impacts can be calculated for just one point in time, or they can be integrated over the whole time horizon.

Pulse and step-pulse metrics can both be used to understand marginal and relative climate change outcomes, but they achieve this through different types of applications.

Pulse-emission metrics primarily provide information about marginal impacts. Each pulse emission metric provides an account of the future climate impacts (as defined by the specific metric) that would be caused by an extra unit of emission of a given gas. For example, GWP_{100} quantifies the radiative forcing over the next 100 years that would result from emitting 1 tonne of CH_4, compared to not emitting this tonne, and expresses this in terms of emitting a specified number of tonnes of CO_2 that would result in the same total radiative forcing over the next 100 years.

By contrast, step/pulse metrics have primarily been used to show the change in temperature over time caused by a particular emissions pathway, relative to warming at a reference date caused by previous emissions. For example, GWP* approximates the temperature change that would result from a change in CH_4 emissions relative to emissions 20 years prior. This is then expressed in terms of the effect of emitting or removing a specified number of tonnes of CO_2 with the same effects on global temperatures. These different perspectives are illustrated in Figure 9 for the contributions to global warming from global net CO_2 emissions, and from global CH_4 emissions from livestock, in a pathway that limits global warming to 1.5 degrees with limited overshoot.

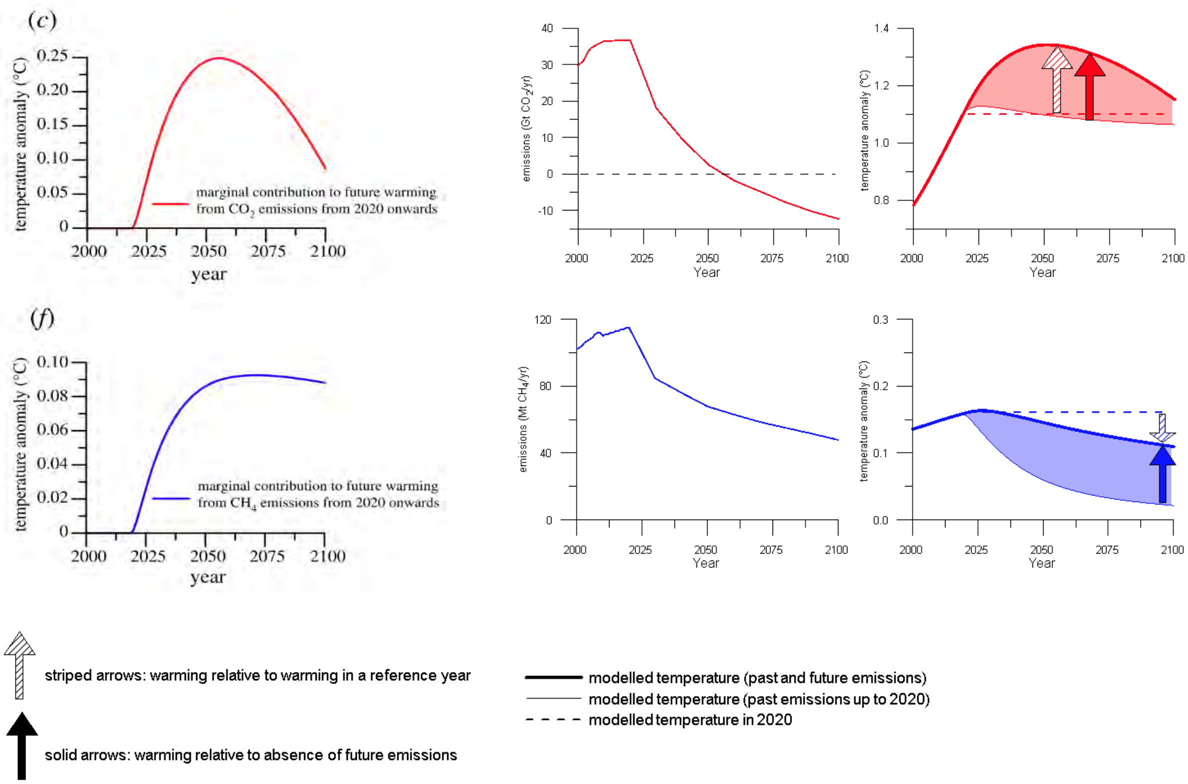

Figure 9. Contributions to global warming from global net CO_2 emissions, and from global CH_4 emissions from livestock, in a pathway that limits global warming to 1.5 degrees with limited overshoot.

Striped arrows illustrate the warming/cooling from future emissions of CO_2 and CH_4 relative to warming in 2020 ('additional' warming), solid arrows and shaded areas indicate the warming from future emissions of CO_2 and CH_4 relative to the absence of those future emissions ('marginal' warming). Marginal warming is shown in the right-hand column of panels. Note that scales are different in the vertical axes, and are showing global net CO_2 emissions and livestock CH_4 emissions. Adapted from Reisinger et al (2021).

Figure 9 provides an illustration of these different perspectives, with the striped arrows showing warming relative to warming in a reference year (or baseline year), and solid arrows showing warming relative to an absence of future emissions. It shows that the choice of defining impacts of emissions relative to a baseline or relative to the absence of ongoing emissions has significant implications for the different gases. The left panels show global CO_2 (upper plot) and CH_4 (lower plot) trajectories in an ambitious mitigation scenario. The right panels show the corresponding contribution to global temperature increase (above pre-industrial temperatures) from either gas, with the thicker line showing temperature-change contribution if gases follow their respective emission pathways, while the thinner line shows the temperature-change contribution if emissions of the gas ceased entirely in 2020.

The relative temperature change resulting from these emission scenarios can be defined from two different perspectives: the effect on global temperatures relative to 2020 (which might be useful to consider how different trajectories would contribute to overall global temperature change, for example), as illustrated by the striped arrows; or the effect on global temperature of these emissions compared to not emitting them (which might be useful to consider the warming caused by future emissions, and the benefits of avoiding different emissions, for example), as illustrated by the solid arrows. The solid arrows are what pulse metrics, such as the GWP or GTP are typically used to express (termed a 'marginal' approach in Reisinger et al. 2021; see AR6 WGIII Chapter 2 and supplementary material for more details), while the striped arrows correspond more to the way step-pulse metrics such as GWP* have been used to date, which we refer to as a 'baselined' approach below.

Due to the different atmospheric lifetimes of the two gases, the consequences of a 'no-emission' pathway differ greatly for CO_2 and CH_4, for the reasons described above. While the 'marginal' (solid arrow) or 'baselined' (striped arrow) approaches are very similar for CO_2, and so either approach could be derived or justified from a perspective of 'CO_2-equivalence', they provide very different perspectives on how to consider the impacts (or avoided impacts) of CH_4 emissions (solid and striped arrows in lower right panel of Figure 9). These differences have important implications for the interpretation and understanding of CO_2 "equivalent" emissions calculated under either type of metric. Which perspective is deemed most appropriate may depend on practical concerns (e.g., cost-effectiveness of mitigating different emissions) or equity considerations (e.g. acknowledging the role of different sectors or activities in overall global warming), as highlighted below.

In the case of step/pulse metrics, the CO_2 emissions that are described as "equivalent" to a given *change* in rate of CH_4 emissions are those that would result in the same *change* in temperature, relative to the baseline year. In other words, in applying step-pulse metrics one must determine reference conditions against which to judge changes, and the step-pulse metric can only describe temperature changes relative to these conditions.

For CO_2, there is (broadly) no further change from a reference temperature when there are no further emissions (or net-zero CO_2 emissions). For short-lived gases, however, if there were prior emissions contributing to the reference temperature then a scenario of ongoing emissions is also effectively embedded in the reference conditions to maintain this temperature (and results in a 'CO_2-equivalent' temperature outcome). Decisions over what reference state to use for step-pulse metrics can therefore have significant implications on the relative valuation of emissions of short-lived gases. For example, a reference year of 2020, 1990, 1900 or 1750 would lead to very different valuations, but could all be applied to step/pulse metrics. It also leads to potential equity impacts that need to be considered, particularly when the approach is applied to emission assessments at sub-global scale (see section 6.5.4).

Step/pulse metrics can directly illustrate the anticipated temperature changes resulting from different emission pathways and incorporate them in 'cumulative emission budgets'. By contrast, pulse metrics answer a different question. They show the relative climate effect at one time horizon that would result from an emission without needing comparison with past emissions. Hence, there is no inconsistency between the different metrics, provided it is recognised they provide different information.

In principle, both pulse and step/pulse metrics can provide marginal or relative information. GWP* can be applied to a time series of emissions, with emissions at the beginning of the time series set to zero. This would provide information on the amount of warming caused by subsequent emissions, compared to the absence of those emissions, and hence the warming that would be avoided if those future emissions were avoided. For example, if one wishes to know the marginal warming caused by CH_4 emissions since 1990 (as opposed to the additional warming) one would set CH_4 emissions prior to 1990 to zero when applying GWP*. "Warming since 1990", and "warming caused by emissions since 1990" are not the same for CH_4 (unlike for CO_2, as shown in Figure 9), hence the policy question seeking an answer needs to be clear, in particular for short lived gases. Conversely, pulse metrics like GWP and GTP can be applied to the difference between a given emission and a baseline emissions level, and would thus be a 'baselined' approach.

6.2.5 Time horizon/Endpoint for metrics

The pulse metrics discussed in section 6.2.3 depend very strongly on the chosen time horizon. The choice of time horizon depends on policy priorities. While particular policy goals may not directly specify a particular time horizon, some possible time horizons could be argued to make more sense than others (Shine et al. 2005; Abernethy and Jackson 2022).

For instance, if the goal is specifically to limit warming to 1.5 degrees with no or limited overshoot, peak warming will occur roughly around 2050 (determined by climate-economic modelling suggesting plausible emission reduction scenarios that will limit warming in accordance with this target). From that perspective, and if the purpose of a metric is to design climate change mitigation strategy based on relative valuation of present-day emissions according to their marginal contribution to this temperature goal, it could therefore make sense to value each emission based on the contribution it makes to warming in the year 2050; i.e. to use the GTP with a time horizon of 30 years for emissions occurring in 2020. Applying this logic consistently would mean that emissions occurring in the year 2030 would be valued with GTP_{20} (although the time frames would likely need to be re-evaluated as the target is approached). It would be inconsistent with this stated policy goal to use GTP_{100}, because the warming in the year 2120 (which is what GTP_{100} describes, for emissions occurring in the year 2020) has no direct significance relative to a policy goal of limiting warming to 1.5 degrees with no or limited overshoot. In practice, there may be multiple policy goals, and not all policy goals can be translated into time horizons and relevant metric choices. For example, if the goal is to limit warming to 1.5 or well below 2 degrees, peak warming could occur as early as 2050 or as late as perhaps 2080, which means there is no single GTP value that satisfies these goals. In addition, stakeholders may not have a clear global policy goal in mind and only want to do their part in limiting their impact on the global climate. In that case, using a metric that is more akin to the Global Damage Potential may be more relevant, although this is path-dependent (see Section 6.3.3).

6.2.6 Discount rates consideration

As climate impacts are experienced at different times in the future, decisions must be made about how to value impacts according to how far into the future they occur . Discount rates are commonly used to quantify future impacts in present value terms. The higher the discount rate, the more impacts are devalued the further into the future they occur. This would shift the mitigation emphasis towards short-lived climate forcers, like CH_4, while reducing the focus on long-lived climate forcers, like CO_2 and N_2O (van den Berg et al., 2015). Whereas a low discount rate will place the emphasis relatively more strongly on long-term climate forcers. The choice of discount rates is hence one of the most critical components of any impact analysis and can be related to the time horizon of GWP or GTP as discussed below. As with time horizons, the choice of discount rates cannot be based solely on an objective scientific basis. Furthermore some authors argue for multiple discount rates depending on the purpose or a declining-in-time discount rate (Arrow et al., 2014).

Different time horizons can be used as proxies for discount rates. By comparing the GWP to the global damage potential (GDamP, see section 6.3.2), it becomes possible to estimate the effective discount rate. Using that approach, the GWP_{100} was estimated to correspond to discount rates between about 3 percent (Mallapragada and Mignone, 2020) and 3.3 percent (with an interquartile range of 2.7 to 4.1 percent in a sensitivity analysis, Sarofim and Giordano, 2018). GWP_{20} corresponded to a discount rate of 7 percent or greater (Mallapragada and Mignone, 2020) and 12.6 percent (interquartile range of 11.1 to 14.6 percent, Sarofim and Giordano, 2018). It should however be noted that such relationships are sensitive to underlying future scenarios, among other assumptions (Mallapragada and Mignone, 2020).

6.2.7 Non radiative forcing impacts

Methane has other important social costs besides its radiative forcing effects, primarily through increasing ground-level ozone concentrations that worsen air quality. This is a major hazard to human health and toxic to plants with impacts on carbon uptake and crop yields (Shindell et al., 2017). Reducing methane emissions would therefore also reduce human mortality due to lower ozone concentrations, and Sarofim et al. (2017) calculated that this health benefit would exceed the climate change mitigation benefit of those emission reductions if they were valued at US$46 per tonne CO_2 eq. The UNEP methane assessment (UNEP and CCAC 2021) found that every Mt reduction in CH_4 emissions prevents approximately 1430 annual premature deaths and avoids annual losses of 145 000 tonnes of wheat, soybeans, maize and rice.

Nitrous oxide emissions deplete stratospheric ozone. This has been estimated to increase its social cost by 20 percent over the pure climate impact (Kanter et al. 2021).

Carbon dioxide emissions also lead to ocean acidification, and all forcing agents will contribute to sea level rise, which acts over many decades after the emission occurs (Sterner et al., 2014b).

Summary points

A metric that establishes equivalence regarding one key measure of the climate system response to emissions does not imply equivalence regarding other key measures. The choice of a metric, including its time horizon, should reflect the policy objectives for which the metric is applied. The most appropriate metric depends on the objective (i.e. on what aspect of climate change is the policy focus, and over which time horizon). {6.2.1}

The large difference in lifetimes for CO_2 and CH_4 mean that the pulse emission metrics vary very strongly with the chosen time horizon {6.2.3}. Step-pulse metrics for forcing and temperature (comparing a change in rate of CH_4 emissions with a one-off emission of CO_2) show much less variation with time horizon {6.2.4}.

A step-pulse metric (GWP*) can be used to calculate an equivalent CO_2 emission time series which gives a good approximation of the temperature time-series that would result from the original CH_4 emissions time-series {6.2.4 and shown in Fig. 6.4}.

There is no solely scientific basis to determine the choice of metric or its time horizon. However, certain policy goals such as cost-effectively deploying emission reduction efforts to keep within temperature limits may implicitly suggest particular metrics and time horizons ranges are more relevant than others {6.2.6}.

Climate metrics for CH_4 include the radiative effects of the resulting increases in ozone (and stratospheric water vapour) but not the human health and crop yield effects. These could double the social cost of methane {6.2.7}.

6.3 The use of GHG metrics in impact and mitigation applications

Emissions metrics allow a quantification of the contribution of specific activities and related GHG emission sources to climate change impacts, or a quantification of the benefits of the avoided climate change impacts by reducing their emissions. The essence of the definition of GHG emission metrics is to allow such quantification to provide objective information about the benefits or trade-offs between specific decisions. Specifically, decision makers may have to decide between the different mitigation options with different costs and benefits, which may involve evaluating the impact of reducing CO_2 andreducing CH_4 emissions. To make an objective choice between these options, decision makers need to be able to quantify the effect of interest of both emission types.

However, metrics are not always needed. Relative metrics only need to be used when there is a need to compare between the effects or contribution of different gases to climate-change impacts or other climate-change effects of interest, such as radiative forcing or temperature changes. At one level, the assessment of all gases is clear. All CH_4 (or other GHG) emissions contribute to global warming. All reductions of CH_4 emissions, therefore, help to reduce global warming. CH_4 and CO_2 differ in their atmospheric lifetimes and consequent radiative properties so that CO_2 has an ongoing warming effect centuries after its initial emission whereas the warming from CH_4 halves after a few decades (Solomon et al., 2010). This implies that global net-zero CO_2 emissions are needed to halt global warming. For CH_4, however, net-zero emissions are not necessarily needed to stabilize the climate in the long-term due to the decay of CH_4 in the atmosphere. Nonetheless, ongoing CH_4 emissions continue to also contribute to higher temperatures than would be the case in the absence of these emissions. Stakeholders may wish to set an individual reduction target for methane emissions, in which case there is no need to use any metric to track progress towards that specific emissions reduction target. Nonetheless, stakeholders may still wish to use metrics to help justify the level of ambition for a specific gas target in comparison with the level of ambition for other gases, so expressing their targets in terms of CO_2 equivalents.

6.3.1 Life-Cycle Assessment and Carbon Footprinting

Life cycle assessment (LCA) is a science-based methodology to quantify the environmental impact over the lifetime of a product or service, covering a broad range of environmental impact categories such as global warming, ecotoxicity, water scarcity and human health. It can inform users about the climate benefit of avoiding or using a given product or service, or about the consequences of substituting one product or service with another one. The ISO 14044:2006 not only specifies requirements and provides guidelines for LCAs overall, but also for life cycle inventory (LCI) studies, which is the data collection portion of LCA. LCI is the accounting for all process inputs and outputs (including resource inputs and emissions to the environment) involved in the system of interest.

In the Life Cycle Impact Assessment (LCIA) stage, LCAs use characterization factors to aggregate the attributed emissions and resource uses of different parts of the system's life cycle into a single value for various impact categories, such as global warming, or fully aggregated into a single score for typically 10-20 mid-point impact categories. The metrics should be chosen to match the user's impact objectives. For characterizing their aggregate climate change impact, LCAs inevitably require the aggregation of the emissions or removals of different greenhouse gases into a common climate change impact, hence necessitating the use of GHG metrics.

Besides specific choices in the LCIA (i.e., how to measure and how to allocate emissions among processes/products), any LCA needs to choose appropriate impact assessment models. Several LCIA methods are available, such as ReCiPe2016 (Huijbregts et al., 2017) or LC-IMPACT (Verones et al., 2020) that consist of a number of environmental impact categories (e.g.., carbon footprint or climate-change impacts, eutrophication, ecotoxicity, and others) and propose Characterization Factors (CFs) to quantitatively link the elementary flows to the selected impact categories.

To provide guidance and standardize procedures, a UNEP working group provided recommendations for specific impact categories. The choices of impact categories and impact assessment methods need to be defined as part of the goal and scope definition of a study. This also includes determination of the temporal scope and selection of an appropriate metric for climate-change impact assessment (or a simple climate model as used in LIME (Inaba and Itsubo, 2018; Tang et al, 2018). Temporal aspects include both the time of a GHG emission (inventory) and the time horizon of the impact assessment (through the chosen metric).

These choices need to be justified for any study. ISO (2006) also suggests that the choice of selecting the impact categories should be based on the specific requirements of the LCA practitioner for meeting the objective of a study (European Commission, 2010), which leaves the choice of metrics open to practitioners. Specifically addressing GHG emissions, ISO 14067 describes the principles, requirements and guidelines for quantifying the carbon footprint according to ISO 14040. All net fossil fuel emissions should be included in the quantification of the carbon footprint while net biogenic emissions should be assigned a lower weighting than fossil-fuel based CO_2 emissions when applying ISO 14067 to an assessment.

Earlier LCA guidance reports of the FAO (FAO 2016a, 2016b, 2016c, 2016d, 2018a, 2018b) were all based on using GWP_{100} but discussed possible reasons for using different climate-change impact metrics for estimating the overall impacts of different GHGs emitted within livestock production systems. More recently, the Global Life Cycle Impact Assessment Method (GLAM) of the Life Cycle Initiative hosted by the United Nations Environment Program (UNEP, 2021) has provided guidance that LCAs should report climate impact assessments with both the GWP_{100} (to represent shorter-term impacts) and GTP_{100} (to represent longer-term impacts), with consideration given to GWP_{20} and GTP_{20} for sensitivity analyses to explore very short-term impacts (Cherubini et al., 2016; Levasseur et al., 2016; Jolliet et al, 2018). These recommendations used metric values from IPCC (2013) and have subsequently been applied in various impact assessments (e.g. Reisinger et al., 2017; Iordan et al., 2018; Tanaka et al., 2019; Tibrewal et al., 2021). These considerations and wider points are also discussed in another recent report on LCA for food items published by the FAO (McLaren et al 2021).

Weighing up methane reductions vs other factors is even more difficult. An LCA can provide the framework to ensure that the analysis is comprehensive and different ways of valuing methane emissions (or reductions) can be used to quantify the benefits of making an emission reduction against potential negative trade-offs through increased emissions of other greenhouse gases or other ecosystem services, such as food production or other environmental benefits. Some studies have also attempted to compare and aggregate different LCA impact-indicator categories to directly quantify the combined overall impact across all different considered individual impacts. They include the so-called endpoint methods (e.g., ReCiPe and LC-Impact), which quantify all impact category results (such as GHG emissions, land use and water consumption) into impacts on human health, ecosystem quality and resource depletion.

They are then followed by an optional normalization and weighting step to arrive at a single score result. Existing methods use different metrics for assessing climate-change impacts, but most methods rely on GWP_{100}. This is very difficult, however, as there is no obvious way to quantitatively compare the impact of greenhouse gas emissions with unrelated, but equally important, impacts such as water yield, erosion control or biodiversity conservation. Ultimately, some judgments must be made in these comparisons when it is not possible to compare impacts on a purely objective scientific basis. The ultimate weighing up needs to reflect different values that need to be derived and agreed on through open discussion. The underlying issues are also further highlighted and discussed in the cross-cutting section of this report. Such aggregation into a single-score LCA result needs to be critically discussed since it involves many additional normative choices and can disguise the complexity and trade-offs in LCA assessments. Modelling from impact category results (e.g. CO_2 equivalents) to endpoint results (e.g. impacts on human health) leads to additional uncertainty since effects of climate change on human health involve additional and highly uncertain models.

In summary, for LCA studies to be in line with ISO standards, there are stated requirements in terms of methodology and reporting metrics. The goal and scope of an LCA needs to clearly define the objective of the study, and this might lead to different metric choices. It is clearly important to reflect on the choice of metrics as they can greatly affect the outcome of any assessment, but no general guidance can be given on the choice of the metrics to use as it will depend on the goals and objectives of the study.

6.3.2 Cost-benefit assessment of climate change mitigation

A cost-benefit analysis requires quantification of the benefit of reducing climate-change related damages. This would allow an evaluation of the trade-offs between greenhouse gas mitigation choices and the resulting climate-related damages (for example, if emissions of one gas increase while those of another decrease), or between several mitigation options that target different gases. Damage metrics are typically based on the cost of damages as a function of changes in radiative forcing or global surface temperature (Deuber et al., 2013), and conventionally, cumulative damages over time are used to assess the losses or costs of climate change.

A weakness of many assessment models is that they may not adequately account for the full effects of catastrophic impacts of climate change (Weitzman, 2012, 2013; Pindyck, 2013). Where included, the impact of catastrophic phenomena such as dangerous rise in sea level or uncontrollable positive climate-forcer feedbacks such as large and rapid release of CH_4 from permafrost, can drastically increase estimated damage values (Weyant, 2017).

One emission metric that is consistent with the cost-benefit framework is the Global Damage Potential (GDamP) (Reilly and Richards 1993; Schmalensee 1993; Fankhauser 1994; Kandlikar 1995; Hammitt et al., 1996; Tol et al., 2012; Kolstad, 2014). It can be interpreted as a more general form of the GWP (Tol et al., 2012; Deuber et al., 2013). It has been derived from an optimal pathway indicated by an Integrated Assessment Model (IAM) under a cost-benefit framework. Under an optimal pathway, the GDamP is defined as the ratio of avoided incremental damages by reducing the emissions of two gases (for example, CO_2 and CH_4). It is thus time-dependent because avoided damages generally vary over time and with the pathway of emission reductions.

On one hand, the GDamP is the most comprehensive available metric in the context of cost-benefit appraisal of emissions as it uses a single framework to consider mitigation and damages as well as the underlying climate physics. On the other hand, the GDamP is highly uncertain because of uncertainty in the many assumptions that are required to translate emissions into damages, including the choice of discount rate and the quantification of climate damages assumed in an IAM. For example, Boucher (2012) estimated the GDamP for CH_4 at 24.3 (mean) but with a large range of uncertainties from 12.5 to 38.0 (5–95 percent interval). As noted in Kolstad (2014), the difficulties in estimating the GDamP are closely related to the large uncertainties in the social cost of CO_2 and non-CO_2 gases in the atmosphere (Marten and Newbold 2012; Waldhoff et al., 2014; Shindell et al., 2017; Errickson et al., 2021). Since damage functions are uncertain, a sensitivity analysis for different damage functions can provide greater insights into the dependence of ultimate outcomes on the assumed damage functions (Kirschbaum, 2014; Kumari et al., 2019).

Kirschbaum (2014) put forward the Climate Change Impact Potential (CCIP), a metric built from damage functions. The CCIP gives equal weight to three categories of damages parameterized through elevated temperature, the rate of warming, and cumulative warming. Background conditions are calculated under the Representative Concentration Pathway (RCP) with the target radiative forcing of 6.0 W/m^2 by the end of this century (RCP 6.0), with CCIP calculating marginal impacts for extra emission units of different gases. A notable difference with the GDamP is that the CCIP does not require an IAM, which means that the CCIP considers solely damages under the specific pathway, without considering the cost of abating greenhouse gas emissions. Damage functions used in the CCIP also partly depend on the future path of background conditions (Kirschbaum 2014).

The cost-benefit or damage metrics, such as the GDamP and CCIP, have not, yet, been applied in the development or assessment of real-world climate policies although CCIPs have been used for impact assessments (Kirschbaum 2017; Brandão et al., 2019). The GDamP has not been used much in recent work, but it is discussed as part of the debate on the social cost of CO_2 and non-CO_2 gases (Marten and Newbold 2012; Waldhoff et al., 2014; Rennert et al. 2022). These metrics are also useful for evaluating and interpreting other more applied metrics such as GWP_{100} from a cost-benefit perspective.

6.3.3 Cost-effectiveness of different mitigation options

A cost-effectiveness analysis is a special case of a more general cost-benefit analysis, with the damage cost function set to zero up to the level of the climate target and to infinity thereafter (Tol et al., 2012). It considers only the cost of mitigation to achieve a specified climate target such as the long-term temperature target of the Paris Agreement. It does not consider the cost associated with climate damages and adaptation, which are generally regarded as being highly uncertain. Another difference between the two frameworks is that, while a cost-benefit analysis simultaneously calculates a target and a pathway, a cost-effectiveness analysis requires a target specification first, and then a cost-effective pathway is calculated to achieve the target. The cost-effectiveness principle is one of the key principles of the United Nations Framework Convention on Climate Change (UNFCCC) [Article 3 of United Nations (1992)] and a guiding principle for climate mitigation pathways presented in previous Intergovernmental Panel on Climate Change (IPCC) Reports.

A metric that is consistent with the cost-effectiveness framework is the Global Cost Potential (GCP) (Manne and Richels, 2001; Johansson, 2012; Tol et al., 2012; Tanaka et al., 2013; Tanaka et al., 2021). GCP can be seen as a more general form of the GTP (Tol et al., 2012). The GCP is defined as the ratio of the willingness to pay for saving the emission of an additional unit of a gas of interest to that of CO_2 at each point in time under a cost-effective pathway. Similar to the GDamP (see Section 6.3.2), a calculation of the GCP requires an IAM (but run under a cost-effectiveness framework), which makes the GCP path- and time-dependent.

Taking CH_4 as an example, the GCP for CH_4 is the ratio of the anticipated future prices of CH_4 and CO_2 on a cost-effective pathway (also called the "price ratio" (Manne and Richels, 2001)) as derived from an IAM for a given climate target (for example, a 2°C warming target). The GCP depends on the climate target, the chosen pathway toward the temperature goal and a range of socio-economic assumptions. The GCP is time-dependent because the prices of CO_2 and CH_4 change over time under a cost-effective pathway. The GCP increases over time up to the point when a temperature target is reached and stays at approximately the same level thereafter (Manne and Richels, 2001; Johansson, 2012; Tanaka et al., 2013). Tanaka et al. (2021) showed that the GCP for CH_4 is relatively close to GWP_{100} up until mid-century under a variety of pathways, but beyond mid-century, GCP starts to significantly deviate from GWP_{100}, depending strongly on the future pathway that will unfold. This analysis supports the use of GWP_{100} for the Paris Agreement at least till the mid-century, with metrics with shorter time horizons becoming more appropriate thereafter.

The temporal change of the GCP value can be approximated by the Cost-Effective Temperature Potential (CETP) (Johansson, 2012). The rising trend of GCP up to the point of stabilization can be captured by a dynamic GTP (Shine et al., 2007) and other dynamic metrics such as the TEMperature Proxy index (TEMP) (Tanaka et al., 2009, 2013). A dynamic metric uses a time horizon with the end point typically being tied to the year of meeting a climate target (Berntsen et al., 2010; Abernethy and Jackson, 2022; McKeough, 2022). In other words, a dynamic time horizon will be shortened as it moves forward to the future, and the metric would have to be adjusted as the emission pathway unfolds. The proximity of the dynamic GTP to GCP justifies the use of the dynamic GTP for analyses of cost-effectiveness, but it has rarely been applied outside of academic research, possibly because there is no commonly agreed year of meeting a temperature target.

The path- and time-dependence of GCP shows that there are limits to the optimality of static metrics such as GWP_{100}. That is, there is an economic cost associated with ongoing use of GWP_{100} instead of use of the GCP or other time-varying metrics. Previous studies showed, however, that the use of GWP_{100} increases global total abatement costs under stabilization pathways by only a few percent (O'Neill 2003; Aaheim et al., 2006; Johansson et al., 2006; van den Berg et al., 2015; Tanaka et al., 2021). Despite relatively small global impacts, there are likely to be more substantial regional and sectoral impacts, including for the agricultural sector, from the choice of metrics (Reisinger et al., 2013; Strefler et al., 2014; Harmsen et al., 2016). The non-optimality of GWP_{100} nevertheless increases in the case of overshoot scenarios (Tanaka et al. 2021). Overshoot scenarios are scenarios under which the temperature target of the Paris Agreement is temporarily exceeded before eventually being achieved.

Similar to the GDamP, the GCP has not been used in climate policies in real-world applications. While the GCP is valuable for quantifying the cost-effectiveness of different metrics, there are conceptual difficulties in operationalizing the GCP because the value of GCP itself requires an assumption on a long-term future emission pathway towards a temperature goal. As a compromise, it has been suggested to use the GCP to guide the choice of emission metrics at certain points in the future as the emission mitigation pathway evolves (Tanaka et al., 2021).

While these studies have shown that the use of GWP_{100} does not guide perfect emission pathways towards selected mitigation goals, the introduced non-optimalities are nonetheless surprisingly small. In other words, if one wants to mitigate methane emissions to cost-effectively achieve some future temperature target, or simply quantify the marginal damages caused by methane emissions, one arrives at CO_2 equivalent metrics for methane somewhere between about 20 and 40. This is roughly consistent with GWP_{100}, but contrasts with values generated with other metrics such as GTP_{100} or GWP_{20}. When more complex net emission patterns are involved, however, then use of different metrics applied to the same net emission patterns can result in very different assessed mitigation outcomes (Brandao et al., 2019). Therefore, even though GWP_{100} was not developed to derive cost-benefit or cost-effective outcomes, it may be adequate for these purposes and is not necessarily incompatible with cost-benefit and cost-effectiveness metrics.

6.3.4 Overall emission reduction policy and the role of agriculture

Any overall emission reduction or temperature targets can be achieved most cost-effectively if all sectors contribute towards the emission reduction effort, including the agricultural sector. Agriculture has an unusual emissions profile as, unlike most other sectors, emissions are dominated by CH_4 and N_2O instead of CO_2. Moreover, emission reduction policies usually involve trade-offs. To satisfy ongoing demand for food, reductions in agricultural production from one sector or region can increase the demand for alternative types of food, or supply from other regions which may ultimately lead to higher or lower emissions than the original food production. Any consequent changes in emissions need to be factored in when assessing the overall effect of any mitigation policy (e.g. by conducting consequential analyses such as Smith et al., 2019).

It is important to also enable cross-sectoral comparisons. This can be done by comparing the contribution of different sectors or countries to past and anticipated future temperature changes. Comparisons also need to be carried out over different timescales, and this is where the different atmospheric life-times of CH_4 and CO_2 create particular challenges. In this context, the role of metrics becomes critically important to be able to assess the relative contributions of agricultural CH_4 and CO_2 in a meaningful way.

The use of metrics becomes necessary

a) if one wants to compare the contributions of different emitters/sectors, etc. that may be emitting different gases, or

b) where there are potential trade-offs of policies that may differentially affect the emissions of different gases so the value of CH_4 reductions must be weighted against possible increases in the emission of other gases.

For a) it comes back to the issues addressed earlier in this chapter, and may factor in to a cost/benefit analysis. For b) we can use an LCA to assess how any CH_4 reduction efforts will affect other GHG emissions, or have other environmental impacts, such as leading to the emission of specific pollutants. They may also have land use impacts as discussed in the 'cross-cutting' section in an earlier chapter.

6.3.4.1 Assessment Boundaries

Defining the boundaries for GHG emissions is as important as choosing the right GHG metric for an assessment. Most agricultural production systems are complex so that many actions to reduce agricultural methane emissions will have impacts on other GHGs, such as increasing fodder diets will lead to higher CO_2 emissions but reduce CH_4 emissions (as discussed in the Section on Feed Additives). Knock-on effects of mitigation actions should, therefore, be considered when developing emissions reduction policies, including, but not limited to, emissions of different GHGs, trade, food security, land use, water consumption as well as water and air pollution.

Direct or indirect leakage of methane emissions across national boundaries is an issue in evaluating any national mitigation strategies. Clearly, a reduction in ruminants in one country by importing ruminant products from an increased ruminant population in another country (with potentially higher emissions) would be direct leakage that could be reported as mitigation in the importing country but as increased emission in the exporting country, with no beneficial effect for the world as a whole. There can also be indirect leakage if one country reduces methane emissions by reducing exports of products from ruminants and if that action leads to increased ruminant numbers in the countries that previously imported ruminant products. This is an issue for all food-production systems since any change in food production in one region will have effects on the production level in different regions and on alternative food products and their likely emission rates of all GHGs. These leakages can be checked crudely by assessing whether the balance between domestic production and consumption is being achieved through either increased imports or decreased exports of livestock products. A more cooperative approach, as allowed under the Paris Agreement, would be to jointly report on mitigation activities of multiple countries that produce livestock products but also have large trade amongst themselves. This would be one approach to ensure that assessments include translocation of livestock emission sources across national boundaries.

Within the system boundaries, the emission of each GHG needs to be reported. Where life cycle inventory data are used as the reference for emissions for components within the system or inputs to or outputs from the system, it is essential to describe emissions of each GHG rather than use an available aggregate GHG equivalent emission calculated with a single metric.

Because there has been no scientific consensus on how to factor out direct human-induced from indirect human-induced and natural effects (e.g., Canadell et al., 2007), all natural emissions that occur on managed land are considered anthropogenic under the UNFCCC (IPCC, 2003; IPCC, 2006). Nevertheless, many countries do not report emissions that they consider natural but that occur on managed land.

However, natural methane emissions from wetlands, inland waters, and wildlife (including insects) that occur on managed land could be included in scenarios of methane fluxes. Many natural emissions are expected to increase with global warming (Dean et al., 2018) making it important to fully develop pathways to reach climate change targets. Some natural emissions could also be affected by livestock management such as change in natural pasture management that could affect wild ruminant populations as well as termite abundance (Manzano and White, 2019).

For example, a comprehensive regional mitigation evaluation in Sweden that included natural emissions, identified the reduction of methane emissions from water bodies rather than reduction of livestock methane emissions as the preferred mitigation action (Skytt et al., 2020). To gain greater clarity and allow better interpretation of the results, it could be useful to separately report indirect from direct emissions where possible to allow for better interpretation of the results.

6.3.4.2 Designing a holistic emission reduction strategy

The successful reduction of overall climate change impacts requires a holistic emission reduction strategy specifically designed to achieve the desired emission reduction goals. The strategy should assess any trade-offs and co-benefits of mitigation choices, with careful consideration of appropriate time horizons so that results can be achieved most cost-effectively and with desirable co-benefits while minimizing any unintended adverse consequences.

Some policy decisions could result in temporarily delaying CO_2 reduction efforts in favour of action on other gases or vice versa if this can achieve specific goals more cost effectively, especially at sub-global levels. What is deemed to be the best approach requires public policy debate in the context of the current overarching policy context, including socio-economic developments, and how different alternative mitigation strategies might contribute to the overall goal of 'sustainability'. The "Do no significant harm" (DNSH) principle of the European Commission may also help in this context. It asserts that a measure should not lead to significant harm to any one of six major environmental objectives"

In this context it is important to assess a variety of potential scenarios and apply different metrics, to achieve an enhanced understanding of the outcome of different policy designs. Any GHG emission metric simplifies the complexity of the climate system response to greenhouse gas emissions. It is, therefore, ambiguous if future mitigation targets are only described in terms of CO_2-equivalents. It would be clearer if emission targets were also specified for individual gases. That could be useful even if only indicative distinctions could be made, or if at least emissions of long-lived and short-lived GHGs were separated in future targets (Denison et al., 2019; Allen et al., 2021).

GWP* or modelling of warming effects over time (i.e., a climate model) can reveal temporal details and trade-offs that are not necessarily apparent with GWP_{100} or other single-pulse metrics. This is especially important if climate targets are not just addressing impacts at a certain point in time, but also assess temporal developments and any trade-offs of warming impacts of different GHG policies before and after this point in time. For integrated policies that account for the whole economy, it is important that goals are not independently defined for different sectors, but also address their effectiveness and any trade-offs between sectors that are required to cost-effectively limit overall GHG emissions.

Additionally, reducing GHG emissions might involve agricultural intensification that could have negative impacts on animal welfare and biodiversity. In some sectors, GHG reductions might lead to fewer trade-offs than in others, or even synergies. In order to assess changes for whole economies and find the most effective GHG mitigation solutions, integrated assessment models should be used. They contain large uncertainties, however, and it would, therefore, be best to apply several models and scenarios, which is currently very resource demanding. Improved scientific tools might be needed for broader applications. The following sections provide some insights based on available studies.

6.3.5 Cross-sector comparisons

Currently, most sectoral comparisons are based on emissions in a given year aggregated using a pulse-emission metric. So, for example, with GWP_{100}, this is simply defined as the marginal radiative forcing integrated over the following 100-years. Any aggregation of the sectoral contribution to overall greenhouse gas emissions are, therefore, highly dependent on the specific metrics used for the integration. For example, the IPCC AR5 synthesis report (IPCC 2014) compared the sectoral contribution to overall emissions in 2010 using three of the most commonly used metrics, GWP_{100}, GWP_{20} and GTP_{100} (Figure 10). The calculated contribution of agriculture to total greenhouse gas emissions differs between 7.2 percent for calculations based on GTP_{100} to 22 percent for calculations based on GWP_{20}. These differences are largely attributable to the differing weight assigned to CH_4 emissions.

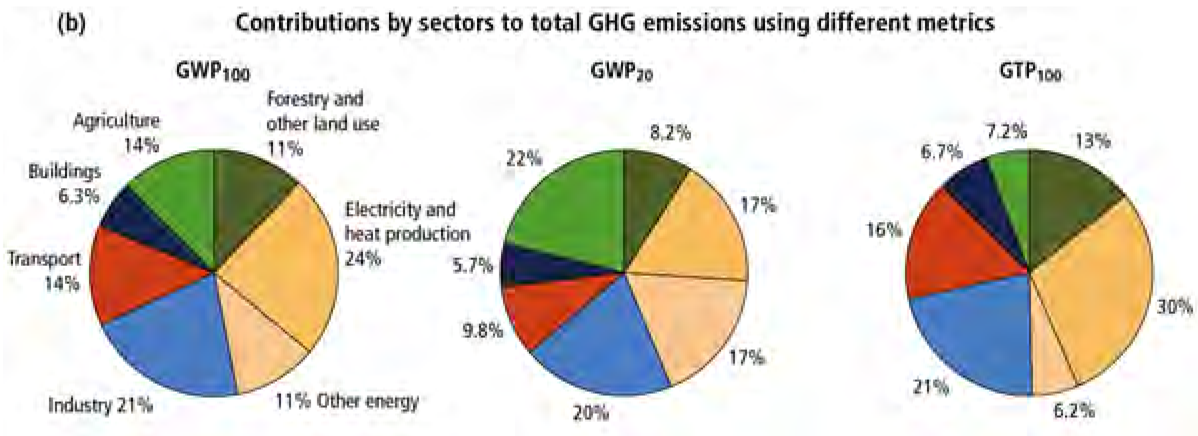

Figure 10. Sectoral contribution to annual total greenhouse gas emissions in 2010 weighted by 3 different greenhouse gas metrics, GWP_{100}, GWP_{20} and GTP_{100}. Reproduced from IPCC (2014).

The climate impacts of different sectors can also be compared by exploring their contribution to global temperature increases from past emissions. This provides an alternative perspective and overcomes the problem of relying on different greenhouse gas metrics to make comparisons between the emissions of different greenhouse gases. Reisinger and Clark (2018) demonstrated this approach for the warming contribution from livestock farming using a simple climate model to calculate the actual contribution of direct livestock-based emissions to global temperature increases up to 2015 (Figure 11).

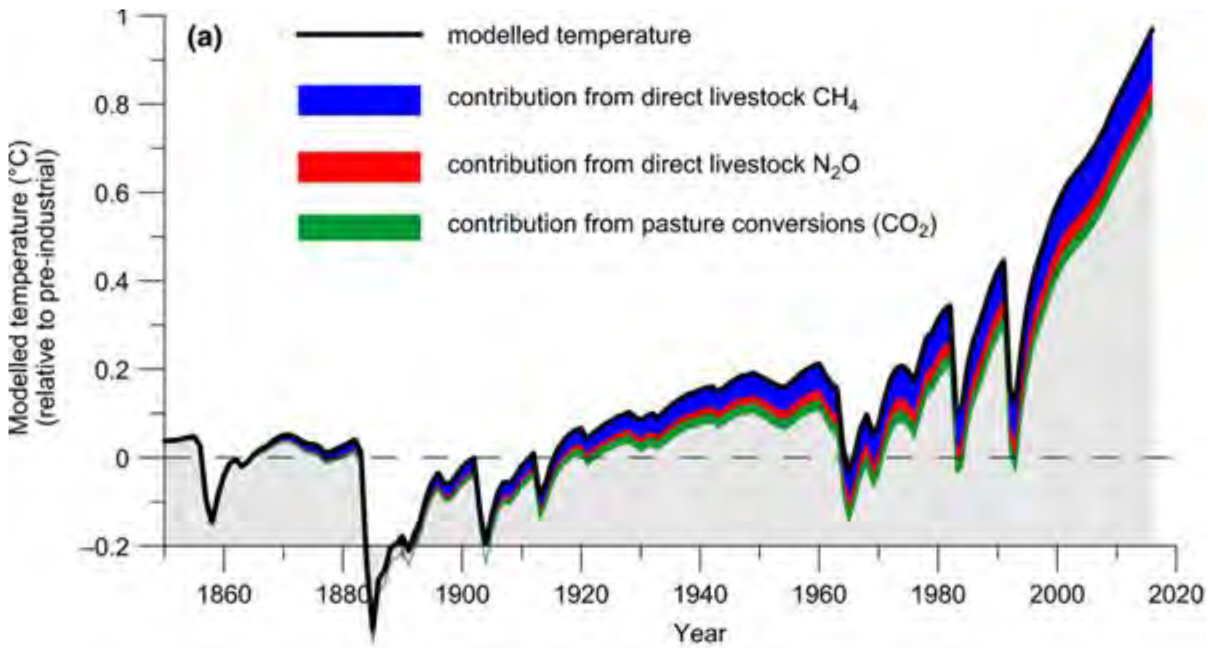

Figure 11. Modelled global temperature anomalies from 1850 to 2015 for all anthropogenic emissions.
The contribution from direct livestock emissions of CH_4 (blue), N_2O (red) and CO_2 from pasture conversions (green) and other anthropogenic emissions (grey). Reproduced from Reisinger and Clark (2018).

A comparison of different sectors based on their contributions to past, present or future global temperature changes provides information that is fundamentally different from the assessed marginal climate-change impact of an individual year's emissions (as defined by whatever metric is used to aggregate or compare different gases) or the marginal impact of all future emissions (Reisinger et al., 2021). All such approaches may be of interest and potentially relevant to policy development, but the calculation method and any emission metric used must be relevant to the question being posed.

6.3.6 Aggregation of different GHGs for reporting and accounting

As reflected in the examples above, how and whether to aggregate the contributions from different GHGs is context-specific and depends on the information a user wants to gain. For some purposes, official guidance specifies the protocol to be used for emission aggregations. For national emission inventory submissions, for example, the emissions of individual greenhouse gases have to be reported without conversion. In addition, the UNFCCC has mandated that nations should use the GWP_{100} to also report aggregated values of these emissions. For 'product carbon footprints' and similar assessments we refer back to the discussion in the Life Cycle Assessment section. The sensitivity to multiple emission metrics, such as GWP_{20}, GWP_{100}, GWP* and GTP_{100}, could be explored to highlight how different impacts vary over a range of timeframes.

It should be emphasized that, irrespective of the method used for data aggregation, it is useful to also report disaggregated data of individual GHG emissions. This ensures full transparency and enables wider analyses beyond the individual metrics and/or aggregation method provided. The time of emission can also be critically important, especially for CH_4 emissions because of its short atmospheric lifetime. CH_4

emissions in later parts of this century can, therefore, have greater climate-change impacts than emissions during the previous century because CH_4's warming contribution in the late 21st century would be felt at a time with higher global background temperatures. The same warming contribution could then have greater climate-change impacts. This wider analysis could include a reanalysis under different metrics or aggregation methods or employ more robust climate modelling approaches.

6.3.7 Biogenic methane - implications for metrics

Any CH_4 released into the atmosphere must have been produced either from carbon laid down in the past from either fossil sources or stored in peat, permafrost or similar deposits or from recently fixed carbon in rice paddies or enteric fermentation (Wiloso et al., 2016).

In the context of climate change, it is important to distinguish between methane indirectly generated from carbon in recently grown biomass and that from old carbon sources, such as fossil deposits. If methane is generated from newly grown biomass, such as by enteric fermentation, carbon is converted to CH_4 whereas the carbon would otherwise be respired as CO_2. This means that the generation of biogenic CH_4 slightly lowers the atmospheric CO_2 concentration, which does not happen when CH_4 is released from a fossil origin. When CH_4 is eventually oxidised, the carbon will transform back into CO_2, a process that is common to both biogenic and fossil CH_4. In the case of methane from fossil origin this oxidation leads to a net increase in atmospheric CO_2 concentrations by adding additional C to the atmosphere that was sequestered in fossil deposits millennia ago (which is the same as direct CO_2 emissions from fossil resources). Biogenic CH_4, instead, does not lead to an eventual net increase in atmospheric CO_2.

For biogenic CH_4 from biomass C that had only recently been produced from CO_2, Varshney and Attri (1999) proposed that GWP_{100} should be reduced by 5 percent from the IPCC values for fossil CH_4. Kirschbaum (2014) and Munoz and Schmidt (2016) proposed that warming potentials of CH_4 should be reduced by 2.75 kg CO_2eq for biogenic CH_4 relative to the value for fossil CH_4 to account for the associated reduction of CO_2 by the formation of CH_4. This conforms to the assumption that one molecule of CO_2 is removed for each molecule of CH_4 generated, with a 1:1 molar ratio converting to a weight ratio of CO_2:CH_4 of 2.75. The default values for metrics in IPCC AR5 were for biogenic methane.

However, this does not account for the difference in temporal developments. IPCC AR6 assumes that only 75 percent of methane oxidation leads to CO_2 while 25 percent of carbon is removed by deposition of reactive intermediates. Accounting for the time taken for methane oxidation slightly decreases the required change of the GWP_{100} of biogenic CH_4 to 1.9 units (Forster et al., 2021). Similarly, Boucher et al., (2009) proposed that the warming potentials of CH_4 from fossil C should be increased by 0.7 to 2.7 units to account for the conversion of oxidized CH_4 to CO_2, while the warming potential of biogenic CH_4 generated from recent biomass should be reduced by 1.4 to 0 units.

The latest IPCC report lists values for GWP_{100} as 29.8 for fossil CH_4 and 27.0 for biogenic CH_4 (IPCC, 2021). These are the most recently calculated global warming potentials under the latest state of the science and atmospheric gas concentrations.

Recommendations (Section 6.3)

It is recommended to report greenhouse gas emissions for individual gases where possible in addition to any emissions aggregation through the use of chosen metrics.

Climate metrics can only provide information about the direct climate consequences of emissions and mitigation actions. Ultimate policy choices need to consider not only the direct climate consequences of any mitigative actions but also other relevant climate and non-climate factors.

Application of a range of metrics can help test the sensitivity of climate-change impact assessments to the choice of metrics. This can be particularly useful if there is no single clearly defined policy objective.

In applying any metric to methane emissions, it is warranted to distinguish between CH_4 derived from fossil versus recent biogenic origin, with warming potentials of CH_4 of fossil origin assigned a higher warming potential by 2.75 units.

6.4 Climate targets and related issues

The livestock sector's use of climate metrics occurs within a larger policy context relating to climate action and sustainable development. This section provides additional discussion in relation to the Paris Agreement, the term *climate neutrality*, sustainable agriculture, and equity. The discussion is provided for informational purposes, so that users can make decisions informed by the wider context including global goals to which every sector contributes. The section does not contain specific recommendations.

6.4.1 The Paris Agreement

6.4.1.1 The goals of the Paris Agreement

The Paris Agreement provides the basis for international climate policy under the United Nations Framework Convention on Climate Change (UNFCCC). It sets out a framework to strengthen the global response to the threat of climate change by, "Holding the increase in the global average temperature to well below 2°C above pre-industrial levels and pursuing efforts to limit the temperature increase to 1.5°C above pre-industrial levels." (Article 2.1)

In order to achieve this long-term temperature goal, the Paris Agreement further describes the "…aim to reach global peaking of greenhouse gas emissions as soon as possible, …to undertake rapid reductions thereafter, …so as to achieve a balance between anthropogenic emissions by sources and removals by sinks of greenhouse gases in the second half of this century." Furthermore, the Agreement describes the need to pursue these goals, "…on the basis of equity, and in the context of sustainable development and efforts to eradicate poverty." (Article 4.1).

6.4.1.2 The Paris Agreement and methane emissions

It is important to note that the Paris Agreement does not specifically discuss methane. The Paris Agreement does not prescribe how much and how quickly the emissions of individual gases must be reduced. Strategies to achieve the Paris Agreement are to be worked out within national contexts. That said, the IPCC sixth assessment report (IPCC, 2021) underscores the need for sustained reductions in greenhouse gas emissions and reaching net zero CO_2 emissions. This report also highlights the importance of limiting the emission of other greenhouse gases and air pollutants, especially methane, as this will have benefits for both human health and the climate. Opportunities to reduce emissions vary across different sources and sectors. It is important to be mindful of trade-offs between reductions of different gases while working toward the goals of the Paris Agreement.

6.4.1.3 Use of climate metrics

The Paris Agreement does not specify metrics. Nor does it use terms such as *net zero* or *carbon neutral*. That said, the common metric used under the UNFCCC since the Kyoto protocol has been GWP_{100} (UNFCCC, 1997). With subsequent IPCC reports, the assessed values of GWP_{100} have undergone changes (Table 8). At the 24th session of the Conference of the Parties (COP24) of the UNFCCC, GWP_{100} was adopted as the common metric for the implementation of the transparency framework of the Paris Agreement (paragraph 37 of the Annex to Decision 18/CMA.1). Later, countries decided to use GWP_{100} values without climate-carbon feedbacks to report aggregate emissions and removals as provided by the IPCC Fifth Assessment Report (AR5) (or a subsequent IPCC report upon future agreement). In addition to the mandatory reporting based on GWP_{100}, the COP24 decision further allows countries to report supplemental information on aggregated CO_2-equivalent emissions also by using other metrics assessed in IPCC reports in addition to GWP_{100}, noting GTP as an example. In addition, Parties to the Paris Agreement agreed to use the same emissions reporting framework for accounting for their Nationally Determined Contributions beyond 2030 (4/CMA.1).

Table 8. GWP values for methane across the different historical IPCC reports.

	SAR (IPCC, 1995)	**TAR (IPCC, 2001)**	**AR4 (2007)**	**AR5 (2014)**	**AR6 (2021)**
	100-year time period				
CH_4 non-fossil origin	21	23	25	28	27.2
CH_4 fossil origin				30	29.8
	20-year time period				
CH_4 non-fossil origin	56	62	72	84	80.8
CH_4 fossil origin				85	82.5

6.4.1.4 Discussion of long- and short-lived greenhouse gases in recent IPCC reports

After the adoption of the Paris Agreement, the UNFCCC invited the IPCC to produce a report specifically on a 1.5°C warmer world. The Special Report on Global Warming of 1.5°C ("SR1.5", IPCC, 2018) observes that "Reaching and sustaining net zero global anthropogenic CO_2 emissions and declining net non-CO_2 radiative forcing would halt anthropogenic global warming on multi-decadal timescales (high confidence)." As such, the report makes a distinction between net zero CO_2 emissions and net zero GHG emissions, which is further reinforced in the IPCC sixth assessment report (IPCC, 2021): "Achieving global net zero CO_2 emissions is a requirement for stabilizing CO_2-induced global surface temperature increase, with anthropogenic CO_2 emissions balanced by anthropogenic removals of CO_2. This is different from achieving net zero GHG emissions, where metric-weighted anthropogenic GHG emissions equal metric-weighted anthropogenic GHG removals… emissions pathways that reach and sustain net zero GHG emissions defined by the 100-year global warming potential are projected to result in a decline in surface temperature." Hence, a distinction is recognised in how different gases contribute to global temperature increases, based on the distinct dynamics between long- and short-lived gases.

For methane, a relatively short-lived greenhouse gas, declining radiative forcing can be achieved with a steady gradual decrease compared to current emission rates. Methane's atmospheric lifetime is sufficiently short that atmospheric concentrations are largely driven by emissions occurring only in recent decades, and so bringing emission rates down below levels experienced a few decades ago will lead to reduced anthropogenic methane concentrations, implied forcing, and contribution to temperature change. For methane, there is therefore not the time-independent relationship between warming and total cumulative emissions as observed for CO_2.

To limit further temperature increases, the requirement for 'net-zero' emissions, where emissions must either be completely removed or offset with additional CO_2 removals, is only strictly necessary for CO_2, due to its cumulative impacts extending into the very long-term. For short-lived gases, an equivalent climate impact to 'net-zero CO_2' can be achieved with some ongoing emissions. It has been demonstrated that net-zero GHG emissions are not necessarily required for temperatures to remain below 1.5°C or 2°C, and it would be possible to achieve this temperature goal without entirely eliminating or offsetting methane emissions (Tanaka and O'Neill, 2018).

The ultimate dynamics of how long- and short-lived gases contribute to overall temperature change are well-understood, and the different gas-specific requirements for reaching any given climate target are widely recognised (Allen et al., 2021). However, the fundamental physical requirements outlined above are not the only consideration for multi-gas climate policy. Cost-effectiveness, equity and technical feasibility are other important considerations.

6.4.2 Climate neutrality

[This section does not have full consensus among the TAG members]

6.4.2.1 Different uses of the term

Climate neutral is a term that is being used with increasing frequency. However, it is used in a variety of ways and with a variety of meanings. Therefore, whenever the term is used it is important that it is clearly defined to avoid misunderstanding.

In some cases, the term has been used more-or-less synonymously with the term *carbon neutral*. For instance, the UN has a Climate Neutral Now Initiative that links the term to a carbon footprint reduction commitment (https://unfccc.int/climate-action/climate-neutral-now). Here it is also important to note ambiguity with use of the term *carbon neutral*. In many cases the term is applied when all GHG emissions are reduced to zero or have been offset by GHG removals in another system. For the aggregation of different GHG emissions and removals, the GWP_{100} climate metric is typically used. However, in its sixth assessment report, the IPCC now defines *carbon neutrality* in relation to CO_2 alone as a "Condition in which anthropogenic CO_2 emissions associated with a subject are balanced by anthropogenic CO_2 removals." The IPCC now uses the term *GHG neutrality* when other non-CO_2 greenhouse gases are included. This contrasts with the draft ISO standard ISO/CD14068, which uses carbon neutral to include all GHGs. In any case, there appears to be little to be gained by using the term *climate neutral* as a synonym for an existing concept.

Climate neutrality is also sometimes thought of as going beyond balancing emissions and removals of GHGs to also include other radiative forcing mechanisms such as aerosols, or changes in albedo that affect the local climate. A definition is provided by IPCC (2018), describing *climate neutrality* as a "Concept of a state in which human activities result in no net effect on the climate system. Achieving such a state would require balancing of residual emissions with emission (CO_2) removal as well as accounting for regional or local biogeophysical effects of human activities that, for example, affect surface albedo or local climate." The practical implementation of this concept is complex as it includes well-mixed greenhouse gases that contribute to global climate change, as well as climate forcers that have only a local climate effect.

Recently, the term *climate neutral* has also been used to describe a system that is making either no net contribution to changes in radiative forcing (Ridoutt, 2021) or no net contribution to additional temperature increases (Costa et al., 2021; Place and Mitloehner, 2021). These definitions stem from analogy with the CO_2-specific outcomes of achieving 'carbon neutrality' as defined above, and an understanding that to stabilize the climate, emissions need to be managed in such a way that radiative forcing and temperatures are not being driven higher and higher. This approach implies a very different role of short-lived gases like methane since an on-going emission of methane at a slowly reducing rate results in a roughly constant climate. However, this approach would only be fulfilling the temperature goals of the Paris Agreement if the climate were stabilized at 1.5°C. This concept of *climate neutrality* can be applied in relation to radiative forcing footprints (Ridoutt and Huang 2019) or any of the various step/pulse metrics (Allen et al., 2016, 2018; Collins et al., 2020; Smith et al., 2021). It is important to note that stabilisation of temperature contribution from each individual gas does not tell us whether this is cost-effective or equitable, or whether the implied emission reductions across all gases are technically feasible.

6.4.2.2 Climate metrics and climate neutrality

If a *carbon neutral* commitment is made, following the IPCC 6th Assessment Report definition, climate metrics are not needed as only CO_2 emissions are considered. The main issues that arise relate to the level of offsetting relative to emissions reduction in the system itself.

However, if a *GHG neutral* commitment is made, metric-weighted anthropogenic GHG emissions associated with a subject are balanced by metric-weighted anthropogenic GHG removals. Neutrality often includes scope 3 emissions. Net-zero GHG emissions are also metric-weighted net anthropogenic GHG emissions, but often do not include scope 3 emissions. Different organizations have different exact definitions. The quantification of GHG emissions and removals depends on the GHG emission metric chosen to compare emissions and removals of different gases, as well as the time horizon chosen for that metric. Consequently, the choice of emission metric to reach and sustain net-zero GHG levels will affect their resulting temperature outcome (IPCC, 2021). In practice and by convention, the GWP_{100} climate metric is used in most programmes. Reaching and sustaining net zero GHG emissions typically leads to a peak and decline in temperatures when quantified with GWP_{100} (IPCC, 2021). However, it is important to be mindful that when organizations make commitments to reduce and/or offset GHG emissions using the GWP_{100} climate metric, it is not immediately clear what will be the impact on future radiative forcing and temperatures as this will vary over time depending on the particular basket of GHG emissions involved (Fuglestvedt et al, 2018; Tanaka et al., 2018, Allen et al., 2021). Net zero GHG emissions defined by CGTP or GWP* imply net zero CO_2 and other long-lived GHG emissions and gradually declining emissions of short-lived gases. The warming evolution resulting from net zero GHG emissions defined with a step-pulse metric corresponds (in terms of radiative forcing and temperature) approximately to reaching net zero CO_2 emissions, and would thus not lead to declining temperatures after net zero GHG emissions are achieved but to an approximate temperature stabilization (IPCC, 2021). The temperature level at stabilization will depend on cumulative CO_2 emissions over the entire historical period and the ongoing emission rate of short lived gases.

The key issue is the very large differences in future climate impact of short and long-lived climate forcers. For long-lived climate forcers, such as CO_2 and N_2O, net zero emissions lead to climate stabilization. The climate impact of a CO_2 emission potentially lasts for millennia. Therefore, ongoing net emissions of CO_2 would be inconsistent with climate stabilization within any human timeframe. Even N_2O has a lifetime and climate impact that exceeds the timeframe by which climate stabilization needs to occur if Paris Agreement temperature targets are to be met, or not far exceeded. Additionally, emission of these GHGs will not only hamper achievement of climate targets by 2100, but also lead to increased temperature over longer timeframes. However, short-lived climate forcers, like methane, with an atmospheric lifetime in the order of a decade or less, do not necessarily need to be reduced to net zero to achieve Paris Agreement goals. With methane, a modestly reducing emissions profile over time, whereby new emissions are balanced by the decay of methane from recent historical emissions, leads to methane-caused climate stabilization. More substantial methane emissions reductions provide a mechanism to lower the temperature, and thus may be an important contributor towards achieving the Paris temperature goal.

Using step-pulse metrics to define GHG neutrality (or net-zero GHG emissions), both short and long-term climate forcers can be aggregated on the basis of future change in warming, resulting in climate stabilization. A key consideration with this approach is that this climate stabilization is required *globally.* It does not follow that GHG neutrality defined in this way is automatically consistent with the Paris Agreement goals at sub-global levels. In particular, the Paris Agreement states the need for equity and sustainable development, and that developing countries will peak and decline their emissions at a later date than developed countries. In addition, we know with high confidence that merely stabilizing warming due to CH_4 would make achieving temperature goals more expensive (because the remaining carbon budget would become much smaller still). An approach using GWP* has been used in a variety of case studies in the livestock sector (Ridoutt, 2021b; Del Prado et al, 2021). A key challenge in applying the method relates to establishing what is an indefinite change in the rate of short-lived climate forcer emissions. This requires defining a baseline, and in many pasture and rangeland-based livestock production systems, emissions can fluctuate quite strongly from year to year. It can therefore be difficult to ascertain a permanent change in emission rates.

Another approach is the radiative forcing climate footprint where the contribution to radiative forcing of current year emissions is summed with the radiative forcing from historical emissions that remain in the atmosphere (Ridoutt and Huang, 2019; ISO 2021). By tracking progress over time, an organisation or sector can assess whether their total contribution to radiative forcing is increasing and take management action to stabilise or reduce it. A situation where an organisation or industry is making no additional contribution to radiative forcing could be regarded as consistent with climate stabilisation and described as *climate neutral*. This does not resolve the question of what an acceptable level of radiative forcing from this organisation or industry is. This approach can be applied to the main GHGs associated with livestock production, i.e., CO_2, CH_4 and N_2O, as demonstrated for sheep production for meat in Australia (Ridoutt, 2021a). It can also be extended to include non-well-mixed GHGs and other drivers of radiative forcing (such as change in albedo) in a regional context. This could be relevant where the burning of biomass occurs, where land transformation and management practices lead to changes in surface albedo, and in sensitive or high-risk environments.

6.4.3 Methane abatement and sustainable agriculture

Climate metrics can help to define and report climate goals and actions from a multi-gas perspective. This helps to assess the impacts of emissions and removals of different greenhouse gases and support understanding of the trade-offs between near-term and longer-term climate effects. However, in pursuing climate action, it is important to also consider wider sustainability goals, such as those described by the UN Sustainable Development Goals, as well as sustainability priorities that are relevant in each local context. For example, different communities and countries have different levels of food security. The socio-economic aspects are especially important for the small-scale livestock sector in developing and emerging economies since it can provide additional income, and support socio-economic developments.

Sustainability is a broad concept with social, environmental, economic, and cultural dimensions. Sustainable agriculture has been variously described. One recent definition of sustainable livestock production states: "Livestock sustainability refers to production approaches that simultaneously meet long-term conditions to ensure society's food and nutrition security, livelihoods and economic growth, animal health and animal welfare and stable climate and efficient resource use (the four livestock sustainability domains) in order to contribute to sustainable food systems" (GASL Secretariat http://www.livestockdialogue.org/). For each dimension of sustainability, a variety of indicators exist. Only by assessing impacts broadly, can trade-offs be evaluated and managed.

6.4.4 Equity considerations

[This section does not have full consensus among the TAG members]

Concern for equity is reflected in the Paris Agreement, and this is a consideration when using climate metrics to define and report climate goals and actions. That said, equity considerations go beyond science and ultimately rest upon value judgements and ethics (Stavins et al., 2014; Robiou du Pont et al., 2016; Klinsky and Winkler, 2018). Equity is not an attribute of climate metrics themselves, but equity considerations can help to determine what metrics are used, how metrics are applied and for what purposes. Certain applications of emission metrics may raise equity concerns if relevant issues are not considered up-front, and there may also be cases where metrics can help illustrate climate equity considerations. A full analysis of relevant climate equity topics is beyond the scope of this LEAP report, and there is relatively little literature exploring the intersection between equity and GHG emission metrics specifically (Rogelj and Schleussner, 2019; Harrison et al., 2021). Modelling studies have shown that the use of different methods to attribute historical responsibility to different nations give different results, due to nonlinearities in the climate system (Trudinger and Enting, 2005; Höhne and Blok, 2005).

Pulse emission metrics (such as GWP_{100}) treat every unit of a given GHG equally and independently from the point in time at which the emission occurred. They cannot directly reflect the overall contribution to global warming made by different emitters from a series of emissions over an extended period of time. Lynch et al., (2020) suggest that use of GWP* could allow emitters to be held accountable for their full historical contributions to global warming, in a way that is not possible using GWP_{100}, but this requires accounting the whole emission trajectory from a sufficiently early baseline (for example, pre-industrial, see Figure 6.4). As step/pulse metrics present an accurate weighting in terms of temperature outcome, they may thus provide an alternative method to include short-lived gases in cumulative emission budgets. Therefore, they could be used to explore national or sectoral 'fair shares' of total warming contributions. Such approaches need to be mindful of the equity considerations if they are not applied to the full historical emissions. Rogelj & Schleussner (2019), in this sense, argued that given the inequality in historic emissions, using GWP* with a present-day baseline could result in highly unequal and unfair outcomes benefiting high historically high emitting countries, sectors, even down to the individual company or farm level. This 'grandfathering' can be avoided by taking a pre-industrial baseline, as noted above, but this raises challenges to equitable allocation of responsibility for this warming within countries. On the other hand, Lynch et al (2020) also argue that GWP100 net-zero targets implicitly set a baseline target for CO_2-induced warming at whatever level was reached prior to reaching net-zero, irrespective of how much warming an emitter may continue to cause through their past emissions and hence ongoing repsonsibility for climate damages. Similar concerns were the basis of the 'Brazilian Proposal' to the Kyoto Protocol to set emission reduction targets based on historical contributions to global warming.

Metric selection and appropriate deployment may be chosen to reflect certain equity considerations, but requires a user to recognise and choose a certain perspective on the fundamental concerns raised, such as whether and how to set effort-sharing expectations based on mitigation potential of contemporary emission reductions or an actor's contribution to overall global warming, or how to allocate responsibility for warming from historical emissions from different gases across today's emitters that may not share the same emissions profile.

6.5 Metric selection guide

This section aims to provide an example of how a practitioner could approach the decision-making process to identify a suitable metric for any particular question or usage, based on the information and learning contained elsewhere within this report. Different metrics incorporate different effects on climate that follow from emissions, and may report these effects covering different time-frames or with respect to different reference conditions. Different metrics may therefore be useful for different purposes. This report does not recommend the sole use of one particular metric for all purposes, as the choice of metric will depend on the specific question being asked, and may also require value judgements based on the priorities of the practitioner or organisation. We recommend practitioners follow the guidance below and consider how each point is relevant to their particular need. Two examples are provided that illustrate the potential appropriate metric depending on the question being framed.

6.5.1 Points to Consider

Example boxes: Section 6.5.2.1 describes an example case study (example 1) which assesses the impact of a dairy farm which could start using a feed additive to reduce methane emissions. Throughout section 6.5.1, we have included the steps we have taken in considering example 1, relevant for each 'point to consider', in boxes such as this one. Full details of the example are in section 6.5.2 and appendix 1 (which contains full details of the modelling work).

6.5.1.1 Define your question

This is the first and most important step. If a particular metric is to be used as the tool for an evaluation, then the objective of the evaluation must be clearly defined. If the end goal is unclear, then an appropriate (or inappropriate) metric cannot be identified. Sometimes the ultimate goal may not be immediately evident. For example, a practitioner may be asked to define an emissions reduction target. However, what is the overarching goal?

These goals might be:

- to minimise emissions of every GHG,
- to minimise or achieve some externally-determined target for aggregated GHG emissions based on a pre-defined metric,
- to limit (at a chosen level) or undo the organisation's overall contribution to global warming,
- to identify a target that incorporates budgetary considerations,
- all of the above
- other.

There could be a hierarchy of goals, such as identifying strategies to reach a climate mitigation target first and then ranking the best strategies based on fairness, equity or effectiveness criteria. If these motivations are made apparent, it can become clearer which approach is the most appropriate to address the particular question. With a multitude of concurrent goals, articulating them can help identify metrics suitable or unsuitable for each goal. This will reveal whether there is one suitable metric, or whether different metrics are needed to address the different goals.

See section 6.2.1.

Example 1: A dairy farmer wants to evaluate the benefits of using a particular feed additive on their herd. The question is defined as: If I start to use the feed additive, what will the impact on climate change be compared to not applying the additive? This is motivated by the desire to reduce environmental impacts of the farm in the coming decades. Present day emissions are known, therefore the question is to assume present day emissions continue and compare this with the emissions generated if the feed additive were introduced.

6.5.1.2 Existing requirements about metrics

This may already be incorporated in the answer to the first question. However, if it is not, are there any regulations which require the use of a particular metric? Although a particular metric may in some cases be mandated, it is worth considering whether that is completely adequate for your needs. If it is not, another metric or modelling exercise is likely required to inform your plans or policies. For example, a sensitivity analysis using several metrics could inform target-setting so that impacts at different timescales, and on temperature outcomes, could be considered. This sensitivity analysis could be especially useful if the overall outcome trying to be achieved is to increase the understanding on the overall environmental impact of different strategies.

See Sections 6.2.2 and 6.2.3.

Example 1: The farmer already uses GWP100 in an existing GHG footprint calculator, but uses other greenhouse gas metrics to inform an internal strategic analysis.

6.5.1.3 Time frame

Alternative emission metrics can differ greatly in how they report greenhouse gases as 'equivalent' to one another (see Section 6.2 for further details). These differences arise primarily because different greenhouse gases show distinct time-dependence in their impacts. Emission metrics typically set a pre-defined time-horizon to constrain comparisons and provide a single measure of 'equivalence', and different time-horizons will result in different valuations. In order to make a judgement on a suitable metric, the time frame under consideration for your question or goal must therefore be explicitly considered. Is your priority for minimizing your operation's contribution to global warming in 2050, 2100 or another specific time, or at all of these times and any intervening years, or over an indefinite period to cover the full impacts anticipated from any emissions?

When there are short-lived climate pollutants being assessed, the use of a pair of time horizons with one for a short (e.g. 20 yr) time horizon with another for a long (e.g. 100 yr) time horizon will show the difference in temporal impacts of the climate pollutants. This improves transparency as no single-term metric can effectively capture the time-dependency of the impacts of SLCPs and LLCPs. Ocko et al. (2017) makes the analogy of using both a short and long-time horizon metric to the conventional reporting of systolic-diastolic blood pressure – each value is meaningful on their own but they are more valuable when reported together. The United Nations Environment Programme (UNEP) and Society of Environmental Toxicology and Chemistry (SETAC) joint Life Cycle Initiative recommend reporting both the GTP100, to indicate longer-term climate impacts, and GWP100, to indicate shorter-term climate impacts, and optionally the GWP20 for very near-term climate impacts (Jolliet et al. 2018). Proposed long-term metrics for the metric pairing include GTP100 (Cherubini and Tanaka, 2016; Cherubini et al., 2016; Levasseur et al., 2016; Jolliet et al., 2018) and GWP100 (Ocko et al., 2017). Proposed short-term metrics for the pairing include GWP100, GWP20, and GTP20 (Cherubini et al., 2016; Cherubini and Tanaka, 2016; Levasseur et a., 2016; Jolliet et al. 2018). CGTP and GWP* evaluating endpoint temperature 100 years in the future would be another potential long-term impact metric, as well as a short-term metric if applied to 20 years in the future.

Use of two or more metrics with different time horizons or formulations can help understand whether a given mitigation strategy is robust across a range of time horizons and underlying motivations. E.g. if a given mitigation strategy delivers climate benefits under, as an extreme example, both GWP20 and GTP100 as alternative metrics, then this would be regarded as a highly robust strategy; whereas if a given mitigation strategy would deliver climate benefits only for one metric but would increase climate change under another metric, then additional thought may be warranted whether the strategy should be adopted. Note that it may still make sense to adopt a strategy even if it does not deliver benefits under all metrics, depending on which metric would be more aligned with an organisation's objective and time horizons for action.

A related concept to consider is 'discounting', whereby future benefits or impacts are valued at a declining rate compared to the present (see Section 6.2.6). The choice of time horizon can also be informed by consistency with discount rates used for other strategic decisions. Different metrics and time horizons effectively correspond to different discount rates (Sarofim and Giordano, 2018; Mallapragada and Mignone, 2020). High discount rates put less value on impacts further into the future, and therefore emphasise the impact of shorter-lived pollutants.

Economic considerations can provide further insights into the choice of metric. As indicated in IPCC AR6 WGIII Chapter 2 and Annex II (Dhakal et al., 2022; Khourdajie et al., 2022), there is increasing evidence that supports the use of GWP100 under pathways toward the Paris Agreement goals as an approximation of economically optimal metrics at least till mid-century (Tanaka et al., 2021). Metrics for CH_4 derived from cost-benefit and cost-effectiveness frameworks lie roughly between 20 and 40, more consistent with GWP100 than GTP100 and GWP20. While this supports the adoption of GWP100 in the Paris Rulebook and the use of GWP100 in this context, it should be noted that this was an inadvertent outcome because GWP100 is by definition not intended to capture economic optimality.

A user may not wish to define any time-horizon or discount rate, but instead try to directly reveal how global warming impacts from emissions will vary over time under a range of mitigation strategies. In this case, approaches such as CGTP or GWP* may be applied, to report relative impacts not just at a pre-defined time, but across any number of years of interest. This is similar to providing multiple metrics and/or alternative time-horizons to provide insight into the temporal evolution of different climate pollutants, but without reporting a full temporal evolution, as described above. When these metrics are used, the starting point for the time series is critical as it provides the baseline level of warming against which any future change in temperature is expressed.

Where the assessment is highly affected by the choice of metric and when appropriate and practical, the use of climate models is a suitable alternative to metrics, to estimate the climate impact (Farquharson et al., 2017), as illustrated in Example 1. This provides a more comprehensive transparent way to describe complex climate impact than a simple metric and that could be used on its own or as a justification for selecting the assessment for a single metric that is most consistent with these more detailed analyses.

See Sections 6.2.5 and 6.2.6, and 6.3.1. to 6.3.3.

Example 1: The farmer primarily wants to know if there are benefits to climate (i.e., lower temperatures) of any intervention in the space of a decade. They would also want to know if this had implications (i.e., higher temperature) at any point before or after that.

6.5.1.4 Context and baseline counterfactual

The context in which the impacts of any emissions are assessed must be considered by the user. Are you interested in the total impacts of an emissions scenario of interest, potentially including the impacts of past emissions with current emissions' impacts, and how these combined impacts might relate to an overall climate objective? Or do you solely want to assess the potentially avoidable future impacts that will occur due to contemporary emissions?

Pulse metrics (e.g. GWP or GTP on any time horizon) capture the impact of an emission relative to no emission. In other words, these metrics tell us the extent to which their specified climate impacts could be avoided if we didn't release any given emission (marginal warming, see Section 6.2.4). They can also be used to compare the climate impacts of alternative mitigation strategies by evaluating the CO_2-equivalent emissions from two scenarios (e.g. with and without a particular mitigation strategy implemented) and determining the difference between those emission scenarios. This is demonstrated in example 1 (section 6.5.2.1).

Step-pulse metrics like GWP* capture the temperature impact of an emission relative to the temperature impact at a baseline year ('additional' warming, see Section 6.2.4). However, if you are calculating equivalent emissions from a baseline year, you should also explicitly consider what that baseline is (also see section 6.5.1.5).

As illustrated in section 6.2.4, this results in quite different perspectives for short- and long-lived GHGs, but also answers a fundamentally different question as it presents the warming impact only relative to warming in a historical reference year. For long-lived GHGs, each individual emission has a broadly additive impact, and so the occurrence of any emission causes further temperature increases beyond the conditions of the baseline year, and the only way to notably reduce temperatures below the baseline would be active GHG removal.

For short-lived GHGs, temperatures drop below those of the baseline year simply as a result of declining warming from prior short-lived gas emissions as they are removed from the atmosphere, and the baseline temperature would be maintained by short-lived gas emissions continuing at virtually the same level from the base year onwards. It is important to clarify that this does not in any sense imply that emissions of a short-lived greenhouse gas (i.e., methane) ever result in an active cooling of the climate. A reduction in emissions of short-lived GHGs can reduce the temperature increases that they had previously caused, up to the point of completely phasing out emissions of this short-lived GHG and thereby reversing most of the temperature contribution that they had made.

Both pulse and step-pulse metrics can use a ‘no emissions’ or a ‘no further policies’ counterfactual by calculating your chosen mitigation scenario CO_2-equivalent emissions, and considering the difference between that and the counterfactual scenario (e.g., ‘no emissions’ or ‘no further policies’). When different metrics lead to the same decision, the robustness of the evidence for that decision is clearer. Where use of different metrics would lead to different decisions, it is worth a reconsideration and articulation of the context, counterfactual scenario and the decision criteria.

See Section 6.2.4

Example 1: We wish to compare a ‘business as usual’ scenario with a ‘feed additive’ scenario. We also wish to know the climate impact of these two scenarios relative to a ‘no farm’ scenario.

6.5.1.5 Comparability and transparency

The comparability of metrics and the transparency of how assessment boundaries affect the metrics is important for selection.

GWP100 is the most commonly used metric including for reporting under the Paris Agreement, so reporting the assessment of impacts using GWP100 can be considered as a way to maintain comparability with many other assessments (Levasseur et al., 2016). If other metrics are chosen, also doing assessment with GWP100 can improve comparability and transparency of the assessment. If the assessment is highly affected by the choice of metric, then an explanation of why the assessment results are different than those using GWP100 can improve the understanding of the users.

The boundaries or counterfactual of the assessment do not change the amount of CO_2-equivalent assigned to a tonne of emitted CH_4 when using GWP and GTP. In these pulse metrics, any unit of emission is accounted for the same way irrespective of the source or the point in time of emissions.

In contrast, the amount of CO_2-warming-equivalent assigned to a tonne of emitted CH_4 using GWP* as defined in Forster et al., (2021) depends on the emissions in the present and 20 years ago. This means it is dependent on the emissions history of an individual emitter (see also Example 2), and if applied only from the present day relative to 20 years prior, will only indicate the additional effect of the emissions on the temperature trend at the present day. In other words, using GWP* to calculate CO_2-warming-equivalent emissions will indicate whether present day methane emissions are causing the temperature to rise or fall, but it will not tell you the absolute level of warming caused by the methane emissions. It may therefore be incomplete or misleading to note only the direction of travel, and not the absolute level of warming.

For example, if methane emissions decline by about 0.3 percent per year, based on GWP*, the CO_2-warming-equivalent emissions would be zero, no matter whether that year’s emission was 10 tonnes or 1 million tonnes. However, the absolute level of methane emissions (the 10 or 1 million tonnes) is relevant for assessing whether that level of emission is acceptable. One straightforward option would be to use GWP* in conjunction with an absolute annual methane emission expressed in terms of a metric reflecting marginal impacts, such as GWP100.

Step/pulse metrics like GWP* depend not only on changes in emissions today, but also the level of emissions 20 years ago. This causes no problem if the historic emission time series is reasonably smooth. However, in real world applications there might be considerable year-to-year variability in methane emissions, which can cause annual CO_2-warming-equivalent emissions using GWP* to be more variable than those calculated with a pulse metric (Meinhausen et al., 2022). The cumulative impact over time would remain accurate despite this year-to-year variability. While this variability would accurately reflect the consequences of a variable time series of emissions, it may have implications for the feasibility of policy based on such emissions that may need to be considered.

6.5.1.6 Other considerations

There may be other relevant influences when choosing metrics, which are unrelated to the underlying climate science of metrics and climate policy objectives (which is our focus here). For example, non-climate impacts like air quality and its impacts on human health and food production (Global Methane Assessment, 2021), the stage of development of a country or region, the importance of a sector to a region relative to other opportunities and the comparative/competitive advantage from an emissions perspective one region has over another. This must be factored in based on the judgement of practitioners and if there is a wider scope covering more than just climate impacts of emissions.

See section 6.2.7.

6.5.2 Examples

This section contains two examples to illustrate some of the concepts around metrics discussed above. Examples 1 and 2 are explored quantitatively to give the reader insights into the implications of using these metrics for analysing these case studies. Example 1 shows under what time scales each metric can represent the temperature outcome by using emissions from farms. The answer is not very obvious from the definition of a metric alone because an actual application may deal with sustained emissions over a certain period (like this example), which is different from pulse emissions used to define metrics such as GWP100. Example 2 illustrate the importance for selecting an appropriate baseline especially when step/pulse metrics such as GWP* is used.

6.5.2.1 Example 1: evaluation of emission metrics in representing the benefits of using a feed additive

A dairy farmer wants to evaluate the benefits of using a particular feed additive on their herd. They wish to use emission metrics to quantify the climate benefit that will result from their use of the feed additive. Their aim is to improve their environmental footprint over the next decade. As they aim to assess their improvement for making this intervention, this indicates that they want to compare the emissions using the feed additive relative to what emissions would be without the feed additive (see Table 9). The farmer already uses GWP100 in a GHG calculator, so there is precedent here.

Table 9. Annual emissions associated with the example farms.

	CH_4 t	N_2O t	CO_2 t
Control farm annual emissions	60	1.68	100
Feed additive farm annual emissions	40	1.68	105

Table 9 shows the emissions associated with the farm today ('control farm'), and to the farm when the feed additive is used. Methane emissions decrease with the introduction of the feed additive, but CO_2 emissions increase (which is caused by the production/distribution of the feed additive, based on current fossil fuel use in energy supply). What is the climate impact of switching from the control farm to using the feed additive? Does the effect of the increase in CO_2 emissions outweigh the effect of reduced methane emissions? We will explore these questions next.

The farm's aggregated annual GHG emissions using GWP100 before deploying the feed additive is 2179 t CO_2eq per year (using AR6 metric values: 60 * 27 + 1.68 * 273 + 100 = 2179 to the nearest round number). The feed additive lowers methane emissions but raises CO_2, with a combined effect of reducing the farm's annual emissions to 1644 t CO_2eq (40 * 27 + 1.68 * 273 + 105 = 1644). These totals may also be divided by the output leaving the farm (e.g., liters of milk) to express the emissions as per-product rather than per-farm footprint (subject to any allocations that may be required as part of Life Cycle Assessment, such as allocating a share of the emissions to other co-products such as beef or leather). To reiterate the context outlined above, these GHG footprints tell us the climate impacts of the farm's annual emissions, relative to if those emissions were not made (the 'marginal' impact of these emissions, as discussed in section 6.2.4). Implementing the feed additive therefore reduces the marginal climate impacts of the farm, as assessed using the GWP100 (i.e., specifically, that the total radiative forcing for 100-years following each year's emissions is reduced by using the feed additive). Below, we consider why and how different metric approaches may provide different quantification of these benefits.

Table 10. Change in annual emissions from using the feed additive compared to the control farm, aggregated using GWP, GTP and GWP.*

Unit	**CH_4**	**N_2O**	**CO_2**	**Aggregated**
Tonnes of each gas saved per year	-20	0	5	N/A
GWP100 CO_2eq tonnes saved per year	-540	0	5	-535
GWP20 CO_2eq tonnes saved per year	-1594	0	5	-1589
GTP100 CO_2eq tonnes saved per year	-94	0	5	-89
GTP20 CO_2eq tonnes saved per year	-1040	0	5	-1035
GWP* CO_2eq tonnes saved per year (for first 20 years; 2020-2039)	-2537	0	5	-2532
GWP* CO_2eq tonnes saved per year (after 20 years steady with the new emissions; 2040 onwards)	-157	0	5	-152

Table 10 shows the change in CO_2eq emissions that occurs when implementing the feed additive across the herd (i.e. the difference between annual emissions under business as usual and implementing the measure), as calculated using different metrics. There is a range of values of CO_2eq emissions from the different metrics (also shown in Figure 12), as each metric captures a different aspect of the impact of those emissions on the climate system. For the pulse metrics, the difference in annual equivalent emissions between the two scenarios is the same every year. Note that CO_2eq emissions are calculated using IPCC AR6 values for GWP and GTP (for example, 27 for GWP100 CH_4), with an exception being CO_2eq emissions based on GWP*, which use the IPCC AR5 value of GWP100 (that is, 28 for GWP100 CH_4) as this is consistent with the GWP* formula used in AR6 (Smith et al., 2021; footnote of Section 7.6.1.4 in IPCC (2021)).

For GWP*, there is a greater value placed on the difference in the first 20 years after the feed additive is introduced[5] (greater than the value placed by GWP20), and then a smaller value placed on the difference beyond that time[6] (more similar to the value placed by GTP100). How can the same difference of 20 tonnes of methane per year vary over time? If we were to tag each molecule of methane from this farm in the atmosphere, when we reduce the methane emission by 20 tonnes, the amount of tagged methane left in the atmosphere would decline over a period of approx. 20-40 years. It would then stabilise at a new equilibrium. In other words, the impact of reducing methane emissions on the atmosphere happens in the few decades immediately after the change in emissions. Later than that, the annual effects are much smaller. GWP* reflects this with its two terms. Pulse metrics either average these time-varying effects over a specified time period (e.g., GWP100), or only assess at a particular time period (e.g. GTP100).

To explore what that means in practice, we have used a simple climate model, the Aggregated Carbon Cycle, Atmospheric Chemistry, and Climate (ACC2) model (Tanaka et al., 2018; Tanaka et al., 2021; see Appendix 1 for details), to demonstrate how these metrics represent the different emissions.

First, we have modelled a scenario where the control farm emissions occur between 2000 and 2100, and then shown the impact of those emissions on global mean surface temperature (black line in Figure 12 I, b). We have then modelled the emissions if the farm starts using the feed additive in 2020 (black lines in Figure 12 II, d). In both cases, there are no emissions before 2000. The modelled temperature from these two simulations are also shown in Figure 12 I, a in the black solid and dashed lines, respectively. The difference between these two scenarios (control emissions and feed additive emissions) is also shown by the black line in Figure 12 III, f. The feed additive lowers the level of global warming caused by this farm by about a quarter by 2100. This clearly demonstrates that the level of global warming caused by the increased CO_2 emissions (from producing the feed additive) is smaller than the amount that the methane reductions lower the temperature by. Using modelled temperature as the metric, there are clear benefits to using the feed additive compared to not using it.

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[5] The CO_2-equivalent emissions in the first 20 years after the switch are calculated using GWP* based on the equation in section 6.2.3 as follows: 28x(4.53x40 - 4.25x60)-28x(4.53x60 - 4.25x60) = -2537 tCO_2eq/year

[6] The CO_2-equivalent emissions more than 20 years after the switch are calculated using GWP* based on the same equation, but with 40 tonnes of methane every year: 28x(4.53x40 - 4.25x40)-28x(4.53x60 - 4.25x60) = -157 tCO_2eq/year

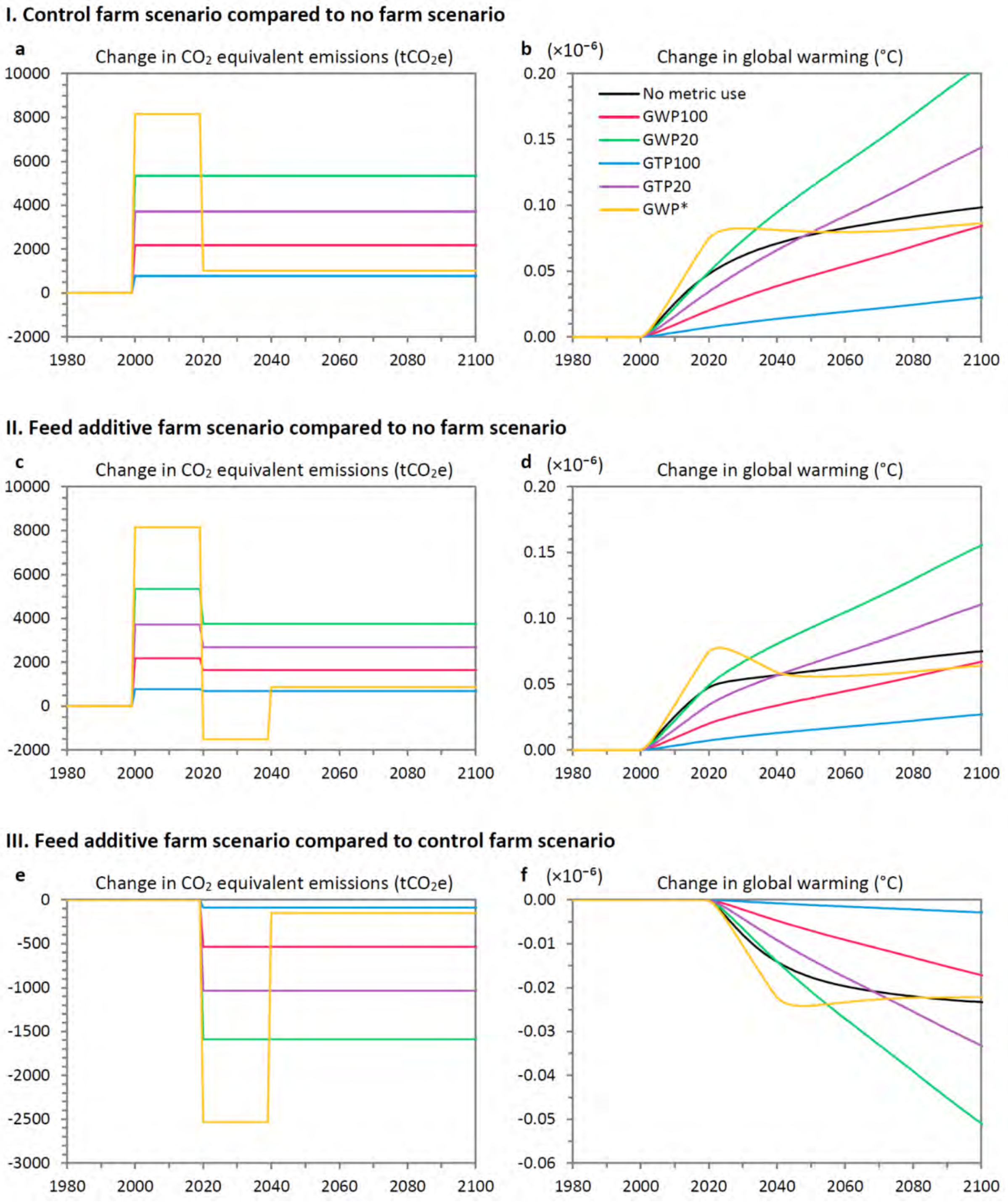

Figure 12. Scenarios computed using Aggregated Carbon Cycle, Atmospheric Chemistry, and Climate (ACC2) model (Tanaka et al., 2018; Tanaka et al., 2021).

Example I: evaluation of metrics based on implied temperatures from each metric-aggregated CO_2-equivalent emission. CO_2-equivalent emissions from the control farm and feed additive farm scenarios aggregated using different emissions metrics (a, c) and resulting changes in global warming calculated through the simple climate model ACC2 using these levels of CO_2 emissions, compared to the case in which no such farms exist (b, d). Black lines show the results by calculating the avoided warming separately from CO_2, methane, and nitrous oxide emissions (that is, emissions are not aggregated into CO_2-equivalent emissions for the 'no metric use' warming *calculations). The corresponding results for the difference between the control farm and the feed additive farm are shown in the last two panels: (e) CO_2-equivalent emissions avoided each year (based on each metric) by using the feed additive beginning in 2020 compared to those from the control farm and (f) avoided warming calculated from ACC2 using CO_2 equivalent emissions based on each metric. N.B. emissions associated with the land in the case of having no farm are not considered here (Manzano and White, 2019; Fløjgaard et al., 2022).*

As the black lines show the temperature change from modelling these emissions scenarios, this will allow us to illustrate how different emissions metrics represent the temperature change by comparing to the black lines. The difference in CO_2eq emissions between the two scenarios are shown in Table 10 and have been converted to CO_2eq. To test how well they approximate the temperature outcomes, we have put the CO_2eq emissions (calculated using each metric) into the simple climate model as CO_2 emissions.

The modelled warming arising from these emissions is shown by the coloured lines in Figure 12 b, d, and f. Note that these are not the temperature outcomes of the feed additive scenario, but what would happen if you emitted the same amount of CO_2 as the CO_2eq emissions defined by each metric. This allows us to illustrate differences between different metrics, and could help to inform an evaluation of which metric is 'best' to use for a question or objective related to global warming.

Figure 12, b and d demonstrate the logic behind why GWP* places a large CO_2eq value on changes to methane emissions for 20 years, and then a small value thereafter. When you model the GWP* emissions (yellow), the temperature curve approximates the modelled temperature from the original emissions (black). Comparing with the other emission metrics, GWP* is closest to using a short-term metric like GWP20 for the first 20 years, and GTP100 beyond that (Figure 12 a and c). It can be interpreted that GWP* approximates the modelled temperature outcomes by using these two values in one metric.

Just as CO_2 emissions act cumulatively, GWP* attempts to report methane emissions in such a way that cumulative emissions also link directly to temperature impacts. Therefore, even though the benefit per year declines after 20 years, the significant reduction in climate impacts achieved over the first 20 years persists. In this example, the total (cumulative) avoided GWP*-calculated CO_2eq emissions over any period (Figure 15c Appendix 1) could also be multiplied by a quantity called the TCRE (Transient Climate Response to cumulative Emissions, which is a factor that scales cumulative CO_2 emissions to the resulting temperature change (MacDougall, 2016)), to estimate the amount of avoided warming from this intervention at the end of that time period. This approach cannot be applied to cumulative emissions of the pulse-emission metrics such as GWP100.

Using GWP, the net avoided emission each year traps the same amount of additional energy in the climate system as would that equivalent amount of CO_2, integrated out to 100 or 20 years, for GWP100 or GWP20 respectively. The difference in magnitude between GWP100 and GWP20 values is because they are averaged over the 100 or 20 year periods, and methane has more radiative forcing impact in the first 20 years after emission. Importantly, GWP100 and GWP20 do not reflect the varying warming effects within these time periods. Also, this example deals with continuous emissions, while GWP is defined using pulse emissions. Thus, the time horizon of GWP is not directly related to the time scale of emissions concerned. Figure 12 b and d show that GWP20 (green), which is designed to focus on the warming potential over a 20 year period, approximates the relative warming over the first 20 years well, but then overestimates the temperature reduction after that. GWP100 (red), that is applied over a 100 year time horizon, underestimates the temperature reduction for the first hundred years. For GWP100, the cumulative relative warming (Figure 15c Appendix 1) is somewhat underestimated (red compared to black).

Using GTP100 or GTP20, the net avoided emissions *each year* would give the same change in temperature as from that equivalent amount of CO_2, at a time point 100 or 20 years after that emission. However, as this example shows a continuous emission and not a pulse emission, the modelled temperature for the original emissions and GTP emissions (blue and black, Figure 12 b and d) do not agree. There is a 'sustained GTP' metric, which is based on the temperature change at a specific time horizon due to a constant 1 kg per year increase in methane emissions (see section 6.2.2.2). Values of this GTP_s are similar to those of GWP (Shine et al 2005). In this example, this is borne out as the GWP100 (red) temperature intersects with the actual temperature (black) around 100 years after the emission change occurs. Similarly to GWP, there is large variation between GTP values across different time horizons, because the climate impacts from methane emissions decline rapidly beyond 20 years after the emission. The cumulative relative warming (Figure 15c Appendix 1) from GTP20 shows good agreement (purple compared to black) for approximately the first century.

In this example, all of the metrics show a benefit to introducing the feed additive, which reduces CH_4 emissions but at the same time increases CO_2 emissions. Considering the relative temperature change between using the feed additive or not (Figure 12 f), GTP100 indicates a very much underestimated benefit, whereas GWP20 indicates a very much overestimated benefit beyond about 40 years. GWP* overestimates the temperature benefit for the first 50 years, but shows an accurate agreement after that. GWP* cannot provide any better agreement here primarily because GWP* represents complex nonlinear climate responses only with two timescales (Allen et al., 2021). However, all metrics do show in this case that there is a clear benefit to using the feed additive, which is borne out when modelling the actual changes in emissions (black). This conclusion will not necessarily hold for every example, for example if the feed additive was associated with significantly higher CO_2 emissions.

In conclusion, each metric provides a different quantification of impacts from the emissions. Hence the importance of clearly defining the question or goal, so that an appropriate metric or metrics could be chosen. If there is no specific time horizon of interest, then multiple time horizons could be included through using GWP20 (to approximate temperature impacts over the first 30 years) and GWP100 (to approximate temperature impacts at a 100 year horizon), as well as GTP100 (to approximate temperature impacts after 100 years, for which the sustained version of the metric GTP_s would be most suitable here as it is a sustained change to emission rate) or over the whole time series using GWP*. Step-pulse metrics capture the time-varying impacts on temperature, however pulse metrics may give an acceptable approximation of benefits (e.g. cumulative relative warming) over specific timescales.

For further detail on this example, please see Appendix 1.

6.5.2.2 Example 2: Illustrating the path dependency of step/pulse metrics

Here, consider that we have three farms which, at present, have the same number of cattle and emissions as the control farm in example 1 (60t CH_4, 1.68t N_2O, 100 t CO_2). Despite the same emissions in 2020, all farms have a different emissions history. Farm A (Abraham) had stable emissions the whole time since being created 2000. Farm B (Bethany) was only started up in 2020, so had zero emissions before that year. Farm C (Chris) had twice the cattle/emissions when it started in 2000, but in 2020 abruptly cut the herd/emissions in half.

In a pulse metric like GWP100, the different history for the three farmers would not affect the valuation of their current emissions: the 2020 CO_2eq emissions of all three farmers would amount to 2178 t CO_2eq, as shown in the red lines from 2020 onwards in Figure 13. The yellow lines in Figure 13 show the CO_2-we emissions calculated using GWP* for the whole time series of emissions, where the amount of CO_2-we emissions calculated in 2020 does depend on the emissions in 2000, which were different for each farm. This reflects the 'additional' nature of applying GWP* to this case - that it is showing the additional impact of emissions at that point in time.

The CO_2-we levels for farm A and farm B are essentially the same but shifted by 20 years, as farm B is established 20 years after farm A. For the first 20 years after establishment, each of these farms are allocated their highest amount of CO_2-we using GWP* (over 8000 tonnes CO_2-we), which is then reduced to around 1000 tonnes CO_2-we in the years after that. As a result of the emissions in our example being dominated by methane, the abrupt reduction in methane emissions in methane for farm C leads to strongly negative CO_2-we values for the 20 years after the halving of emissions. This does not mean that the physical temperature contribution of the remaining emissions are negative, it just means that the temperature increase caused by emissions up to the year 2020 are partly reversed.

Without considering prior warming, applying GWP* could give seemingly contradictory results for farms with the same emissions at present, but different histories. This is because GWP*, in this example, is showing the 'additional' effects of the farms and not the 'marginal' effects (see Section 6.2.4 for explanation of terms). This example illustrates that accounting for the full temperature increase caused by emissions should be acknowledged to avoid potentially misleading or inequitable outcomes. This could be done by applying GWP* to the full historical timeseries of emissions, or applying it relative to the case where there was previously no farm (i.e. farm B) to show the maximum impact of the farm. To avoid neglecting this important context, it is suggested to not use a single year value for step/pulse metrics in isolation. Using a longer timeseries will give a more complete assessment. Alternatively, CO_2-we emissions could be evaluated alongside absolute annual methane emissions, to indicate both the additional effect of these emissions and the level which they are increasing or decreasing from.

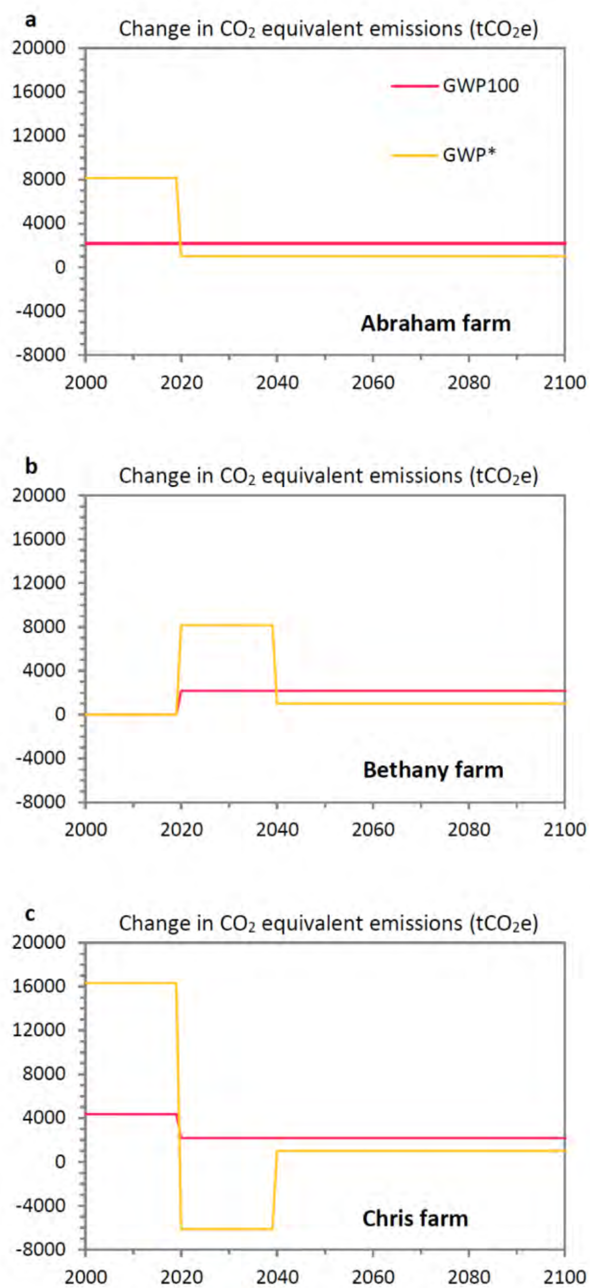

Figure 13. CO_2eq and CO_2-we emissions from the three farms calculated using GWP100 and GWP.*
Full results based on all metrics considered here, including temperature calculations, can be found in *Figure 16* of Appendix 1.

6.5.3 Summary of key features and limitations of GWPs, GWP and GTP*

One aim of this section is to summarise some of the key features and limitations of GWP100, GWP* and GTP. This is not fully comprehensive, however it serves as an outline or reminder to the agricultural practitioner, with the issues explored in more detail throughout this report.

6.5.3.1 GWP100

GWP100 is in common use, including as the specified metric for reporting emissions to the UNFCCC. It provides an estimate of how much energy is accumulated over the 100 year time period compared to an absence of the emission. It is therefore useful if the question is to compare the impacts to the climate system as a whole over the coming century compared to not emitting those GHGs (marginal impacts in the discussion above). The same could be said of GWP20, for the following 20 years after the emission only. GWP100 and GWP20 are derived from pulse emissions, and when applying them to sustained changes to emissions over multiple years, the metric time horizons (100 and 20 years) are not indicative of the impact (e.g. example 1 in Section 6.5.2.1). GWP100 has the disadvantage that it does not relate directly to how much methane emissions change the surface temperature over time (additional impacts in the the discussion above). In particular, it does not reflect that reducing methane emissions does not lead to additional warming. It also under-represents the strong warming caused by introducing new methane emissions. Therefore, the temperature outcomes over time from any trade-offs being considered would not be clear, because 1 tCO_2eq of methane does not cause the same amount of warming as 1 tCO_2eq of N_2O or CO_2.

Also see section 6.2.2.1.

*6.5.3.2 GWP**

GWP* is not a single-number metric, like GWP and GTP. GWP* approximates the warming that arises from a time-series of short-lived emissions like methane, relative to the warming at the starting point of that time series (termed additional warming). The minimum time-series required is two data points separated by 20 years, where it can be used to evaluate the effect of those methane emissions relative to the emissions 20 years prior. This may be a disadvantage for some applications if emissions from 20 years prior are not available, or assumptions cannot be made. This requirement forces the user to enter a thought process to define the question being asked more specifically, to ensure it is correctly reflected in the assumptions of past emissions, which relate to questions of responsibility, equity and fairness. If a baseline of the present day is used, then pathways are assessed relative to the present day level of warming from methane. We suggest that this level of warming is explicitly evaluated, otherwise the omission of this information could lead to misinterpretations that methane emissions could cause cooling. Rather, the baseline level of warming can be reversed through methane emission reductions, which is reflected by CO2-warming-equivalent emissions calculated using GWP*. Methane emissions which are reducing year-on-year will cause temperature to reduce compared to the baseline level of warming (additional impacts), but at the same time cause higher temperatures than if the emissions never occurred (marginal impacts). To give a more complete analysis, emissions could be evaluated starting from a time prior to the present day, e.g. at the time when the organisation or farm was established, or from a point in time onwards that may be relevant for climate policy, e.g. since 1990. GWP* is a useful metric if a time-series of emissions is being evaluated, or compared to another emissions scenario, based on impact on temperature e.g. comparison of benefits from several competing mitigation pathways.

As 1 tCO_2-we calculated with GWP* generates approximately the same temperature change over time no matter which gas it relates to, trade-offs can be assessed with respect to their effect on global warming using GWP*.

Also see section 6.2.3.

6.5.3.3 GTP

GTP can be used to estimate the amount of warming that would arise from an emission at a particular time horizon, compared to an absence of that emission. It is therefore useful if the question is to compare the temperature change at a specified time, compared to not emitting those GHGs. The disadvantage is that the time horizon must be specified, and therefore multiple calculations would be required if multiple time horizons were of interest, or if multiple years of emission related to the same end-point year was of interest. If trading off a one-off 1 tCO_2eq emission calculated using GTP, the temperature impact would be equivalent at that specific time horizon only.

When GTP100 is applied to emissions occurring over multiple years or decades, it does not represent the temperature impact at 100 years from the start of the example. For this, the 'sustained GTP' metric would be more indicative of temperature outcomes.

Also see section 6.2.2.2.

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APPENDICES

Appendix 1. Details of case studies

This section provides technical details of the model-based temperature calculations in Section 6.5 and presents the results from the two examples more comprehensively. We used a simple climate model to calculate the global-mean temperature changes from CO_2-equivalent emissions aggregated using each emission metric considered. The simple climate model we employed is the Aggregated Carbon Cycle, Atmospheric Chemistry, and Climate (ACC2)[7] model (Tanaka et al., 2007), with two recent applications to metric studies (Tanaka and O'Neill, 2018; Tanaka et al., 2021). ACC2 is one of the simple climate models that have been evaluated within a recent intermodel comparison project (Nicholls et al., 2020). Simple climate models are generally intended to calculate global-annual-mean changes in key aspects of the earth system (for example, surface temperatures and atmospheric CO_2 concentration) on annual, decadal, and centennial time scales. Such models do not deal with interannual and decadal variability of the earth system, as well as the seasonal cycle within a year. They do not generally provide projections at regional scales.

ACC2 consists of carbon cycle, atmospheric chemistry, physical climate, and economy modules. In the examples here, ACC2 is used as a simple climate model, without using the economy module, which is required when it is used as an Integrated Assessment model. The input to ACC2 is the emissions scenarios of greenhouse gases and air pollutants. The output from the model is the projections of atmospheric concentrations and radiative forcing of CO_2, CH_4, and N_2O, among others, and global-annual-mean temperature changes relative to preindustrial levels.

The physical climate module of ACC2 is an energy balance model coupled with an ocean heat diffusion model DOECLIM (Kriegler, 2005). The carbon cycle module is a box model comprising three ocean boxes, four land boxes, and a coupled atmosphere-mixed layer box. The model captures key nonlinearities of the global carbon cycle. Namely, the ocean CO_2 uptake saturates with rising atmospheric CO_2 concentration due to the thermodynamic balance involving carbonate species (Hooss et al., 2001; Bruckner et al., 2003). The land CO_2 uptake from the biosphere increases under rising atmospheric CO2 concentration due to the CO_2 fertilization effect. The atmospheric chemistry module accounts for the tropospheric O3 production from CH_4 emissions. The lifetime of CH_4 is related to the OH concentration, which further depends on the CH_4 concentration and pollutant emissions, providing a positive feedback to the CH_4 lifetime itself. The lifetime of N_2O is inversely related to the N_2O concentration, providing a negative feedback to the N_2O lifetime. It is important to note that each forcing term (or specifically atmospheric CO2, CH_4, and N_2O concentrations) is calculated separately without any gas aggregation using emission metrics, unless indicated otherwise. The equilibrium climate sensitivity is assumed to be 3 deg C, the best estimate of IPCC AR6 WGI (2021). Other uncertain parameters are optimized by using historical data and observations based on a Bayesian approach (Tanaka et al., 2009b).

[7] Note that the metrics and formula for GWP* have been developed using different simple climate models, and not ACC2. The discrepancy between the models (i.e. IPCC impulse response functions) used to derive metric values (including the GWP* equation) and the model (i.e. ACC2) used to investigate the temperature implications of metrics may explain some of the differences between the temperatures relying on metrics (colored lines) and the temperatures purely derived from the model (black lines).

To calculate the temperature effects of emissions from individual small farms in our examples, an assumption is required for the background emissions. We adopted the Representative Concentration Pathway (RCP) 4.5 W/m2 pathway, an emissions scenario in which the radiative forcing is stabilized at 4.5 W/m2 in the year 2100 (Moss et al., 2010). Thus, in our examples, emissions from individual farms are assumed on top of the RCP4.5 scenario. Emission data for RCP4.5 used in our analysis are consistent with those used in the intercomparison project for simple climate models (Nicholls et al., 2020). When we added farm emissions to the RCP4.5 scenario, we assumed 1,000 times larger farm emissions than the original magnitudes. Then the temperature difference due to farm emissions calculated from the model was divided by 1,000. We have checked the sensitivity of the results with respect to the scaling factor and have confirmed that the results do not depend on the scaling factor within a large range including 1,000.

Table 11. Absolute emissions when using the feed additive, relative to no emissions, aggregated using GWP, GTP and GWP.*

Note that CO_2eq emissions are calculated using IPCC AR6 values for GWP and GTP (for example, 27 for GWP100 CH_4), with an exception being CO_2eq emissions based on GWP, which use the IPCC AR5 value of GWP100 (that is, 28 for GWP100 CH_4) as described in the GWP* formula (Smith et al., 2021; footnote of Section 7.6.1.4 in IPCC (2021)). Note that a change to using the AR6 value of GWP100 in the GWP* formula would remain well within the uncertainties and not affect the results in any meaningful way.*

Unit	**CH_4**	**N_2O**	**CO_2**	**Aggregated**
Tonnes of each gas per year	40	1.68	105	N/A
GWP100 CO_2eq tonnes per year	1080	458	105	1644
GWP20 CO_2eq tonnes per year	3188	458	105	3751
GTP100 CO_2eq tonnes per year	188	391	105	684
GTP20 CO_2eq tonnes per year	2080	498	105	2683
GWP* CO_2eq tonnes per year (for first 20 years)	5074	458	105	5637
GWP* CO_2eq tonnes per year (after 20 years steady with the new emissions)	314	458	105	877

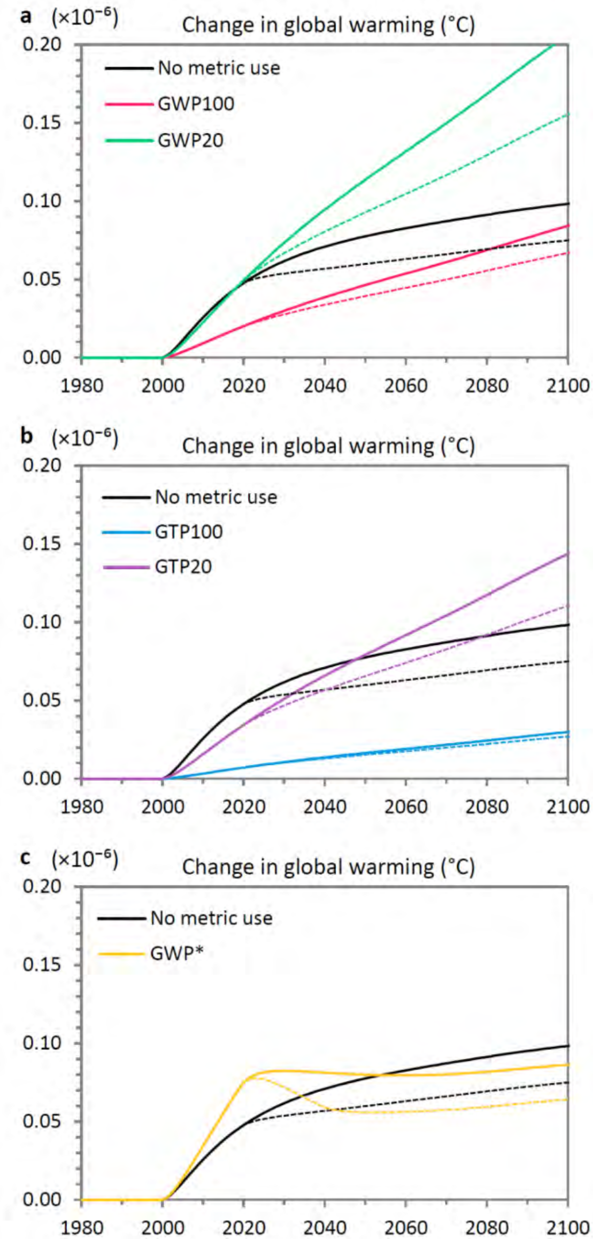

Figure 14. Additional results for Example 1.

Modelled global warming from the control farm (solid) and the feed additive farm (dashed) scenarios (black). Modelled global warming from CO2 emissions derived using different metrics of equivalence are shown by coloured lines. Panel a shows GWP based equivalence, b shows GTP based equivalence, and panel c shows GWP based equivalence.*

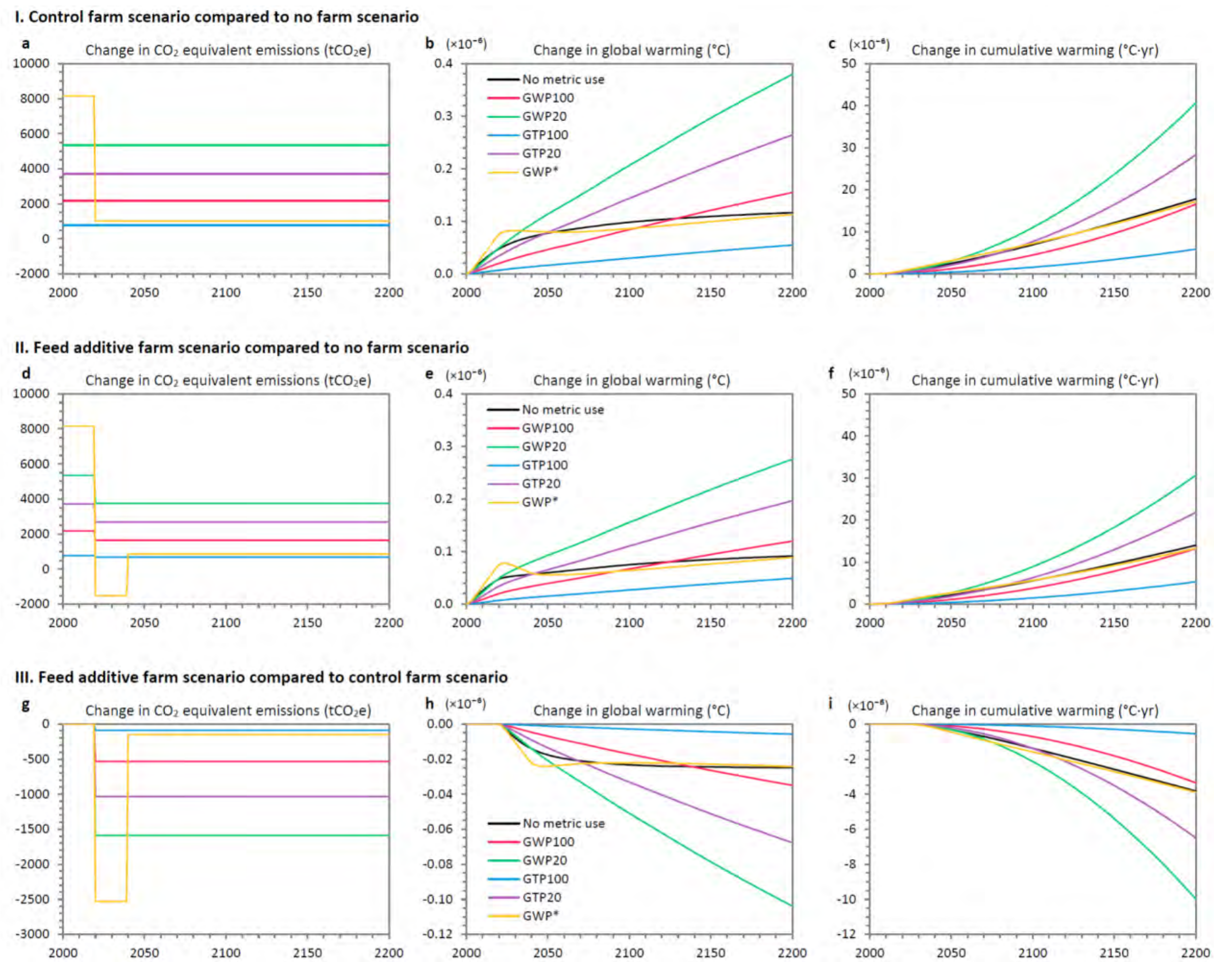

Figure 15. Detailed results for Example 1 (evaluation of emission metrics in representing the benefits of using a feed additive).

This figure shows the results for a longer time scale (until 2200), including cumulative warming, a proxy of climate damage. See text for further details.

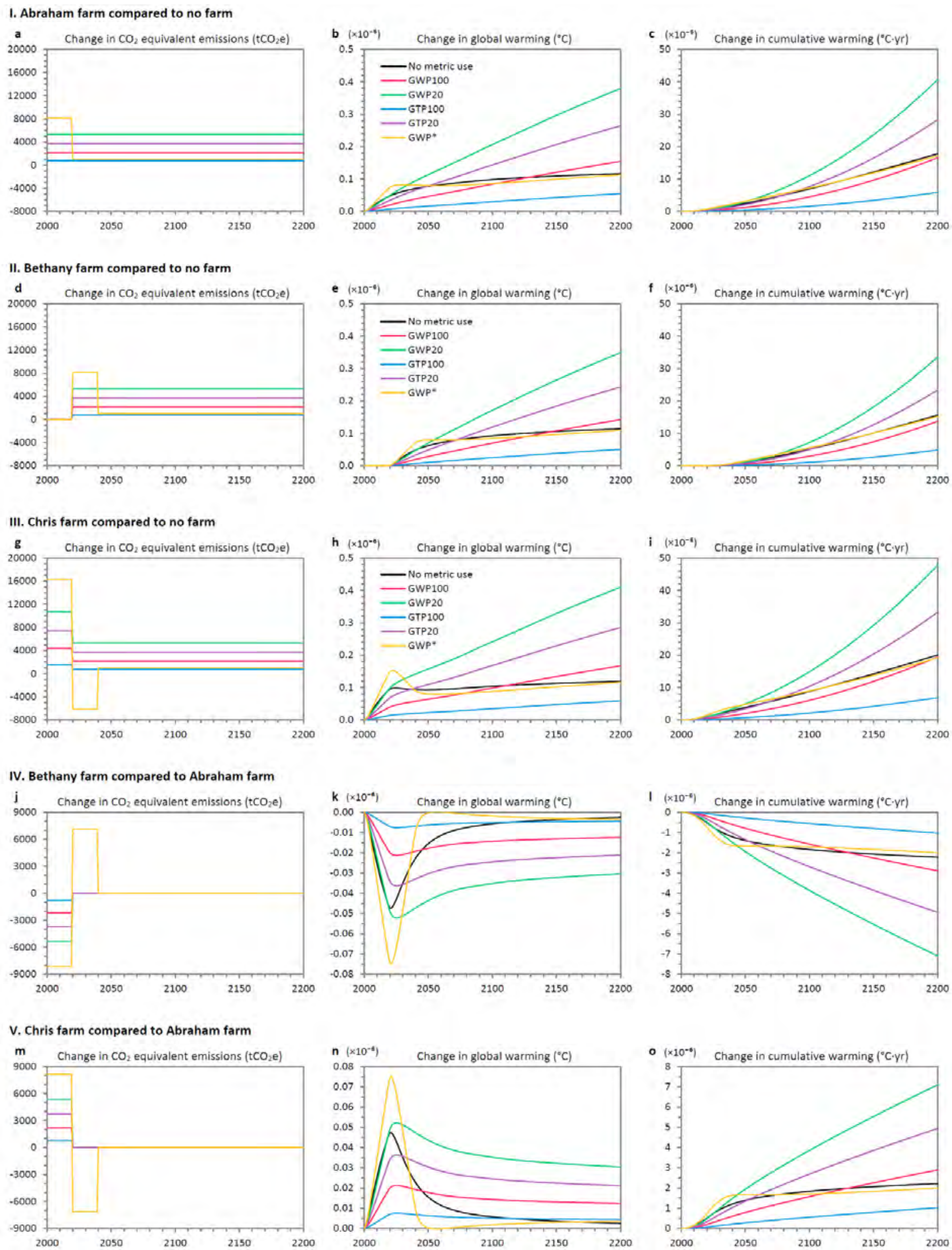

Figure 16. Detailed results for Example 2 (illustrating the path dependency of step/pulse metrics in representing the impacts of three farmers with different historical emissions).

This figure shows the results for a longer time scale (until 2200), as well as the temperature outcome based on the same method used in Example 1. See text for further details.

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