Food and Agriculture
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United Nations

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

Methane is a short-lived gas with an atmospheric lifetime of around a decade, whereas the dominant greenhouse gas, carbon dioxide, affects the global climate for hundreds of years. According to the 2021 Sixth Assessment Report of the Intergovernmental Panel on Climate Change, methane emissions from anthropogenic activities currently contribute about 0.5 °C to observed global warming. Reducing methane levels has been identified as a crucial – and rapid - step towards slowing down global warming.

Most anthropogenic methane emissions from agrifood systems result from the enteric fermentation of ruminant livestock and the anaerobic digestion of animal manure as well as other organic wastes, which involve complex metabolic interactions between microbial groups. By joining our efforts to reduce methane emissions from livestock and rice systems, agrifood systems will contribute to the Global Methane Pledge, a non-binding initiative signed by 150 countries at the 26th UN Climate Conference in 2021. Curbing methane emissions is an integral part of the strategies aimed at limiting the global temperature increase to well below 2 °C and preferably to 1.5 °C above the preindustrial level, in line with the Paris Agreement and with Sustainable Development Goal 13 on climate action. This goal also chimes with the call to raise ambitions for the mitigation commitments and targets outlined in nationally determined contributions.

For the first time, FAO, through the Livestock Environmental Assessment and Performance Partnership (LEAP), provides a comprehensive picture and robust analysis of methane emissions in livestock and rice systems. Developed by the FAO LEAP Partnership technical advisory group on methane, a multidisciplinary team composed of 54 international scientists and experts, this report analyses the sources and sinks of methane related to livestock as well as rice production systems, summarizes existing technical and innovative mitigation solutions, and evaluates metrics to quantify the impacts of methane emissions on the climate. The group analysed a wide range of scientific papers to offer valuable insights and provide scientific evidence that policy makers and stakeholders – including the public, the private sector, non-state entities and producers' organizations – can use to design and implement technical mitigation strategies and formulate policy frameworks to enhance climate actions in the context of livestock and rice systems.

Reducing greenhouse gas emissions is an important component of the FAO Strategy on Climate Change and of the Organization's Strategic Framework 2022-2031 founded on better production, better life, better nutrition and better environment. This report contributes to a better environment and supports Members in integrating specific methane mitigation interventions and targets into national climate actions as requested at the first session of the Sub-Committee on Livestock of FAO's Committee on Agriculture (COAG) (https://www.fao.org/3/ni966en/ni966en.pdf, paragraph 25).

I hope that the results and recommendations of this report bolster the efforts of countries and stakeholders committed to reducing methane emissions and, in doing so, moves us towards more efficient, inclusive, resilient, low-emission and sustainable agrifood systems.

Maria Helena Semedo

Maria Helena Semedo
Deputy Director-General

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Development process of the methane report

The Methane TAG was composed of 54 international experts in animal sciences, climate sciences, physics, plant sciences, soil sciences and environmental sciences. The Methane TAG was established in December 2020 and met virtually in the course of several meetings. It was co-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 of Great Britain and Northern Ireland) and Jun Murase (Co-Chair, Graduate School of Bioagricultural Sciences, Nagoya University, Japan), with the technical support of Aimable Uwizeye (Coordinator of FAO LEAP Secretariat, Italy).

PART 1. SOURCES AND SINKS OF METHANE EMISSIONS IN AGRICULTURE

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PART 2. QUANTIFICATION OF METHANE EMISSIONS

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PART 3. MITIGATION OF METHANE EMISSIONS

Karen A. Beauchemin (Agriculture and Agri-Food Canada, Lethbridge Research and Development Centre, Canada) and Emilio M. Ungerfeld (Centro Regional de Investigación Carillanca, Instituto de Investigaciones Agropecuarias [INIA], Chile) led the development of this part. It was co-authored by Adibe L. Abdalla (Center for Nuclear Energy in Agriculture, University of São Paulo, Brazil), Clementina Álvarez (Department of Research, TINE SA, Norway), Claudia Arndt (International Livestock Research Institute [IRLI], Kenya), Philippe Becquet (International Feed Industry Federation, Germany), Chaouki Benchaar (Sherbrooke Research and Development Centre, Agriculture and Agri-Food Canada, Canada), Alexandre Berndt (Embrapa Southeast Livestock, Brazil), Agustín del Prado (Basque Centre for Climate Change [BC3], Ikerbasque, Basque Foundation for Science, Spain), David Kenny (Teagasc, Animal and Grassland Research and Innovation Centre, Ireland), John Lynch (University of Oxford, United Kingdom), Rogerio M. Mauricio (Department of Bioengineering, Federal University of São João del-Rei, Brazil), Tim A. McAllister (Agriculture and Agri-Food Canada, Lethbridge Research and Development Centre, Canada), Mutian Niu (Institute of Agricultural Sciences, ETH Zürich, Switzerland), Walter Oyhantçabal (Facultad de Agronomía, Universidad de la República, Uruguay), Andy Reisinger (Independent consultant, New Zealand), Saheed A. Salami (Mootral Ltd, United Kingdom), Laurence Shalloo (Teagasc, Animal and Grassland Research and Innovation Department, Ireland), Yan Sun (Cargill Inc., United States), Maria Paz Tieri (FONTAGRO, Spain), Juan M. Tricarico (Innovation Center for U.S. Dairy, United States), Aimable Uwizeye (Animal Production and Health Division, FAO, Italy), Camillo De Camillis (Animal Production and Health Division, FAO, Italy), Martial Bernoux (Office of Climate Change, Biodiversity and Environment, FAO, Italy), Timothy Robinson (Animal Production and Health Division, FAO, Italy), Jun Murase (Nagoya University, Japan), and Ermias Kebreab (Department of Animal Science, University of California, Davis, United States). Part 3 was summarized and published: Beauchemin *et al.* 2022. Invited review: Current enteric methane mitigation options. *Journal of Dairy Science*, 105 (12): 9297-9326. https://doi.org/10.3168/jds.2022-22091.

PART 4. METRICS FOR QUANTIFYING THE IMPACT OF METHANE EMISSIONS

The development of this chapter was led by Michelle Cain (Cranfield University, United Kingdom), with sections co-led by John Lynch (University of Oxford, United Kingdom) for Section 9, William J. Collins (Department of Meteorology, University of Reading, United Kingdom) for Section 9.1, Miko Kirschbaum (Manaaki Whenua – Landcare Research, New Zealand) for Section 9.2, Brad Ridoutt (Commonwealth Scientific and Industrial Research Organisation [CSIRO], Agriculture and Food, Australia and Department of Agricultural Economics, University of the Free State, Bloemfontein, South Africa) and Agustín del Prado (Basque Centre for Climate Change [BC3] and Ikerbasque, Basque Foundation for Science, Spain) for Section 9.3, while Katsumasa Tanaka (Laboratoire des Sciences du Climat et de l'Environnement [LSCE], France and National Institute for Environmental Studies [NIES], Japan) co-led Section 9.4. This part was co-authored by Juan M. Tricarico (Innovation Center for U.S. Dairy, United States), Adibe L. Abdalla (Center for Nuclear Energy in Agriculture, University of São Paulo, Brazil), Alexandre Berndt (Embrapa Southeast Livestock, Brazil). Javier M. Echazarreta (Centro Carnes – Instituto Nacional de Tecnología Industrial [INTI], Argentina) Clementina Álvarez (Department of Research, TINE SA, Norway), Luiz G. Ribeiro (Embrapa, Brazil), Luis O. Tedeschi (Department of Animal Sciences, Texas A&M University, United States), Emilio M. Ungerfeld (Centro Regional de Investigación Carillanca, Instituto de Investigaciones Agropecuarias [INIA], Chile), Ermias Kebreab (University of California, Davis, United States), Munavar Zhumanova (Center for Global Change and Earth Observations, Michigan State University, United States), Brian G. McConkey (Viresco Solutions Inc., Canada), Karen A. Beauchemin (Agriculture and Agri-Food Canada, Lethbridge Research and Development Centre, Canada), Andy Reisinger (Independent consultant, New Zealand), Anna Flysjö (Arla Foods, Sweden), Dipti Pitta (University of Pennsylvania, United States), Jean-Baptiste Dollé (IDELE, France), Julián Chará (Center for Research on Sustainable Agriculture [CIPAV], Colombia), Maria Paz Tieri (FONTAGRO, Spain), Frank Mitloehner (University of California, Davis, United States), Samuel Wenifa Anuga (European University Institute, Italy), Saheed A. Salami (Mootral Ltd, United Kingdom), André Mazzetto (AgResearch, New Zealand), Claudia Arndt (International Livestock Research Institute [IRLI], Kenya), Chaouki Benchaar (Agriculture and Agri-Food Canada, Canada), Jacobo Arango (International Center for Tropical Agriculture [CIAT], Colombia), Joeri Rogelj (Centre for Environmental Policy, Imperial College London, United Kindom), Mutian Niu (Institute of Agricultural Sciences, ETH Zürich, Switzerland), Stephan Pfister (ETH Zürich, Switzerland), Carl-Friedrich Schleussner (Humboldt Universität zu Berlin, Germany), Walter H. Oyhantçabal (Cironi, Ministry of Livestock, Agriculture and Fisheries, Uruguay), Ranjan Parajuli (EcoEngineers, University of Arkansas, United States), David Kenny (Teagasc, Animal and Grassland Research and Innovation Department, Ireland), Jacob P. Muhondwa (Ardhi University, Tanzania), Mélynda Hassouna (National Research Institute for Agriculture, Food and Environment [INRAE], Institut Agro Rennes, France), Hongmin Dong (Chinese Academy of Agricultural Sciences [CAAS], China), Jun Murase (Nagoya University, Japan), Tim McAllister (Agriculture and Agri-Food Canada, Canada), Michaël Mathot (Agricultural Systems Unit, Walloon Agricultural Research Centre, Belgium), Philippe Becquet (International Feed Industry Federation, Germany), Robert T. Burns (University of Tennessee, Knoxville, United States), Rogerio M. Mauricio (Department of Bioengineering, Federal University of São João del-Rei, Brazil), Stephen Wiedemann (Integrity Ag & Environment, Australia), Alice Anyango Onyango (International Livestock Research Institute, Kenya), Laurence Shalloo (Teagasc, Animal and Grassland Research and Innovation Department, Ireland), Yan Sun (Cargill, United States), and Aimable Uwizeye (Animal Production and Health Division, FAO, Italy).

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Multi-step review process

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The Livestock Environmental Assessment and Performance (LEAP) Partnership

The FAO LEAP is a multi-stakeholder initiative launched in July 2012 to improve the environmental performance of livestock supply chains. Hosted by the Food and Agriculture Organization of the United Nations, the partnership 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 FAO LEAP Partnership provides state-of-the-art methods and metrics to assess environmental impacts and benchmark performance across livestock supply chains.[1]

THE STEERING COMMITTEE OF THE FAO LEAP PARTNERSHIP

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THE LEAP SECRETARIAT

FAO LEAP Secretariat has coordinated and facilitated the work of this TAG. The Secretariat guided and contributed to the content development, as well as ensuring coherence with other existing guidelines. The LEAP Secretariat, hosted at FAO, is composed of Aimable Uwizeye (FAO Technical Officer and Coordinator of

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

the Secretariat since August 2022), Camillo De Camillis (Partnership Manager until April 2022), Xiangyu Song (Partnership Manager since December 2022), Monica Rulli (Technical Specialist), María Soledad Fernández González (Communication Specialist until January 2021), Emmie Wachira (Programme and Outreach Specialist until December 2021), Sara Giuliani (Programme and Outreach Specialist since March 2022), Tim Robinson (Secretariat Coordinator until July 2022), Henning Steinfeld (Partnership Co-Chair until July 2022), Thanawat Tiensin (Partnership Co-chair since January 2023).

ADDITIONAL CONTRIBUTIONS

Professional editing and proofreading were done by Agnieszka Gratza. Sara Giuliani looked after the communication and publication management. Claudia Ciarlantini (FAO) and Enrico Masci were responsible for the design and layout of this publication. Administrative support was provided by Eva María Pardo Navarro and Isabel Burgos.

Abbreviations

AD	anaerobic digestion
AFOLU	agriculture, forestry and other land use
AR4	Fourth Assessment Report of the Intergovernmental Panel on Climate Change
AR5	Fifth Assessment Report of the Intergovernmental Panel on Climate Change
AR6	Sixth Assessment Report of the Intergovernmental Panel on Climate Change
ATP	adenosine triphosphate
AWD	alternative wetting and drying
BAT	best available technology
BCM	bromochloromethane
BES	bromoethanesulfonate
CF	characterization factor
CGTP	combined global temperature change potential
CO_2eq	carbon dioxide equivalent
CP	crude protein
CT	condensed tannins
DM	dry matter
DMI	dry matter intake
DNDC	DeNitrification-DeComposition
EF	emission factor
FAO	Food and Agriculture Organization of the United Nations
GCP	global cost potential
GDamP	global damage potential

GE	gross energy
GEI	gross energy intake
GHG	greenhouse gas
GIS	geographic information system
GIT	gastrointestinal tract
GTP	global temperature potential
GWP	global warming potential
GWP*	GWP-star
HT	hydrolysable tannins
IAM	integrated assessment model
IDF	International Dairy Federation
IRRI	International Rice Research Institute
IgG	immunoglobulin G
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
JRC	Joint Research Centre of the European Commission
LCA	life cycle assessment
LEAP	FAO Livestock Environmental Assessment and Performance Partnership
LMD	laser methane detector
LUC	land use change
MCFA	medium-chain fatty acids
MCR	methyl-coenzyme M reductase
NASEM	National Academies of Sciences, Engineering, and Medicine
NDF	neutral detergent fibre
NOPA	3-nitrooxypropionic acid

OM	organic matter
OMD	organic matter digested
PUFA	polyunsaturated fatty acids
PGPR	plant growth-promoting rhizobacteria
SETAC	Society of Environmental Toxicology and Chemistry
SPS	silvopastoral system
SRI	system of rice intensification
TAG	technical advisory group
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
VFA	volatile fatty acid
VR	ventilation rate
WSC	water soluble carbohydrates
Ym	methane conversion factor (percent)

CHEMICAL ELEMENTS AND FORMULAE

3-NOP	3-nitrooxypropanol
C	carbon
CH_4	methane
Cl	chlorine
CO_2	carbon dioxide
FeS	iron(II) sulphide
H^+	proton, cationic form of atomic hydrogen
H_2	hydrogen

N	nitrogen
NAD^+	oxidized nicotinamide adenine dinucleotide
NADH	reduced nicotinamide adenine dinucleotide
NH_3	ammonia
N_2O	nitrous oxide
O_3	ozone
OH	hydroxyl
SF_6	sulphur hexafluoride, tracer gas

UNITS

°C	degree Celsius
Gt	gigatonne, metric unit equivalent to 1 billion (10^9) tonnes
MJ	megaJoule
Mt	megatonne, metric unit equivalent to 1 million (10^6) tonnes
nm	nanometre
Ppm	parts per million
Tg	teragrsam, metric unit equivalent to 1 million (10^6) tonnes
W	Watt

Executive summary

The report contains four parts, addressing 1. the sources and sinks of methane (CH_4) emissions in agriculture; 2. the quantification of CH_4 emissions; 3. the mitigation of CH_4 emissions and 4. the metrics for quantifying the impact of CH_4 emissions. The majority of CH_4 emissions from the agricultural sector are a consequence of microbial-mediated enteric fermentation processes in ruminant livestock, which make up about 30 percent of global anthropogenic CH_4 emissions. Anaerobic digestion of animal manure and other organic wastes, which involves complex metabolic interactions between microbial groups, contributes to about 4.5 percent of the world's anthropogenic CH_4 emissions. Rice paddies, meanwhile, are estimated to contribute 8 percent of total human-caused CH_4 emissions. Global CH_4 emissions are largely offset by the atmospheric and soil CH_4 sinks. The atmospheric sink occurs through the chemical degradation of CH_4 by hydroxyl (OH) and chlorine (Cl) radicals in the troposphere and stratosphere and is responsible for 90 to 96 percent of the global CH_4 sink. The soil accounts for about 4 to 10 percent of the CH_4 degraded. The ocean acts as a small CH_4 sink for atmospheric CH_4.

Methane is a short-lived gas which has an atmospheric lifetime of around a decade, whereas the dominant greenhouse gas (GHG), carbon dioxide, affects the climate for hundreds of years, if not longer. Because of this difference in their respective lifetimes, the GHG emission metrics used to compare CH_4 with CO_2 (carbon dioxide) vary depending on what time frame they consider. This is not an issue for nitrous oxide (N_2O), for example, as its lifetime extends beyond a century and metrics typically compare time frames of a century or less. The appropriate quantification of GHG emissions, specifically CH_4, has raised questions about how GHG emission inventories are reported and, perhaps more importantly, how best to mitigate CH_4 emissions. This review documents existing methods and methodologies to measure and estimate CH_4 emissions from ruminant animals and the manure produced therein according to various scales and conditions.

Measurements of CH_4 have frequently been conducted in research settings using classical methodologies developed for bioenergetic purposes, such as gas exchange techniques (respiration chambers, headboxes). While very precise, these techniques are limited to research settings as they are expensive, labour-intensive and applicable only to a few animals. Head-stalls, as exemplified by the GreenFeed system, have been used to measure expired CH_4 for individual animals housed alone or in groups, in confinement or grazing. This technique requires frequent animal visitation over the diurnal measurement period and an adequate number of collection days. The tracer gas technique can be used to measure CH_4 from individual animals housed outdoors, as low background concentrations of methane make it easier to detect emissions. Micrometeorological techniques, such as open-path lasers, can measure CH_4 emissions over larger areas and from many animals, but limitations exist, including the need to measure over more extensive periods.

The measurement of CH_4 emissions from manure depends on the type of storage, animal housing, CH_4 concentration inside and outside the boundaries of the target area, and ventilation rate (VR), which is likely the variable that contributes the most to measurement uncertainty. Chamber (open/closed) and micrometeorological methods are used to collect CH_4 fluxes from rice paddy soils *in situ*. For large-scale areas, aircraft, drones and satellites have been used in association with the tracer flux method, inverse modelling, imagery, and light detection and ranging (LiDAR), but research is lagging behind in validating these methods. Bottom-up approaches to estimating CH_4 emissions rely on empirical or mechanistic modelling to quantify the contribution of individual sources (enteric and manure). In contrast, top-down approaches estimate the amount of CH_4 in the atmosphere using spatial and temporal models to account for transportation from an emitter to an observation point. While these two estimation methods rarely agree, in practice they help identify knowledge gaps and research requirements.

The Sixth Assessment Report of the Intergovernmental Panel of Experts on Climate Change found that methane emissions from all human activities contributed about 0.5 °C to the present observed warming. Decreasing the emissions of enteric CH_4 from ruminant production is a key element of the strategies designed to limit the global temperature increase to 1.5 °C. Research in the area of enteric CH_4 mitigation has grown exponentially in the last two decades, and various strategies for enteric CH_4 abatement have been investigated: production intensification, dietary manipulation (including supplementation and processing of concentrates and lipids, management of forage and pastures), rumen manipulation (supplementation of ionophores, 3-nitrooxypropanol, macroalgae, alternative electron acceptors and phytochemicals), and the selection of low CH_4-producing animals. Other enteric CH_4 mitigation strategies, although at less advanced stages of research, are rapidly developing.

The report discusses and analyses the currently available enteric CH_4 mitigation strategies with an emphasis on opportunities and barriers to their implementation in confined and partial grazing production systems, as well as in extensive and fully grazing production systems. For each enteric CH_4 mitigation strategy, the report discusses its effectiveness in decreasing the total CH_4 emissions and emissions calculated on a per animal product basis, safety aspects, impacts on the emissions of other GHGs, and other economic, regulatory and societal issues that are key to implementation. Most research has been conducted with confined animals, and considerably more research is needed to develop, adapt and evaluate anti-methanogenic strategies for grazing systems. In general, few options are currently available for extensive production systems that do not use feed supplements. Further research is needed to develop enteric CH_4 mitigation strategies that are locally applicable. There is a lack of information required to calculate carbon footprints of interventions on a regional basis, which would make it possible to evaluate the impact of mitigation strategies on net GHG emissions. Economically affordable enteric CH_4 mitigation solutions are also in short supply. Several agricultural practices, including water management, organic amendment, fertilizer management and crop management, can mitigate CH_4 emissions from rice paddies. Locally appropriate options that consider rice yields and the risk of other GHG emissions, like N_2O, should be adopted. A successful implementation of safe and effective anti-methanogenic strategies will depend on delivery mechanisms and adequate technical support for producers but also on consumer involvement and acceptance. It calls for a holistic approach and buy-in at all levels of the supply chain.

Part 4 of this review focuses on metrics used to quantify the impact of methane emissions, and the mitigation thereof. The primary purpose of a GHG emission metric is to provide information on how different GHG emissions (or emission reductions) contribute to climate change and the resulting impacts. The metric can then be used to aggregate different GHG emissions into a total "CO_2-equivalent" emission. Each GHG emission metric is defined by a given climate impact (e.g. temperature, radiative forcing) over a given time. A metric that establishes equivalence as regards one key measure of the climate system's response to emissions does not imply equivalence with respect to other key measures. The most appropriate metric depends on the policy objectives (i.e. what aspect of climate change does the policy focus on and over which time horizon). The most common GHG emission metric is the global warming potential (GWP) integrated over 100 years, which is used for reporting national emissions inventories to the UNFCCC.

The context in which GHG metrics are used is reviewed, alongside technical descriptions of pulse-emission metrics and step-pulse metrics, including a discussion of time horizons, discount rates and non-radiative forcing impacts. Pulse-emission metrics compare 1 kg of a gas to 1 kg of another gas, usually over a chosen time frame (e.g. a 100-year period after the emission) or at a particular time in the future (e.g. 50 years after the emission occurs). For short-lived pollutants like CH_4, this means that the choice of time horizon has a large impact on their metric value. The GWP and the global temperature potential (GTP) are pulse-emission metrics. Taking a different approach to comparing gases, step-pulse metrics account specifically for the effects of sustained short-lived emissions. The equivalence is based on working backwards from the temperature or radiative forcing outcome of a time series of CH_4 emissions, and approximating what CO_2 emissions would lead to the same temperature or radiative forcing outcome. The step-pulse metrics explored in this report are the GWP* (GWP-star) and the combined global temperature change potential (CGTP).

Pulse-emission metrics primarily provide information about the future climate impacts (as defined by the specific metric) that would be caused by an extra unit of emission of a given gas, compared to having no emission. In this review, we call these impacts "marginal", e.g. "marginal warming". In 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 resulting from previous emissions. We call these impacts "additional" since the reference date, e.g. "additional warming". It is important to note that there is no purely scientific or universal basis to determine metric choice.

The use of metrics in assessments of impact and mitigation is reviewed. Metrics are needed if potential trade-offs with emissions of other GHGs are to be evaluated, or to compare different sectors or emitters with a variety of GHGs being emitted. If a reduction target for CH_4 alone is under consideration, a metric would not be required to track progress, although it could be used to evaluate or justify the level of ambition. Metrics used in the life cycle assessment (LCA) should be chosen to match the user's impact objectives, which could encompass a variety of environmental impact objectives. Cost-benefit and cost-effectiveness assessment are discussed, and this includes a range of different metrics which factor in associated costs. Irrespective of the method used to aggregate different GHGs, reporting the emissions of individual GHGs is recommended to ensure clarity and transparency. Using a range of metrics can help to test the sensitivity of climate change impact assessments to the choice of metric.

Metrics are used within a wider policy framework relating to climate action and sustainable development. Hence the key themes relevant for metrics outlined in the final part of the report, which discusses the Paris Agreement, differing definitions of climate neutrality and the complexity this introduces, sustainable agriculture and equity considerations, among other issues. These are of crucial importance when applying metrics and making decisions about targets and climate action.

Introduction

Achieving the sustainability of agrifood systems is urgent and the global community is expecting each sector of the economy to undertake the necessary transformative actions. Sustainability remains a challenge for the agrifood systems sector because of the sheer volume of food, and livestock products in particular, produced to meet the nutritional needs of a growing population in the context of climate change and other environmental impacts. In 2017, agrifood systems, including agriculture, forestry and other land use (AFOLU), were responsible for 23 percent of total anthropogenic greenhouse gas (GHG) emissions globally, assessed using a global warming potential (GWP) for 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, above all the energy sector. For instance, the US Environmental Protection Agency (EPA) reports that, although agrifood systems are responsible for 9 to 10 percent of total GHG emissions, livestock contributes less than 4 percent of direct emissions (Dillon *et al.*, 2021; Tedeschi, 2022). Most of the emissions from AFOLU are in the 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 global ruminant population increased by 66 percent from 1960 to 2017, whereas the population of non-ruminants increased even more rapidly by 435 percent over the same period. Both ruminant and non-ruminant populations are projected to further increase, which will only exacerbate GHG emissions, in particular CH_4, from livestock systems (FAO, 2018b). Meat and milk from ruminant livestock provide an important source of protein and other nutrients destined for human consumption. Although ruminants have a unique advantage of being able to consume forages and graze on 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 in manure management systems.

Over 150 countries and supporters have endorsed the Global Methane Pledge (www.globalmethanepledge.org), a voluntary commitment initiated by the European Union and the United States of America to collectively decrease CH_4 emissions by 30 percent from 2020 levels by 2030. Reducing CH_4 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, decreasing CH_4 emissions is seen as a rapid way of helping to 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 was developed by an international group of scientists and experts working on the sources and sinks of CH_4, the quantification of CH_4 emissions, and related mitigations and climate metrics. The report aims to provide a comprehensive review and analysis of CH_4 sources and sinks in agrifood systems, existing mitigation solutions and those that are at an experimental stage, and metrics used to quantify the impacts of CH_4 emissions on climate. In the context of the Global Methane Pledge and the Paris Agreement goals, this report provides comprehensive scientific information that can be used by different stakeholders – including the public, the private sector, non-state entities and producers' organizations – to design and implement technical mitigation strategies and programmes aimed at cutting CH_4 emissions in livestock and rice systems. It also contains useful information designed to facilitate policy work and enhance national climate actions. The report complements the previous FAO LEAP guidelines with detailed information needed to conduct mitigation scenario analysis, using the highest tier from the IPCC guidelines. This will continuously improve the accuracy, transparency, consistency, comparability and completeness of the inventory of greenhouse gases, including CH_4, as well as the monitoring of mitigation programmes in livestock.

The report is divided into four parts:

- Part 1: Sources and sinks of methane emissions from food and agriculture
- Part 2: Quantification of methane emissions
- Part 3: Mitigation of methane emissions
- Part 4: Metrics for quantifying the impact of methane emissions

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Part 3 was published as Beauchemin *et al.* 2022. Invited review: Current enteric methane mitigation options. *Journal of Dairy Science*, 105(12): 9297-9326. https://doi.org/10.3168/jds.2022-22091.

PART 1

Sources and sinks of methane emissions from food and 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 to 70 percent, with developing countries accounting for the 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 of ruminant livestock and, to a lesser extent, storage of manure. The livestock sector is the largest land-use system on earth, occupying between 30 (Herrero *et al.*, 2013) and 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.*, 2013a), and it is estimated that about 80 percent of the GHG emissions from livestock and 90 percent of CH_4 emissions is derived from ruminant livestock (Scholtz, Neser and Makgahlela, 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 graze on 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, depending on the type of feed offered, contributing to approximately 6 percent of global anthropogenic GHG emissions (Beauchemin *et al.*, 2020).

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. Among ruminants, the highest daily emitters on a per animal basis are cattle, followed by sheep, then goats and buffalo, which have similar emissions (Seijan *et al.*, 2011). Emissions of GHG, including CH_4, produced by both large herbivorous non-ruminants and the sizeable population of small farm animals, such as swine, remain substantial (Patra, 2014). Indeed, Clauss *et al.* (2020) stated that CH_4 emissions from some non-ruminants, when expressed in terms of intensity, remain comparable to those of ruminants. Misiukiewicz *et al.* (2021) recently produced a comprehensive review of methanogens living in the gastrointestinal tract (GIT) of various non-ruminants, such as swine, horses, donkeys, rabbits and poultry.

Enteric CH_4 emissions can vary substantially between animals, even within the same species, and there is increasing evidence that host genetics play an important role in this (see Section 1.2). Notwithstanding, factors such as the chemical composition of the diet, the level of feeding above maintenance, and the inclusion of certain feed additives have a greater influence on individual animal emissions than genetic makeup per se. Indeed, the manipulation of environmental factors can be harnessed as mitigation strategies to reduce CH_4 emissions (see Part 3).

The rumen is a complex ecosystem composed of bacteria, fungi, protozoa, archaea 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 structural plant carbohydrates that cannot be digested by humans and other animals, while 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 utilized to reduce CO_2 by methanogens, which belong to the domain Archaea. Methane produced in the rumen accounts for up to 90 percent of ruminant enteric CH_4 emissions, whereas microbial fermentation in the large intestine accounts for the remainder of emissions. Large intestine fermentation is also a characteristic of non-ruminants such as swine and hind-gut fermenters such as horses, which also produce CH_4 but to a much lesser extent.

1.2 BIOCHEMISTRY OF METHANE PRODUCTION IN MICROBIAL ANAEROBIC ECOSYSTEMS

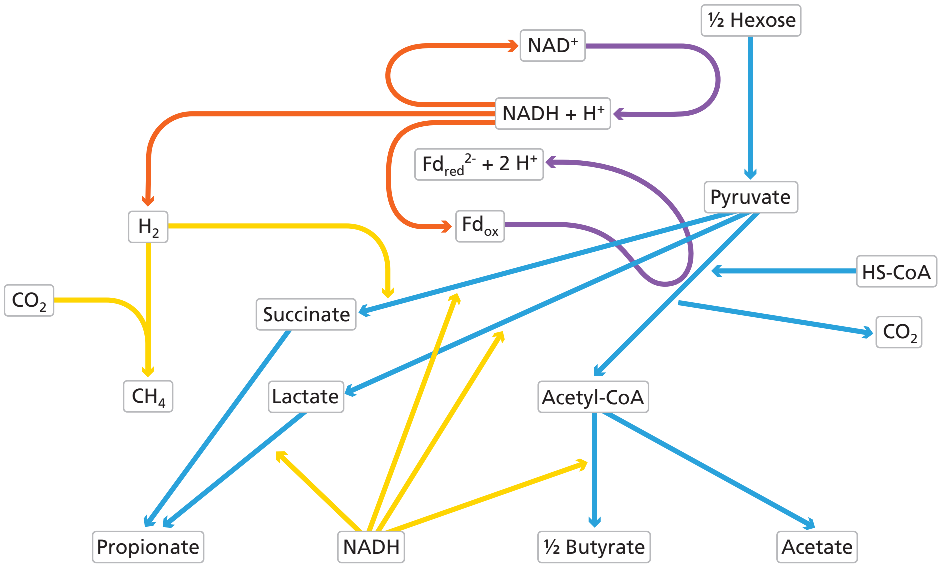
In anaerobic environments with low oxygen concentrations and limited mineral electron acceptors, fermentation can provide Gibbs energy to generate adenosine triphosphate (ATP) necessary for microbial maintenance and growth. Fermentation is an incomplete oxidation and carbon compounds formed in the process 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 into volatile fatty acids (VFA; mainly acetate, propionate and butyrate), CO_2 and CH_4 as the main final products of fermentation, with dihydrogen, formate, lactate and succinate as important electron carrier intermediates (Figure 1; Russell and Wallace, 1997; Ungerfeld, 2020).

Blue arrows signal transformations of carbon compounds. Note that smaller flows of VFA interconversion also exist (see summary by Ungerfeld and Kohn [2006] and more recent work by Markantonatos, Green and Varga [2008], Markantonatos *et al.* [2009], Markantonatos and Varga [2017], Nolan *et al.* [2014] and Gleason, Beckett and White [2022]). Purple arrows point to semi-reactions of cofactor reduction. Red arrows indicate 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). Lastly, the 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 are marked with yellow arrows. Metabolic hydrogen in reduced

Figure 1
Main biochemical pathways in rumen fermentation



Source: Adapted from Russell, J.B. & Wallace, R.J. 1997. Energy-yielding and energy-consuming reactions. In: P.N. Hobson & C.S. Stewart, eds. The rumen microbial ecosystem, pp. 246– 282. London, Blackie Academic & Professional. https://doi.org/10.1007/978-94- 009-1453-7_6 and Ungerfeld, E.M. 2020. Metabolic hydrogen flows in rumen fermentation: Principles and possibilities of interventions. *Frontiers in Microbiology*, 11: 589. https://doi.org/10.3389/fmicb.2020.00589

nicotinamide adenine dinucleotide (NADH) at a particular point in time may or may not have been part of the dihydrogen pool prior to its incorporation into a particular pathway, therefore NADH shown as a direct electron donor in propionate and butyrate formation may or may not have been formed through reduction of oxidized nicotinamide adenine dinucleotide (NAD^+) with H_2 in bifurcation with ferredoxin.

As with 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 dihydrogen transfer. 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 reoxidized for fermentation to continue (Wolin, Miller and Stewart, 1997). Cofactor reoxidation occurs mostly through hydrogen-evolving hydrogenases which transfer electrons to protons to form dihydrogen (Frey, 2002) and formate (Russell and Wallace, 1997). Greening *et al.* (2019) showed the pivotal role in rumen fermentation of flavin-based electron confurcation and bifurcation (Buckel and Thauer, 2013, 2018a, 2018b) in the formation and incorporation of dihydrogen.

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 favours reoxidation of reduced cofactors, and thus allows fermentation to continue (Wolin, Miller and Stewart, 1997). Elegant experiments demonstrating how pure cultures of rumen microorganisms stopped or decreased the production of rumen fermentation intermediates such as dihydrogen, formate and ethanol as final fermentation products, when cocultivated with methanogens or other hydrogenotrophs, illustrated the role of the interspecies transfer of dihydrogen in shaping rumen fermentation (e.g. Marvin-Sikkema *et al.*, 1990). The close proximity between hydrogenogens and hydrogenotrophs favours the kinetics of dihydrogen transfer and its rapid utilization, as in microbial biofilms (Leng, 2014) and in protozoal-methanogen symbiosis (Newbold *et al.*, 2015).

Methane is generally 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, a more reduced than fermented substrate, constitutes another electron sink. Mineral electron acceptors such as nitrate and sulfate thermodynamically outcompete methanogenesis, yet their availability in 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 considered thermodynamically unfeasible in the rumen (Ungerfeld and Kohn, 2006), and yet more recent findings have revealed 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), in which formate is the second electron donor in importance (Hungate *et al.*, 1970). Formate must be oxidized to CO_2 and dihydrogen by archaea 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 also uses as substrates methanol, methylamines and methylated sulphur compounds, which can accumulate in the rumen following the ingestion of some diets, for example those containing pectin (Söllinger *et al.*, 2018).

The production of acetate and, to a lesser extent, butyrate lead to the net release of metabolic hydrogen and the resulting formation of dihydrogen. Acetate production is thus associated with methanogenesis. The replacement of roughages with concentrates typically decreases CH_4 formed per unit of fermented organic matter (OM) – not necessarily the total amount of CH_4 produced, as the intake of rumen-fermented OM often increases when feeding concentrates – and shifts rumen fermentation from acetate to propionate. A mechanism explaining this fermentation shift has been proposed by Janssen (2010), drawing on the Monod model of microbial growth, which relates actual and theoretical maximal microbial growth rate to the concentration of the substrate most limiting for the growth of microorganisms – dihydrogen in the case of most rumen methanogens, since they are hydrogenotrophs. As concentrates form an increasing part of a ruminant's diet, the passage rate increases, resulting in an increased growth rate of methanogens that are not washed out and continue to produce CH_4. A greater growth rate in turn leads to an elevated dihydrogen concentration, according to the Monod function. Likewise, feeding ruminants concentrates generally causes rapid fermentation and rumen pH decreases, resulting in decreased theoretical maximal growth rates of methanogens. Consequently, there is an increase in the concentration of dihydrogen, which in turn thermodynamically inhibits acetate and favours propionate production (Janssen, 2010).

Similarly, when methanogenesis is inhibited by chemical compounds, the maximal growth rate of methanogens decreases and dihydrogen accumulates (Janssen, 2010). In this regard, it is worth bearing in mind that the inhibition of methanogenesis is not an isolated intervention in rumen fermentation and that it will have profound consequences on the flows of metabolic hydrogen. When strategies to mitigate CH_4 emissions through the use of chemical inhibitors are considered, inhibiting CH_4 production should therefore not be viewed as the sole objective of the intervention; another approach, one which would redirect 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 channel part of dihydrogen typically accumulating when methanogenesis is inhibited to VFA production by adding electron acceptors that are intermediates of VFA formation or specific microbial additives (Ungerfeld, 2020).

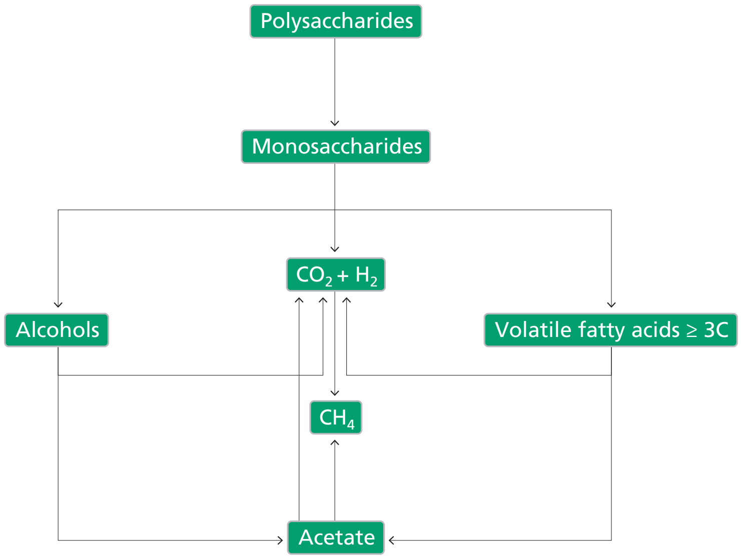
1.2.2 Manure

Anaerobic digestion (AD) of animal manure and other organic wastes into CO_2 and CH_4 also involves complex metabolic interactions between microbial groups. While the physicochemical principles controlling both systems are the same, conditions such as 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 into 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, provided they are present at high concentration.

Hydrolysis of carbohydrates is a relatively slow process carried out by a very diverse group of bacteria. In biodigesters fed with cattle manure, bacteria fermenting hydrolysed monomers predominantly belong to the genera *Clostridium*, *Eubacterium* and *Bacteroides* (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 chemical processes thermodynamically feasible (Schink, 2002). Imbalance between fermentation and syntrophy can result in increased concentration of VFA and acidification, which inhibits fermentation. For the oxidation of VFA longer than two carbons to acetate to be thermodynamically feasible, the concentration of dihydrogen has to be kept very low, which requires functional methanogenesis (Schink, 2002; Ferry, 2011; Alvarado *et al.*, 2014). A low concentration of dihydrogen (as well as a 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).

Figure 2

Simplified scheme of the main pathways of anaerobic digestion



Source: Adapted from Ferry, J.G. 2015. Acetate metabolism in anaerobes from the domain Archaea. Life, 5(2): 1454–1471. https://doi.org/10.3390/life5021454

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. Methanobacteriales and Methanomicrobiales use dihydrogen as an electron donor, along with formate, ethanol and isopropanol in some species. Except for the genus *Methanosphaera* (Methanobacteriales), Methanobacteriales and Methanomicrobiales cannot use acetate as a 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 various substrates have their last step in common, the reduction of methyl-coenzyme M to CH_4. In the 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 with 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 a 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, 2002). 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 a higher yield and achieve a faster rate of growth to match rumen outflow rates.

1.2.3 Soil

Rice is a major crop for the human population globally, one for which there is an increasing demand. It is therefore important to understand the mechanisms behind CH_4 production and oxidation in rice soils to devise CH_4 mitigation strategies for the cultivation of this crop (Liesack, Schnell and Revsbech, 2000). Knowing how to control major flows of carbon and metabolic hydrogen can help design more appropriate interventions and cultivation practices aimed at mitigating 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 saturation level. The presence of oxygen and other electron acceptors, carbon substrates, water, redox potential and pH has an impact on methane production in soils. This section will focus mainly on rice field soils, which are seasonally flooded, alternating oxidizing and reducing conditions. Rice paddies are estimated to contribute to 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, the degradation of polymers, mainly polysaccharides such as cellulose and hemicellulose, is the first step to releasing fermentable monomers. Rice straw is ploughed under the soil after harvest, thus setting the degradation of polysaccharides in motion (Liesack, Schnell and Revsbech, 2000). Between 80 and 90 percent of rice straw is degraded within the first growth season (Conrad, 2020a). The amount of CH_4 emitted from rice straw – either after soil incorporation or from open field burning – will depend on the chosen type of rice management on the farm (see Section 7). Moreover, OM provided by the roots of rice plants is always the primary carbon source of CH_4 produced in rice field soil (Kimura, Murase and Lu, 2004).

After rice paddies are flooded, 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 such as dihydrogen and acetate at too low a 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 the 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 which is added to the soil as a fertilizer. Nitrate can be reduced to dinitrogen, nitrite, nitrous oxide (another greenhouse gas which has higher GWP than CO_2) or ammonia. In rice soils, iron content is usually high enough to prevent the accumulation of hydrogen sulfide (Liesack, Schnell and Revsbech, 2000). Importantly, reduced inorganic ions are reoxidized when the soil is aerated, or if a strong inorganic electron acceptor is added to the soil (for example, adding nitrate will regenerate ferric iron and sulfate; Conrad, 2020b). Also, the reoxidation 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, Schnell and Revsbech, 2000).

Once all of the inorganic electron acceptors are reduced to such an extent that each process reaches a 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, MacCubbin and Hodson, 1984). The 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. While the 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 is being produced (Liesack, Schnell and Revsbech, 2000; Conrad, 1999, 2020a, 2020b).

As in other anaerobic environments, dihydrogen turnover is very high. A low concentration of dihydrogen approaching the thermodynamic threshold of methanogenesis makes dihydrogen-releasing reactions thermodynamically feasible (Conrad, 1999). Microorganisms involved in the different phases of the OM oxidation in anaerobic degradation must accommodate their activities to the thermodynamic feasibility of chemical processes, but they contribute significantly to accelerating the kinetics of those processes. Theoretically, the anaerobic degradation of cellulose would result in equimolar amounts of CH_4 and CO_2 as well as 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 also further influence the stoichiometry of the final products of anaerobic digestion in soils (Conrad, 1999, 2020a, 2020b).

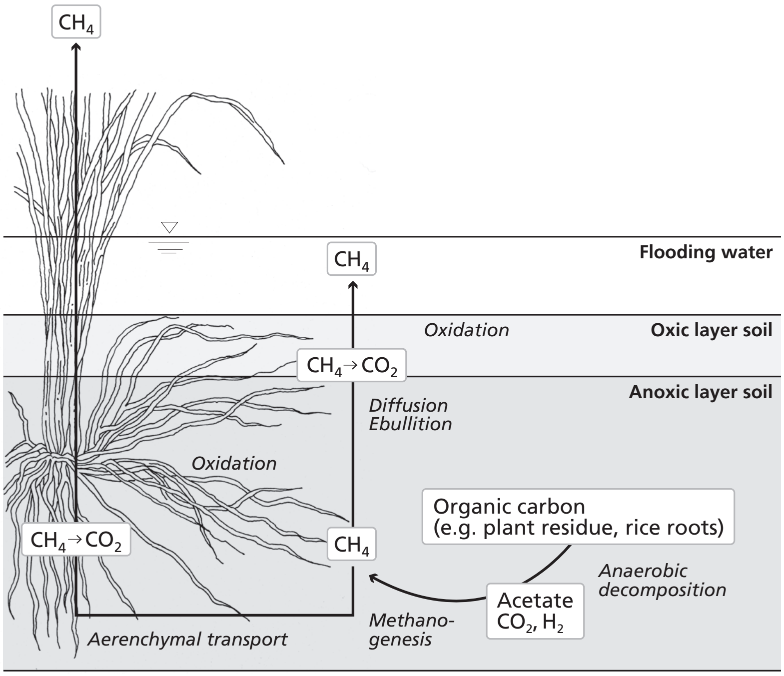
Temperature influences the predominant substrates for methanogenesis, with a relative increase in acetate and a decrease in dihydrogen at low temperatures (Chin and Conrad, 1995; Conrad, 2020a). Furthermore, at low temperatures, acetate production increases relative to CO_2 and, as a consequence of reductive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

Soil methanogenesis is strongly inhibited by oxygen, and yet soil methanogens have evolved to adapt to succeeding events of flooding and desiccation, and can tolerate the presence of oxygen, despite not forming spores or cysts. Methanogens have generally been found to decline, but not disappear, with soil desiccation (Conrad, 2020b). Methanogenesis has been noted to occur even in anoxic microniches in oxic soils. Methanogens belonging to the 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 found in such environments 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 in 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). And yet aerobic methanotrophic bacteria situated in oxic/anoxic interphases can oxidize up to 80 percent of soil-produced CH_4 before it is released into 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 into the atmosphere as bubbles and above all through the plants' aerenchyma (Liesack, Schnell and Revsbech, 2000).

Anaerobic CH_4 oxidation, conducted by both bacteria and archaea, can also eliminate substantial amounts of CH_4 formed in some soils before it reaches the atmosphere. Methane oxidation coupled with sulfate reduction is important in marine sediments, but it may also be important in terrestrial soils subjected to cycles of sulphur 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



Source: Authors' own elaboration.

1.3 METHANE EMISSIONS DURING THE 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 manure 0.2 Gt CO_2eq; the 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 CH_4 (Steinfeld *et al.*, 2006). In the United States, the Environmental Protection Agency (EPA, 2006) estimated the CH_4 emission from manure at between 470 and 523 megatonnes (Mt) CO_2eq per year. For the European Union, manure CH_4 emissions represent 44 Mt CO_2eq (Eurostat, 2018).

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

The magnitude of CH_4 emissions generated during storage of manure mainly depend on the duration of manure storage, the storage system used, temperature and manure composition (Dennehy *et al.*, 2017; Philippe and Nicks, 2015). For instance, Petersen *et al.* (2013b) found that the cumulative CH_4 emissions from stored pig manure in summer were over 100 times greater than those in winter; however, this depends on the geographical context. Emissions of N_2O typically range from less than 1 to 4.3 percent of the total nitrogen 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 THE APPLICATION OF MANURE

While the vast majority 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 dry matter (DM) content, the application technique and the 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, with varying DM contents, spread by mimicking trailing-shoe and splash-plate applications. The study varied the DM contents by mixing different ratios of faeces and urine. Although ammonia (NH_3) volitalization losses were substantially increased on plots spread with slurry, the cumulative direct N_2O emissions and corresponding emission factors were significantly higher when calcium ammonium nitrate had been applied. In terms of GHG field balance, the potential decrease in indirect N_2O emissions, calculated from a reduction of NH_3 volitalization losses using trailing-shoe as opposed to splash-plate methods, 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 in mitigating 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, resulting in an overall positive effect on reactive nitrogen losses and in agronomic benefits for farmers. However, a recent analysis of a large dataset of CH_4 fluxes from agricultural sites across Ireland and the United Kingdom of Great Britain and Northern Ireland indicate that these soils are small net emitters of CH_4 rather than sinks, with fluxes occurring following animal manure applications especially (Cowan *et al.*, 2021).

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 to manure) can range from less than 0.1 to 3 percent. Higher emissions (of 7.3 to 13.9 percent) have been measured during land application of pig slurry (Velthof, Kuikman and Oenema, 2003). The range in N_2O emission factors following slurry and solid manure applications reflects differences in soil type, soil conditions (temperature or water-filled pore space), manure composition (NH_4 +-N, C content and form) and the measurement period.

Other investigators have examined the potential CH_4 oxidizing capacity of soils, particularly as regards removing CH_4 emissions from manure and digestate slurry. The magnitude of the effect depends on the soil's chemical and physical properties 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

According to a review of CH_4 emission factors by O'Brien and Shalloo (2016), several countries recognize that CH_4 emissions from cattle and livestock manure are linked to other GHG emissions from manure, such as N_2O. Some countries use a process-based model, one that simultaneously quantifies GHG and NH_3 emissions from livestock, for consistency. Reduced NH_3 losses from manure spreading are likely to increase N availability in agricultural soils and this, in turn, may affect the production and release of N_2O (Brink, Kroeze and Klimont, 2001). As N_2O is a much more potent GHG (IPCC, 2007), this could be regarded as pollution swapping (Stevens and Quinton, 2009) since 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 across the whole manure management chain. In their meta-analysis, Hou, Velthof and Oenema (2015) found that lowering the crude protein (CP) content of feed and acidifying slurry are strategies that consistently reduce NH_3 and GHG emissions in the whole chain.

1.6 SPATIO-TEMPORAL VARIATION IN METHANE EMISSIONS

Reliable, high-resolution spatio-temporal inventories of CH_4 emission from livestock production systems are required to ensure an 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 the vectoral zonation of production systems (for both pastoral and mixed systems). As a consequence of the temporal and spatial variability of emissions from livestock housing and manure management, measurements and monitoring that cover both the daily and seasonal variations in emissions need to occur over longer periods of time 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 emphasize the large spatial variations of emissions due to differences in soil type, climatic parameters and water conditions, or indeed varied soil fertilization as well as manure management practices and composition. A detailed discussion on quantification methods is presented in Section 2. In addition, the conditions for manure are far less well controlled than in the case of enteric emissions, where the physiological regulatory mechanisms of the ruminant are 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, Abdullah and Greetham (2018) point out that, historically, livestock animals functioned as bioprocessors, converting materials inedible by humans into nutritious meat, eggs and milk. The authors consider that contemporary treatment technologies can assist in converting food into safe, nutritious and value-added feed products, instead of wasting it, and deem recovering unconsumed food for animal feeding to be a viable solution that simultaneously addresses the reduction of food waste, food security, resource conservation, and pollution and climate-change mitigation.

The reduction and sustainable management of food waste are fundamental tenets of the circular bioeconomy concept. Globally, around 1.3×10^9 tonnes of food waste are disposed of in landfills (Hao, Karthikeyan and Heimann, 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, the 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 (Mak *et al.*, 2020). The recycling of food waste that could not be reduced by a more efficient food supply chain and informed consumer behaviour provides employment, reduces GHG emissions, decreases disposal costs, mitigates the negative environmental impacts and supports sustainable waste management practices that fall under the biocircular 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 it has not yet been fully applied due to a number of technical and social challenges (Xu *et al.*, 2018). Indeed, in the United States, less than 2 percent of food waste is anaerobically digested. The management of food waste through biological processing is a more environmentally sustainable approach than thermo-chemical conversion or landfilling. The composition and CH_4 generating the potential of some common food waste streams has been summarized by Xu *et al.* (2018). However, the composition, and the physico-chemical and biological characteristics of food waste, can affect the overall biological process when it comes to product yield and degradation rate. The pretreatment (i.e. grinding or drying) of food waste ahead of anaerobic digestion has been proposed in order to overcome this major bottleneck in the system. Codigesting food waste with manure, sewage sludge and lignocellulosic biomass could be beneficial due to the dilution of toxic chemicals, enhanced balance of nutrients and synergistic effects of microorganisms.

There are major variations in legislation addressing the inclusion of unconsumed human food into the diets of livestock between different countries and continents. These variations range from reducing the food waste entering landfills to fears over the health implications of its 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 reach minimum thresholds of GHG emissions savings to count towards renewable targets and 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 by-products which, following biologically based processing, are converted to electrical, heat and power energies through a relatively easy-to-manage process in small industrial and agricultural units (Antoni, Zverlov and Schwarz, 2007). Digester designs vary widely in size, function and operational parameters, and have been reviewed in the context of different production systems by Hristov *et al.* (2013b). While strongly recommending the use of anaerobic manure digesters as a CH_4-mitigation strategy for the agriculture sector, Gerber *et al.* (2013b) caution that careful management is necessary, so that they do not become emitters of CH_4 into the atmosphere. The authors suggest that the 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 the 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, Hellweg and Guillen-Gosalbez, 2014). This is largely due to the emissions that are mitigated through 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 prioritize the digestion of manures to maximize GHG mitigation (Styles, Dominguez and Chadwick, 2016). While manures have a lower biomethane potential in comparison with other feedstocks, it has been suggested that focusing on smaller biogas plants with lower energy conversion efficiency may be preferable as a strategy for waste management to renewable energy generation, where there are more efficient alternatives in terms of cost and land requirements, such as wind and solar energy (Styles, Dominguez and Chadwick, 2016).

The methodology applied by the Joint Research Centre (JRC) of the European Commission to calculate the GHG emissions associated with bioenergy pathways outlined in the Renewable Energy Directive is a simplified attributional life cycle assessment (LCA) for Giuntoli *et al.* (2017). According to the JRC, biogas produced from manure can receive emission credits for emissions avoided from the traditional management of manure, including CH_4 and N_2O, provided that manure is not stored for too long. Using manure in AD systems is considered an improved agricultural management technique and the emissions avoided through the management of the raw manure are credited to the bioenergy pathway. The value of the credit is equal to -45 g CO_2eq per MJ of manure used (Giuntoli *et al.*, 2017). However, the JRC recognizes 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 digesters utilize the energetic potential contained in the manure for the production of heat and electricity, which reduces N_2O emission during the treatment through a relatively closed system, and results in biogas digestate that makes for a valuable fertilizer still containing most of the nitrogen (Kreidenweis *et al.*, 2021). However, comparisons between open and closed digestion systems of ammonia emissions are lacking in the literature. Existing publications compare digested to raw manure, which is not the right comparison for evaluating the overall effect of a digester. Closed digesters do not address the concerns for ammonia emission levels in neighbouring communities that tend to be near large animal feeding operations to a greater extent than open systems.

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, Pipatti and Hashimoto, 2008), and for about 12 percent of total global anthropogenic CH_4 emissions for the 2008–2017 period. Bakkaloglu *et al.* (2021) suggest that CH_4 emissions resulting from biogas generation may be between 0.4 and 3.8 percent of the total gas production, and could account for 1.9 percent of the total CH_4 emissions in the United Kingdom, 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 utilization. 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 than those from the smaller facilities. In general, CH_4 losses were higher in wastewater treatment biogas plants (7.5 percent on average) compared to agricultural biogas plants (2.4 percent on 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 through the chemical degradation of CH_4 by hydroxyl (OH) and chlorine (Cl) radicals in the troposphere and stratosphere (IPCC, 2007) and is responsible for 90 to 96 percent of global CH_4 sink (Wuebbles and Hayhoe, 2002; Shukla, Pandey and Mishra, 2013; Saunois *et al.*, 2019), equivalent to 550 Tg per year. The soil accounts for about 4 to 10 percent of the CH_4 degraded (Born, Dorr and Levin, 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 per year (Shukla, Pandey and Mishra, 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 per year (IPCC, 2001; Knief, Lipski and Dunfield, 2003; Tian *et al.*, 2016) with an uncertainty of 11 Tg to 49 Tg per 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, Pandey and Mishra, 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, Iversen and Henriksen, 1997).

Among upland soils, forest soils are the most efficient CH_4 sink in both temperate and tropical regions (Henckel *et al.*, 2000; Steinkamp, Butterbach-Bahl and Papen, 2001; Singh *et al.*, 1997), with a global annual average uptake rate of 5.7 kg, 3.3 kg and 2.64 kg CH_4/ha for temperate, tropical and boreal forest biomes, respectively (Dutaur and Verchot, 2007). Grasslands, shrub lands, and steppe and savanna biomes have an average annual uptake of 2.32 kg, 2.25 kg and 1.49 kg CH_4/ha (Dutaur and Verchot, 2007). Cropland and desert have the lowest uptake rate with an annual mean rate of 1.23 kg 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 of 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, Van Cleemput and Villaralvo, 1997). 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 CH_4 consumption but insufficient soil moisture reduces methanotroph activity.

Changing climate and climatic factors, particularly seasonal precipitation variations in semi-arid regions and drylands, likewise affect the soil CH_4 sink capacity, directly and indirectly. Soil OM also increases CH_4 consumption through both pathways – pore space and pore size increase with increasing soil OM, while carbon and nutrients in soil OM increase methanotroph numbers (Gatica *et al.*, 2020; Tang *et al.*, 2019b). Physical factors include 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, Pandey and Mishra, 2013). The land degradation that reduces soil OM and increases soil bulk density consequently reduces the soil sink capacity, whereas restoration increases the sink; however, the increase in sink capacity with restoration is slower than the loss of sink capacity with degradation (Wu *et al.*, 2020). The addition of inorganic nitrogen 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 mixed with organic amendments such as manure, nitrogen has a lesser effect on CH_4 uptake. Pesticides and herbicides, metal pollution and land use patterns can also have a significant effect on CH_4 oxidation and methanotrophic community (Boeckx, Van Cleemput and Meyer, 1998; Priemé and Ekelund, 2001; Shukla, Pandey and Mishra, 2013).

2.1.2 Land management effects on the soil methane sink

I. Pasture

A global meta-analysis showed that adding nitrogen to pastures reduced the soil methane sink capacity by more than 10 percent, but that the application of phosphorus with the nitrogen roughly halved that reduction (Zhang, L. *et al.*, 2020). The livestock stocking rate, has an important but yet to be quantified effect on CH_4 uptake. Compared with moderate or light grazing, heavy grazing intensity reduces the sink capacity by 12 percent globally, due to the effect of heavy grazing pressure which reduces plant productivity and soil OM while increasing soil bulk density from hoof action (Tang *et al.*, 2019b). For low productivity grazing lands with low livestock stocking rates, the soil sink can be an important part of the grazing system's 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). There is much interest in adaptive multi-paddock grazing, but careful analysis of the literature reveals a failure to control key factors, particularly stocking rates. Further research is needed to ensure that studies control stocking rates, repeat soil carbon and methane flux measurements, and collect other pertinent field data.

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, Xu and Beese, 2003). The effects of tree species 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 (Gatica *et al.*, 2020).

III. Cropland

Cropland typically has nitrogen addition and 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 the tillage system (Venterea, Burger and Spokas, 2005; Jacinthe and Lal, 2005; Kessavalou *et al.*, 1998), biochar addition (Cong, Meng and Ying, 2018) or cover crops (Singh, Abagandura and Kumar, 2020) on CH_4 uptake.

IV. Agroforestry

Because soil under trees typically has a greater CH_4 uptake rate, the treed portion of the land has a higher sink than the untreed cropland (Amadi, Van Rees and Farrell, 2016). In an experiment carried out 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, Chará and Barahona, 2019). In addition, the carbon sequestered in the shrubs and/or trees of silvopastoral systems provides an opportunity to offset some (Monjardino, Revell and Pannell, 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

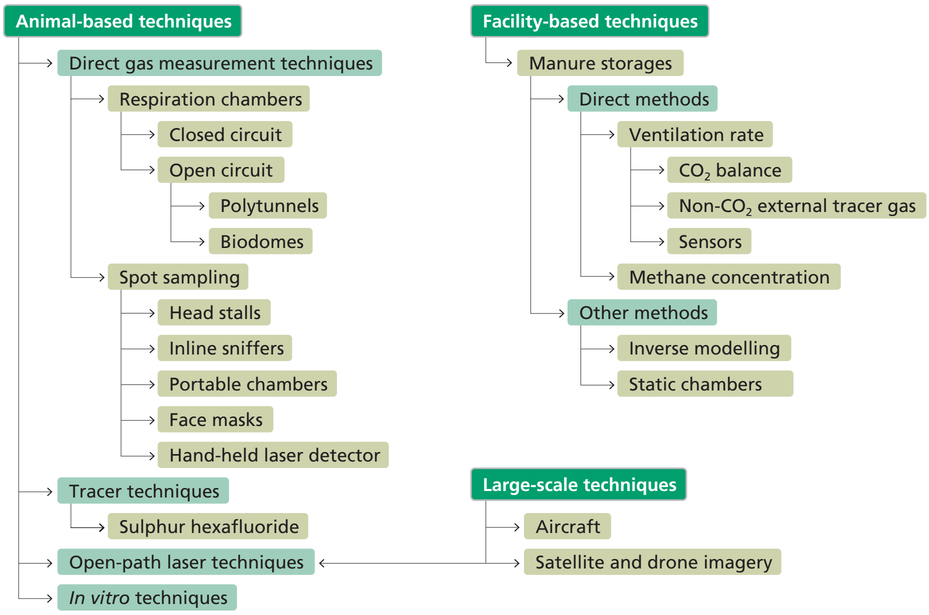
3. Measurement

Figure 4 depicts the flow chart indicating the categorization of current techniques used to determine methane emissions at the animal, facility and large-scale levels.

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 outlines the 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 exacerbate CH_4 measurements if the conditions are not consistent with the original assumptions. For instance, some techniques are more suitable for grazing animals (e.g. sulphur 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 the most critical measurements to derive a CH_4 emissions rate.

Figure 4
A schematic flow chart of current techniques used to determine methane emissions at the animal, facility and large-scale levels



Source: Adapted fromTedeschi, L.O., Abdalla, A.L., Álvarez, C., Anuga, S.W., Arango, J., Beauchemin, K.A., Becquet, P., Berndt, A., Burns, R., De Camillis, C., Chará, J., Echazarreta, J.M., Hassouna, M., Kenny, D., Mathot, M., Mauricio, R.M., McClelland, S.C., Niu, M., Onyango, A.A., Parajuli, R., Pereira, L.G.R., del Prado, A., Tieri, M.P., Uwizeye, A. & Kebreab, E. 2022. Quantification of methane emitted by ruminants: A review of methods. *Journal of Animal Science*, 100(7): 1–22. https://doi.org/10.1093/jas/skac197

Table 1. Characteristics of different techniques used to measure methane

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; need 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; require 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 tracers and, therefore, relies on spot concentration measurements; high contact with animal, which can disrupt normal behaviour; 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 rather than 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/bar/ building		Research	Information about many animals; data produced in a 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/region		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	They 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; detects CO_2 and CH_4 concurrently	

Source: Based on Hill, J., McSweeney, C., Wright, A.-D.G., Bishop-Hurley, G. & Kalantarzadeh, K. 2016. Measuring methane production from ruminants. *Trends in Biotechnology*, 34(1): 26–35. https://doi.org/10.1016/j.tibtech.2015.10.004

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 the gas exchange of mainly O_2, CO_2 and CH_4, either using open-circuit chambers that analyse the composition of inflow and outflow air or closed-circuit chambers that analyse the composition of air accumulated over some time (Johnson and Johnson, 1995). A limitation of respiration chambers is that animals may not exhibit normal behaviours – they might, for instance, decrease their feed consumption – thus resulting in an underestimation of actual CH_4 emissions when compared to free-ranging animals under farm conditions (Huhtanen, Ramin and Hristov, 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 (fewer than 20). Furthermore, emissions from manure, if accumulated in the chamber, must be accounted for (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 they employ locally available materials for construction and the air-conditioning systems that are simpler than those described for other open-circuit systems. The system is typically located in the daily environment of cows (Canul Solis *et al.*, 2017; Hellwing *et al.*, 2012) or sheep (Abdalla *et al.*, 2012). It 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 in air sampled from the outlet, using an infrared analyser 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 polyvinyl chloride 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, Chang and Tomkins, 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

Headstalls, also referred to as Automated Head-Chamber Systems (AHCS) (Hristov *et al.*, 2015b) (e.g. GreenFeed Emission Monitoring™ system), and sniffers (e.g. GASMET 4030 system) are based on spot sampling of eructated and exhaled gases from the animals' mouth and nostrils. Sniffers only measure concentration. Headstalls are usually programmed to deliver a small amount of feed to lure animals to insert 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 the observed mean) for dairy cows (Huhtanen, Ramin and Hristov, 2019). The adequacy of headstalls (e.g. GreenFeed) in measuring CH_4 highly depends on the daily frequency of the visits by the animal (within-day variation), animal behaviour (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.* (2015b) proposed sampling eight times during a 24-hour feeding cycle, staggered in time over 3 days. Arbre *et al.* (2016) measured daily values and obtained a repeatability of 70 percent in 17 days; they 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 that he increased 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 depending on animal size. The system is suitable for pasture (i.e. grazing conditions) and indoor 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 (three 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.*, 2015b). Benefits include its lower cost compared to respiration chambers but, similar to sniffers and gas tracers, it does not consider CH_4 emissions 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 (Bell *et al.*, 2014) is lower than that of respiration chambers (Yan *et al.*, 2010), likely because CH_4 concentration depends on the distance between the sniffer and the 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). Made with plexiglass on the sides and top, these are bottomless boxes that are lowered down on animals and sealed (Thompson and Rowntree, 2020). There are three sampling ports on the top of these boxes, which allow to follow gas accumulation in time. Comparisons with respiration chamber measurements have indicated moderate correlations (up to 0.6) for up to two-hour sampling durations (Goopy *et al.*, 2011; Goopy *et al.*, 2015). One limitation of portable accumulation chambers is that measurements are short term, and represent only a portion of the 24-hour emission cycle.

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 emissions from the breath of ruminant animals. Some recovery tests may be needed. Methane has two strong groups of absorption lines, centred at 3.3 micrometres (v3 band) and 7.6 micrometres (v4 band). Most laser-based devices operate at near-infrared wavelengths, limited to below 2.2 micrometres. The most robust absorption band of CH_4 is located at 1.64 to 1.70 micrometres (2v3 band). This corresponds to the single-mode, single-frequency emission wavelength of indium gallium arsenide-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 metres, it does not disturb animal behaviour. The instrument accounts for the thickness of any CH_4 plumes, and the result is expressed in terms of CH_4 concentration. The LMD thus enables real-time measurement with a fast response. Furthermore, the LMD can segregate the CH_4 concentration from dairy cows performing different physiological activities, such as ruminating, feeding or sleeping. As the LMD only spot samples an animal's breath, attempts have been made to use these measurements to calculate total emissions (g per day). One study reported that the CH_4 measurement with the LMD had a strong agreement with measurements in respiration chambers (r = 0.8) (Chagunda and Yan, 2011). One of the challenges of using the 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 respond to this challenge, a threshold value for separating the two events is being tested. Applying this approach to grazing animals is also challenging 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 is negatively correlated with CH_4 concentration (r = -0.41). An additional limitation is that the device should be at the right distance from the animal (Sorg, 2022) to avoid contamination from a neighbouring animal. The LMD instrument is relatively novel when it comes to ruminant animals, and extensive studies will be required to determine the measurements' repeatability (Chagunda, 2013), which could be used to develop standard protocols for data measurement and analysis (Sorg, 2022). That said, such techniques may help improve the accuracy of the 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. 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 the SF_6 method and respiration chambers can be greater than 10 percent (Storm *et al.*, 2012; Ramírez-Restrepo *et al.*, 2020), likely on account of the inconsistent release of SF_6 from the permeation tubes deposited within the rumen, variations in the animal's breath collection efficiency, interruption of normal behaviour due to the sampling equipment harness, and inability to collect CH_4 emissions produced in the hindgut (Lassey, 2007). To improve the predictability of the SF_6 method, some modifications have been proposed, such as the continuous collection at a constant rate for 24 hours 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 three-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 for 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, Clark and Muetzel, 2016) systems. However, Hristov *et al.* (2016) noted that SF_6 is better suited for open spaces or well-ventilated buildings because, in poorly ventilated buildings, background CH_4 could affect the interpretation of results. The technique cannot be used close to other CH_4 sources (e.g. slurry, manure, other animals and wet areas) and SF_6 sources (e.g. electricity transformers and industrial sites) (Jonker and Waghorn, eds., 2020). The SF_6 technique is relatively inexpensive, but only one animal per unit can be measured. An adequate calibration of the release rate of the tracer gas from the permeation tube should be conducted in advance of placement in the rumen, and the experiment should be carried out soon after this calibration since there is a decrease in the permeation rate of the tube. Adjustments for the changing permeation rate should be performed in long-term trials (Jonker and Waghorn, eds., 2020).

Madsen *et al.* (2010) proposed predicting CH_4 from CO_2 calculated based on body weight, energy-corrected milk yield and the days of pregnancy, provided that the efficient energy use for maintenance and production is constant for dairy cows. Individual CH_4 concentration was recorded for three days in an automatic milking system with a portable air sampler and analyser unit, based on Fourier transform infrared detection and using CO_2 as a tracer gas (Lassen, Løvendahl and Madsen, 2012). Air was analysed every 20 seconds when the animals were milked, and the ratio between CH_4 and CO_2 was used to measure CH_4 emissions. The repeatability of the measurement (CH_4:CO_2 ratio) was 0.39 and 0.34 for Holstein and Jersey cows, respectively (Lassen, Løvendahl and Madsen, 2012). These results suggested that the CH_4:CO_2 ratio could be used for genetic evaluations of dairy cows (Lassen, Løvendahl and Madsen, 2012). Unfortunately, efficient cows (i.e. those yielding more milk per feed consumed) produce less heat and consequently less CO_2 per unit of metabolic body weight and energy-corrected milk; thus, there is a risk of overestimating their CH_4 production. Hence, the genetic selection for low CH_4 emitters using CO_2 production rate as a reference will favour 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 by adopting an "inverse dispersion" approach (McGinn *et al.*, 2006). The technique has been used for CH_4 (McGinn *et al.*, 2006) and NH_3 (McGinn *et al.*, 2007) emissions. Validation assays have shown its limitations with regards to the time of data collection (McGinn *et al.*, 2006, 2008). The open-path laser technique has been updated by having different analysers and atmospheric parameters integrated into a flying platform, thus showing more reliable and promising results (Hacker *et al.*, 2016). These authors indicated that with the revised 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 the open-path laser technique with an atmospheric dispersion model for grazing animals to animals with respiration chambers consuming freshly cut *Chloris gayana*. Daily estimates were of 136 g and 114 g CH_4, respectively, and the authors considered that further comparisons using different forages and herds were needed. Subsequently, Tomkins and Charmley (2015) tested the open-path laser technique relying on the expected behaviour of herding animals around water points during the day. The measurement was conducted over 4 to 16 days for 78 hours, with data collected every 10 minutes. Historical meteorological data for wind direction determined the physical arrangement of equipment at each tested site. The data needed to be filtered based on environmental conditions, including 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 works well when used on aggregations of grazing cattle for 7 to 8 hours per day over 7 to 14 days, and that it is also an 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 in mitigating CH_4 production under standardized conditions (Yáñez-Ruiz *et al.*, 2016). Due to the complexity and cost of methodologies for evaluating enteric CH_4 emissions directly from animals, the possibility of obtaining results through *in vitro* systems would be an alternative, mainly in providing an initial screening of a larger number of samples with different methanogenesis-reducing options, such as tannins, plant secondary metabolites and 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 the 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 a rumen inoculum (strained rumen fluid), a buffer to avoid significant pH variation, and media to provide necessary nutrients to rumen microbiota. The CH_4 production is usually expressed per incubated unit or on a digested DM or OM basis.

3.2 FACILITY-BASED TECHNIQUES

3.2.1 Manure storage

Three different approaches to the quantification of manure CH_4 emissions from animal housing facilities are commonly used: direct measurement methods, inverse modelling (manure and animals) and the chamber technique (manure emissions) (Hassouna and Eglin, 2016). At the barn level, the removal of cattle to estimate emissions from manure has been performed (Edouard *et al.*, 2019; Mathot *et al.*, 2012, 2016). The measurement methods that exist today were developed for scientific purposes, which is why some of these 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. Since there is no international standardization of these methods to date, it is yet to be 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, which would be easier to implement, less expensive (Robin *et al.*, 2010; Hassouna *et al.*, 2010) and adaptable to different contexts, so as to meet objectives such as the certification of emission reductions in real conditions or the quantification of emission factors taking into account intracategory 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 and the in-house CH_4 concentration, minus the background concentration (Hassouna *et al.*, 2021). A methodology to quantify the uncertainty of aerial emissions for the direct methods has been outlined by

Gates *et al.* (2009), one which combines the statistical uncertainty of the emissions concentration measurement and the ventilation rate measurement. Measurements associated with the ventilation rate have been demonstrated to be the major contributor to the emissions rate uncertainty when relying on the direct methods.

3.2.1.1.1 Ventilation rate

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

i. Carbon dioxide balance

For this method (Barreto-Mendes *et al.*, 2014; Liu, Powers and Harmon, 2016), which uses CO_2 as the tracer gas, the main hypothesis is that the 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). 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 the ventilation rate in an 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, the CO_2 production per heat unit and animal activity. In many studies, these parameters are calculated with models given by the International Commission of Agricultural Engineering (CIGR, 2002). According to Zhang, Pedersen and Kai (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 the daily variation of CO_2 production that depends on animal activity should be considered to have an accurate estimation of the daily variation of the 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 in 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 not produced in the building. The barn ventilation rate is calculated using the tracer gas injection rate and the tracer concentration gradient, assuming the 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 using a sensor with a low detection limit. In livestock buildings, this method could be implemented by means of two different approaches: a constant dosing of the tracer gas or with spot injections (concentration decay method). For the constant injection method, the tracer gas is dosed into the barn or, more generally, close to an emitting area or point source. This tracer gas mimics the dynamic flow and dilution of CH_4 or other target gases 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 unit until the desired threshold is achieved and a uniform distribution of the tracer gas is reached. Then the injection is stopped, and the decrease of tracer gas concentration is monitored for a given period to calculate the barn's 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 such as 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 yielded similar results, with the CO_2 mass balance method being quantitatively 10 to 12 percent lower in the estimate of emissions compared to the SF_6 tracer methods.

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 VR of the house can be estimated by summing up 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 of 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). For 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 to be 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 of the barn. Most of the time, the same device is implemented for both measurements, which implies 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 techniques with a very selective detection system (such as lasers) are preferred for continuous measurements. Hassouna *et al.* (2013) highlighted interference problems with non-selective methods such as the (commonly used) photoacoustic infrared spectroscopy that can lead to overestimation CH_4 emissions. Gas chromatography can also be implemented, but the continuous measurement is more complicated on commercial farms because regular calibration is required. Nevertheless, not all sensors and gas analysers 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 require 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 draw on chamber and micrometeorological methods that come in manifold designs and have 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 labour. These techniques also rely on the deployment of different gas analysers 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; 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 at given time intervals (e.g. at 0, 10, 20, 30 and up to 45 minutes), depending on the rate of CH_4 evolution, the kinds of manual chambers employed or the type of equations used to derive the emission rates (Tiwari *et al.*, 2015). For non-CO_2 trace gases like N_2O and CH_4, longer time intervals 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 CH_4. The enclosure interval itself is clearly limited during daytime to protect the plants from the stress caused by increasing temperature and CO_2 depletion. Alternatively, to limit these, closed chambers may also be exposed at nighttime (Wassmann, 2019). Although emission rates are lower at night, the diurnal patterns may be taken into account for intercomparisons of varieties and treatments. One or two small fans are typically installed inside the chamber to thoroughly mix the atmospheric gases (Tiwari *et al.*, 2015). Gas samples can be collected via syringe and transferred into vials for offsite analysis (Sass *et al.*, 1990, 1991) or *in situ* analysis if using a dynamic system with automated sampling devices (Wassmann, Papen and Rennenberg, 1993; Wassmann *et al.* 2000; Hall, Winters and Rogers, 2014). The obvious advantages of dynamic systems are the high temporal resolutions and seamless observation periods, as in when the emission measurements encompass two-hour intervals over entire 24-cycles and stretch over the entire cropping season (Wassmann, Neue and Lantin, 2000). These direct measurement systems can be valuable in combination with modelling approaches, namely the validation of Tier 2 regional equations created using such measurements or simulation models as DayCent or Landscape DNDC applied to the specific conditions of rice fields (Weller *et al.*, 2016; Kraus *et al.*, 2016; Janz *et al.*, 2019). It is vital to note that sometimes simulation models require a lot of input data which is not available for smallholder farms in Asia and Africa.

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. A literature search in Google Scholar for the terms “rice” and “closed chamber” has yielded 23 hits for the year 1991, 101 for 2001, 241 for 2011 and 632 for 2021, illustrating the growing number of these measurements. 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 United Nations Framework Convention on Climate Change (UNFCCC). The caveat of these wide-ranging applications is that the measurement results often remain as “grey literature” with no peer-reviewed publications and are not always available to an international audience; for example, the IPCC emission factor database shows only 24 emission factors based on measurements for CH_4 in rice production (as of January 2022). While there are a lot more peer-reviewed publications than included in the database, the process of entering emission factors into the IPCC database is not straightforward and hence there are very few entries in this database.

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). As with 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 advantages come into play for highly reactive gases such as the NO-NO_2-O_3 triad (Breuninger *et al.*, 2012). Given the complexity of the gas sampling patterns, however, dynamic chamber systems are rarely used for non-reactive gases like CH_4, i.e. the current spike in available emission measurements in rice fields is exclusively based on closed chamber systems.

Gas chromatography is the conventional method used to analyse CH_4 concentrations in gas samples from soils and manure handling systems. As for CH_4 analysis, the flame ionization detector (FID) (Weiss, 1981) is the gas chromatography detector of choice, whereas other detectors may be deployed for specific purposes, such as mass spectrometry (Ekeberg *et al.*, 2004) to determine isotopic composition of one or multiple gas analysis systems for the parallel assessment of several GHGs (Hedley, Saggar and Tate, 2006; Sitaula, Luo and Bakken, 1992). Laser technologies, Fourier-transform infrared and other optical techniques continue to grow in popularity for analysing 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 the quantum cascade laser (Cowan *et al.*, 2014; Nelson *et al.*, 2002) and other spectroscopic techniques with quantum cascade laser such as cavity ring-down spectroscopy (Brannon *et al.*, 2016; Christiansen *et al.*, 2015) as well as off-axis integrated cavity output (Waldo *et al.*, 2019; Brannon *et al.*, 2016) (Harvey *et al.*, 2020). Infrared absorption measurement detectors are well suited for situations that require frequent, high precision measurements, e.g. to capture diel variation and for 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 the

CH_4 flux. Collections rings or collars should be installed well in advance of sample collection, i.e. more than 24 hours, to allow the diffusion of gas from the soil or litter layer to the atmosphere sufficient time to equilibrate following the disturbance event. More details about robust trace gas estimation with closed 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 depends on cost consideration, labour availability, experimental design and sampling conditions (e.g. site accessibility, climate, soil type). Closed chambers with manual sampling are advantageous because they require only low investment and are simple to deploy, but they involve greater manual labour costs (Savage, Phillips and Davidson, 2014). Both non-flow-through and flow-through chambers can alter the 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 time needed for measurement with closed chambers can also alter the diffusion gradients (Davidson *et al.*, 2002; Savage, Phillips and Davidson, 2014). Open chambers, particularly flow-through systems, allow for more frequent, and less time-and labour-intensive measurements (Ueyama *et al.*, 2015; Savage, Phillips and Davidson, 2014). Furthermore, open chambers may be more appropriate for manure handling systems given the differences in the 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). Detailed instructions on how to customize manual chamber design for tropical or semi-arid regions of the world and other low infrastructure contexts are provided by Tiwari *et al.* (2015).

3.2.2.2 Micrometeorological techniques

The main micrometeorological technique for measuring CH_4 fluxes from soils is by eddy covariance. 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 the atmospheric boundary layer (Baldocchi, 2014, 2003; Baldochi, Hinks and Meyers, 1988). Samples are taken rapidly (more than 10 times per second) for long durations (of more than 30 minutes) to calculate GHG flux density between the soil and/or vegetation and the atmosphere, thus 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 that 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, Hinks and Meyers, 1988). However, eddy covariance is less well suited for small-scale manipulation experiments, and exhibits some bias with respect to spatially heterogeneous gases like CH_4 and N_2O (Baldocchi *et al.*, 2012). Eddy covariance may therefore be more appropriate for ecosystem monitoring of CH_4 fluxes and, when applied in experimental contexts, it should be combined with chamber-based methods rather than completely substituted for them (Eugster and Merbold, 2015). Another aspect to consider is the large area ("fetch") required for eddy covariance measurements that constitutes a major impediment for intercomparisons of different agronomic treatments. While the minimum fetch for eddy covariance measurements depends on the height where the sensors are placed, the typical setup of a two-metre-high mast in a rice field translates into a 100-metre radius and a coherent experimental field of 4 ha (Alberto *et al.*, 2009). Given that these measurement systems are relatively expensive, a practical solution can be a "roving tower" that is routinely shifted from one experimental field to another (Alberto *et al.*, 2012). While the flux records that integrate over larger areas do not present the artificial patchiness of chamber measurements, the need for a steady horizontal air flow within the fetch puts additional constraints on eddy covariance measurements. This crucial requirement often leads to greater data gaps during night-time and effectively precludes eddy covariance measurements during periods of high turbulence, which are often the case in tropical regions during the rainy season. Eddy covariance measurements to determine CH_4 emissions from rice fields have been applied in several countries, e.g. the United States (Reba *et al.*, 2020), China (Ge *et al.*, 2018), India (Swain *et al.*, 2018) and the Philippines (Alberto *et al.*, 2014). Additional research is needed to understand differences in seasonal flux estimates for rice paddies measured with chambers versus eddy covariance (Reba *et al.*, 2020).

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, the emission rates being estimated by applying Gauss's theorem (Conley *et al.*, 2017). At the barn level, the 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 CH_4 emissions for the whole facility. In Californian dairies, aircraft measurements were compared to open-path measurements using inverse dispersion modelling and vehicle measurements made with the tracer flux ratio method. The estimated CH_4 emission rates were compared on a whole-farm level and for primary sources within a farm, such as 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 to estimate the yield of crops given the good correlation between the leaf area index and the normalized difference vegetation index (Lamb *et al.*, 2011; Nagy, Fehér and Tamás, 2018; Wahab, Hall and Jirström, 2018). Drones and satellites have also been used to track and count animals (Laradji *et al.*, 2020) and to detect CH_4 leaks from oil and gas facilities in natural gas pipelines (Barchyn, Hugenholtz and Fox, 2019; Lauvaux *et al.*, 2022; Tannant *et al.*, 2018; Varon *et al.*, 2018). There is a potential for adapting these technologies to assess and benchmark livestock-related CH_4 emissions on farms.

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 measurements of CH_4 emissions provide better spatio-temporal 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 (Frankenburg *et al.*, 2006), and later with GOSAT (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 (Varon *et al.*, 2018), GOSAT-2 (Glumb, Davis and Lietzke, 2014), geoCARB (Polonsky *et al.*, 2014), and MethaneSAT (Staebell *et al.*, 2021). Even though some missions qualified as hyperspectral imagers or known as imaging spectrometers are not optimized for CH_4 mapping, they sample the strong CH_4 absorption at 2300nm with tens of spectral channels, which can be exploited for CH_4 retrieval (Varon *et al.*, 2021; Guanter *et al.*, 2021). Satellite-based measurements rely on inverse modelling to understand and quantify CH_4 emissions at regional and global scales (UNEP and CCAC, 2021). Under inverse modelling, 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 G. *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-prone Asia. They found a strong correlation between areas where rice is produced 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 and satellite measurements could help reduce the spatial uncertainties associated with rice CH_4 estimates in empirical and process-based models (Zhang G. *et al.*, 2020). However, Zeng *et al.* (2021) reanalysed the same atmospheric CH_4 concentration data with CH_4 simulations from a chemical transport model, and found insufficient evidence to support the claim that spatial areas of rice production and atmospheric CH_4 concentrations are correlated. These authors caution against the use of correlation-based inference to estimate CH_4 emissions from rice production at regional and continental scales, and point out that more work combining satellite observations and model simulations is needed to parse out different CH_4 emissions sources (Zeng *et al.*, 2021).

Airborne and ground-based *in situ* measurements continue to be the main methods for measuring CH_4 concentrations despite their limitations. Work previously carried out in California on rice (Peischl *et al.*, 2012) and dairy (Arndt *et al.*, 2018) production systems shows how remote sensing techniques can capture seasonal CH_4 emissions dynamics for those regional production systems that are 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.

3.4 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, identify emission factors or certify emission mitigation. Gates *et al.* (2009) demonstrated how component error analysis could be used to quantify uncertainties such as the 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 animal inventory, dry-matter feed intake, the chemical composition of the diets, CH_4 emission factors and predictions of enteric CH_4 emissions are the main culprit. Unfortunately, until now, uncertainty has not been evaluated for all published emissions values, which makes comparing results between the different papers, evaluating the quality of the results and certifying the emission reductions difficult. One future challenge will be providing 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 the 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 ill-quantified and there is a lack of agreement between distinct estimates. The "bottom-up" approaches seem to follow a more mechanistic, conceptual, build-up rather than a reconciliatory approach (e.g. "top-down") that may be unsuitable if the actual sources are not known, and lead to incorrectly assigning estimate shares to known sources. Vibart *et al.* (2021) provided an extensive discussion of mathematical models that can predict on-farm CH_4 and N_2O emissions.

4.1.1 Modelling to estimate enteric methane

There are many different types of mathematical modelling 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). For predictability purposes, some mathematical models of nutrition may incorporate different (and sometimes complementary) methods, often called levels or tiers of solutions (Tedeschi and Fox, 2020a). The simplicity of empirical models is commonly the dominant factor in the decision-making process when selecting models to predict CH_4 emissions. In part, the models' simplicity is a function of the inputs required for the execution of the model (essentially derived from statistical regression models and methods), which favours the selection of empirical models over more complex (and sometimes more complete) types of modelling 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 making 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 which 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. Cautionary notes should therefore 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 modelling methods should be available and considered. 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 its 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 to support a specific predictive goal (Tedeschi and Fox, 2020a). Unfortunately, few mathematical nutrition models have explicitly modelled 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 at all, to ruminant animal performance and production.

4.1.1.1 Empirical models

Bottom-up models for predicting emissions have been used in lieu of actual measurement. These models draw on 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. It 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 in 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 in percent of GEI converted to CH_4). Tier 3 is 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 involving 16 dietary and animal variables, and found 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 suggested that increased complexity improved prediction. They also reported that models with DMI only had a good accuracy of prediction while other dietary variables further improved the prediction of the models. These findings are consistent with those discussed by Appuhamy, France and Kebreab (2016), who reviewed 40 models with 20 variables and found that 43 percent of the models used DMI to predict CH_4 production.

Determining DMI for stall-fed and confined animals is straightforward, but many livestock systems involve ruminants grazing on native pastures, their diet supplemented with crop residues and cultivated fodder/forage in mixed crop-livestock systems. Working out the dietary amounts and composition in these systems is complicated. In part, voluntary feed intake depends on the digestibility of the diet (or the digestible energy), which, in turn, depends on the level of intake (Tedeschi *et al.*, 2019). This problematic nature becomes more involved still owing to a lack of proper characterization of the prevailing livestock systems (i.e. numbers, breeds, herd structures, body weight, physiological states and level of productivity) as well as of the terrestrial characterization of the pastureland. Typical methods for estimating DMI include employing empirical models such as those based on the net energy system (NASEM, 2016; NRC, 2001; NRC, 2007) or those factoring in animal characteristics, pasture conditions and supplementation (CSIRO, 2007), the 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 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 recourse to a "feed basket", a term referring to proportions of feeds on offer in a given season and 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 towards the mitigation of emissions (Bodansky *et al.*, 2016).

Additional, targeted inputs might further improve the adequacy and predictability of empirical models. An example is the mid-infrared spectrum of milk components as a proxy for estimating individual CH_4 emissions with chemometric 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 milk-infrared spectra can be obtained routinely at a reasonable cost (already collected for milk payment, milk recording or both). This proxy presents a significant interest for large-scale studies (that compare animals, herds, periods, geographical regions, genetic studies) (Vanlierde *et al.*, 2020), but the information about the limitation and applicability of milk mid-infrared spectra of milk components is still 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 (propionate and valerate). The remaining hydrogen is used for the reduction of CO_2 to CH_4. 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, Ominski and Kebreab, 2011; 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 the fermentation of soluble sugars and starch (Bannink, Reijs and Dijkstra, 2008) is used as a Tier 3 method for CH_4 inventory in the Kingdom of the Netherlands (Bannink, van Schijndel and Dijkstra, 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, France and Gill, 1987; Baldwin, France and Gill, 1987; Baldwin, Thornley and Beaver, 1987; 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, Baldwin and Koong (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 proposed by Dijkstra *et al.* (1992) uses three pools (amylolytic, fibrolytic and protozoal).

Several studies have evaluated the predictive potential of empirical and mechanistic models for enteric CH_4 production from cattle using independent data sources (Alemu, Ominski and Kebreab, 2011; Benchaar *et al.*, 1998; Kebreab *et al.*, 2006, 2008). Benchaar *et al.* (1998) compared the predictive capacity of two mechanistic and two linear models with a database constructed from existing 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, Ominski and Kebreab (2011) 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 would then serve as input for IPCC models for inventory purposes.

Another mathematical model for predicting VFA and ruminal pH that can be used to forecast CH_4 emission was developed by Pitt *et al.* (1996) and Pitt and Pell (1997) within the Cornell Net Carbohydrate and Protein System framework. Based on the mass balance approach, the assumptions in developing the model included: i.) the 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 non-carbohydrate sources (Tedeschi and Fox, 2020a, 2020b). Further additions to Pitt's model were discussed by Tedeschi and Fox (2020a, 2020b) and incorporated into the NASEM (2016), including the impact of pectin 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 Modelling to estimate manure methane

4.1.2.1 Empirical models

Similar to enteric CH_4, IPCC's guidelines for national greenhouse gas inventories (2019) indicate three tiers of complexity in estimating 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 volatile solid (VS) unit 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; Vigan *et al.*, 2019; van der Weerden *et al.*, 2020). Finally, Tier 3 requires specific modelling approaches tailored to country-specific methodologies or measurement-based approaches to quantifying emission factors. Likewise, several models have been used to estimate the CH_4 emissions from manure storage systems, but they possess a higher degree of uncertainty. 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 modelling of CH_4 emissions is challenging because of the complex data requirement and model parameterization (Li *et al.*, 2012), which limits their use for regional or country estimates. Furthermore, the use of mechanistic models in LCA analysis remains elusive. As in the case of enteric emissions, mechanistic models of manure emissions are scarce. One such model, Manure-DNDC (Li *et al.*, 2012) is an extended version of the DeNitrification-DeComposition (DNDC) model (Li, Frolking, S. and Frolking, T., 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 Californian cropping systems and used for developing Californian CH_4 emissions inventory from rice paddies and N_2O emissions inventory from synthetic fertilizers and crop residue (Deng *et al.*, 2018a, 2018b).

4.1.3 Soil/crop modelling

4.1.3.1 Empirical models/IPCC methodology

The IPCC methodology for estimating CH_4 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 the 2019 refinement (IPCC, 2019). The guidelines for rice cultivation comprise a fairly simple empirical model based on emission 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 the 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, ranging from the local to the global scale, and thus developed into a standard approach for calculating CH_4 emissions from rice production.

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

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: i.) rice ecosystem type (irrigated, rainfed, deep water, and upland rice production); ii.) flooding pattern before and during rice cultivation period; and iii.) type and amount of organic amendments. Other conditions such as soil type and rice cultivar can be considered for the detailed estimation if the specific information about the relationship between these conditions and CH_4 emissions is available.

Three tiers can be used depending on data availability. Tier 1 applies to countries where either CH_4 emissions from rice production are not a key category or where country-specific emission factors do not exist. In Tier 1, CH_4 emissions are estimated based on the available data regarding the annual harvest area of rice after the disaggregation of the area according to its water regime: irrigated, rainfed and upland. The calculations are done separately for each water regime and organic amendment. 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 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 subnational 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 circumstances of rice cultivation.

4.1.3.1.1 Daily emission factor and scaling factors

A global CH_4 baseline emission factor proposed in the 2019 refinement is 1.19 kg CH_4 ha^{-1} d^{-1}, with a confidence interval of 0.80 to 1.76. Regional CH_4 baseline emission factors, ranging from 0.65 to 1.32 kg CH_4 ha^{-1} d^{-1} (IPCC, 2019), are also proposed to enable the collection of more disaggregated activity data. The emission factor is adjusted to different scaling factors in order 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, the scaling factors for soil type and rice cultivar can be included.

In the case of continuously flooded fields, the scaling factor for water regimes during the cultivation period ranges from 0.06 for deep-water rice to 0.71 for a field with a single drainage period (IPCC, 2019). The scaling factor for upland rice cultivation is zero. For water regimes before the rice cultivation period, it ranges from 0.59 in case of fields without a flooded preseason over one year to 2.41 for those with a 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 0.17, the lowest value, for compost (IPCC, 2019).

4.1.3.1.2 Activity data

Estimation of CH_4 emissions from rice cultivation by empirical models is primarily 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 rice variety. In the refinement of 2019, the default cultivation period of rice is estimated on a global scale (113 days with an error range of 74 to 152 days) as well as on a subcontinental scale (102 to 139 days) (IPCC, 2019). A correlation of locally verified cultivation areas with available data for the emission factors would be invaluable. International data sources are also available for the 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 used to estimate methane emissions. Data relating to the rice area harvested can be obtained from FAOSTAT on the FAO website (www.fao.org/faostat). The Ricepedia online source provided by the International Rice Research Institute (IRRI, https://ricepedia.org/rice-around-the-world) features harvested areas of rice by ecosystem type for major rice-producing countries, alongside other useful information such as 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 used 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 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, 2014). By means of a specific methanogenesis submodel, DayCent integrates soil redox potential, 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, Li and Saggar, 2010), and it is used as a Tier 3 method in 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 modelling 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 the DNDC simulations of CH_4 emissions was able to better represent in both Japan (Katayanagi *et al.*, 2017) and India (Pathak, Li and Wassmann, 2005) the management factors that influence CH_4 production in rice systems – that is, organic matter inputs, total production area, drainage class types and water management. In Japan, the DNDC model simulations were used to generate revised emission 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 below-ground plant C and N inputs, these models were not developed with the intent of rigorously modelling the impacts of varying cultivar types and certain environmental conditions such as pest outbreaks on crop yields, and 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). In order to overcome this challenge, Tian *et al.* (2021) combined the decision support system for agrotechnology transfer (DSSAT) (Jones *et al.*, 2003; Sarkar, 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, and to identify best management practices for minimizing the food-water-GHG emissions trade-offs in China. Future efforts to combine crop growth and production models with soil biogeochemical models could help improve GHG emission estimates from rice paddy systems but also identify co-benefits and trade-offs associated with management decisions.

Unlike in other scientific fields, the use of ensemble modelling is still not common in soil science. An ensemble modelling 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 processes (Parker 2013), but also the use of different model input datasets (Tian *et al.*, 2019). While some research in this area has addressed crop production and yields (Asseng *et al.*, 2013), there is limited work on applying ensemble model simulations to soil N_2O (Ehrhardt *et al.*, 2018; Tian *et al.*, 2019) and soil C dynamics (Sándor *et al.*, 2020). The subject of 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 for individual sources to global assessments of annual CH_4 emissions. Bottom-up approaches typically involve measuring at the scale of individual CH_4 emitters, such as livestock or manure storage facilities. These approaches use emissions factors developed based on data collected from individual, activity and sometimes mechanistic models. Top-down approaches, in contrast, estimate emission by 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 emissions may provide powerful discrimination between sources (Nisbet *et al.*, 2020). The proportion of biogenic emissions (from wetlands, ruminants or wastes) results in a shift to negative values of $\delta^{13}C_{CH4}$ (atmospheric CH_4 changing the carbon isotope ratio) (Nisbet *et al.*, 2019). Various top-down techniques are used for measuring CH_4 emissions, including remote observation (e.g. atmospheric CH_4 by infrared spectrometry), towers, aircraft and satellites. Many modelling 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, as in the case of Hristov *et al.* (2013a).

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 re-examination of estimates from both approaches (NASEM, 2018). The challenge for top-down approaches is that estimates include emissions from all sources and may have difficulty in 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, Johnson and Kebreab (2014) challenged top-down estimates made by Miller *et al.* (2013), 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 the atmospheric transport models. Furthermore, NASEM (2018) reports that current global and regional atmospheric transport models are unlikely to be able to accurately represent small-scale processes, making it difficult for them to accurately simulate observed CH_4 at continental sites. Contemporaneous top-down and bottom-up measurements were conducted by Arndt *et al.* (2018). The authors showed that whole-facility CH_4 emission estimates were comparable in 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. Short-term measurements should therefore not replace long-term measurements. Top-down and bottom-up methods could be complementary in identifying gaps and may lead to a better characterization of CH_4 emissions.

PART 3

Mitigation of methane emissions

5. Mitigation strategies for methane emissions

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: i.) animal breeding and management; ii.) feed management, diet formulation and precision feeding; iii.) forages; and iv.) 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 depends on the production system and the regional or local conditions; hence the need for numerous approaches. Strategies that differ in mode of action may have potentially 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 any possible limitations.

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 important variables 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 decreases higher than 25 percent.

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

Respiratory CO_2 and CO_2 of 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 in the atmosphere.

5.1 ANIMAL BREEDING AND MANAGEMENT: INCREASED ANIMAL PRODUCTION

5.1.1 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 (Knapp *et al.*, 2014; Beauchemin *et al.*, 2020).

5.1.2 Mode of action

Increased animal production reduces CH_4 emission intensity by the dilution effect 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.

5.1.3 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.*, 2013a). 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, Kebreab and Wattiaux, 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 decrease 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 reduction in the number of animals to decrease the total emissions of the country or region.

Replacing specialized beef herds or some portion thereof with dairy herds producing beef deserves consideration. By making maintaining or even increasing beef production possible with fewer animals, this could decrease absolute emissions and emission intensity. But such a solution may not be applicable to all situations, as in many countries or regions beef calves are raised on pastures with lower quality soils that can only meet the energy requirements of beef cows for gestation and lactation, rather than for fattening animals. Thus, semi-intensive or intensive dairy production may not be possible under those conditions.

5.1.4 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 for combining 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).

5.1.5 Effects on other emissions

Increasing animal production may increase CH_4 and N_2O emissions from manure storage and land application, owing to the increase in feed intake (Gerber *et al.*, 2013b). In addition, upstream CO_2 emissions may also rise 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 reoccupy the ecological niches of livestock, causing a net increase in CH_4 emissions (Manzano and White, 2019).

5.1.6 Productivity and the quality of meat, milk, manure, crop, and air

Animal production is increased along with manure production and crop cultivation due to the increased feed intake by individual animals. However, resource use efficiency and emissions increase 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.

5.1.7 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).

5.1.8 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. Education and knowledge transfer, availability of natural and technological resources, and a positive return on investment for producers are needed to implement these strategies. Furthermore, successful adoption requires the identification and breakdown of barriers for different livestock systems and regions, as demonstrated by failures and successes in adopting recognized best practices for increasing animal production in low-income countries (Owen, Smith and Makkar, 2012).

5.1.9 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 so 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 centres 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 greater productivity increases CH_4 emissions. Reducing emission intensity becomes more important when expanding ruminant production to meet the demand for food of a growing population.

5.2 ANIMAL BREEDING AND MANAGEMENT: SELECTION FOR LOW METHANE-PRODUCING ANIMALS

5.2.1 Description

Animal breeding that exploits natural animal variation in CH_4 emissions is an inexpensive, permanent and cumulative mitigation strategy (Hayes, Lewin and Goddard, 2013). At present there are only a few instances where CH_4 is taken into consideration in breeding programs around the world, including a large-scale commercial trial with sheep farmers currently underway in New Zealand and a program in the Kingdom of the Netherlands which integrates CH_4 emissions into the breeding dairy values (Rowe *et al.*, 2019; de Haas *et al.*, 2021).

5.2.2 Mode of action

Animal breeding exploits natural between-animal variation in CH_4 emissions (de Haas *et al.*, 2017). Various possible modes of action 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.

5.2.3 Efficacy

The magnitude of possible CH_4 mitigation is not fully understood. Earlier studies have been relatively small-scale (Chagunda, Ross and Roberts, 2009; Garnsworthy *et al.*, 2012; Lassen and Løvendahl, 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 ranging from 13 to 24 percent are possible between 2018 and 2050, their magnitude depending on the economic weight of CH_4 production (de Haas *et al.*, 2021).

5.2.4 Potential to combine with other mitigation strategies

Given that genetic selection is both complementary and additional to other mitigation strategies, genetic selection for CH_4 can be combined with other mitigation strategies. A challenging aspect is that selection for a CH_4 trait takes selection pressure from other economically important traits of interest.

5.2.5 Effects on other emissions

Selection for decreased CH_4 may alter OM digestibility.

5.2.6 Productivity and the quality of meat, milk, manure, crop, and air

Selecting solely for a low total CH_4 production may simply select for lower DMI and can result in lower production (Lassen and Løvendahl, 2016; de Haas *et al.*, 2017; Breider, Wall and Garnsworthy, 2019). Also, low CH_4-producing animals should in theory 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 a targeted selection for CH_4 production within a breeding program, the link between CH_4, animal productivity and economics needs to be considered.

5.2.7 Safety and health aspects

No adverse issues related to breeding have been reported in the literature.

5.2.8 Adoption potential

The adoption potential is high but requires considerable investment by industry to measure and identify low CH_4 phenotypes. Assessing an animal's CH_4 phenotype is difficult because CH_4 must be measured over an extended period of time (weeks), 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 an alternative means of phenotyping low-CH_4 animals. Once the trait is integrated into the breeding program, there should be little impediment for adoption. A considerable difference in the adoption potential in low-income and high-income countries is to be expected. An investigation of genotype by environment interaction would determine whether optimum genetics identified in one country is suitable for another country or region. Interactions with diet types need to be explored.

5.2.9 Research required

Information is needed on low-CH_4 animal phenotypes, which will involve measuring CH_4 production on a large cohort of animals (more than 2000) (de Haas *et al.*, 2017). Substantial 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 against the CH_4 trait of relevance. The final step is to include the trait of interest in the selection index. This calls for a linkage between the CH_4 trait of interest and economics, which could be done by placing a price on CH_4 emissions.

5.3 ANIMAL BREEDING AND MANAGEMENT: IMPROVED FEED EFFICIENCY

5.3.1 Description

Improving feed efficiency, defined as the ratio of animal product to feed intake (i.e. kg of meat or milk per kg DMI), reduces CH_4 emission intensity. Feed efficiency may be improved by increasing the nutrient density or feed digestibility, altering the rumen microbial composition, enhancing feed management practices (Knapp *et al.*, 2014), and selectively breeding for animals with negative residual feed intake[2] (Løvendahl *et al.*, 2018; Beauchemin *et al.*, 2020) and smaller metabolic body weight (VandeHaar *et al.*, 2016), or a combination of the above.

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[2] 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.

5.3.2 Mode of action

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

5.3.3 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 genetic variability (Hristov *et al.*, 2013a).

5.3.4 Potential to combine with other mitigation strategies

Improving feed efficiency can potentially be combined with other mitigation strategies.

5.3.5 Effects on other emissions

Improving feed efficiency will reduce absolute CH_4 emissions, CH_4 intensity and 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. Depending on the magnitude of the natural CH_4 emission baseline (Manzano and White, 2019) this switch may not result in a net reduced warming effect.

5.3.6 Productivity and the quality of meat, milk, manure, crop, and air

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

5.3.7 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 a greater inclusion of starch or fat in ruminant diets (Knapp *et al.*, 2014). Caution should also be exercised when using an unbalanced selection for negative residual feed intake as it could lead to undesirable effects due to negatively correlated traits (Løvendahl *et al.*, 2018).

5.3.8 Adoption potential

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

5.3.9 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). Understanding how the biological factors that influence feed efficiency and enteric CH_4 emissions interact (Cantalapiedra-Hijar *et al.*, 2018; Løvendahl *et al.*, 2018) requires further research. Research is also needed to study the effects on enteric CH_4 emissions (both intensity and absolute emissions) of improving feed efficiency under various genotype by environment by dietary conditions. The potential for cumulative or synergistic effects of improved feed efficiency and strategic dietary management as well as supplementation should be examined. A holistic bioeconomic evaluation of improving herd feed efficiency over time is called for. Genetic selection for feed efficiency is not yet a breeding objective in most systems due to the lack of genomic tools designed to predict feed efficiency.

5.4 ANIMAL BREEDING AND MANAGEMENT: IMPROVED ANIMAL HEALTH

5.4.1 Description

Animal health improved through breeding, disease prevention and treatment, enhanced nutrition or husbandry is bound to reduce CH_4 emission intensity.

5.4.2 Mode of action

Improved animal health typically increases animal production (Dürr *et al.*, 2008; Hand, Godkin and Kelton, 2012) and improves feed efficiency (Potter, Arndt and Hristov, 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).

5.4.3 Efficacy

Efficacy depends on whether the disease itself negatively affects feed intake, digestibility and/or animal productivity. Improved health is likely to increase absolute enteric CH_4 emissions but to decrease CH_4 emission intensity (Potter, Arndt and Hristov, 2018). A review that modelled the increased intake and production as well as animal longevity resulting from improved animal health showed a 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.*, 2015a; Özkan Gülzari, Vosough Ahmadi and Stott, 2018; Potter, Arndt and Hristov, 2018; von Soosten *et al.*, 2020). The overall effect of improving animal health on CH_4 emissions will thus depend on whether animal performance is negatively affected by disease, and whether improved health increases productivity.

5.4.4 Potential to combine with other mitigation strategies

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

5.4.5 Effects on other emissions

Improved animal health is likely to increase upstream emissions associated with crop production if the feed intake and animal performance increase. Nitrous oxide emissions from manure might decrease if the animals produce more, as more dietary nitrogen would be retained in meat and milk (Arndt *et al.*, 2015a). 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.

5.4.6 Productivity and the quality of meat, milk, manure, crop, and air

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 the stage of lactation at the 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 range from 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 USD 95.31 and USD 211.03 for the treatment, the discarded milk, labour 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 the mortality of young animals will lessen 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 need for growing replacements (Hristov *et al.*, 2013b).

5.4.7 Safety and health aspects

No adverse effect has been reported in the literature.

5.4.8 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, it is low to medium because of the costs of treatments and preventive care, and of access to treatments.

5.4.9 Research required

Most of the research on the effect of animal health on CH_4 production is based on modelling (Özkan Gülzari, Vosough Ahmadi and Stott, 2018; von Soosten *et al.*, 2020) and only a few studies have measured directly the effect of health on enteric CH_4 emissions (Arndt *et al.*, 2015a; Houdijk *et al.*, 2018). In general, it is possible to calculate the impact of the 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 impact on enteric CH_4 emission of individual animals by affecting DMI, including its metabolism and digestive aspects.

5.5 ANIMAL BREEDING AND MANAGEMENT: IMPROVED ANIMAL REPRODUCTION

5.5.1 Description

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

5.5.2 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, and this increases the total daily emissions from the herd.

5.5.3 Efficacy

The magnitude of CH_4 mitigation depends on the reproductive status of the herd. Research conducted generally involves modelling at herd level rather than the farm system level which takes into account the growing and non-productive animals required for each productive animal in the herd (Lovett *et al.*, 2006a, 2008; O'Brien *et al.*, 2010; Lahart *et al.*, 2021).

5.5.4 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).

5.5.5 Effects on other emissions

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

5.5.6 Productivity and the quality of meat, milk, manure, crop, and air

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, Shalloo and Butler, 2013). Increasing animal fertility should increase farm profitability (Shalloo, Cromie and McHugh, 2014) given that fewer replacement animals would be required to maintain the herd. If excess replacement heifers are used to produce beef, the 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.

5.5.7 Safety and health aspects

No adverse effect has been reported in the literature.

5.5.8 Adoption potential

A balanced approach is needed for genetic selection programs to incorporate reproductive traits in addition to other economically important traits. Selecting solely 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, the requirements for successful adoption are education, knowledge transfer, availability of and access to resources, and a 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.

5.5.9 Research required

Studies quantifying the effects of improved reproduction on CH_4 emissions are needed. This valuable information will enable farmers to make management decisions based on both economic and environmental outcomes. The impact on CH_4 emissions of using sexed semen and embryo transfer to increase the beef merit of animals from the dairy herd should be quantified. The 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.

5.6 FEED MANAGEMENT, DIET FORMULATION AND PRECISION FEEDING: INCREASED FEEDING LEVEL

5.6.1 Description

In this section, we discuss the isolated effects of increasing the feed intake of animals (i.e. feeding level) without altering diet composition. That said, in practice, there may be few production situations in which animals can be fed extra feed without altering diet composition. For example, supplementing grazing animals with concentrate will decrease the forage to concentrate ratio. A taller pasture with greater grass availability will likely be less digestible. Altering feed intake and diet composition affects animal production.

5.6.2 Mode of action

Increasing the feed intake of ruminants decreases the 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) and leading to a decline in CH_4 losses per unit of DMI or as a percentage of gross energy intake (GEI). In addition, a rapid passage rate increases the growth rate of methanogens and H_2 concentration, inhibiting acetate, H_2 and CH_4 production and favouring propionate production, which is a competitive pathway for the use of H_2 (Janssen, 2010). Importantly, increased feed intake decreases the proportion of ingested and absorbed nutrients and of energy associated with animal maintenance. As a result, increased feed intake dilutes CH_4 production due to maintenance, and a greater proportion of CH_4 emitted is associated with animal production (Capper, Cady and Bauman, 2009). The consequence of this is that the 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.

5.6.3 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 the level of intake above maintenance, while Hammond *et al.* (2013) observed a decline in the 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 a 2 to 6 percent decrease of CH_4 per energy-corrected 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, France and Kebreab, 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).

5.6.4 Potential to combine with other mitigation strategies

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

5.6.5 Effects on other emissions

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

5.6.6 Productivity and the quality of meat, milk, manure, crop, and air

Increasing the 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 requirements. In addition, greater intake increases the excretion of feces and urine, potentially affecting manure composition and emissions (Hristov *et al.*, 2013b), although perhaps not per unit of animal product.

5.6.7 Safety and health aspects

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

5.6.8 Adoption potential

This mitigation strategy is easily adoptable in production systems where it is possible to offer increasing feed. 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.

5.6.9 Research required

The general principles that determine the effects of increasing feed intake on digestion, fermentation and CH_4 production are well established. However, it is important to refine existing prediction models that estimate CH_4 production in response to DMI, and to develop new models for particular regions or diets. It is recommended that this mitigation strategy be 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 GHGs.

5.7 FEED MANAGEMENT, DIET FORMULATION AND PRECISION FEEDING: DECREASED FORAGE TO CONCENTRATE RATIO

5.7.1 Description

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

5.7.2 Mode of action

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

5.7.3 Efficacy

There is general agreement that feeding concentrates to ruminants reduces CH_4 emissions, expressed relative to GEI, DMI and product, although the adduced magnitude varies. Johnson and Johnson (1995) reported a 2 to 3 percentage unit decrease in GE lost 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 percentage unit less CH_4 (4.5 percent vs 6.0 percent of GEI) from beef cattle fed primarily grain as opposed to 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-fibre carbohydrates in the diet, up to a maximum of 15 percent decrease. Sauvant and Nozière (2016) quantified the effects of concentrate percentage on CH_4/OMD from the results of calorimetric measurements gathered in the "Rumener" database, concluding that energy lost as CH_4 is minimized with a high percentage of concentrate fed at high intake levels. The difference in magnitude of the 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, Givens and Garnsworthy, 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 vs pasture), pasture characteristics or differences in methodology used to estimate DMI.

Although increasing concentrate supplementation decreases CH_4 production per kilogram of DMI, OMD 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.

5.7.4 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-nitrooxypropanol (3-NOP) show synergy with concentrates, whereby the mitigation potential of inhibitors in high concentrate diets is increased (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*.

5.7.5 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 fossil fuels and nitrogen fertilizer used to produce the grain (Boadi *et al.*, 2004; Beauchemin, McAllister and McGinn, 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 increased concentrate (Johnson, Phetteplace and Seidl, 2002; Lovett *et al.*, 2006a). This emphasizes the need to evaluate total CO_2eq emissions using an LCA for individual farms and geographical regions (Beauchemin *et al.*, 2008). Changes in soil carbon need to be incorporated into the LCA.

5.7.6 Productivity and the quality of meat, milk, manure, crop, and air

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 fewer polyunsaturated, rumenic and vaccenic acids compared to animals fed conserved forages, especially grazing animals. If increasing concentrate percentage in the diet increases intake, the amount of manure may also be increased depending on digestibility.

5.7.7 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 should therefore be implemented and monitored carefully.

5.7.8 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 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 co-products 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, McAllister and McGinn, 2009). However, ruminants consume considerable quantities of food waste and co-products, converting these low-value materials into high-quality products. There is an opportunity to increase the use of these materials from grains and oilseeds not suitable for human consumption (e.g. frozen, off-grade, distillers grains and so on) (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.

5.7.9 Research required

Given that the general scientific concepts are well established, further research should be focused on the adoption potential at a regional scale, using the LCA approach. A quantification of natural baseline CH_4 emissions in natural or rewilded grazing ecosystems is needed to assess how effective an increased ratio of concentrate is in mitigating global warming.

5.8 FEED MANAGEMENT, DIET FORMULATION AND PRECISION FEEDING: STARCH CONCENTRATE SOURCES AND PROCESSING

5.8.1 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.

5.8.2 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) while decreasing the abundance of rumen protozoa (Franzolin and Dehority, 2010). The inhibitory effect on protozoa limits their symbiotic role in protecting methanogens from oxygen toxicity and reduces the generation of H_2 as substrate for methanogens in CH_4 formation (Newbold *et al.*, 2015). In addition, the processing method and source of grain can affect DM and starch degradability in the rumen. Slower rates of ruminal OM degradability will allow a greater proportion of OM digestion to occur in the intestines, decreasing the availability of substrate for CH_4 production in the rumen.

5.8.3 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 dairy cows a wheat-based diet reduces CH_4 emissions, yield and intensity by an average of 30, 48 and 41 percent, respectively, compared to corn-based and barley-based diets. Similarly, Ramin *et al.* (2021) reported that an oat-based diet decreased 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 reduced CH_4 yield by 30 percent in comparison with a barley-based diet (Beauchemin and McGinn, 2005), possibly due to decreased ruminal starch digestibility (Yang *et al.*, 1997). Furthermore, grain processing methods (the application of various combinations of heat, moisture, time and mechanical actions) can modify the 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, Cole and MacDonald, 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).

5.8.4 Potential to combine with other mitigation strategies

There is limited information on the synergistic effect of combining this mitigation strategy with others. However, it appears feasible to combine this 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).

5.8.5 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 the grain source and processing method, which might increase the excretion of nutrients such as fermentable OM and nitrogen (Beauchemin and McGinn, 2005; Hales, Cole and MacDonald, 2012), and the amount of CH_4, ammonia and N_2O emissions from manure (Gerber *et al.*, 2013b).

5.8.6 Productivity and the quality of meat, milk, manure, crop, and air

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, 2017; Ramin, Fant and Huhtanen, 2021) if rumen pH declines. The decrease in milk components could reduce the profitability of dairy producers.

5.8.7 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 subacute acidosis and other metabolic incidences, such as laminitis and liver abscesses, which could impair animal health.

5.8.8 Adoption potential

This CH_4 mitigation strategy is readily available and can be easily implemented in intensive or confined feeding systems but has limited potential for application in grazing systems. Processing and feeding various grain sources is easily carried out by farmers and does not require government approval. Formulating diets with grain sources calls for 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, the price volatility of grains and the cost of processing grains. The combination of these factors could increase feed costs and limit the potential for adoption of this mitigation strategy. Moreover, as well as modifying the forage to concentrate ratio (see Section 5.9), this strategy can increase food-feed competition and may contrast with the positive image of ruminants as utilizing human-inedible feed resources.

5.8.9 Research required

While a considerable amount of research on dairy cows has been conducted, more research is required to characterize how the grain source and processing method could influence enteric CH_4 emission in beef cattle and small ruminants. The effect of the grain processing method and degree of processing on the rate and extent of starch digestion needs to be clarified with regard to the impact on metabolic disorders such as acidosis. Although the magnitude of CH_4 abatement of wheat-based diets is attractive compared with 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.

5.9 FEED MANAGEMENT, DIET FORMULATION AND PRECISION FEEDING: SUPPLEMENTATION OF LIPIDS

5.9.1 Description

Dietary supplementation of lipids.

5.9.2 Mode of action

Dietary lipids produce 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 the ruminal fermentation process to promote the production of propionate resulting in lower CH_4 production; and decreasing feed fermentability in the rumen (Newbold *et al.*, 2015; Honan *et al.*, 2021). Lipids can encapsulate feed particles, which reduces rumen fermentation, leaving digestion to occur in the small intestine. In addition, as lipids are largely unfermentable (except for the glycerol moiety), the replacement of carbohydrates with lipids reduces fermentable OM, thus contributing to a decrease in enteric CH_4 emissions.

5.9.3 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, 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 in 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 proved effective in reducing CH_4 emissions (Fievez *et al.*, 2003; Jordan *et al.*, 2006a; Martin *et al.*, 2008; 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 this depends on the extent to which the oilseeds have been processed. 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; while 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 is 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 as opposed 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).

5.9.4 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).

5.9.5 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 greater 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, 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).

5.9.6 Productivity and the quality of meat, milk, manure, crop, and air

Supplementing fats for 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 have detrimental effects on rumen fermentation, feed digestion and animal performance (Grainger and Beauchemin, 2011; Patra, 2013, 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. 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, Ibrahim and AbuGhazaleh, 2008; Bayat *et al.*, 2015).

5.9.7 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.

5.9.8 Adoption potential

This CH_4 mitigation strategy is readily available and can be easily implemented in intensive and confined feeding systems. 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 DM in the diet. Feeding refined oils can be costly, with limited potential for commercial application. As an alternative, processed oilseeds can be less expensive and might stimulate the adoption of supplementing dietary lipids.

Although limited options exist to implement this strategy in grazing systems, promising efforts have been made 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).

5.9.9 Research required

To stimulate uptake, further research is required that would identify cost-effective fat sources and their respective supplemental level to 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) should be better understood, particularly as regards the CH_4 inhibitory effect of dietary lipids. It must be ensured that CH_4 inhibition due to lipid supplementation of diets is not the result of a decrease in fibre digestibility. Studies are also needed to ascertain the long-term effect of supplemental fats in suppressing CH_4 emissions. Given its potential impact on feed emissions and nutrient excretion, the effectiveness of this mitigation strategy should be addressed using LCA.

5.10 FORAGES: FORAGE STORAGE AND PROCESSING

5.10.1 Description

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

5.10.2 Mode of action

More than one mode of action may be involved. Compared to its preservation as hay, ensiling forage can decrease CH_4 production because the soluble carbohydrate fractions ferment during silage making, thereby reducing rumen fermentation (McDonald, Henderson and Heron, 1991). Processing strategies such as pelleting increase the rumen outflow rate. The greater passage rate decreases OM degradation in the rumen (Thomson, 1972; Huhtanen and Jaakkola, 1993; Hironaka *et al.*, 1996; Le Liboux and Peyraud, 1999), which results in lower CH_4 production. Moreover, an increased passage rate increases the growth rate of methanogens, and consequently the H_2 concentration increases according to the Monod function. A greater H_2 concentration thermodynamically inhibits H_2 production, with the result that acetate production, which releases H_2, is also inhibited. Less H_2 being produced means less H_2 being incorporated into the CH_4 production. Fermentation is shifted towards propionate production (Janssen, 2010).

5.10.3 Efficacy

Johnson, Ward and Ramsey (1996) reported a decrease of CH_4 yield between 20 to 40 percent when forage was ground or pelleted, compared with feeding animals long forage. Benchaar, Pomar and Chiquette (2001) reported similar findings in a simulation study, with approximately a 20 percent reduction of CH_4 production (g/day and percentage of GEI) for pelleted in contrast to long alfalfa hay. The efficacy of forage processing in 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, Bouvier and Demarquilly, 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 the forage preservation method on CH_4 production (Knapp *et al.*, 2014). Benchaar, Pomar and Chiquette (2001) simulated 33 percent less CH_4 (g/day, percent of GEI) for alfalfa silage compared with alfalfa hay, due to a lower ruminal degradation of OM as carbohydrates are partly fermented in silage making (McDonald, Henderson and Heron, 1991). However, a decrease in CH_4 production due to the reduced 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 the preservation method will depend on forage species and the stage of maturity of the harvested forage (Evans, 2018).

5.10.4 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 and whether the effects are additive will have to be evaluated in each case.

5.10.5 Effects on other emissions

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

5.10.6 Productivity and the quality of meat, milk, manure, crop, and air

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

5.10.7 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 the intake of individual animals.

5.10.8 Adoption potential

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

5.10.9 Research required

While storage and processing of forage has been shown to decrease CH_4 yield, it is not clear if the CH_4 per unit of animal product is also decreased as there is limited literature on this subject. 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. Since this can vary widely between systems and regions, studies parameterizing local production systems are needed to develop predictive models.

5.11 FORAGES: INCREASED FORAGE DIGESTIBILITY

5.11.1 Description

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

5.11.2 Mode of action

Forages are more digestible when in a vegetative phenological stage of maturity. In pastoral systems, forage digestibility can be increased by optimizing grazing management so that the pre-grazing herbal mass and height are not excessive. The digestibility of OM is often higher for low than for high herbal mass swards. The 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 also been investigated to see whether they increase the digestibility of mature forages (Adesogan *et al.*, 2019). Increasing forage digestibility raises animal productivity, forage intake and digestion. Responses to increased forage digestibility in terms of absolute CH_4 production can be variable, but absolute CH_4 production usually increases with greater DMI and increased OM fermentation in the rumen. When measured *in vitro*, concentrations of neutral detergent fibre and indigestible fibre reduce CH_4 production, while concentrations of water-soluble carbohydrates and OM digestibility of the forage increase CH_4 production (Weiby *et al.*, 2022). Thus, selecting forages that increase *in vitro* CH_4 production will likely decrease CH_4 intensity (CH_4 per kg of DMI, milk or meat). The amounts of additional CH_4 produced when feeding low-fibre, highly digestible forages are relatively lower than those produced by the animal product (Beauchemin, McAllister and McGinn, 2009).

5.11.3 Efficacy

Greater forage digestibility can increase absolute CH_4 emissions, but it generally results in low to moderate decreases in CH_4 emission intensity (Beauchemin *et al.*, 2020). Dairy cows grazing on swards differing in pregrazing 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 a 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 was 12 percent lower 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 DM intensity, DM digestibility and milk production. Absolute CH_4 production was 6 percent greater with the earliest cut grass, but CH_4 intensity was 24 percent lower. Macome *et al.* (2018) evaluated grass ensiled at four different stages of maturity, and concluded 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 lower, respectively, for the youngest compared to the oldest cut grass.

5.11.4 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 increased forage digestibility is combined with other CH_4 mitigation strategies.

5.11.5 Effects on other emissions

Emissions of GHG other than enteric CH_4 will be altered by grazing (e.g. changes in stocking rates) or changes in cutting management that affect the digestibility of conserved forages. Earlier herbage cutting for ensilage or hay-making will result in a lower grass biomass available, thus affecting the fossil fuel emissions of CO_2 per kilogram of DM conserved, although less fossil fuel per hectare may be needed to harvest and ensile or bale the forage. Downstream emissions will also be affected, as a greater digestibility will decrease the output of manure as well as changing its composition, and perhaps decrease the emissions of CH_4 from manure accordingly. 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. An LCA will be needed, and local research is recommended to establish reliable practices for each region.

5.11.6 Productivity and the quality of meat, milk, manure, crop, and air

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

5.11.7 Safety and health aspects

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

5.11.8 Adoption potential

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

5.11.9 Research required

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

5.12 FORAGES: PERENNIAL LEGUMES

5.12.1 Description

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

5.12.2 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 fibre (NDF) than grasses, and although the fibre in legumes is more lignified, the decline in fibre digestibility with advancing maturity is much greater for grasses than for legumes. Fibre that is more digestible results in a ruminal fermentation that decreases the acetate to propionate ratio and methanogenesis. In addition, legumes contain secondary compounds that decrease CH_4 production (i.e. condensed tannins and saponins; see Section 5.25 and Section 5.26), although concentrations of these compounds are 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). Lastly, animal performance is often increased with the inclusion of legumes in ruminant diets, which decreases CH_4 intensity.

5.12.3 Efficacy

The fact that the reduction in CH_4 production due to the dietary inclusion of legumes depends on the quality of the forages being compared makes it difficult to quantify, as differences in feed intake and digestibility render the results confusing. 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.*, 2015b). When it comes to 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 account for 5.4 to 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 to 6.5 percent of GEI (8.6–13.0 percent of digestible energy intake). The notable exception was the legume *Leucaena leucocephala*, which decreased CH_4 yield by 11 percent when its inclusion rate was doubled; similar effects were not observed 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, the mitigation effect being highly dependent on forage species and quality. When the nutritive value of the diet (digestibility, CP) increases with the incorporation of legumes, animal performance would be expected to increase, thereby decreasing CH_4 intensity.

5.12.4 Potential to combine with other mitigation strategies

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

5.12.5 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 the N available for associate and subsequent crops (Schultze-Kraft *et al.*, 2018). The nitrogen 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 those generated by 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 the dietary forage source can affect the physicochemical characteristics of manure. For example, CH_4 emissions from manure slurry were lower 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 lower 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 of associated costs and emissions (Schultze-Kraft *et al.*, 2018).

5.12.6 Productivity and the quality of meat, milk, manure, crop, and air

The effects on animal productivity of increasing the proportion of forage legumes in the diet are 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 in the United Kingdom and the European Union. Johansen, Lund and Weisbjerg (2018) conducted a meta-analysis of temperate forages in dairy cows' diets and concluded that, overall, legume-based diets resulted in higher DMI 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, and thus not all legumes are equally effective. Both animal and forage productivity need to be considered when conducting a system analysis.

5.12.7 Safety and health aspects

No major concerns. Grazed clovers and alfalfa can cause bloat (timpanism), but this aspect is known and can generally 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.

5.12.8 Adoption potential

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

5.12.9 Research required

Life cycle assessment studies that consider climate, soil type, land use and production systems are necessary to determine the optimum use of legumes in different locations. These assessments should compare animal productivity under different forage management systems to identify the optimum legume for inclusion, one which minimizes emission intensity. Forage productivity and persistence must also be considered. There is a need for controlled animal research studies that account for differences in intake, digestibility and plant secondary compounds to examine the true CH_4 mitigation potential of tropical and temperate legumes. Plant secondary compounds in different legume species should be quantified, factoring in the stage of maturity of the plants and the duration of their storage.

5.13 FORAGES: HIGH-STARCH FORAGES

5.13.1 Description

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

5.13.2 Mode of action

With high-starch forage there is an increase in starch and a decrease in the fibre concentration of the diet, resulting in a rumen fermentation that promotes propionate production (Arndt *et al.*, 2015a), 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 a high total digestible nutrient content and a greater DMI (Benchaar *et al.*, 2014; Gislon *et al.*, 2020).

5.13.3 Efficacy

Methane production may increase, remain stable or decrease, depending on 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 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, the effects on CH_4 per unit of animal productivity are highly variable, and may differ according to nutritional values of harvested forages (Arndt *et al.*, 2015b). The efficacy of high-starch forages in reducing CH_4 depends on the stage of maturity of the various forages (i.e. time of harvest) and the relative differences in starch concentration.

5.13.4 Potential to combine with other mitigation strategies

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

5.13.5 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 on 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, Montes and Chianese (2010) reported that increasing the ratio of maize silage to alfalfa silage in dairy cow diets resulted in N being used more efficiently, which brought about a small decrease in excreted manure N that reduced the emission of N_2O from cropland. Maize silage production led to fewer CO_2 emissions from machinery and fuel compared with alfalfa, with the net result of a 13 percent decrease in 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, Montes and Chianese (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 for 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.

5.13.6 Productivity and the quality of meat, milk, manure, crop, and air

The effects of high-starch forages on animal productivity depend on the nutritive value of forages. The 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). The use of maize silage can decrease the N content of diets, and has been associated with greater N use efficiency in animals, decreased manure N excretion, and reduced ammonia-N and N_2O emissions from manure (Ardnt *et al.*, 2015a). The effects on CH_4 emissions from manure are not well known.

5.13.7 Safety and health aspects

No major concerns and accepted by regulatory officials.

5.13.8 Adoption potential

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

5.13.9 Research required

Feeding high-starch forages to reduce enteric CH_4 emissions is not recommended unless accompanied by an LCA indicating that the net emissions of meat and milk production are also decreased. The 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 for using locally adapted high-starch forages to increase animal productivity and lower the CO_2eq emission intensity of animal products. The research needs to take into account local agronomical and animal production conditions. A comparison of the yields of forages and the feeding value they produce per ha is needed.

5.14 FORAGES: HIGH-SUGAR GRASSES

5.14.1 Description

The use of high-sugar grasses (mainly cultivars of perennial ryegrass, *Lolium perenne* L.) with an 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.*, 2006b; 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.*, 2006b; Rivero *et al.*, 2020).

5.14.2 Mode of action

The greater concentration of readily available carbohydrates decreases the acetate to propionate ratio in rumen fermentation, and consequently CH_4 production is reduced (Rivero *et al.*, 2020). High-WSC grasses also improve the rate of fermentation and 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.

5.14.3 Efficacy

In vitro studies generally report less CH_4 for high- versus low-sugar grasses (Lovett *et al.*, 2006b; 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. The mitigation potential also depends on the concomitant changes in CP and NDF concentration and digestibility. Using a modelling approach in which high-sugar grasses were 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. Yet the simulated CH_4 intensity decreased by up to 17 percent when DMI increased due to feeding high-sugar grasses. Zhao, O'Connell and Yan (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. 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.

5.14.4 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.

5.14.5 Effects on other emissions

High-sugar grass cultivars have been shown to decrease the 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 the total CO_2eq per kilogram of milk was reduced by 3 percent when dairy cows were fed on high-sugar compared with conventional ryegrass pastures (Soteriades *et al.*, 2018).

5.14.6 Productivity and the quality of meat, milk, manure, crop, and air

In theory, an increased supply of readily fermentable carbohydrates should raise animal productivity in a manner similar to supplementation in the case of concentrates. Increased (+9 percent) DMI due to increased digestibility was reported for dairy cows in early lactation that had been fed fresh high-sugar grass (243 vs 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 the productivity of high producing ruminants if protein requirements are not met. For example, milk production was 18 percent lower when dairy cows were fed dried high-sugar compared with controlled 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). The productivity and other agronomic characteristics of high-sugar grasses will also have to be considered, as they may impact the area of grassland necessary to sustain a certain level of production. Any differences in persistence could affect how soon a pasture needs to be resown, which will affect the emissions of CO_2 and N_2O associated with the use of fossil fuels and fertilizers.

5.14.7 Safety and health aspects

No major concerns and accepted by regulatory officials.

5.14.8 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 (Rivera, Chara and Barahona, 2019). The adoption potential of high-sugar ryegrass cultivars is considerable in temperate areas where ryegrass is commonly grown. Prior to recommending high-sugar grasses for CH_4 mitigation, the climate, soil, the growing environment and yield potential must be considered. Perennial high-sugar grasses are currently not available in tropical or subtropical areas.

5.14.9 Research required

Most of the research to date on high-sugar grass cultivars has been limited to the United Kingdom, the Kingdom of the Netherlands and New Zealand, and an expanded geographical analysis is thus 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 should be examined. A more in-depth understanding of the chemical composition and digestibility of high-sugar grasses is also lacking. Finally, LCA studies that consider climate, soil type, land use and production systems are needed to determine the optimum use of high-sugar grasses in different geographical locations.

5.15 FORAGES: PASTURES AND GRAZING MANAGEMENT

5.15.1 Description

Grasslands are important sources of feed 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 with forage availability (both in terms of quantity and quality), while promoting rapid pasture regrowth 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 (Congio *et al.*, 2018; Savian *et al.*, 2018), with increased soil carbon stocks and decreased CH_4 intensity (Guyader *et al.*, 2016; de Oliveira Silva *et al.*, 2016; Makkar, 2018; Savian *et al.*, 2018). The use of pastures for sustainable production and the production of animal protein sources contributes to 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, including increasing biodiversity, fire control and water management (Murgueitio *et al.*, 2011). Silvopastoral systems promote a 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). The use of SPS can be a viable option especially in Latin America. Vandermeulen *et al.* (2018) showed that SPS with multipurpose shrubs and trees were beneficial for the ecosystem while the woody fodder improved ruminal protein digestion, reduced parasitic infestation and decreased CH_4 emissions, but limitations such as toxins can restrict their use. Mauricio *et al.* (2019) demonstrated that SPS based on different forage species, shrubs and trees enhanced the capacity to produce meat and milk without the use of grain.

5.15.2 Mode of action

This strategy is based on the intensification of grazing systems. The intention 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, 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).

5.15.3 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 the expanded forage production entails greater stocking rates. The extent to which grazing management lowers CH_4 intensity varies extremely, depending on the production system and local conditions. For example, rotational grazing based on sward pre- and post-grazing heights increased the digestible OM intake of sheep grazing on Italian ryegrass (*Lolium multiflorum*), thus reducing CH_4 intensity by

17 percent, although the 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 it did not alter the absolute CH_4 production. In the case of beef cattle, CH_4 intensity (g/kg carcass) was 10 percent lower for heavy versus light, continuous grazing, although the soil carbon sequestration was lower 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 (Vandermeulen *et al.*, 2018; Stewart *et al.*, 2019; Ku-Vera *et al.*, 2020). The presence of shrubs or legume forages (e.g. *Macrotyloma axillare*) 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). The inclusion of diversified forage species in pastures can therefore increase the quantity of biomass for animals while decreasing enteric CH_4 emissions.

5.15.4 Potential to combine with other mitigation strategies

It is expected that grazing management and use of SPS would add to the effects of other CH_4 mitigation strategies.

5.15.5 Effects on other emissions

Grazing management affects the 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, Xu and Beese, 2003) due to complex mechanisms having to do with the chemical composition, moisture and microbiology of the soil (Dunfield, 2007). Higher feed intake may increase manure emissions unless forage digestibility is also increased. However, the intensification of animal production would be expected to decrease the total CO_2eq emissions per unit of livestock product (Capper, Cady and Bauman, 2009); therefore, an LCA is needed when assessing the strategy's efficacy.

5.15.6 Productivity and the quality of meat, milk, manure, crop, and air

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 conducted by Congio *et al.* (2018) improved milk production efficiency by 51 percent, while decreasing CH_4 emission intensity and CH_4 yield by 20 percent and 18 percent, respectively. Greater milk production efficiency increased CH_4 emissions per hectare by 29 percent. The authors considered that strategic grazing was cost effective.

5.15.7 Safety and health aspects

No major concerns and accepted by regulatory officials.

5.15.8 Adoption potential

It is possible to implement immediately the improved management of pastures in extensive and intensive livestock systems. The selection of forages, shrubs and fodder species to be used needs to be tailored to each region and grazing management system. The high cost of implementing rotational systems (fences, water troughs) limits adoption prospects. Nevertheless, pasture management can be implemented at farm level, is suitable for all grazing ruminant categories, and has high farmer and consumer acceptance. 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 a negative impact on animal welfare (e.g. heat stress). Implementing the SPS is an objective of the Global Agenda for Sustainable Livestock (GASL, 2021).

5.15.9 Research required

A holistic approach involving multidisciplinary research teams and 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. A life cycle assessment of pasture-based systems needs to encompass soil carbon in addition to accurate estimates of enteric CH_4 emissions and excreta as well as other aspects of the landscape and environment. Long-term, regionally focused research is needed. Extension services supported by public policies (i.e. payment for environmental services) may be needed to encourage adoption.

5.16 RUMEN MANIPULATION: IONOPHORES

5.16.1 Description

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

5.16.2 Mode of action

Ionophores are carboxylic polyether substances that interfere with the ion transport across cell membranes of gram-positive bacteria and protozoa. Ionophores modify the ion transport flux across cell membranes of microorganisms, increasing the concentration of protons (H^+) in the cytoplasm (Duffield and Bagg, 2000; Duffield, Rabiee and Lean, 2008a, 2008b; Hersom and Thrift, 2012; Azzaz, Murad and Morsy, 2015). For maintaining cell equilibrium, the bacterial cells use energy to extrude H^+, which results in reduced growth and the death of cells (Duffield and Bagg, 2000). Due to the structure of cell membranes, ionophores are mainly active against gram-positive bacteria and protozoa (Beauchemin *et al.*, 2009; Hersom and Thrift, 2012; Azzaz, Murad and Morsy, 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 the VFA production from acetate (H_2 source) to propionate (H_2 sink), thus leading to reduced methanogenesis (Mathison *et al.*, 1998; Duffield and Bagg, 2000; Duffield, Rabiee and Lean, 2008a; Hristov *et al.*, 2013; Azzaz, Murad and Morsy, 2015). An increased feed efficiency (Hersom and Thrift, 2012; Hristov *et al.*, 2013) also reduces CH_4 emission intensity. The potential for the rumen microbes to adapt 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.

5.16.3 Efficacy

In their meta-analysis of 22 published studies, Appuhamy *et al.* (2013) demonstrated that monensin reduced Ym, the CH_4 conversion factor, by 0.5 percent units (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 adjusted to the monensin dose, the CH_4 mitigation effects were similar for dairy cows and beef steers (-12 ± 6 g/d and -14 ± 6 g/d, respectively). When factoring in DMI differences, monensin reduced Ym in dairy cows and beef steers by 0.23 ± 014 and 0.33 ± 0.16, respectively. The duration of the treatment period did not significantly modify the monensin effect.

5.16.4 Potential to combine with other mitigation strategies

Using combinations of ionophores or the rotational feeding of ionophores may help avoid microbial adaptation (Mathison *et al.*, 1998). It has a good potential to complement 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).

5.16.5 Effects on other emissions

With improvements in feed conversion efficiency due to monensin, the quantity of OM excreted in the manure might be reduced, further reducing CH_4 emissions from a farm. A nitrogen metabolism improved through ionophores reduces urinary N excretion and associated potential emissions of NH_3 and N_2O. The monensin dose in the diet being small, the increase in CO_2 emissions from the use of fossil fuels in monensin manufacture and transport is consequently thought to be rather small.

5.16.6 Productivity and the quality of meat, milk, manure, crop, and air

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

5.16.7 Safety and health aspects

The concentration of ionophores in the diet should be limited to avoid toxicity (Novilla, 1992; Hall, 2000) and, as with any feed additive, care should be exerted during handling. The use of monensin is subject to approval by regulatory agencies, and it is banned in some countries, including in the European Union. It has been questioned whether the widespread use of ionophores contributes to the cross-resistance to other antibiotics (Wong, 2019).

5.16.8 Adoption potential

When the use of ionophores is authorized, the adoption potential 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. The 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.

5.16.9 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.

5.17 RUMEN MANIPULATION: CHEMICAL INHIBITORS OF METHANE PRODUCTION

5.17.1 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 (5.18). Studies investigating the use of chemical inhibitors of methanogenesis in preruminant animals are discussed in Section 5.19.

5.17.2 Mode of action

Chemical inhibitors target methanogens but not all of them through directly inhibiting methanogenesis. Halogenated methane 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 the cobamide-dependent methyl transfer (Wood *et al.*, 1968). Coenzyme M analogues bromoethanesulfonate (BES; Gunsalus, Romesser and Wolfe, 1978) and 3-NOP (Duin *et al.*, 2016) also inhibit the last step of methanogenesis by blocking methyl-coenzyme 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 (Garcia-Lopez, Kung, Odom, 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, Scott and Parish, 1972), among others, is evidenced by the accumulation of H_2, but their exact mechanisms of action in the methanogen cell has not been demonstrated.

5.17.3 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 of all the various anti-methanogenic strategies.

In some *in vivo* studies, absolute CH_4 production was inhibited by more than 90 percent compared with controlled 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 days *in vivo* (Immig *et al.*, 1995). Conversely, a long-term inhibition of methanogenesis *in vivo* by different chemical compounds has been observed in other studies (e.g. Trei *et al.*, 1971; Trei, Scott and Parish, 1972; Clapperton, 1974, 1977; Davies *et al.*, 1982; Tomkins, Colegate and Hunter, 2009).

5.17.4 Potential to combine with other mitigation strategies

The 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 coupled with other anti-methanogenic strategies that have different mechanisms of action. Different methanogens are inhibited by chemical compounds to different extents (Ungerfeld *et al.*, 2004; Duin *et al.*, 2016), and thus combining or rotating different inhibitors of methanogenesis is an interesting research direction. *In vitro* experiments as a proof of concept are recommended (e.g. Zhang and Yang, 2012; Patra and Yu, 2013).

5.17.5 Effects on other emissions

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

5.17.6 Productivity and the quality of meat, milk, manure, crop, and air

In general, inhibiting 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). The amount and chemical composition of manure is probably little affected, but the passage of chemical inhibitors to manure has not been evaluated (except for 3-NOP).

5.17.7 Safety and health aspects

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

5.17.8 Adoption potential

Chemical inhibitors can allow a strong and consistent decrease in enteric CH_4 emissions with minimal effects on other GHG emissions but the research is yet to demonstrate this clearly. The inclusion of an inhibitor in a diet is bound to increase costs and will be unattractive to producers, unless accompanied by the higher price that products associated with a lower carbon footprint command. It may also be possible to take advantage of a 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.

5.17.9 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 additional inhibitors are being investigated (e.g. Zhang Z.-W. *et al.*, 2019a, 2019b). At the same time, there are older reports in which some compounds were shown to have long-lasting effects on methanogenesis; we are not aware of further investigations regarding the toxicity, passage to animal products or damage to the environment of those compounds, and those lines of research were probably abandoned because of the difficulties involved in handling these compounds in the feed or food chain. Moreover, further research to understand changes in rumen microbiome and whole animal metabolism induced by inhibition of methanogenesis is recommended in order to optimize the results of the interventions (Ungerfeld, 2018; Ungerfeld and Hackmann, 2020).

5.18 RUMEN MANIPULATION: 3-NITROOXYPROPANOL (3-NOP)

5.18.1 Description

3-nitrooxypropanol (3-NOP) is a CH_4 inhibitor developed and commercialized by DSM Nutritional Products (Basel, Switzerland). This mononitrate ester of 1, 3-propanediol has the $HOCH_2CH_2CH_2ONO_2$ formula (Duin *et al.*, 2016; Yu, Beauchemin and Dong, 2021).

5.18.2 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 cofactor F_{430}. Additionally, 3-NOP is cleaved by an electron transfer process into nitrite and 1, 3-propanediol, which also inactivates MCR. It is worth noting that the totality of the mode of action is reversible upon removal of 3-NOP (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).

5.18.3 Efficacy

There is a growing number of scientific publications (greater than 50) describing its efficacy for dairy and beef cattle in a range of different diets and management systems, with several reviews and meta-analyses (Dijkstra *et al.*, 2018, Jayenagara *et al.*, 2018; Kim *et al.*, 2020; Arndt *et al.*, 2021; Yu *et al.*, 2021; Kebreab *et al.*, 2023). 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 to register 3-NOP in Europe, allowed the evaluating panel to assess and conclude that 3-NOP has the potential to be efficacious in the case of all ruminant species (Bampidis *et al.*, 2021). The meta-analyses by Dijkstra *et al.* (2018) and Kim *et al.* (2020) established linear decreases in CH_4 production corresponding to 3-NOP dosage. In the meta-analysis by Dijkstra *et al.* (2018), 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). Those meta-analyses were lately updated for dairy cows to include the most recent studies (Kebreab *et al.*, 2023) and to take into account the effect of diet composition (NDF, EE and starch content). Models including only the 3-NOP dose indicated a decrease of 32.7, 30.9 and 32.6 percent for CH_4 production (g/d), yield (g/kg DMI) and intensity (g/kg energy-corrected milk), respectively, at an average 3-NOP dose of 70.5 mg/kg DM. The response to 3-NOP declined with increasing dietary NDF and EE. Although most long-term studies have shown that 3-NOP effectiveness remained constant, a couple of studies reported that 3-NOP effectiveness declined slightly over time, which might be related to the low dosage used (Yu, Beauchemin and Dong, 2021).

5.18.4 Potential to combine with other mitigation strategies

Good potential to combine with other strategies that have different modes of action. Incremental mitigation effects on CH_4 yield were reported for 3-NOP when combined with unsaturated lipids (Zhang *et al.*, 2021), higher concentrate proportion (Schilde *et al.*, 2021) and monensin ionophores (Vyas *et al.*, 2018).

5.18.5 Effects on other emissions

The emissions from producing 3-NOP in small-scale conditions were reported as 48 to 52 kg CO_2eq/kg 3-NOP (Alvarez-Hess *et al.*, 2019; Feng and Kebreab, 2020. This would represent approximately 6 g CO_2eq/kg diet DM, assuming a dose of 118 mg 3-NOP/kg DM. For example, for a dairy cow daily consuming 25 kg DM and emitting 274 g of CH_4 (~100 kg per year), 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), not counting emissions of CO_2eq from manure, N_2O and fossil fuels. Nkemka, Beauchemin and Hao (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.* (2020) likewise 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.

5.18.6 Productivity and the quality of meat, milk, manure, crop, and air

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). According to most dairy studies, supplementing diets with 3-NOP (40 to 80 mg/kg DM) has not improved animal performance (Arndt *et al.*, 2021; Jayanegara *et al.*, 2018), nor did it affect DMI, milk yield, milk component yield or feed efficiency. It did slightly increase body weight gain (Haisan *et al.*, 2014; Hristov *et al.*, 2015a; van Gastelen *et al.*, 2020), and small changes in milk components (Jayanegara *et al.*, 2018; Schilde *et al.*, 2021) were noted in some studies. Depending on the diet and the 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 a high 3-NOP dose (200 mg/kg DM). An enhanced gain to feed ratio (by 2.5 percent to 5 percent) was reported in some (Alemu *et al.*, 2021; Vyas *et al.*, 2016, 2018) but not all beef cattle studies (K. Beauchemin, personal communication). The impact of 3-NOP on rumen fermentation was assessed for both beef and dairy cattle. A distinct shift towards greater propionate and butyrate concentration and a reduction in the acetate to propionate concentration ratio upon supplementation with 3-NOP were observed (Jayanegara *et al.*, 2018). It has been hypothesized that this shift might lead to higher energy and glucose availability for the animal (Ungerfeld, 2018; Ungerfeld, Beauchemin and Muñoz, 2022). Ruminal pH was also shown to be higher with 3-NOP supplementation, indicating a reduced risk of rumen acidosis (Jayanegara *et al.*, 2018).

5.18.7 Safety and health aspects

The safety of 3-NOP for animal use and humans consuming meat and milk from animals fed 3-NOP was assessed and its use approved on the basis of an extensive set of studies, initially by regulatory officials in Brazil and Chile, followed by the European Union (Bampidis *et al.*, 2021), Argentina, Australia, Pakistan, Switzerland, Türkiye and Uruguay, and it is currently being assessed by regulatory officials in other jurisdictions. The EU market authorization process for feed additives declared the product to be safe for dairy cows and cows intended for reproduction, the consumer and the environment, when administered 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 hydrolysed in the rumen post-dosing (2 to 3 hours; Thiel *et al.*, 2019a) to 1, 3-propanediol and nitrate, which are low-toxicity compounds naturally occurring in the rumen of non-3-NOP supplemented cows. 1, 3 propanediol is further hydrolysed 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 radioactivity of dosed ^{14}C-labelled 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 were 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. Based on extensive examination of genotoxicity, Bampidis *et al.* (2021) did not rule out that 3-NOP may be genotoxic. However, those findings were not considered relevant for the safety of the target species and consumers, because 3-NOP is rapidly metabolized in the rumen and post-absorption. Bampidis *et al.* (2021) stated that feeding 3-NOP exposes the consumer to 3-nitrooxypropionic acid (NOPA), a non-genotoxic compound. However, NOPA is an intermediary metabolite of absorbed 3-NOP in rodents, and not a main end product for 3-NOP metabolism and elimination. Furthermore, in ruminants, rumen metabolism of 3-NOP sharply decreases the plasma concentration of NOPA after ingestion of 3-NOP in comparison to rodents. Even with doses of 3-NOP at twice the maximal recommended dose, the passage of NOPA to the milk of dairy cows was almost always undetectable or occurred at a very low concentration, and human intake of NOPA from milk produced by cows fed 3-NOP was estimated to be negligible and safe for consumers (Bampidis *et al.*, 2021). Just as other feed additives, 3-NOP should be handled with the necessary care, both in the supply chain and by farmers and farm workers, to avoid potential irritation.

5.18.8 Adoption potential

3-NOP is already approved for use in Brazil, Chile and the European Union, 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 CH_4. Preliminary studies using a prototype slow-release form of 3-NOP have proven successful (Muetzel *et al.*, 2019) but will require further testing in larger scale studies. 3-NOP requires approval by regulatory officials. 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. A survey regarding farmer or sector experience with 3-NOP in dairy diets will start in the Kingdom of the Netherlands in 2022. We are not aware of the existence of surveys regarding consumer acceptance of 3-NOP.

5.18.9 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 refining. The efficacy of using 3-NOP in long-term beef and dairy cow studies under various conditions requires additional validation. Studies that combine 3-NOP with other mitigation strategies are lacking. Further evaluation of the GHG emissions from manure of animals fed 3-NOP is needed, although no negative impacts on digestibility have been observed. More research is also needed to achieve a better understanding of the changes in the rumen and the whole animal metabolism that result from inhibiting rumen methanogenesis, which may potentially translate into an improved animal productivity.

5.19 RUMEN MANIPULATION: IMMUNIZATION AGAINST METHANOGENS

5.19.1 Description

Vaccination against rumen methanogens.

5.19.2 Mode of action

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

5.19.3 Efficacy

The effects on CH_4 yield have been mild or non-existent in sheep (Wright *et al.*, 2004; Leslie, Aspin and Clark, 2008; Williams *et al.*, 2009) and goats (Zhang *et al.*, 2015). The antibodies against methanogens decreased CH_4 production in mixed rumen cultures, according to a non-peer reviewed study (Baker and Perth, 2000), and effects were variable as stated by 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 concentrations of antibodies in saliva, estimated to result in up to 10^4 molecules of antigen-specific IgG per methanogen cell in the rumen (Subharat *et al.*, 2016).

5.19.4 Potential to combine with other mitigation strategies

It appears feasible, but experiments have not been conducted to investigate synergisms among mitigation strategies. Provided that vaccines can be efficacious, the use of other additives that directly target methanogens could amount to duplicative mitigation.

5.19.5 Effects on other emissions

It seems likely that CO_2 emissions from fossil fuels generated by manufacturing, packaging, transporting and storing vaccines would be minimal. It is assumed that digestibility and nutrient excretion will not be impacted by vaccination.

5.19.6 Productivity and the quality of meat, milk, manure, crop, and air

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, 2000). If an efficacious vaccine that clearly demonstrates a reduction in CH_4 emissions is developed, the effects on animal performance and product quality would need to be thoroughly evaluated, as other CH_4-mitigating measures do not normally have such beneficial effects.

5.19.7 Safety and health aspects

Unknown at present, but it can be supposed to pose a low risk, as antibodies naturally occur in animal tissues eaten by humans. Once developed, the vaccines would need to go through appropriate regulatory approval processes.

5.19.8 Adoption potential

This enteric CH_4 mitigation strategy is attractive for extensive grazing production systems that use supplements infrequently or not at all and that have a limited potential for intensification. The fact that it is unlikely to significantly affect the emissions of other GHGs, that it would be easy and safe to apply, not require specialized technical skills and likely be acceptable to government agencies and consumers also makes it interesting. If proven effective, the development of vaccines against methanogens is perhaps the most desirable approach for controlling CH_4 emissions due to extensive ruminant production systems.

5.19.9 Research required

This CH_4 mitigation strategy is at present at a proof-of-concept stage, as shown by antibody responses that the vaccination induced in serum, saliva and rumen fluid (Wright *et al.*, 2004; Zhang *et al.*, 2015; Subharat *et al.*, 2015, 2016). The 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. The genome sequencing 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). It has been demonstrated that antibodies against methanogens have some stability in rumen fluid (Subharat *et al.*, 2015) and that they 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 minor 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, Aspin and Clark, 2008; Williams *et al.*, 2009; Zhang *et al.*, 2015) than in mixed cultures (Cook *et al.*, 2008). Vaccination did not affect the abundance of methanogens but it increased their diversity, suggesting that the lack of effects on CH_4 production might be due to a lack of broad-spectrum vaccines used against the rumen methanogenic community (Williams *et al.*, 2009). More work is required to select appropriate antigens present across diverse rumen methanogens clades, determine their efficacy against cultivable rumen methanogens as well as *in vitro* mixed batch and continuous cultures, develop adequate adjuvants and assess the persistence of immune responses across ruminant populations.

5.20 RUMEN MANIPULATION: BROMOFORM-CONTAINING SEAWEEDS (*ASPARAGOPSIS* SP.)

5.20.1 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 *Asparagopsis 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.*, 2019a, 2021; Stefenoni *et al.*, 2021).

5.20.2 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 the cobamide-dependent methyl transfer (Wood *et al.*, 1968) into mercaptoethanesulfonate (coenzyme M) to produce methyl-coenzyme M, which is itself the methyl donor in the last step of methanogenesis (Harms and Thauer, 1996).

5.20.3 Efficacy

In vivo studies with sheep, steers and dairy cows reported dose-dependent decreases between 9 to 98 percent of CH_4 production when supplementing the diet with *Asparagopsis* (Li *et al.*, 2016; Kinley *et al.*, 2020; Roque *et al.*, 2019a, 2021; Stefenoni *et al.*, 2021). A severe inhibition of methanogenesis (>50 percent) was observed with 1 percent or less *Asparagopsis* in the diet (Li *et al.*, 2016; Kinley *et al.*, 2020; Roque *et al.*, 2019a, 2021). The efficacy of *Asparagopsis* on CH_4 mitigation depends on its concentration of bromoform, which ranged from 3.28 to 39 × 10^{-3} µg/kg DMI in different studies (Kinley *et al.*, 2020; Roque *et al.*, 2019a, 2021). Additionally, *Asparagopsis* may be more effective at decreasing CH_4 production in high-concentrate than in high-forage diets (Roque *et al.*, 2021). Stefenoni *et al.* (2021) concluded that the efficacy of *Asparagopsis* gradually diminished 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, Roque *et al.* (2021) did not report a loss of efficacy from the product over a five-month period.

5.20.4 Potential to combine with other mitigation strategies

How the use of *Asparagopsis* combines with other mitigation strategies has not been experimentally examined, but it is expected to have potential when different bioactive components or modes of action are involved. Combining it with other CH_4 mitigation strategies may allow decreasing the concentration of bromoform in the diet to alleviate potentially detrimental effects on DMI, health and safety (see section 5.20.7 below).

5.20.5 Effects on other emissions

The CO_2eq emissions from growing, harvesting, processing (drying), storing and transporting *Asparagopsis* on a large scale need to be considered in an LCA to determine the net impact on GHG intensity of meat and milk production. The assessment of ozone-related environmental impacts probably deserves consideration as bromoform is documented as an ozone-depleting substance (Papanastasiou, McKeen and Burkholder, 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.*, 2022).

5.20.6 Productivity and the quality of meat, milk, manure, crop, and air

A dietary supplementation with *Asparagopsis* reduced feed intake in most (Roque *et al.*, 2019a, 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 sheep (Li *et al.*, 2016) and dairy cows (Muizelaar *et al.*, 2021) that had been 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 the reduction of feed intake. The inclusion of *Asparagopsis* in the diet had no effects on carcass or meat quality (Kinley *et al.*, 2020; Roque *et al.*, 2021). Milk production was decreased due to supplementation with 1 percent DM *Asparagopsis* because of the DMI reduction (Roque *et al.*, 2019a; Stefenoni *et al.*, 2021). The effect of *Asparagopsis* on manure GHG emission is unknown.

5.20.7 Safety and health aspects

Long-term oral exposure of animals to bromoform can cause liver and intestinal tumors. It is therefore classified in the United States as a Group B2, probable human carcinogen (Environmental Protection Agency [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 an accumulation of iodine in meat was reported (Roque *et al.*, 2021). Roque *et al.* (2019a) and Stefenoni *et al.* (2021) did not find any passage of bromoform to milk from dairy cows fed *Asparagopsis*, although Muizelaar *et al.* (2021) with no control animals in their study, reported passage of bromoform to milk in non-adapted dairy cows. According to Stefenoni *et al.* (2021), iodine and bromide accumulated in milk from cows fed *Asparagopsis*. The ruminal mucosa of animals that consumed *Asparagopsis* showed pathological signs in sheep (Li *et al.*, 2016) and cows (Muizelaar *et al.*, 2021). As in the case of other feed additives, *Asparagopsis* should be handled with the necessary care.

5.20.8 Adoption potential

Bromoform-containing seaweed will not be ready for adoption until various challenges are successfully met, notably the potential safety risks it poses for animals and humans. So far, *in vivo* studies have used wild-harvested *Asparagopsis* with variable bromoform content (Vijn *et al.*, 2020). A successful adoption will require consistently growing and processing seaweed species to accumulate halogenated compounds and maintain their concentrations when transporting, handling and feeding animals. Bromoform and other halogenated alkanes are a concern for animal, food and environmental safety that will need to be resolved, if this strategy is to be adopted. The passage of bromoform to milk in animals fed *Asparagopsis* is inconsistent (Roque *et al.*, 2019a; Stefenoni *et al.*, 2021; Muizelaar *et al.*, 2021). Animal and food safety concerns relative to the transfer and accumulation of iodine and bromide in milk and meat will also need to be examined and successfully addressed for seaweed-based mitigants to be adopted. If the inclusion of bromoform-containing seaweeds at low levels could be regarded as acceptable, the administration of pure bromoform in other forms (e.g. slow release) could equally be considered. This criterion might be extended to other methanogenesis-inhibiting haloalkanes, such as chloroform and bromochloromethane, although an encapsulated form of bromochloromethane was deemed unlikely to be approved for commercial use in Australia (Tomkins, Colegate and Hunter, 2009), as the production of bromochloromethane was banned in 2002 under the Montreal Protocol because of its ozone-destroying properties. Thus, it currently cannot be considered for use as a CH_4 mitigation technology. Finally, as with any other additive, the inclusion of *Asparagopsis* in animal feed will involve an extra expenditure, and therefore its cost-effectiveness must be considered.

5.20.9 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, and yet the potential threat that bromoform poses to animals, farmers, consumers and the environment are a concern when it comes to its use. The metabolic fate of bromoform ingested by animals and the distribution 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) should 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.

5.21 RUMEN MANIPULATION: OTHER SEAWEEDS

5.21.1 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.*, 2020).

5.21.2 Mode of action

Seaweeds have a highly variable chemical composition, depending on 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 (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 – including *Laminaria digitata* (brown), *Macrocystis pyrifera* (brown), *Pterocladia capillacea* (red), *Rhodymenia californica* (red), *Ulva intestinalis* (green) and *Ulva* spp. (green) – produce bromoform, but they store less in biomass than *Asparagopsis* (Carpenter and Liss, 2000).

5.21.3 Efficacy

Several seaweeds have been identified as having a 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 not known and needs to be investigated.

5.21.4 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 that have similar modes of action.

5.21.5 Effects on other emissions

The CO_2eq emissions of growing, harvesting, processing (drying), storing and transporting seaweed at a large scale need to be considered in an 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 the possibility of purifying or extracting seaweed bioactives, which would decrease emissions related to drying and transportation. The potential fixation of CO_2 through photosynthesis was deemed to contribute towards mitigation in the emission of GHGs (McCauley *et al.*, 2020); however, this is likely a minor benefit as most of the CO_2 would be released into the atmosphere by the animals or humans consuming animal products, as is the case with other feedstuffs.

5.21.6 Productivity and the quality of meat, milk, manure, crop, and air

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

5.21.7 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). A high-iodine concentration was found in the milk of cows fed the brown seaweed *Ascophyllum nodosum* (Antaya *et al.*, 2015), a finding that limits the adoption potential for dairy cows. Health problems were reported in sheep accustomed 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 seaweed inclusion in the diet.

5.21.8 Adoption potential

The prospects for immediate adoption are low, but there is a good potential for adoption in the future, especially in coastal areas with native seaweeds where these may be consumed wet. Otherwise, seaweed needs to be dried rapidly, before it becomes mouldy. Low-temperature drying reduces the inactivation of biochemical compounds (Makkar *et al.*, 2016). A sustainable production of seaweeds will be required to meet the demand (Abbott *et al.*, 2020). Poor palatability due to high-salt content and toxicity may be limitations, particularly when offered as free choice to animals or to grazing ruminants. It might be more 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 be contingent on cost-benefit analysis and regional availability. Government agencies' approval will depend on the content of potentially toxic minerals, which may have to be analysed from batch to batch unless a consistent composition can be shown. Inclusion of seaweeds in ruminant diets may be acceptable to consumers, provided there is no risk of toxicity and no off-flavours in meat or milk.

5.21.9 Research required

Several aspects still need to be considered if seaweed use is to reduce enteric CH_4 emissions (Vijn *et al.*, 2020). Substantial *in vivo* research is needed to establish the CH_4 mitigation potential and the environmental impacts of seaweed farming. Bioactive compounds and the growth conditions that promote these bioactives are key. Product palatability, best additive delivery method, quality control and the potential to extract bioactive compounds will need to be determined. 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 to be carried out.

5.22 RUMEN MANIPULATION: DEFAUNATION

5.22.1 Description

Some rumen methanogens are ectosymbionts (Vogels, Hoppe and Stumme, 1980) or endosymbionts (Finlay *et al.*, 1994) of protozoa, which supply them with H_2 and formate. It has been proposed that the 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 by using chemicals or 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.

5.22.2 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 favours carbohydrate fermentation. It has been estimated that protozoa-associated methanogens contribute between 9 and 37 percent of CH_4 produced in rumen fermentation (Newbold, Lassalas and Jouany, 1995; Newbold *et al.*, 2015). Since the presence of protozoa is not strictly necessary to rumen function and animal survival (Morgavi, *et al.*, 2010; Newbold *et al.*, 2015), 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, Williams and McEwan, 2020), but protozoa-associated methanogens seem to be more active CH_4 producers than free-living methanogens (Jami and Mizrahi, 2020). Protozoa may also favour methanogens by protecting them from oxygen toxicity (Morgavi *et al.*, 2010).

5.22.3 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 the 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 confidence interval) decrease in CH_4 yield. The recent meta-analysis by Arndt *et al.* (2021) concluded 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 established (Morgavi *et al.*, 2010; Guyader *et al.*, 2014). The meta-analysis by Li *et al.* (2018) suggested a long-term adaptation of CH_4 production to defaunation. Similarly, Morgavi *et al.* (2012) showed a numerical decrease in CH_4 production of wethers defaunated in the short term, 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 evidence of a long-term adaptation to defaunation (Morgavi, Jouany and Martin, 2008).

5.22.4 Potential to combine with other mitigation strategies

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

5.22.5 Effects on other emissions

Because it can improve the efficiency of N utilization and decrease N elimination in urine (Eugène, Archimède and Sauvant, 2004; Newbold *et al.*, 2015), defaunation may decrease the emissions of N_2O from N voided into the environment in animal urine. The fibre excreted in manure may increase, as defaunation has been shown to decrease fibre digestibility (Eugène, Archimède and Sauvant, 2004; Newbold *et al.*, 2015; Li *et al.*, 2018).

5.22.6 Productivity and the quality of meat, milk, manure, crop, and air

The meta-analyses by Eugène, Archimède and Sauvant (2004), and Newbold *et al.* (2015), reported beneficial effects of defaunation on weight gain, feed conversion efficiency and wool production, with either no effects on DMI (Eugène, Archimède and Sauvant, 2004) or decreased DMI (Newbold *et al.*, 2015). The meta-analysis by Arndt *et al.* (2021) did not suggest any effects of defaunation on DMI or weight gain. There were consistent decreases in rumen, overall tract OM and NDF digestibility, rumen VFA and ammonia concentration, an increased microbial nitrogen production, and a shift in nitrogen excretion from urine to feces (Eugène, Archimède and Sauvant, 2004; Newbold *et al.*, 2015; Li *et al.*, 2018). Decreases in fibre 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 (Eugène, Archimède and Sauvant, 2004). Protozoal numbers were found to associate positively with DMI and NDF digestibility (Guyader *et al.*, 2014). According to Newbold *et al.* (2015), defaunation decreases biohydrogenation of polyunsaturated fatty acids.

5.22.7 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 (Eugène, Archimède and Sauvant, 2004; Newbold *et al.*, 2015). There is no evidence to suggest that defaunation could harm the animal's health and the environment, or put those who consume animal products at risk.

5.22.8 Adoption potential

Defaunation results in mild decreases in CH_4 emissions. Moreover, defaunating and maintaining defaunated animals in production settings poses a challenge. Consequently, defaunation has not been recommended as a CH_4 mitigation strategy for practical reasons (Hristov *et al.*, 2013a; Newbold *et al.*, 2015; Huws, Williams and McEwan, 2020).

5.22.9 Research required

There are differences among protozoa with regard to their associated methanogens and contribution to CH_4 production, as well as 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-fibre degrading activity of this order (Huws, Williams and McEwan, 2020), but this kind 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, fibre digestion, utilization of soluble carbohydrates, oxygen scavenging, as well as their rumen sequestration and passage (Firkins and Mackie, 2020).

5.23 RUMEN MANIPULATION: ALTERNATIVE ELECTRON ACCEPTORS

5.23.1 Description

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

5.23.2 Mode of action

Organic alternative electron acceptors are carboxylic acid intermediates of pathways in rumen fermentation that either incorporate metabolic hydrogen themselves (fumarate, which is reduced to succinate in the propionate randomizing pathway) or can be metabolized into 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 dissociate when added as salts to the diet and, when they are reduced, draw electrons away from CH_4 formation. A complete nitrate reduction yields primarily ammonium, which can be incorporated into microbial N or absorbed through the rumen wall. Nitrate reduction via intermediate nitrite also exerts a direct inhibition of methanogens (Hulshof *et al.*, 2012; Latham *et al.*, 2016; Yang *et al.*, 2016). The reduction of sulfate yields hydrogen sulfide, which can be expelled as a gas (dissimilatory reduction) or incorporated into microbial amino acids and cofactors (assimilatory reduction; Drewnoski, Pogge and Hansen, 2014).

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

5.23.3 Efficacy

The mode of action of alternative electron acceptors imposes a theoretical limitation on their efficacy, due to the stoichiometry of metabolic hydrogen incorporation 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 were below the theoretical expectation for fumarate and malate, because fumarate and malate were apparently partially converted to acetate rather than to propionate, thus releasing [2H] instead of incorporating it [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 in. 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 for.

Stoichiometrically, 4 moles of hydrogen are redirected towards the 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 a lower CH_4 decrease, and the direct toxicity of nitrate reduction via 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 lasting 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 subsequent 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 the nitrate dose, type of animal (greater efficacy in dairy than in beef cattle) and DMI. Its efficacy diminished with increasing DMI. On average, their findings amounted to 364 g of sodium nitrate decreasing CH_4 production by 10 percent in a cow consuming 24 kg of DM per day and producing 328 g of CH_4 per day, 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.

5.23.4 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 it interacted synergistically with canola oil (Villar *et al.*, 2020). Nitrate interacted negatively with defaunation on CH_4 production (Nguyen, Barnett and Hegarty, 2015). The addition of nitrite-reducing bacterium *Propionibacterium acidipropionici* did not interact with nitrate addition (nor had an effect) on CH_4 emissions from sheep (de Raphélis-Soissan, 2014). To compensate for the small reduction in feed consumption, diets containing nitrate could be associated with supplementation in oils and fats, increasing both the energy density and the mitigation potential of the diet.

5.23.5 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 be considerable due to these compounds's relatively important dietary concentrations 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 prevent additional CO_2 emissions.

Apart from the emissions related 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 it is supplemented to a N-deficient diet, nitrate should isonitrogenously replace another N source in order not to increase N voided into the environment, which can potentially increase N_2O emissions (Beauchemin *et al.*, 2020).

5.23.6 Productivity and the quality of meat, milk, manure, crop, and air

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 the production of rumenic acid, which may potentially improve the nutritional qualities of animal products. Most studies have not shown any effects of a moderate inclusion of malate on DMI, whereas fumarate effects have been more inconsistent, with decreased DMI in some studies and lack of effects in others. Malate supplementation did not affect weight gain or milk production in some studies and improved them in others, while fumarate supplementation has not affected milk production (Carro and Ungerfeld, 2015).

Overall, the benefits of nitrate supplementation on animal productivity have not been demonstrated (Yang *et al.*, 2016), except when 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).

5.23.7 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 European Union and the United States (Carro and Ungerfeld, 2015). Nitrate fermentation via 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 an increase in the reduction rate 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. The inclusion of nitrate in animal feeds is not approved in the United States and in 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). The high dietary sulfate results in hydrogen sulfide production, which can cause polioencephalomalacia (Drewnoski, Pogge and Hansen, 2014).

5.23.8 Adoption potential

Feeding fumarate and malate to ruminants is largely limited by cost, given 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 the gradual adaptation of animals and can only be recommended for farms in which feed intake is carefully managed. In addition, the nitrate content of herbage and forages needs to be taken into account to prevent excessively high levels. The potential rise in N_2O emissions as a result 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).

5.23.9 Research required

In vivo experiments with combinations of methanogenic inhibitors and fumarate or malate to examine the incorporation of accumulated dihydrogen into propionate production would be of interest. The selection of grasses with a malate content that stays high throughout 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, 2005b), but they only numerically lowered nitrite and methaemoglobin concentration in plasma *in vivo* (de Raphélis-Soissan *et al.*, 2014). More efforts to examine different doses and frequencies of administration of nitrite-reducing bacteria are recommended, in addition to isolating new nitrite reducers from the rumen environment.

5.24 RUMEN MANIPULATION: ESSENTIAL OILS

5.24.1 Description

Essential oils are complex mixtures of volatile lipophilic secondary metabolites, traditionally extracted from plants by boiling water and steam distillation; other methods include solvent extraction, supercritical CO_2 extraction and expression extraction. They are specific to plants and responsible for each plant's characteristic flavour and fragrance (Benchaar and Greathead, 2011). Essential oils can be extracted from many parts of a plant, such as the leaves, flowers, stem, seeds, roots and bark (Benchaar *et al.*, 2008). When extracted and concentrated, essential oils may exert antimicrobial activities on a wide variety of microorganisms, including bacteria, protozoa and fungi (Deans and Ritchie, 1987; Sivropoulou *et al.*, 1996; Chao, Young and Oberg, 2000). In addition to plant sources, essential oils can be chemically synthesized for commercial use. Chemically, essential oils are variable mixtures of terpenoids, mainly monoterpenes and sesquiterpenes, although diterpenes may also be present, as well as 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).

5.24.2 Mode of action

Most essential oils exert their antimicrobial activities by interacting with processes related to the bacterial cell membrane, including electron transport, ion gradients, protein translocation, phosphorylation and other enzyme-dependent reactions (Ultee, Kets and Smid, 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, such as 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, a reductive step required for the disposing of reducing equivalents mainly produced by this group of bacteria (Owens and Goetsch, 1988). On the other hand, gram-negative bacteria are involved in the fermentation pathways associated with the production of propionate and succinate (Russell and Strobel, 1989; Russell, 1996). When this group of bacteria is predominant in the rumen, rumen fermentation patterns shift towards more propionate (H_2-using pathway) and less acetate (H_2-producing pathway) production, thus reducing the availability of hydrogen for ruminal methanogenesis (Bergen and Bates, 1984). Neither methanogens nor protozoa, which are in a symbiotic relationship with methanogens, appear to be sensitive to the activity of essential oils (Benchaar and Greathead, 2011).

5.24.3 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, Trabalza-Marinucci and You, 2016). The addition of Mootral as an ingredient at 9.9 or 18.0 percent of the substrate incubated in semi-continuous cultures markedly decreased CH_4 production by 95 percent and by more than 99 percent, respectively, while the total VFA production was increased (Eger *et al.*, 2018). In another Rusitec study, the inclusion of Mootral – once more at levels of a dietary ingredient (17.7 percent DM) – in semi-continuous cultures (Rusitec) eliminated CH_4 production after 4 days of Mootral inclusion, following which CH_4 production resumed. Yet again, the total VFA production strongly increased with Mootral inclusion (Brede *et al.*, 2021). However, the results from *in vivo* studies have been less conclusive (Benchaar and Greathead, 2011). Essential oils with a 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* when added at high levels relative to feed substrate, although their efficacy was not confirmed or less pronounced *in vivo* (Klevenhusen *et al.*, 2011; Benchaar, 2020, 2021). Commercial products containing various essential oils have been shown in a very limited number of studies to have a potential to decrease CH_4 production. For instance, a commercial product of oregano oil (Orego Stim®, Anpario plc, Nottinghamshire, United Kingdom) fed to lactating dairy cows was reported to reduce CH_4 yield by 22 percent (Kolling *et al.*, 2018). Feeding 15 g per 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 in the final week (week 12) of the study (Roque *et al.*, 2019b). In another study, Mootral did not affect CH_4 emissions by feedlot steers in week 8 of the trial, but decreased the total CH_4 and CH_4 yield in week 29 by 26 and 30 percent, respectively, when feeding a high-concentrate diet (Bitsie *et al.*, 2022). A 10 percent decrease in CH_4 yield was reported for a commercial mixture of coriander, eugenol, geranyl acetate and geraniol (Agolin® Ruminant; Agolin S.A., Bière, Switzerland) when fed to dairy cows at the rate of 1 g per day (Belanche *et al.*, 2020). Based on the literature available to date, it appears that essential oils and their compounds may hold promise for CH_4 mitigation, but further work on animal feeding – especially long-term studies – is required to determine their efficacy.

5.24.4 Potential to combine with other mitigation strategies

Opportunities exist to combine this with other mitigation strategies that have different or similar mechanisms of action. For instance, given that essential oils have no effect on protozoa, combining these substances with other phytocompounds known for their antiprotozoal activity (e.g. saponins) may increase their mitigating action. Monensin is known for its inhibitory effect on ruminal methanogenesis, due to negative effect on gram-positive bacteria, which increases propionate production at the expense of acetate. 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 the mitigating effects.

5.24.5 Effects on other emissions

Some essential oils and their compounds have been reported to reduce dietary protein degradation *in vitro*, although *in vivo* studies have been inconsistent (Cobellis, Trabalza-Marinucci and Yu, 2016). If this is accompanied by a reduction in urinary N excretion, the presence of N_2O and ammonia may potentially be reduced.

5.24.6 Productivity and the quality of meat, milk, manure, crop, and air

In general, feeding essential oils to ruminants does not affect animal productivity or product (milk, meat) quality (Benchaar, Hristov and Greathead, 2009). Adverse effects of essential oils on feed digestion were reported (Benchaar, Hirstov and Greathead, 2009; Cobellis, Trabalza-Marinucci and Yu, 2016) and, if such effects occur in animals, it would have a negative impact on productivity. There is a potential for the transfer of compounds present in essential oils (e.g. terpenes) to milk (Lejonklev *et al.*, 2013) and meat (de Oliveira Monteschio *et al.*, 2017), which can positively or negatively affect the quality and organoleptic properties of meat and milk. The amount and chemical composition of manure are unlikely to be affected, but if the feed digestion in the rumen is depressed, the amount of manure excreted and associated emissions could increase.

5.24.7 Safety and health aspects

Little is known about how safe the use of essential oils is in ruminant nutrition. At the doses generally recommended by the feed industry, the probability of essential oils being toxic to animals is low. However, caution should be exercised, especially if the 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. Yet essential oil compounds such as estragole and methyl eugenol were deleted from the list in 2001 due to their genotoxic properties (Burt, 2004). The 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), which suggests that caution must be taken by users when handling such feed additives.

5.24.8 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 essential oils or adapt to them over time. It remains challenging 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 in a way that controls the release of the active ingredient once added to the animal's diet. However, the long-term stability of products and the need for controlled storage conditions can be limiting factors. Finally, unless there are clear productivity benefits, the additional costs involved may discourage some from adopting this particular strategy.

5.24.9 Research required

The potential of essential oils to mitigate enteric CH_4 emission has been mostly examined *in vitro* and there is a 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 required to identify the ones that are most effective in reducing 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 specific dietary conditions that lend themselves to CH_4 mitigation without negatively affecting animal productivity. Moreover, the favourable effects obtained *in vitro* may be due to microbial adaptation *in vivo*. Consequently, additional long-term animal studies are needed to investigate the extent to which microbes adapt to these substances. Further work is also required to assess the transfer of essential oils into animal products and the potential impact this may have on the quality of animal products.

5.25 RUMEN MANIPULATION: TANNIN EXTRACTS

5.25.1 Description

A dietary supplement of tannin-rich extracts.

5.25.2 Mode of action

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

5.25.3 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's maturity stage and growing conditions. The anti-methanogenic effect of tannin-containing feeds is variable due to factors such as the plant source, structural complexity (CT and HT have high and low molecular weights, respectively), dose, the types 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 that CH_4 production decreased with increasing dietary tannin levels, with a more consistent, discernible effect observed when tannin inclusion was greater than 20 g/kg dietary DM (Jayanegara, Leiber and Kreuzer, 2012). Studies conducted on 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, Dall-Orsoletta and Ribeiro-Filho, 2017; Denninger *et al.*, 2020), CT from *Sericea lespedeza* with the addition of 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 to 45 percent and the CH_4 mitigation effects were observed in both forage- 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 to suggest that feeding tannins could exhibit long-term CH_4 mitigating effects (Staerfl *et al.*, 2012a; Duval *et al.*, 2016; Aboagye *et al.*, 2018).

5.25.4 Potential to combine with other mitigation strategies

Combining tannin extracts with other CH_4 inhibitors appears feasible, 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 a tannin extract from *Swietenia mahogani* was combined with a *Sapindus* saponin extract *in vitro* (Jayanegara *et al.*, 2020), and when supplementing a 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 a tannin extract from *Acacia mearnsii* was combined with nitrate (Adejoro *et al.*, 2020), when a *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).

5.25.5 Effects on other emissions

If tannin supplementation decreases fibre digestibility, the excretion of fermentable OM would be expected to increase, which might increase CH_4 losses from manure (Gerber *et al.*, 2013b). However, Staerfl *et al.* (2012a) showed that feeding acacia tannin extracts reduced fibre digestibility without affecting CH_4 emission from manure. Tannins have been shown to inhibit manure CH_4 emission when ingested or added directly to manure (Whitehead, Spence and Cotta, 2013; Pham *et al.*, 2017). The anti-methanogenic effect of ingested tannins may thus persist in manure. In addition, numerous studies (especially those involving high-protein diets) have demonstrated 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 lowered (Powell, Aguerre and Wattiaux, 2011; Duval *et al.*, 2016).

5.25.6 Productivity and the quality of meat, milk, manure, crop, and air

Tannin-containing feeds can be less palatable because tannins bind to salivary glycoproteins, which results in an astringent taste (Mueller-Harvey, 2006). Moreover, feeding high concentrations of tannins (i.e. >50 g/kg DM) can activate antinutritional properties that negatively impact intake, fibre and protein digestibility, and animal performance (Aboagye and Beauchemin, 2019). Supplementing purified tannin extracts 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 avert the negative effects of tannins, feeding a low to moderate threshold dose has been recommended (i.e. <30 to 50 g/kg DM diet), as this 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). Lastly, the dietary supplementation of tannins can improve the fatty acid composition, oxidative stability, and the sensory qualities of meat and milk (Salami *et al.*, 2019; Frutos *et al.*, 2020).

5.25.7 Safety and health aspects

Compared with CTs, HTs are more susceptible to microbial hydrolysis in the gut, since the metabolites they produce may have potentially toxic effects on the animal post-absorption (Reed, 1995; McSweeney *et al.*, 2001). Feeding high concentrations (i.e. >50 g/kg DM diet) of HTs may cause such adverse effects as 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 could in turn increase the risk of toxicity to plant compounds such as cyanogenic glycosides (Reed, 1995). The negative effect of tannins, particularly HTs, can be prevented through 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 safety risk to animal products destined for human consumption.

5.25.8 Adoption potential

Tannins are secondary metabolites naturally present in plants. The 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 and confined feeding systems. Tannins are safe to apply and this strategy does not require specialized technical skills to be implemented; 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 onerous regulatory approval process compared with chemical feed additives, despite there being some risks of negative side effects.

5.25.9 Research required

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

5.26 RUMEN MANIPULATION: SAPONINS

5.26.1 Description

The dietary supplementation of saponin-containing plants or saponin-rich extracts.

5.26.2 Mode of action

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

5.26.3 Efficacy

The CH_4-mitigating effect of saponins is highly variable depending on the source, chemical structure and dosage of saponins, diet composition, and the adaptation of 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; Jafari, *et al.*, 2019). A meta-analysis of *in vitro* studies found that CH_4 production decreased with increasing levels of saponins, and that the anti-methanogenic effectiveness of saponin sources was as follows: *Yucca* > tea > *Quillaja*, in receding order, although no statistical difference between them was observed (Jayanegara, Wina and Takahashi, 2014). The variability of the anti-methanogenic effect of saponins may be partly linked to the transient nature of their anti-protozoal activity (Wina, Muetzel and Becker, 2005), due to the inactivation of saponins through the 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 for improving 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 modifying the chemical structure of saponins to prevent enzymatic cleavage for microbial degradation (Ramos-Morales *et al.*, 2017a).

5.26.4 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- and concentrate-based diets exhibited an additive CH_4 mitigating effect when combined with garlic oil, nitrate or both, without adverse effects on feed digestion and rumen fermentation (Patra and Yu, 2013, 2014, 2015a, 2015b). Moreover, additive anti-methanogenic effects were observed *in vitro* for *Quillaja* saponin combined with nitrate and sulfate (Patra and Yu, 2014) and for *Sapindus* saponin combined with the 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.

5.26.5 Effects on other emissions

Saponins could reduce rumen NH_3 concentration and improve N use 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, Muetzel and Becker, 2005; Patra and Saxena, 2009a). Consequently, feeding saponins – particularly *Yucca* saponins – has the potential 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 in improving N use efficiency could reduce N losses and N_2O emissions from manure (Yurtseven *et al.*, 2018).

5.26.6 Productivity and the quality of meat, milk, manure, crop, and air

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 the dietary inclusion of higher saponin levels did not have adverse effects on feed digestion and rumen fermentation (Jayanegara, Wina and Takahashi, 2014). Although the benefits of saponins on animal productivity are uneven, their anti-protozoal effect could increase the efficiency of microbial protein synthesis and enhance the supply of metabolizable protein, thus improving animal performance especially for roughage-based diets (Wina, Muetzel and Becker, 2005; Patra and Saxena, 2009a). Moreover, there are indications that dietary saponins could have 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 only limited improvements have been observed in the quality of milk (Vasta and Luciano, 2011; Szczechowiak *et al.*, 2016; Toral *et al.*, 2018).

5.26.7 Safety and health aspects

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

5.26.8 Adoption potential

Supplementing ruminant diets with saponin-containing plants or extracts could readily be adopted as a strategy. The production of saponin extracts is scalable and some saponin extracts (e.g. *Yucca* and *Quillaja* bark saponins) have been commercially produced for application in the pharmaceutical, food and cosmetic industries (Güçlü-Üstündağ and Mazza, 2007). At least one patent exists which involves the use of saponins in ruminant feeding (Aoun *et al.*, 2003). Saponins are safe to apply and do not require specialized technical skills for the formulation of diets.

5.26.9 Research required

Yucca, tea and *Quillaja* saponins have shown potential for reducing CH_4 emissions but further studies are required to establish the optimum dosage and their interaction with basal diets with a view to improving our understanding of their anti-methanogenic effects in the long term. The combination of *Quillaja* saponin with other methanogenesis inhibitors (particularly nitrate) promises 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, 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.

5.27 RUMEN MANIPULATION: BIOCHAR

5.27.1 Description

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

5.27.2 Mode of action

It has been proposed that biochar enhances biofilm formation (Leng, 2014) and hydrogen transfer among members within the microbial communities (Chen *et al.*, 2014). The transfer of dihydrogen to acceptors other than CO_2 could result in a reduction of enteric CH_4 emissions.

5.27.3 Efficacy

The 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 continuous culture systems (Tamayao *et al.*, 2021a, 2021b). Subsequent studies also failed to detect any impact of biochar on CH_4 emissions in finishing beef cattle (Terry *et al.*, 2019; Sperber *et al.*, 2021). Biomass sources as well as pyrolysis conditions and the secondary treatment of biochar with acidic or alkali solutions may affect the efficacy of biochar. Since biochar appears to be largely indigestible by mixed rumen cultures (Tamayao *et al.*, 2021), reductions in CH_4 emissions could be related to a depression in digestibility, provided that biochar constitutes a significant proportion of the diet.

5.27.4 Potential to combine with other mitigation strategies

Synergistic responses to combining biochar with biofat, an industrial by-product of cashew nut shells, have been shown to reduce CH_4 emissions *in vitro* (Saenab *et al.*, 2020), but a synergistic interaction with other mitigation strategies has not been reported *in vivo*.

5.27.5 Effects on other emissions

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

5.27.6 Productivity and the quality of meat, milk, manure, crop, and air

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.

5.27.7 Safety and health aspects

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

5.27.8 Adoption potential

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

5.27.9 Research required

Biochar appears to have limited potential to lower enteric CH_4 emissions. Alternative biomass sources for pyrolysis and secondary chemical treatments of biochar could still be explored to determine 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; for example, by increasing the level of stable carbon, it can contribute to foster OM accumulation and the retention of manure nutrients within the plant root profile. The use of biochar to lower GHG emissions from livestock should be explored from an LCA perspective, taking into consideration all emissions and sinks throughout the production chain. Basic, long-term research could seek a better understanding of how channelling dihydrogen to different hydrogenotrophic microbial groups is controlled.

5.28 RUMEN MANIPULATION: DIRECT-FED MICROBIALS

5.28.1 Description

Direct-fed microbials, or live microbial additives, are viable microorganisms (e.g. fungi, yeasts, bacteria) that can modify rumen fermentation when ingested by a ruminant. For present purposes, we will focus exclusively on the decrease of CH_4 production, despite there being other objectives, such as stabilizing rumen pH and improving lactate utilization or fibre digestion.

5.28.2 Mode of action

There can be various modes of action. Generally, the purpose of a live microbial additive is 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, the stimulation of pathways that do not produce dihydrogen, or through anaerobic CH_4 oxidation (Jeyanathan, Martin and Morgavi, 2013). For a live microbial additive to be successful, the added microorganism must follow a thermodynamically feasible pathway and the affinity for the reaction substrates must be high. Supplying additional enzyme activity in the form of a live microorganism to a thermodynamically non-spontaneous process is ineffective. For example, a hydrogenotroph should have a low dihydrogen threshold and a high affinity for dihydrogen to compete with hydrogenotrophic methanogens (Ungerfeld, 2020). Another possibility would be to use live microbial additives that produce bacteriocins capable of directly inhibiting methanogens (Gilbert *et al.*, 2009; Jeyanathan, Martin and Morgavi, 2013).

5.28.3 Efficacy

The effects of yeasts, *Aspergillus oryzae*, and of lactic acid bacteria on rumen fermentation and CH_4 production have been inconsistent, and therefore they have not been selected to decrease CH_4 production (Jeyanathan, Martin and Morgavi, 2013; Weimer, 2015). A strategy that has been investigated is the stimulation of propionate production as a pathway incorporating metabolic hydrogen (Jeyanathan, Martin and Morgavi, 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 observed in those fed a diet higher in starch (Berger *et al.*, 2014). Adding nitrate and sulfate-reducing bacteria has also been found to decrease CH_4 production *in vitro* (Jeyanathan, Martin and Morgavi, 2013). The use of reductive acetogens, which have the ability to reduce CO_2 with dihydrogen to produce acetate, in *in vitro* rumen fermentation, had minimal or no effects on CH_4 production unless accompanied by a chemical inhibitor of methanogenesis (Nollet, Demeyer and Verstraete, 1997; Le Van *et al.*, 1998; Lopez *et al.*, 1999). The magnitude of CH_4 oxidation in the rumen was estimated to be minimal (Jeyanathan, Martin and Morgavi, 2013).

5.28.4 Potential to combine with other mitigation strategies

The physicochemical mode of action of live microbial additives charts possible combinations with other CH_4 mitigation strategies. Live microbial additives can enhance the flow of metabolic hydrogen through desirable, thermodynamically feasible metabolic pathways, whose rates are 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, Demeyer and Verstraete, 1997; Le Van *et al.*, 1998; Lopez *et al.*, 1999). In various studies of *in vitro* cultures, nitrite- and nitrate-reducing bacteria have been successfully tested with added nitrate to decrease CH_4 production, enhance the rate of nitrite reduction to ammonium, and prevent the accumulation of nitrite (Jeyanathan, Martin and Morgavi, 2014), as the reductions of nitrate and sulfate are thermodynamically more favourable than methanogenesis in the rumen (Ungerfeld and Kohn, 2006). In an experiment conducted with sheep fed nitrate as a CH_4-mitigation strategy, supplementation with the nitrite-reducing bacterium *Propionibacterium acidipropionici* decreased plasma nitrite concentration only numerically (de Raphélis-Soissan, 2014). Using live succinate or propionate producers could also improve the conversion of fumarate or malate added to propionate.

5.28.5 Effects on other emissions

Growing, storing and transporting live microbial additives would generate some CO_2 emissions from fossil fuels. The impact on the animal's efficiency of using N would have to be evaluated. Overall, the additional emissions of CO_2eq would presumably be low.

5.28.6 Productivity and the quality of meat, milk, manure, crop, and air

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

5.28.7 Safety and health aspects

Approval by regulatory agencies is usually required. In order to secure it, the microorganism in question must be characterized and described in great detail, and the potential for pathogenicity must be discarded. Live microbial additives have been studied to prevent disorders such as acidosis and to decrease the load of pathogens in cattle (Jeyanathan, Martin and Morgavi, 2013; Elghandour *et al.*, 2015). The commercial availability and use of probiotics to meet the nutritional and health requirements of humans and domestic animals is widespread.

5.28.8 Adoption potential

The potential is good, so long as consistent *in vivo* results can be obtained. Live microbial additives in particular stand a very good chance of being adopted as a companion species of chemical inhibitors of methanogenesis; they may be able to improve productivity in certain animal categories and diets by directing metabolic hydrogen accumulated as dihydrogen towards desirable products. Possible changes in the absorption of metabolites would improve animal performance, which would offset the additional feeding costs incurred. Preparations of live microbial additives should remain viable for prolonged periods of time, and be easy to use, store and transport. With a few exceptions, live microbial additives do not persist in the rumen and need to be dosed frequently to have an effect on digestion and fermentation (Weimer, 2015). Live microbial additives may thus not always be suitable for extensive beef production systems, in which the animals have only sporadic contact with their human keepers.

5.28.9 Research required

There is a dearth of *in vitro* and *in vivo* research addressing the optimization of rumen fermentation with live microbial additives, especially when combining them with other CH_4 mitigation strategies such as chemical inhibitors of methanogenesis and alternative electron acceptors. A reflective approach that considers the possible physicochemical limitations of fermentation pathways is recommended. If live microbial additives can be shown to be consistently effective, there will be a need for applied research to determine the optimal frequency, dose and mode of administration. An understanding of physiological and metabolic changes that can occur in the animal will be required to optimize the production and absorption of metabolites so as to improve animal productivity.

5.29 RUMEN MANIPULATION: EARLY LIFE INTERVENTIONS

5.29.1 Description

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

5.29.2 Mode of action

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

5.29.3 Efficacy

Abecia *et al.* (2013) supplemented does with the methanogenesis inhibitor BCM for two months, after they gave birth to twin goat kids. 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 occurred 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.

Conversely, Debruyne *et al.* (2018) did not find long-lasting effects on CH_4 production of coconut oil supplementation to goat kids up to 11 weeks of age 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 any 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.

5.29.4 Potential to combine with other mitigation strategies

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

5.29.5 Effects on other emissions

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

5.29.6 Productivity and the quality of meat, milk, manure, crop, and air

The 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; the performance of the animals later in life was not reported. Supplementation of goat kids with coconut oil was shown to decrease body weight at 28 weeks of age (Debruyne *et al.*, 2018). Saro *et al.* (2018) did not observe any effects of supplementing lambs during 10 weeks with a linseed–garlic oil combination on weight gain at 10 and 20 weeks of age. Meale *et al.* (2021) did not report any 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 favour of control animals, the same as with preweaning concentrate intake. Given that early life treatments are applied for a relatively short period of time, it is possible that negative effects on animal performance, should they occur, might be offset by compensatory growth.

5.29.7 Safety and health aspects

Potential consequences of early life interventions affecting safety and health will depend on the strategy used. However, there will be wash-out periods of several months during the animals' growing phase, before they produce milk or meat. Furthermore, doses of any additives would be much diminished in comparison with an adult animal of a much greater body size, which would also diminish potentially negative environmental effects. Therefore, it is likely that additives that could pose unacceptable levels of risk for the environment or for consumers when fed to adult animals would 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. The long-term efficacy of early life interventions in adult animals may also need to be demonstrated for their usage as a mitigating measure to be approved by regulatory agencies.

5.29.8 Adoption potential

The concept of early life interventions is very attractive, given 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 for 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 as to the efficacy of early life interventions in decreasing CH_4 production later in life and the persistency of the effects observed, although some recent results are encouraging (Meale *et al.*, 2021). The efficacy of early life interventions likely depends on the additive or dietary modification used, the dose, mode and duration of the administration, and the animal species, among other factors.

5.29.9 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 be very important to study the effects of each early life intervention on future animal performance and health, and 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, as well as possible changes in the immune system (Yáñez-Ruiz, Abecia and Newbold, 2015).

5.30 RUMEN MANIPULATION: PHAGE AND LYTIC ENZYMES ACTIVE AGAINST METHANOGENS

5.30.1 Description

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

5.30.2 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.

5.30.3 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). The 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.

5.30.4 Potential to combine with other mitigation strategies

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

5.30.5 Effects on other emissions

Phage or enzyme production would necessitate the establishment of manufacturing facilities that would likely require the use of fossil fuels. Producing phage or lytic enzymes on a commercial scale could prove challenging. It is assumed that phage would not alter N_2O emissions and the efficiency of milk or meat production, however, there is no evidence to suggest either way because the technology has not been assessed outside of the laboratory.

5.30.6 Productivity and the quality of meat, milk, manure, crop, and air

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

5.30.7 Safety and health aspects

This strategy is presumed to be low risk since there are already some therapeutic applications of phage in medicine and food safety, and none of the 65 known archaeal viruses have been linked to animal pathogenesis (Wirth and Young, 2020).

5.30.8 Adoption potential

This enteric CH_4 mitigation strategy would require administration of 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 the self-propagating biocontrol of ruminal methanogens. However, to date lytic phages active against rumen methanogens have not been identified, unlike current candidate lytic enzymes that have been identified as a result of the sequencing of prophage within the methanogen genome (Leahy *et al.*, 2010).

5.30.9 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 harbours a rich and diverse virome (Gilbert *et al.*, 2020), there is but 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 phages 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 their role in the ecology of rumen methanogens is equally vital. More work is required to define the diversity of archaeal viruses (Coutinho, Edwards and Rodriguez-Valera, 2019), as they are likely underrepresented in genomic approaches that characterize the rumen virome. Striving 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 whole range of methanogens that reside in the rumen.

5.31 SUMMARY TABLES

In the following summary tables (Table 2 to Table 4), we delineate the possibilities and barriers for the application of various mitigation strategies aiming to reduce enteric CH_4 emissions from 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. There can be many feed ingredients, including cereal grains, oilseeds and meals, conserved forages, by-products, and premixes containing minerals, vitamins and additives. The feeding frequency and management (i.e. total mixed ration or feed components offered separately) is decided on 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 ruminant 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 through grazing pastures versus the proportion of supplemented feed 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, subtropical, temperate), eco-zone, and so forth. As shown in the tables, the application of each enteric CH_4 mitigation strategy for each of the three production systems is based on the following qualitative assessment:

1. Available knowledge generated by applied research, indicating the number of existing peer-reviewed *in vivo* 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 in 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, while 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 or distribution may still be pending, but those aspects are considered in the last column (as Government and Accessibility, respectively). Moreover, 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 outcomes 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 across countries, regions and farms.

Table 2. Summary of enteric methane mitigation strategies for confined ruminant (dairy, beef or other) systems

Mitigation strategy	*In vivo* studies conducted on CH_4 mitigating F = few (< 5); S = some (5-10); M = many (>10)	Expected CH_4 decrease range H = ≥25 %; M = 15-24 %; L = ≤15 %; 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
Starch concentrate 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	N/A	-	-	-	-	-	-	-
Species (use of forbs, diverse mixtures)	N/A	-	-	-	-	-	-	-
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

(Cont.)

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 the aversion to financial risk is considered under Cost (C); only the 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 on the market 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.

Source: Authors' own elaboration.

Table 3. Summary of enteric methane mitigation strategies for extensive pastoral/ranching systems (beef, dairy and other) based on grazing without supplementation

Mitigation strategy	*In vivo* studies conducted on CH_4 mitigation F = few (< 5); S = some (5-10); M = many (>10)	Expected CH4 decrease range H = ≥25 %; M = 15-24 %; L = ≤15 %; I = increase may be observed; U = unknown (not examined); V = variable g/d	g/kg meat or milk	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
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	N/A	-	-	-	-	-	-	-
Starch concentrate sources and processing	N/A	-	-	-	-	-	-	-
Supplementation of lipids	N/A	-	-	-	-	-	-	-
Forages								
Forage storage and processing	N/A	-	-	-	-	-	-	-
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	N/A	-	-	-	-	-	-	-
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

(Cont.)

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 it is advised to consider both aspects for decision-making;

[3] Resistance to change because of the aversion to financial risk is considered under Cost (C); only the aversion to technical change is considered under Resistance to change (F);

[4] Some are currently available on the market, 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.

Source: Authors' own elaboration.

Table 4. Summary of enteric methane mitigation strategies for mixed grazing with supplementation of concentrates, by-products and conserved forages

Mitigation strategy	*In vivo* studies conducted on CH_4 mitigation F = few (< 5); S = some (5-10); M = many (>10)	Expected CH4 decrease range H = ≥25 %; M = 15-24 %; L = ≤15 %; 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
Starch concentrate 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

(Cont.)

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 Technical support (T) are highly subjective evaluations and will vary considerably among particular producers, but it is advised to consider both aspects for decision-making;

[3] Resistance to change because of the aversion to financial risk is considered under Cost (C); only the 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 on the market, 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.

Source: Authors' own elaboration.

6. Mitigation strategies for methane emissions from animal 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 target CH_4 emissions from animal housing systems as well. Numerous strategies have been put forward to mitigate CH_4 emissions arising from manure. These strategies include the collection and capture of biogas (Clemens and Ahlgrimm, 2001), the employment 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, Andersen and Eriksen, 2012), the addition of amendments that inhibit CH_4 production (Andersen *et al.*, 2018), the separation of solids, the 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 factors such as temperature, pH, retention time and favourable anaerobic conditions for methanogenic bacteria activity result in increased CH_4 production, while the presence of inhibitory compounds or environments that inhibit the growth of CH_4-producing bacteria, can reduce CH_4 production (Andersen *et al*., 2018).

Anaerobic digestion followed by biogas collection and utilization is one of the most effective means of reducing CH_4 emissions from manure, provided that fugitive emissions are well controlled. Anaerobic digestion reduces the carbon (C) content of manure (Parajuli, Dalgaard and Birkved, 2018). Lowering the C content of manure means that there is less energy to support the denitrifying bacteria, which reduces the potential for N_2O formation of digested manure 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 its mode of action, efficacy potential, current adoption potential and antagonistic effects on N_2O production. The potential efficacy ratings of Low, Medium and High are provided in Table 5, where Low represents a reported CH_4-mitigation efficacy of up to 33 percent, Medium ranges between 33 and 66 percent, while it is greater than 66 percent for High. This classification system follows the method 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”. More detailed information about each strategy, including quantitative information, potential changes in ammonia (NH_3) emissions (increase or decrease), and reference publications for further study are provided below Table 5. 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 compared to other areas due to local regulations, the availability or cost of the technology. Where this is the case, it is discussed in more depth in the section regarding the mitigation strategy in question.

The listing of currently available mitigation strategies does not reflect best management practices. A specific strategy may work well in one situation and be a poor choice in another. While the focus of this report is on CH_4, some strategies that mitigate CH_4 result in the formation of other GHG emissions, such as N_2O, as well as increasing NH_3 emissions. When that is the case, it is mentioned in Table 5 alongside the description of the listed strategy. It should also be noted that some mitigation techniques may be combined for increased efficacy (e.g. anaerobic digestion with a subsurface manure injection land application), either at the same manure management stage or at farm level.

Table 5. Mitigation strategies for methane emissions from animal housing, manure storage and land application

Strategy	Mode of action	Efficacy potential	Current adoption potential	Antagonistic GHG emission effects
Biogas collection and utilization	System engineered to collect and use biogas	High if fugitive emissions are controlled	High	No
Decreasing manure storage temperature	Reduction in growth rate of methanogenic bacteria	Low to Medium 5% 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	High	No
Addition of methane inhibitors to manure (narasin, monensin, etc.)	Compounds cause changes to the microbial community that can inhibit CH_4 production	Medium to High Efficacy increases with increased 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. The 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 soil conditions
Manure application timing	Soil temperature and moisture content impact methanogenic bacteria activity	Low	Medium	Yes. N_2O emissions may increase under some soil conditions, but they may be decreased under others
Nutritional strategies	Reduction of the quantity of manure through an improved feed conversion rate, linked to increased feed digestibility	Medium	Medium	No

Source: Authors' own elaboration.

6.1 BIOGAS COLLECTION AND UTILIZATION

6.1.1 Description

The reduction of CH_4 emissions from animal manure storage can be achieved through the enhanced production and engineered collection of CH_4 via biogas from manure. Biogas can be collected in traditional manure storages or purpose-built anaerobic digestion systems to increase the production of CH_4 for its use as an energy source.

6.1.2 Mode of action

The 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.

6.1.3 Efficacy

It is worth bearing in mind 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). If manure is stored in a gas-tight structure preventing fugitive emissions, all CH_4 emissions from stored manure can be eliminated through the use of anaerobic digester systems (Clemens *et al.*, 2006). Similarly, Maurer *et al.* (2016) report CH_4 mitigation from anaerobic digestion to be “High”, meaning greater than 66 percent.

6.1.4 Potential to combine with other mitigation strategies

The use of some CH_4 mitigation strategies, such as manure acidification and the 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 lessen the efficacy of biogas collection, strategies antagonistic to CH_4 production that are implemented upstream of anaerobic digestion should be avoided when it takes place. That said, this technology can be used in combination with most other mitigation strategies. The use 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).

6.1.5 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 biogas (Li *et al*., 2017). Tapping CH_4 (as a fuel) or converting it (upgraded CH_4) can mitigate the GHG emissions. Digestate (i.e. material remaining following anaerobic digestion) can also contribute to indirect GHG credits with respect to chemical fertilizers that it substitutes.

6.1.6 Productivity and the quality of meat, milk, manure, crop, and air

There is no impact on meat or milk production. Although the anaerobic digestion of manure does not remove nutrients, it will end up transposing manure nutrients from inorganic to more readily available organic plant forms. The presence of sulphur in manure will result in the hydrogen sulfide formation in biogas, which has foul odor and human health hazard.

6.1.7 Safety and health aspects

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

6.1.8 Adoption potential

Manure anaerobic digestion technology is well developed and ready for use. It is easily adopted to liquid manure slurries and has a long history of being used for both cattle and swine manures. The principle obstacle to the adoption of manure anaerobic digestion has been the relatively high cost of biogas production compared to other available energy sources (Beddoes *et al.*, 2007; Torrijos, 2016).

6.1.9 Research required

There are no major research gaps.

6.2 DECREASED MANURE STORAGE TEMPERATURE

6.2.1 Description

Active cooling of slurry areas can significantly reduce CH_4 emissions.

6.2.2 Mode of action

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

6.2.3 Efficacy

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

6.2.4 Potential to combine with other mitigation strategies

Manure cooling can be combined with other mitigation strategies.

6.2.5 Effects on other emissions

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

6.2.6 Productivity and the quality of meat, milk, manure, crop, and air

No impact on meat or milk production. Manure cooling can assist in mitigating NH_3 emissions.

6.2.7 Safety and health aspects

No safety or health concerns.

6.2.8 Adoption potential

Controlling manure storage temperature is technically feasible, albeit potentially expensive (depending on climate). It may be a cost-effective option, if the exchanged heat can be harnessed to produce electricity or heat. Decreasing manure temperature to less than 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).

6.2.9 Research required

Most of the research has been carried out in the context of reducing NH_3 emissions in-house, and the measured impact on CH_4 emissions has consequently been limited. An additional demonstration of efficacy through the evaluation of CH_4 emissions at this specific scale may be necessary. Cooling systems that could be easily implemented in different types of housing still need to be developed.

6.3 MANURE ACIDIFICATION THROUGH DIETARY MEASURES

6.3.1 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).

6.3.2 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, which the increased concentration in urine further reduces.

6.3.3 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. 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 the diet of pigs for fattening (6.4 ± 0.6 vs 7.3 ± 0.2 for the test and control animals), while the reduction was not significant at an incorporation rate of 0.5 percent (Guingand, Demerson and Broz, 2005). The addition of 1 percent benzoic acid in the diet of pigs for fattening reduced urinary pH by one pH unit, regardless of the protein level – 7.93 vs 7.09 (low protein diets) and 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 to 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 the direct acidification of manure slurry using sulphuric acid has been shown to substantially reduce CH_4 emissions.

6.3.4 Potential to combine with other mitigation strategies

Due to its unique mode of action, benzoic acid can be used alongside 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 pH of manure. The use of benzoic acid in feed may negatively influence anaerobic digestion.

6.3.5 Effects on other emissions

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

6.3.6 Productivity and the quality of meat, milk, manure, crop, and air

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

6.3.7 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 emissions in animal housing provides additional safety and welfare benefits for the animals and farmers.

6.3.8 Adoption potential

Since benzoic acid can easily be incorporated into pig feed, it can be adopted when farmers are using compound feed or when producing feed on farm. Its positive impact on animal productivity and welfare usually compensates 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.

6.3.9 Research required

Most of the research has been done in the context of reducing NH_3 emissions from farms and the impact on CH_4 emissions has not been measured. It might be necessary to demonstrate efficacy through the evaluation of CH_4 emissions.

6.4 MANURE ACIDIFICATION THROUGH DIRECT AMENDMENT

6.4.1 Description

The reduction of manure pH by directly adding acids to manure slurries or stockpiles.

6.4.2 Mode of action

Methanogenic bacteria are inhibited as pH decreases.

6.4.3 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.*, 2013a), whereas Sokolov *et al.* (2020) reported CH_4 reductions of 77 percent in dairy cattle manure.

6.4.4 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.

6.4.5 Effects on other emissions

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

6.4.6 Productivity and the quality of meat, milk, manure, crop, and air

No impact on meat, milk or manure quality. The acidification of manure to a pH in the 5.5 range generally does not pose problems for crop production. Acidification reduces the loss of N in the form of NH_3, which results in increased N available to crops. It also reduces NH_3 emissions during and after the land application of manure. Surface application of acidified slurry is a good alternative to slurry injection (Fangueiro *et al.*, 2017).

6.4.7 Safety and health aspects

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

6.4.8 Adoption potential

This is a highly developed technology, which is even listed as a best available technique (BAT) for NH_3 mitigations. Nevertheless, technical barriers (risks related to the storage and handling of acid or the corrosion of materials) and psychological barriers (consumer distrust) exist in some countries.

6.4.9 Research required

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

6.5 METHANE INHIBITORS

6.5.1 Description

When added directly to manure storage, amendments such as tannins (Whitehead, Spence and Cotta, 2013), monensin (Clanton, Jacobson and Schmidt, 2012) and narasin (Andersen *et al.*, 2018) have been found to limit the formation of CH_4 in stored manures.

6.5.2 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.

6.5.3 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 per kg of manure. Andersen *et al.* (2018) reported that 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 reduced CH_4 production by over 85 percent for up to 28 days.

6.5.4 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. This technology can otherwise be used alongside most other mitigation strategies.

6.5.5 Effects on other emissions

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

6.5.6 Productivity and the quality of meat, milk, manure, crop, and air

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

6.5.7 Safety and health aspects

None.

6.5.8 Adoption potential

This technology is fully developed and ready for use. However, the adoption of this particular mitigation strategy will depend on the registration status in different jurisdictions of the substances involved. The main obstacle to adoption lies in the additional cost incurred when purchasing these compounds with no associated increase in production.

6.5.9 Research required

There are no major research gaps.

6.6 DECREASED MANURE STORAGE INTERVAL

6.6.1 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 outdoor storage**.**

6.6.2 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).

6.6.3 Efficacy

The highest efficacy will be achieved for animal production systems that have the greatest CH_4 yield from stored manure, such as deep-pit swine production systems (Park *et al.*, 2006). Petersen *et al.* (2013b) reported a 40 to 50 percent reduction of CH_4 emissions 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 have limited effectiveness.

6.6.4 Potential to combine with other mitigation strategies

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

6.6.5 Effects on other emissions

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

6.6.6 Productivity and the quality of meat, milk, manure, crop, and air

None.

6.6.7 Safety and health aspects

None.

6.6.8 Adoption potential

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

6.6.9 Research required

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

6.7 SOLID–LIQUID SEPARATION

6.7.1 Description

Solid–liquid separation has become a complementary management option for manure management systems, particularly for anaerobic systems. The separation process can help divert solids with a high phosphorous-to-nitrogen ratio to nutrient deficient areas. This can help reduce the GHG emissions produced during manure storage and manure application. Reducing CH_4 emissions is possible because volatile solids are separated along with the solid waste stream. Solid separation also reduces crust formation, which is useful to limit the anaerobic conditions during manure storage.

6.7.2 Mode of action

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

6.7.3 Efficacy

The CH_4 reduction ranges from 7.0 to 49.0 percent depending on several factors, including system design (e.g. screen size), the concentration of solids in processed manure, manure flow rate, and the type and configuration of the manure processing pit before it gets to the mechanical separator (Zhang, R. *et al.*, 2019).

6.7.4 Potential to combine with other mitigation strategies

This mitigation strategy may be combined with other mitigation strategies.

6.7.5 Effects on other emissions

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

6.7.6 Productivity and the quality of meat, milk, manure, crop, and air

None.

6.7.7 Safety and health aspects

Generally safe with potential hazards associated with moving parts.

6.7.8 Adoption potential

There are several designs that could be applied on farms of different sizes. These designs could also be integrated into current manure management systems with little modification. The adoption will depend on the cost of retrofitting existing management systems.

6.7.9 Research required

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

6.8 MANURE COMPOSTING/AERATION

6.8.1 Description

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

6.8.2 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.

6.8.3 Efficacy

Maurer *et al.* (2016) find composting to be 70 percent effective at all scales for dairy manure but note a reduction by 34 percent of CH_4 emissions for swine manure composting at all scales. This large discrepancy reflects the difference in CH_4 emissions for composting systems with varying aerobic or anaerobic conditions.

6.8.4 Potential to combine with other mitigation strategies

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

6.8.5 Effects on other emissions

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

6.8.6 Productivity and the quality of meat, milk, manure, crop, and air

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

6.8.7 Safety and health aspects

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

6.8.8 Adoption potential

Composting and aeration technologies are highly developed and ready for use. Composting can easily be adopted for solid manures and slurry with the addition of a carbon source.

6.8.9 Research required

No additional research is required to implement this strategy.

6.9 BIOFILTERS AND SCRUBBERS

6.9.1 Description

Biofilters and biofilter/scrubber combinations have been found 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).

6.9.2 Mode of action

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

6.9.3 Efficacy

Maurer *et al.* (2016) report a CH_4 mitigation effect of 17 to 24 percent across all species and at all scales in their summary of performance data for technologies used to control gaseous emissions from livestock operations.

6.9.4 Potential to combine with other mitigation strategies

This mitigation strategy may be combined with any others.

6.9.5 Effects on other emissions

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

6.9.6 Productivity and the quality of meat, milk, manure, crop, and air

None.

6.9.7 Safety and health aspects

None.

6.9.8 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. The cost of this retrofit can be prohibitive for many operations.

6.9.9 Research required

Additional research on limiting N_2O production in biofilters is needed.

6.10 MANURE INCORPORATION AND INJECTION

6.10.1 Description

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

6.10.2 Mode of action

Soils can serve either as a source or a sink for CH_4, depending on the conditions and whether the 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 favourable for methanogenic bacteria activity, CH_4 emissions can increase following the incorporation or injection of manure.

6.10.3 Efficacy

The mitigation efficacy is high when soil conditions favour methanotrophic bacteria growth. When soil conditions favour 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 they quickly fall to very low levels following incorporation or injection (Montes *et al.*, 2013). Lovanh, Warren and Sistani (2008) reported that the injection of swine manure resulted in an order of magnitude reduction of CH_4 as 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. For instance, Sistani *et al.* (2010) report that CH_4 emissions from cropland fertilized with injected swine were significantly higher than when a commercial fertilizer control had been used.

6.10.4 Potential to combine with other mitigation strategies

The potential to combine other strategies, such as anaerobic digestion or solids separation, with manure incorporation or injection is excellent. 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.

6.10.5 Effects on other emissions

Manure incorporation and especially its injection 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 differences in N_2O emissions between the surface application and injection of swine manure. The inconsistency in reported N_2O emissions, calculated following the land application of manure, is likely due to the diversity of soil conditions in which the emissions were measured.

6.10.6 Productivity and the quality of meat, milk, manure, crop, and air

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

6.10.7 Safety and health aspects

None.

6.10.8 Adoption potential

This technology is fully developed and ready for use. It is worth pointing out that, in order to make use of this technology, producers will have to purchase the purpose-specific equipment required to pump, transfer and make a subsurface injection of manure. The cost of this equipment may prove a barrier to adoption for some farmers.

6.10.9 Research required

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

6.11 MANURE APPLICATION TIMING

6.11.1 Description

Manure application, at various times of the day and in different seasons, using a range of methods that are currently employed for incorporation and surface applications.

6.11.2 Mode of action

Soil temperature and moisture content affect methanogenic bacteria activity.

6.11.3 Efficacy

In their table of possible mitigation strategies, Montes *et al.* (2013) list application timing as having an efficacy of ≤10 percent.

6.11.4 Potential to combine with other mitigation strategies

Combining this strategy with other manure treatment technologies, such as manure storage and the production of stable manure products (e.g. composted manure), could give application timing greater flexibility.

6.11.5 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 so 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.

6.11.6 Productivity and the quality of meat, milk, manure, crop, and air

None.

6.11.7 Safety and health aspects

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

6.11.8 Adoption potential

It can be achieved in practice, when storage volume and weather conditions allow for it.

6.11.9 Research required

The measurements of N_2O and NH_3 emissions in different weather conditions and cropping systems need further research.

6.12 NUTRITIONAL STRATEGIES

6.12.1 Description

Reducing the amount of excreted OM decreases the emission of CH_4 from manure.

6.12.2 Mode of action

Nutritional mitigation options that improve the 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, the preparation of feed in the form of pellets may also reduce feed losses on pig farms.

6.12.3 Efficacy

The efficacy of this strategy depends on the different mitigation options that are available and the status of the farm. An improvement between 2 and 5 percent of the feed conversion ratio can be achieved under typical farm conditions.

6.12.4 Potential to combine with other mitigation strategies

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

6.12.5 Effects on other emissions

Usually, an 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.

6.12.6 Productivity and the quality of meat, milk, manure, crop, and air

The feed conversion ratio is an important parameter of productivity for farmers. The effects on animal productivity are covered elsewhere in this document (Section 5).

6.12.7 Safety and health aspects

The nutritional mitigation solutions used to improve the feed conversion ratio of animals are typically considered to be GRAS (generally recognized as safe) or their safety is evaluated by regulatory authorities.

6.12.8 Adoption potential

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

6.12.9 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.

6.13 GRAZING PRACTICES – PRODUCTION SYSTEM

Modifying grazing systems has repercussions on the entire production system, unlike applying a single mitigation method for stored or land applied manure. As such, they have not been included in Table 5. Modifying grazing systems to decrease CH_4 emissions can affect the amount and composition of manure excreted by animals. Methane emissions from urine and dung droppings of grazing animals are minimal compared to those emitted through animal confinement in manure storage systems (Pellerin *et al.*, 2017). The reduction in CH_4 emissions from animal excreta is substantial, especially in warm climates.

Owing to the high-fibre concentration of the herbage, grazing animals tend to produce greater enteric CH_4 emissions compared with animals in confinement systems that are fed mixed diets. Grazing systems, when intensively managed, have larger N_2O emissions at the field level. Conversely, NH_3 emissions (a precursor to N_2O) are generally reduced in grazing vs confined systems. Grazing systems reduce the amount of manure produced on the farm because the bedding is not being used, and excreta are delivered directly to the pasture. There can be differences within grazing systems that affect the potential for soil organic carbon sequestration.

7. Mitigation of methane emission from rice paddies

Methane is emitted from rice paddies due to the 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 partly oxidized in the oxic rhizosphere and surface soil. Thus, the balance of CH_4 production and oxidation controls CH_4 emission (Figure 3). Several management practices that induce the increased redox potential of soil suppress CH_4 production and hence the emissions from rice fields.

7.1 WATER MANAGEMENT

Modifications of the water management have a proven track record to reduce CH_4 emissions from rice fields and are deemed the most promising way to mitigate CH_4 emissions from rice paddies (Wassmann, 2019). The drainage of rice fields increases the redox potential, which strongly suppresses the microbial processes of CH_4 production and concomitantly stimulates CH_4 oxidation. However, the drawback of floodwater generates short-term spikes of gaseous CH_4 that has been entrapped in the flooded soils (Wassmann *et al.*, 1994). Nevertheless, the overall amount of CH_4 emitted from the soil through the course of the cropping season is significantly reduced as demonstrated in numerous field measurements (Sander, Wassmann and Siopongo, 2014). Either single or multiple drainage approaches, like alternative wet and dry (AWD) management, have shown consistently significant mitigation potential – although the magnitude of reduction given by different studies ranges widely (Yagi *et al.*, 2020).

While the baseline is defined as continuous flooding, the scaling factors for other water regimes in the IPCC guideline vary 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, 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 from 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.*, 2018) – do not reverse the trend of GHG savings through AWD (Majumdar, 2003; Yagi *et al.*, 2020). Although 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, 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 USD 100 per hectare (Frith, Wassmann and Sander, 2021).

Water management before the cultivation period also affects CH_4 emissions during rice cultivation. A prolonged non-flooded preseason over one year had a significantly lower CH_4 emission scaling factor (0.41-0.84), while a flooded preseason over 30 days before cropping had a more than doubled scaling factor (2.13-2.73) (IPCC, 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 CH_4 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.*, 2015). However, if available, appropriate water management promotes rice production while effectively mitigating CH_4 emissions (Yagi *et al.*, 2020). Land levelling allows for a spatially homogeneous water management, and would contribute to effective CH_4 migration. Better water management in rice paddies to mitigate CH_4 emission could also contribute to sustainable water, an important goal for agriculture (FAO, 2020). On the other hand, the prolonged aerated conditions could cause an enhanced decomposition of soil organic matter, lowering the carbon storage and fertility of rice field soils in the long term. A meta-analysis by Livsey *et al.* (2019) showed that while mild forms of AWD reduce emissions of CH_4 by up to 52 percent, such management can increase CO_2 emissions by 45 percent while increasing soil-to-atmosphere carbon flux by 25 percent when compared to continuous flooding. AWD was also found to have a negative effect on both soil organic carbon – reducing concentrations by 5.2 percent – and soil organic nitrogen – potentially depleting stocks by more than 100 kg N ha^{-1} y^{-1}. While significant negative effects of AWD on rice yield may not be visible in short-term experiments (1- to 3-year-long studies), care should be taken when assessing the long-term benefit of AWD-like irrigation practices because they can decrease soil fertility and hence yields in the long term (Livsey *et al.*, 2019).

7.2 ORGANIC AMENDMENTS

More CH_4 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 the soils. If rice straw is incorporated into the soil after harvest, the timing of rice straw application significantly affects CH_4 emissions. A long interval between straw incorporation and flooding lowers CH_4 emissions during the rice-growing season, as compared to incorporating rice straw just before flooding (IPCC, 2019). Either removing or burning rice straw drastically reduces CH_4 emissions, but it has adverse effects on the local air quality (in case of burning) and may decrease 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 routinely been removed in 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 – as opposed to a baseline of incorporating fresh straw – is another option for reducing CH_4 emissions from rice fields (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, CH_4 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 factor than the incorporation of fresh rice straw (Buendia *et al.*, 2019), thus offering the additional 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 its long-term effect remains unclear, it has often been shown that biochar application is an effective way of reducing CH_4 emissions from flooded rice fields (Jeffery *et al.*, 2016; Mohammadi *et al.*, 2020; Yagi *et al.*, 2020). Environmental life cycle assessment studies revealed 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 of rice grain, implying a significant reduction relative to the 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 stage of pilot studies, as the practicability and environmental impacts of available stoves are still uncertain. A combination of AWD water management with biochar application may further reduce CH_4 emissions (Sriphirom *et al.*, 2020), though there is not enough data to draw conclusions about how biochar production and application affect whole-system GHG budgets (Gurwick *et al.*, 2013).

7.3 FERTILIZER AND OTHER AMENDMENTS

The application of sulfate-containing fertilizer, such as ammonium sulfate and phosphogypsum, reduces CH_4 emissions (Yagi *et al.*, 2020; Kumar *et al.*, 2020), as sulfate ion supports sulfate reduction that outcompetes CH_4 production in flooded rice field soils (Achtnich, Bak and Conrad, 1995).

Biofertilizers, 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 CH_4 emissions 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 CH_4 emissions from rice fields (Malyan *et al.*, 2016). Nitrification inhibitors promote rice plant growth through increased nutrient uptake, and they increase the redox potential in the rhizosphere, which reduces CH_4 emission (Boeckx, Xu and Van Cleemput, 2005).

The reduction of ferric iron likewise competes with methanogenesis (Achtnich, Bak and Conrad, 1995). Adding steel slag can mitigate CH_4 emission from paddy fields (Kumar *et al.*, 2020). Silica oxide in steel slag can also mitigate CH_4 emissions from rice by promoting the development of aerenchyma in rice roots, which increases oxygen transportation from the atmosphere to the root region and enhances rhizospheric CH_4 oxidation (Kumar *et al.*, 2020).

7.4 PLANTING METHODS AND CROP MANAGEMENT PACKAGES

Compared to the traditional transplanting of rice seedlings, direct seeding has been reported to reduce CH_4 emissions (per m^2 and day) (Yagi *et al.*, 2020; Malyan *et al.*, 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 the labour savings involved. It could also be optimized as a mitigation potential in many rice-growing areas.

In preseason conditions, CH_4 emissions are reduced by prolonged periods without flooding (Yagi *et al.*, 2020), caused by a long fallow season or a crop rotation with an upland crop. This effect is considered in the IPCC guidelines in the form of a preseason 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, labour-intensive management, which presents certain features of low-emission management (Malyan *et al.*, 2016; Yagi *et al.*, 2020). The term "SRI", however, has been used in the literature with reference to a very wide range of crop management practices, in particular those regarding the application of organic manure (Ly *et al.*, 2013). The original SRI concept encompasses a high amount of organic inputs that result in a high background level of CH_4 emissions. As the SRI prescribes intermittent flooding, the actual increment in emissions will be lower than in the case of continuous flooding, due to the suppressed methanogenesis. The SRI has been considered as a mitigation strategy, which can be justified when compared to continuous flooding (Ly *et al.*, 2013) or as long as organic amendments are omitted (Jain *et al.*, 2014). The calculated mitigation effect by the SRI will ultimately depend on the definition of the baseline management as well as on the SRI subtype used for the comparison.

7.5 SELECTING/BREEDING RICE VARIETIES

The difference in CH_4 emissions for different rice varieties has been documented in several case studies. The underlying mechanisms to reduce CH_4 emissions from rice paddies through variety selection still remain unclear – except for the straightforward approach of replacing long-duration with short-duration varieties, which was proposed back in the year 2000 by Setyanto *et al.* All other possible changes to plant morphology and physiology showed inconsistent results across different studies, due to complex genetics, environment and management interactions that directly or indirectly alter the CH_4 budget (Wassmann, Neue and Lantin, 2000). Derived from plant morphology, the low permeability of the aerenchyma constrains the CH_4 transfer from the soil to the atmosphere (Butterbach-Bahl, Papen and Rennenberg, 1997; Aulakh, Wassmann and Rennenberg, 2002). Since this trait will also limit the transfer of oxygen into the root system, which has the opposite effect on CH_4 fluxes, the net impact on CH_4 emissions will vary according to specific circumstances, e.g. water regime and fertilizer management.

From a physiological perspective, root exudation determines the amount of methanogenic material and is therefore strongly linked to CH_4 emissions (Lu *et al.*, 1999). The actual amount of root exudation, however, is chiefly affected by the nutrient status of the rice plant (Lu *et al.*, 2000), and consequently its impact on CH_4 emissions may be concealed by other factors. A high efficiency of the physiological carbon sink, i.e. the allocation of metabolites in the grain, has been shown to be favourable for low-emission plants in greenhouse experiments (Denier van der Gon *et al.*, 2002) as well as by means of genetically modified organisms (Su *et al.*, 2015). Broadly speaking, early-maturing rice cultivars with few unproductive tillers, small root systems, a high root oxidative activity and harvest index, and a low root exudation were proposed for mitigating CH_4 emissions in rice fields (Malyan *et al.*, 2016). However, a mechanistic understanding is still needed to select and breed varieties that emit less CH_4 (Balakrishnan *et al.*, 2018; Yagi *et al.*, 2020).

7.6 REDUCING METHANE FROM STRAW BURNING

Although CH_4 emissions from rice production are generally equated with biogenic emissions from flooded fields, common farming practices in many Asian countries also generate sizeable amounts of pyrogenic CH_4. Open field burning entails an incomplete combustion of rice straw and this generates CH_4 as well as, to a lesser extent, nitrous oxide (Romasanta *et al.*, 2017). Despite considerable efforts to eliminate this practice, straw burning is still rampant in many parts of Asia, causing enormous problems related to local air pollution (Gadde, Menke and Wassmann, 2009). While rice straw is typically kept in piles on the fields after harvest, the proportion of incomplete combustion is a function of the moisture content in these piles and therefore of local rainfall events (Romasanta *et al.*, 2017). An alternative management practice to straw burning is soil incorporation, but straw amendments increase the CH_4 emissions once the field gets flooded. Based on the 2019 IPCC 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 constitutes a relatively laborious activity given the typically low levels of mechanization in most rice-producing regions at present. Straw could be used to make compost and then 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, such as animal manure, for making compost (Nguyen-Van-Hung, 2020). Straw could also be fed to cattle, but its low nutritional value will cause sizeable CH_4 emissions from the animals. In principle, straw represents a valuable feedstock for bioenergy as is shown by the use of wheat straw in many industrialized countries. Rice straw, however, has a high silica content that tends to cause technical problems ("slagging") in combustion devices (Chieng and Kuan, 2020). Moreover, a commercial use will require for it to be available in a compact form, which facilitates transport and storage. To this end, the prevailing trend towards the mechanization of rice production, and the new baling machines in particular, may transform straw into a readily available commodity that can be traded (Nguyen-Van-Hung *et al.*, 2020).

7.7 CHOICE OF OPTIONS

Appropriate water management, including mid-season drainage, AWD and SRI, is the most promising option for mitigating CH_4 emissions from flooded rice fields. It would thus be our first choice if practicable, and provided that such water management did not increase N_2O emissions and/or decrease soil organic carbon. Preventing the introduction of fresh organic matter like rice straw into the soil is an effective remedy against excessive CH_4 production and the resulting high emissions in rice fields.

Some of the fertilization techniques have been considered for the mitigation of CH_4 – either as an additional measure or in case the water management options did not prove practicable. Sulfate-containing fertilizers may help to reduce CH_4 emissions, but they are not suitable for soils with a low amount of reducible iron that forms insoluble iron sulfide (FeS), because in that case the reduced sulphur ion (S^{2-}) would damage the rice roots. Biofertilizers (*Azolla* and blue-green algae) can oxygenate the surface soil, reducing CH_4 production and promoting CH_4 oxidation, but whether they will have a discernible impact under field conditions is merely speculative at this point. The application of iron and silica-containing materials that can maintain higher redox conditions in soil and the rhizosphere can also play a part in reducing CH_4 emissions. A meta-analysis of biochar treatments demonstrated the mitigation potential of biochar (Jeffrey *et al*., 2016), but there is no evidence as to the applicability of this option at a larger scale.

Many of the above-mentioned methods could contribute to better plant growth and higher yields, thus reducing GHG emission per yield, i.e. "GHG intensity" expressed as tonnes of CO_2eq per ha, as well as a carbon footprint defined as emissions per amount of product, and expressed as kg CO_2eq per kg of rice product. As for the private sector, the production-based emissions are much more relevant than the area-based emissions. Future efforts to reduce CH_4 emissions from rice may be driven by user-friendly and transparent calculation tools and labels indicating the product's carbon footprint (Wassmann, Neue and Lantin, 2022). This will then encompass a wide range of production-enhancing approaches, e.g. rice hybrid technologies. Such efficiency gains in terms of food production have routinely been considered as a mitigation option in animal systems, but they have hardly been mentioned in the context of rice production.

7.8 NEWLY EMERGING TECHNOLOGIES

In addition to the currently available methods for reducing CH_4 emissions from rice, several new technologies under ongoing investigation show a high potential as mitigation options. For example, research suggests that plant growth-promoting rhizobacteria (PGPR) such as diazotrophs could increase root mass, thereby promoting molecular oxygen (O_2) release to the soil, and suppress methanogenesis (Singh and Strong, 2016). The potential of transgenics to decrease CH_4 emissions is demonstrated by using a barley transcription gene (Su *et al.*, 2015). Microbial fuel cells (MFCs), which generate electricity in rice field soil, compete with CH_4 production and can thus mitigate CH_4 emission from the rhizosphere (Kouzuma, Kaku and Watanabe, 2014). These new technologies are still in their infancy and further investigations and verifications are needed before they can be applied at a field scale (Pratt and Tate, 2018).

8. Cross-cutting methane mitigation

8.1 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 that they 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 designed to ensure holistic appraisals and provide some illustrative examples of CH_4 reduction strategies that are being considered.

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 prompt 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 GHGs other than CH_4, including N_2O and CO_2, alongside reductions in resource use and wider environmental impacts (Capper, 2011). In other cases, trade-offs may be needed. Efforts to reduce CH_4 emissions, for instance, may increase other GHGs (Cardoso *et al.*, 2016) or raise concerns with respect to animal welfare (Llonch *et al.*, 2017). Similarly, in the case of rice systems, one must consider the changes in net GHGs, paying attention to N_2O and CO_2 emissions in addition to CH_4 (Kritee *et al.*, 2018).

Comprehensive assessments that cover multiple impact categories are typically provided by an LCA, with an attributional LCA being the most common one. An attributional LCA tracks energy, material uses and pollutant releases occurring along the supply chain and production process in order to report the total inventory or “footprint” that can be attributed to a given output or functional unit (ISO, 2006). The functional unit may either be a product or commodity of a certain quality (e.g. milk, for dairy production), or a more specific aspect of the outputs (e.g. the protein or calorie content). The choice of a functional unit depends on the nature of the assessment being made and the intended use of the life-cycle information.

The boundaries of a life-cycle system relevant to a given question are extended as far as possible, ideally starting from the point of production of all inputs (“cradle”), capturing the impacts that have occurred before the agricultural production phase, such as energy-use in the case of a 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 have the greatest ability to reduce impacts, but the chain can also be followed through to processing, consumption and disposal (for a complete “cradle-to-grave” LCA). Disposal from production, such as infrastructure or manure, are part of the production system.

In this way, an LCA provides a useful methodology to explore CH_4 reduction strategies in a broader context. When adopting a life-cycle perspective, we look beyond the mere reduction in CH_4 emissions that might be achieved through different measures to consider the wider associated impacts, both positive and negative, such as those associated with the manufacturing of CH_4-reducing feed additives. It also offers the whole production-system vantage that makes it possible to identify wider co-benefits or potential trade-offs, as noted above. A life cycle assessment generally captures emissions per functional unit, which is often a product. However, from a global perspective, it is the absolute emissions that matter for assessing the most extreme climate consequences. Some mitigation strategies may reduce emission intensity by increasing efficiency, which may then facilitate greater production, resulting in increased absolute emissions. Whether emission intensity or total emissions are the most relevant aspects to characterize mitigation outcomes depends on the wider policy and development objectives.

In addition to its role in setting a comprehensive framework for compiling a life-cycle inventory, an LCA is also commonly used to assess the impacts resulting from this inventory. This is done by translating the inventory data into 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. A more detailed discussion of various metrics that can be considered for LCA is found in Chapter 6.

It must be noted that GWP_{100} is just one potential climate impact indicator. It is a "midpoint" indicator, only part of the way along the chain of translating GHG emissions into an eventual contribution to climate change and resulting damages. Depending on, for example, the time frame or the aspect of climate change that is of interest, other indicators may be equally justified yet give a different answer as to whether a specific intervention has an overall positive or negative impact. Recent guidance recommends considering multiple metric choices in Life Cycle Impact Assessments (Levasseur *et al.*, 2016), including the GTP. The relative valuation of CH_4, a short-lived GHG, is particularly sensitive to the metric choice and time horizon. Part 4 of this report discusses the usage to which different GHG metrics are put and how to interpret contributions to climate change. The metrics section of this document explores alternative ways of quantifying the impact of CH_4 mitigation.

Climate impact is merely one component of a total impact assessment in an LCA; water scarcity, land use, biodiversity loss, air and water pollution are some of the other common outcomes. Interventions aiming to reduce CH_4 emissions can be weighed against these other impacts, not unlike when exploring the influence they may have on other GHG emissions, as described in Section 5.

These wider impact categories come with standard, simplified indicators designed 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, along with guidance on how to explore the sensitivity to different metrics and ensure the method used can be sensibly applied to the question at hand (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 the GHG emissions is ultimately at the discretion of the user/investigator. Separate impact categories can also be confronted and combined into aggregated indicators, as in the "disability-adjusted life years" estimating the total burden on human health, the financial valuation providing a common currency for all impacts and outputs, or abstract scores acting as a simple communication device for total impacts. There is, however, no universally agreed upon method of indicator weighing or aggregating, and doing so can obscure individual results; it is therefore standard practice to retain separate reporting categories in addition to fully aggregated indicator results.

These challenges result in limitations and potentially subjective appraisals in agricultural LCAs, as is widely acknowledged. For example, van der Werf, Knudsen and Cederberg (2020) argued that an 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 broader context of assessing CH_4 reductions, it is important to note that an LCA may give us some insight into as well as a means of quantifying wider benefits and/or trade-offs, but the LCA results depend on methodological choices that make the outcome highly uncertain. There may be other considerations that determine how policymakers and society at large come to view certain system transitions as positive.

Some of these broader issues may be addressed through a 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 the production method or the type or quantity of functional unit produced), rather than just comparing individual system impacts. Where an attributional LCA allocates elementary flows to individual products, which may then be compared, a consequential LCA estimates the deviations in the elementary flows resulting from a system change (Rebitzer *et al.*, 2004; Ekvall and Weidema, 2004).

Consequential LCA may be particularly relevant where proposed CH_4-reduction measures entail major, systemic impacts, such as a global shift towards more intensive ruminant production, or reductions in total ruminant production. A full review of the consequential LCA, which has its own challenges and limitations (Yang and Heijungs, 2018), is beyond the scope of this report. It may, however, provide another useful approach which has not been hitherto sufficiently explored when it comes to assessing 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 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 practice.

In summary, the complex and interlinked nature of agricultural production means that we must consider CH_4 reductions in a wider context, as further explored in the examples presented below. An 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. It can also provide guidance and useful frameworks for carrying out an environmental impact assessment. However, an exhaustive analysis of the impacts of agricultural production systems, the extent to which they may be deemed “sustainable”, and the full range of considerations required for decision-making, may need a deeper appraisal and interpretation than what the LCA impact indicators alone can provide. This may include an assessment of whether reductions in emission intensity or absolute emissions are the relevant measure of success. There is a growing body of literature that continues to develop the Life-Cycle Impact Assessment methodologies, and suggests refinements as to how they are applied. Given the focus of this report on CH_4 reduction, we provide an extended discussion of how CH_4 emissions in particular are reported, but the wider context outlined here remains important.

8.2 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 fossil fuel use for operating machinery 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. It is therefore 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, 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 Californian 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, and so only the additional emission in producing 3-NOP was taken into account 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. 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 (Figure 5), reported that the average net reduction rates with supplementation of 3-NOP and nitrate in the Californian 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 not to be affected by the inclusion of feed additive.

The implementation of mitigation options reviewed in the previous sections has 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 the diet, should be analysed for downstream emission effects. To analyse the effect of more complex mitigation strategies, 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). This type of framework has been shown to identify in particular how effective GHG mitigation methods may, in some cases, alter the 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 its lack of availability beyond academia and a 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.*, 2013a). 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. the effectiveness of the combined mitigation methods may not be equal to the sum of the individual methods when applied on their own (del Prado *et al.*, 2010).

8.3 LCA SCENARIO ANALYSIS FOR LESS INTENSIVE SYSTEMS

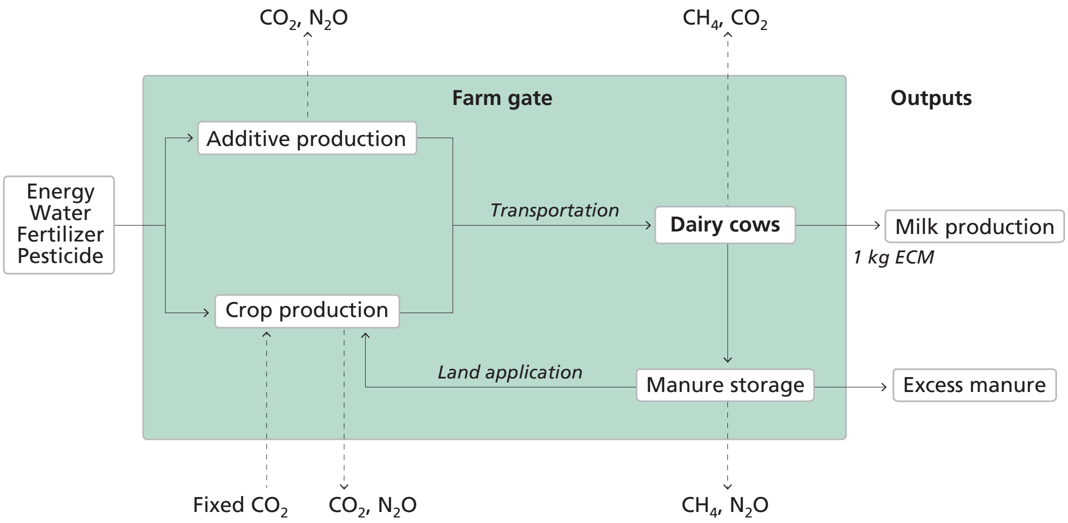
Less intensive livestock production systems tend to have a greater share of their total carbon footprint as CH_4, especially from enteric fermentation. In contrast to intensive ruminant production systems where enteric CH_4 typically comprises less than 40 percent of the total CO_2eq/kg of product based on a GWP_{100} (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 fewer emissions from use of fossil fuel. In some extensive systems, enteric CH_4 can comprise more than 70 percent of the carbon footprint of meat and milk due to the use of less digestible, fibrous feed and a reduced level of animal productivity (Flysjö *et al.*, 2011; Chobtang *et al.*, 2016; Sánchez 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, 2019). Increasing the digestibility of forage has been identified as a strategy to decrease enteric CH_4 emission intensity. However, the mitigation effect needs to be analysed case by case. For example, a shift from feeding less grass to more whole-plant maize silage 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 in some systems silage may not be 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 could be much greater than emission reductions at animal level (Vellinga and Hoving, 2011). Yan, Humphries and Holden (2013) conducted an 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 from grazing dairy cows. Similarly, Schils *et al.* (2005) found that GHG intensity from a grass-clover system was 10 percent lower than that from a grass-fertilizer N system.

Lahart *et al.* (2021) compared the effect of genetic merit in Holstein-Friesian dairy cows across three contrasting feeding pasture-based production systems (extensive to intensive). The authors reported that improved genetic merit combined with reducing concentrate supplementation led to a general improvement in GHG intensity as well as an 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 reduce 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 an increased average daily gain and shorter finishing periods reduced CH_4 emissions per unit of product (Lovett *et al.*, 2005). Both Pelletier, Pirog and Rasmussen (2010) and Murphy *et al.* (2017) reported that GHG emission intensities were greater for beef finished

Figure 5
System boundary of the life cycle assessment for the Californian milk production



ECM = energy-corrected milk. It should be noted that CO_2 from animal origin is considered a net zero carbon balance with plant sequestration (see Section 5).

Source: Adapted fromFeng, X.Y. & Kebreab, E. 2020. Net reductions in greenhouse gas emissions from feed additive use in California dairy cattle. *PLoS ONE*, 15(9). https://doi.org/10.1371/journal.pone.0234289 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) contended that earlier age at slaughter did not necessarily lead to the greatest profitability owing to the lower gross output value achieved. In improved pasture-based systems, the ability to slaughter animals at a younger age often leads to a greater stocking density, thus resulting in increased GHG emissions per hectare compared with more extensive production systems, although there would be fewer emissions per kg of beef. 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 emission intensity of food production but increase total emissions per hectare. Whether reducing emission 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 analyse these consequential land uses too.

Grasslands can be a carbon source or sink depending on climate, site characteristics such as soil type, and management practices like 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 performance of pasture relative to grain-based production systems from a net GHG emission perspective (Soussana, Tallec and Blanfort, 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 the impact of methane emissions

9. Introduction

The distinct chemical and physical properties of different GHGs, and their ultimate effects on global warming, both in terms of the strength and duration of any climate impacts, are generally well-understood and scientifically uncontested. For the purposes of 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 global warming and other climate impacts, or to quantify the benefits of potential emission reductions.

Emission metrics can provide a means of comparing different greenhouse gas emissions by putting them onto 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 the global, national, regional or institutional levels; trading emissions of different GHGs against each other; aiding mitigation decision-making, especially in trade-off situations when reducing one gas is very costly but reducing another one is much less so, or when decreasing the emissions of one GHG contributes to increasing the emissions of another.

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 that of greenhouse gas emissions. However, there are some 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 emission metrics" terminology.

The following definition comes from the glossary of the IPCC's Sixth Assessment Report:

> **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 (IPCC, 2021b, p. 2232).

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 rest on particular aspects of the physical response that are used as proxies to represent climate change over a time horizon that needs to be determined. 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 of 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 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 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 the conceptual consistency between metrics and policy objectives, relevant considerations can also include the scientific uncertainty of metric values, the ease of communication and the 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 be between using a large number of different metrics for the sake of scientific or policy completeness and opting for a small set of metrics that may be imperfect but could be considered good enough for a range of applications 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 making baseline assessments and greenhouse gas mitigation choices within agricultural supply chains.

9.1 CONTEXT AND DEFINITIONS

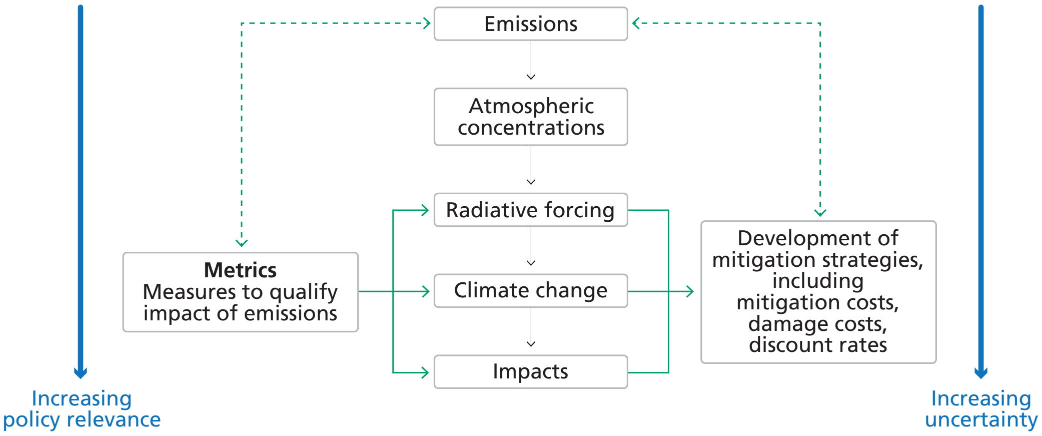
9.1.1 Key principles of GHG emission metrics

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 associated impacts (e.g. Fuglestvedt *et al.*, 2010) or, conversely, on the benefits that avoiding any given emission(s) would bring by not contributing to climate change and its impacts. This may take the form of describing how different activities or sectors contribute to overall climate change or climate change impacts, assessing the priorities and trade-offs associated with emitting or mitigating different GHGs, or aiding decision-making and identifying the most efficient ways to meet overarching climate targets. Emission metrics provide a shortcut in the cause–effect chain and translate emissions to impacts, as shown in the figure below.

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

Figure 6

The cause–effect chain from emissions to climate change impacts

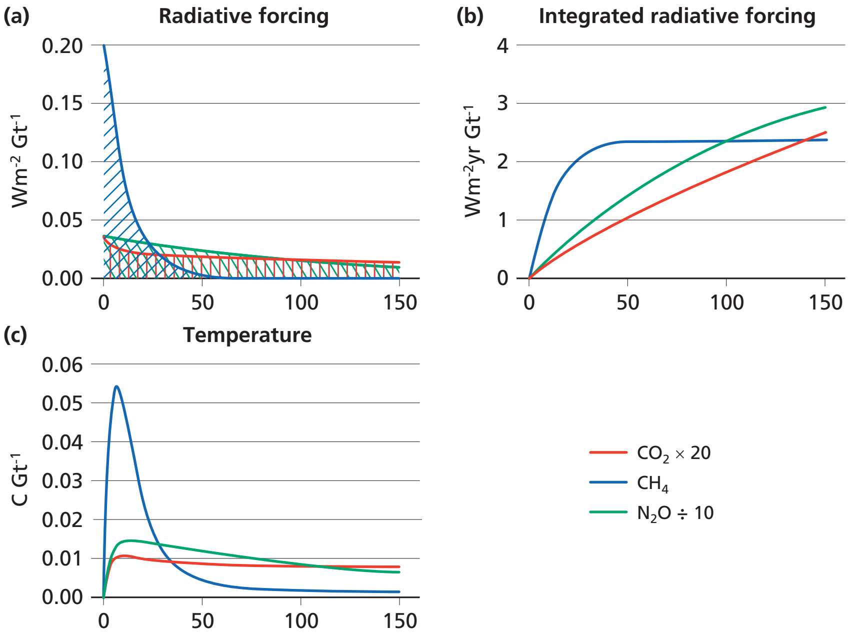


It illustrates the role of metrics in defining the estimated responses to emissions (left) and in the development of multicomponent mitigation strategies (right). The relevance of the various effects increases from emissions to impacts but the uncertainty increases as well. The dotted line on the left 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, J.S., Berntsen, T.K., Godal, O., Sausen, R., Shine, K.P. & Skodvin, T. 2003. Metrics of climate change: Assessing radiative forcing and emission indices. Climatic Change, *58: 267–331. https://doi.org/10.1023/A:1023905326842 and Plattner G.-K., Stocker, T., Midgley, P. & Tigno, M. 2009. IPCC Expert meeting on the science of alternative metrics, Oslo, Norway, 18-20 March 2009. www.ipcc.ch/site/assets/uploads/2018/05/expert-meeting-metrics-oslo.pdf).*

Source: Figure reproduced from Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T. & Zhang, H. 2013. Anthropogenic and natural radiative forcing. In: T.F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P.M. Midgley, eds. *Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK & New York, USA, Cambridge University Press. www.ipcc.ch/site/assets/uploads/2018/02/ WG1AR5_Chapter08_FINAL.pdf break down or dissipate in the atmosphere.[3] 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. Carbon dioxide has a complex atmospheric lifetime as it is removed from the atmosphere by various processes with different rates, but it can largely be considered an extremely long-lived gas, with a significant fraction of emissions remaining in the atmosphere for millennia (Archer *et al.*, 2009; Joos *et al.*, 2013). Figure 7 shows the different effects on radiative forcing and temperature change for one gigatonne (Gt) of CO_2, CH_4 and N_2O pulse emissions. The remainder of this section will explain the key principles which underlie GHG emission metrics.

Figure 7

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



(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.

Source: Authors' own elaboration.

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

Changes in greenhouse gas concentrations impact the climate by changing the atmospheric energy balance (radiative forcing). The extent to which a given change in the concentration of a gas leads to radiative forcing is known as its “radiative efficiency”, and can be regarded 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’s AR6 (Forster *et al.*, 2021) therefore now include the carbon cycle response by default.

The impact of the emissions of chemically-reactive gases on other greenhouse gases also needs 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 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, Section 9.1.2.1). It is also possible to continue along the cause–effect chain (see Figure 6), 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, Section 9.1.2.2), takes this approach, comparing emissions on the basis of their relative contribution to global temperature change at a specific point in time following the emission.

Metrics can progress further still to quantify impacts as the damages resulting from climate change, for example economic damages (Hammitt *et al.*, 1996) or individual environmental impacts such as precipitation and sea-level rise (Shine *et al.*, 2015; Sterner, Johansson and Azar, 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 needed 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 Section 9.2.2 and Section 9.2.3 for details).

9.1.2 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” for comparing 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_2eq) quantity; the values in Table 6 and Table 7 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. The 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 alone, it can serve as a proxy for metrics that evaluate the damage due to emissions from an economic perspective (Tol *et al.*, 2012). Global damage potentials are discussed in Section 9.2.3.

9.1.2.1 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 an equal mass of CO_2. The most frequently used, and effectively "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, p. 711) put it as follows: "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. Nitrous oxide 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 in the case of 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 as to the atmospheric longevity of CO_2 and any gases that CO_2 is compared with. The categorization of CH_4 as either "fossil" or "non-fossil" depends on whether the carbon introduced into the atmosphere is considered new or not (or already included in budgets) (see Section 9.2.7).

Table 6. GWP values from the IPCC's Sixth Assessment Report (AR6)

	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

Source: Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M. & Zhang, H. 2021. The Earth's energy budget, climate feedbacks, and climate sensitivity. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu & B. Zhou, eds. *Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 923–1054. Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157896.001

Table 7. GTP values based on formulae from the IPCC's Sixth Assessment Report (AR6)

	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

Source: Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M. & Zhang, H. 2021. The Earth's energy budget, climate feedbacks, and climate sensitivity. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu & B. Zhou, eds. *Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 923–1054. Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157896.001

9.1.2.2 GTP

Another relatively common metric 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). 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 2023, it compares gases based on the temperature increase resulting from these emissions in the year 2123). As shown in Table 7, the GTP for short-lived gases is highly sensitive to the choice of time horizon.

The global temperature change potential 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 for the individual year specified. A key application of GTP would be for the quantification of the contributions to the emission of different gases with the goal of not exceeding any set temperature target at a specific future point in time.

The global temperature change potential 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 the AR6.

9.1.3 Step-pulse metrics

Due to the strong influence of the chosen time horizon on the pulse-emission metrics for the 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).

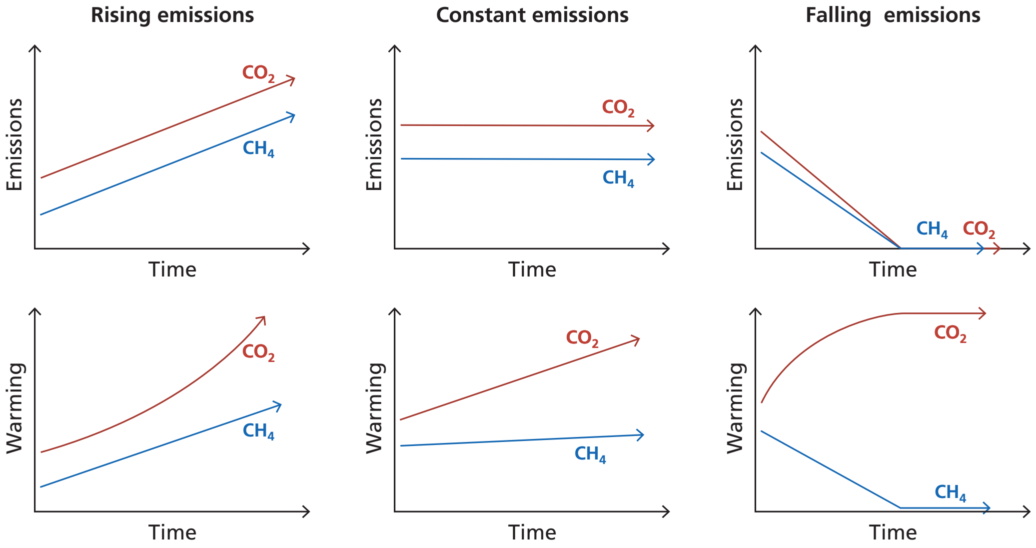
As this type of equivalence is based on matching eventual warming outcomes of the emissions being described, it has been suggested that step-pulse metrics can report a CO_2-warming equivalence (CO_2-we), in contrast to CO_2eq from pulse-emission comparisons (Cain *et al.*, 2019). There were earlier attempts to match warming or forcing outcomes under a scenario based on pulse- or step-based metrics (Wigley, 1998; Tanaka *et al.*, 2009a, 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 resulting in significant additional warming. This new sustained CH_4 source will drive an increase in temperature over the first few decades after its introduction. After this, the temperature will gradually stabilize, but at a higher temperature than before, as ongoing emissions will be balanced by the chemical reactions which will destroy atmospheric CH_4 following a few decades of stable CH_4 emission rates. This, in turn, will cause 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, Cain and Allen, 2021). This scenario is shown by the middle of the three panels in Figure 8, which illustrates constant emissions of CO_2 and CH_4 and the level of warming they each generate.

If the 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 8). To similarly reduce the level of warming from an earlier CO_2 emission, it would have to be actively removed from the atmosphere. Assumptions about past emissions and the climatic impacts they may still be exerting, and how to define existing or new sources, thus have a large impact on the calculated equivalent CO_2 emissions that would result in the same temperature change (i.e. CO_2-warming equivalence).

"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 a 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

Figure 8
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.

Source: Figure reproduced from Allen, M.R., Lynch, J., Cain, M. & Frame, D. 2022b. Climate metrics for ruminant livestock. Oxford, UK, Oxford Martin Programme on Climate Pollutants. https://www.oxfordmartin.ox.ac.uk/downloads/reports/ClimateMetricsforRuminent Livestock_Brief_July2022_FINAL.pdf

modelled temperature (Cain *et al.*, 2019; Smith, Cain and Allen, 2021). Lynch *et al.* (2020) demonstrated the validity of GWP* in a wider range of scenarios, exploring its use to estimate temperature responses to non-global emission trajectories, while Cain *et al.* (2021) used GWP* to evaluate scenarios which aim to achieve the Paris Agreement temperature goals.

The equation to convert a CH_4 emission – $CH_4(t)$ – to a CO_2-warming equivalent (CO_2-we(t)) – emission, using GWP*, is:

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

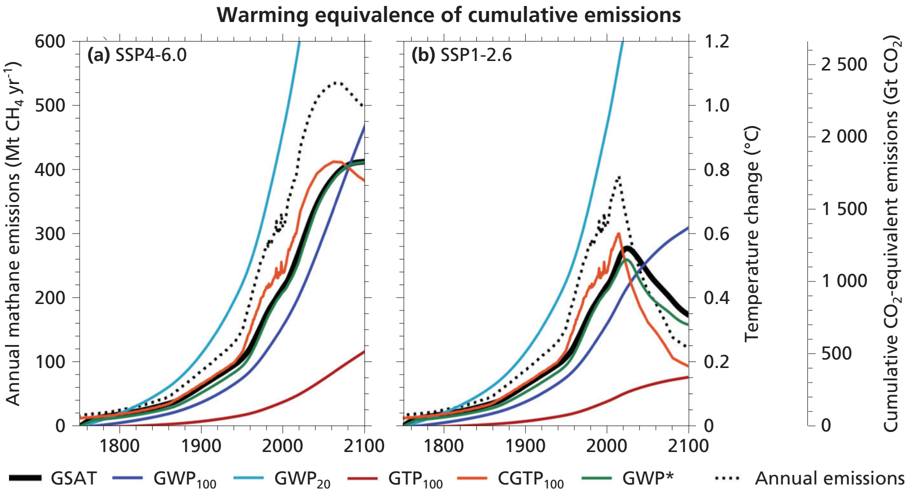
which simplifies to:

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

where GWP_{100} is the normal GWP for pulse emissions of CH_4 and CO_2 from AR5 (following Smith, Cain and Allen, 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 $\Delta CH_4(t) = CH_4(t) - CH_4(t-20)$ is the difference in CH_4-emission rate between time t and 20 years prior (Smith *et al.*, 2021).

Figure 9

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)



The temperature response from these emissions, calculated using an emulator, is shown with the black line (labelled GSAT for global surface air temperature).

Source: Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M. & Zhang, H. 2021. The Earth's energy budget, climate feedbacks, and climate sensitivity. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu & B. Zhou, eds. *Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 923–1054. Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157896.001

This formula for GWP* allows us to calculate 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 9. The first panel (a) shows a lower ambition scenario for CH_4 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 two-term approximation intended to find the CO_2-equivalent emissions that would generate the same radiative forcing time series as that 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 the combined global temperature change potential (CGTP), respectively. The CGTP metric is similar to GWP*, in that it compares the warming resulting from a step-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 the 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 $CGTP_{100}$ for the two scenarios are marked with an orange line in Figure 9, and show good agreement with the modelled warming (heavy black line) for both. The two step-pulse metrics ($CGTP_{100}$ and GWP*) are able to capture the reduction in warming resulting from CH_4 emission cuts, which cannot be captured with GWP_{100} (dark blue) or GWP_{20} (light blue). GWP* also represents the historical period more closely. Further discussion of both can be found in Forster *et al.* (2021).

9.1.4 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. 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, Minx and Toth, 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 10). 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.

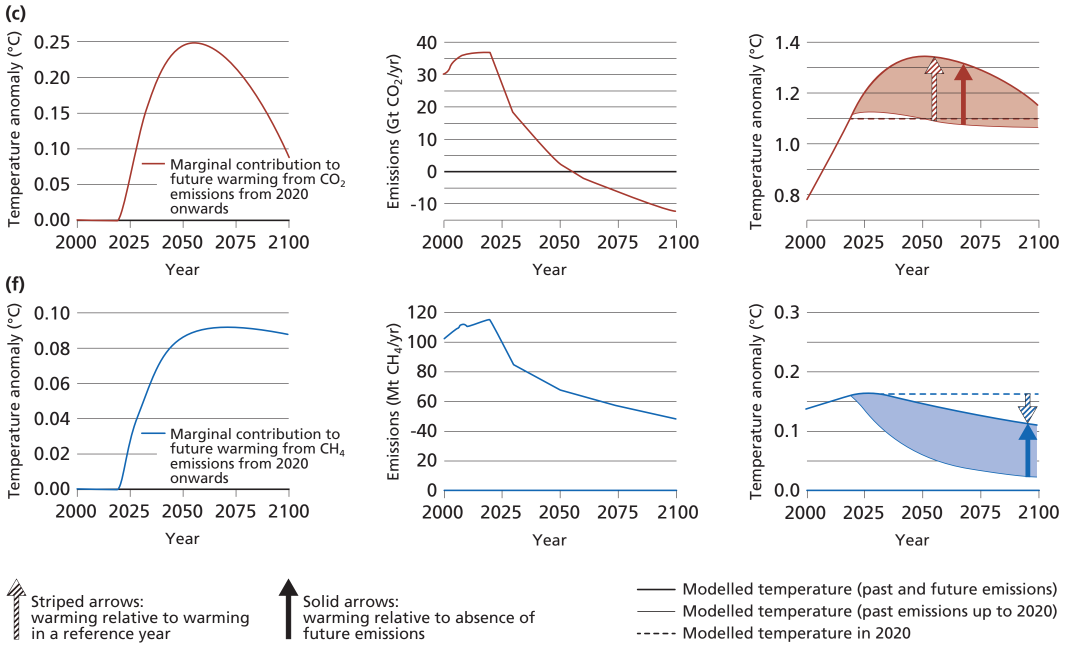
Pulse and step-pulse metrics can both be used to understand marginal and additional climate change outcomes, but they achieve this through different types of applications.

Pulse-emission metrics primarily provide information about marginal impacts. Each pulse-emission metric gives 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 (“additional” warming since the reference date). 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 10.

Figure 10

Contributions to global warming from global net CO_2 emissions and 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), while 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 the panels. Note that the scales, showing global net CO_2 emissions and livestock CH_4 emissions, are different in the vertical axes.

Source: Adapted from Reisinger, A., Clark, H., Cowie, A.L., Emmet-Booth, J., Gonzalez Fischer, C., Herrero, M., Howden, M. & Leahy, S. 2021. How necessary and feasible are reductions of methane emissions from livestock to support stringent temperature goals? *Philosophical Transactions of the Royal Society. Series A – Mathematical, Physical and Engineering Sciences*, 379(2210): 20200452. https://doi.org/10.1098/ rsta.2020.0452

Figure 10 provides an illustration of these different perspectives, with the striped arrows showing warming relative to a reference (or baseline) year ("additional warming"), and solid arrows showing warming relative to an absence of future emissions ("marginal warming"). It shows that the choice of defining impacts of emissions relative to a baseline/reference year 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 preindustrial temperatures) from either gas, with the thicker line indicating temperature-change contribution if the 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 shaded areas show the marginal warming from the two gases (i.e. the contribution to global warming from future emissions of those gases and, conversely, the amount of global warming that could be prevented if future emissions of those gases were avoided).

The relative temperature change resulting from these emission scenarios can be considered from two different perspectives: the effect on global temperatures relative to 2020 (which might be useful to assess 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 assess 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 Chapter 2 of IPCC, 2022 and supplementary material for more details), while the striped arrows correspond to the way step-pulse metrics such as GWP* have been used to date – what we refer to as a "baselined" approach below – which represents "additional warming" relative to that baseline year.

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 additional (striped arrow) approaches are very similar for CO_2, they provide very different perspectives on how to consider the impacts (or avoided impacts) of CH_4 emissions (solid and striped arrows in the lower right panel of Figure 10). These differences have important consequences 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. the 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 in subsequent sections.

In the case of step-pulse metrics, the CO_2 emissions that are described as equivalent to a given change in the 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 the 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 it 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 all these years could 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 subglobal scale (see Section 9.3.4).

Step-pulse metrics can directly illustrate the anticipated temperature changes resulting from different emission pathways and incorporate them into "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 a comparison with past emissions. Hence, there is no inconsistency between the different metrics, so long as it is recognized that they provide different information.

In principle, both pulse and step-pulse metrics can yield marginal or additional information. GWP* can be applied to a time series of emissions, with emissions at the beginning of the time series set to zero (e.g. Rogelj and Schleussner, 2019). 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 did not occur. For example, if one wished 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 10), 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 could thus be used in an additional approach.

9.1.5 Time horizon/endpoint for metrics

The pulse metrics discussed in section 9.1.3 depend very strongly on the chosen time horizon. The choice of time horizon depends on policy priorities. While given 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 a climate change mitigation strategy based on a relative valuation of present-day emissions according to their marginal contribution to this temperature goal, it could thus make sense to value each emission based on the contribution it makes to global 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). This approach is also referred to as the dynamic GTP (Shine *et al.*, 2007). 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 (GDamP) may be more relevant, although this is path-dependent (see Section 9.2.3).

9.1.6 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 if a metric is intended to reflect the future damages caused by each emission.

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, such as CO_2 and N_2O (van den Berg *et al.*, 2015). In contrast, 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 rely 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 used for example in GWP and GTP can be used as proxies for discount rates. By comparing the GWP to GDamP (see section 9.2.2), 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).

9.1.7 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 as well as being toxic to plants, with impacts on carbon uptake and crop yields (Shindell, Fuglestvedt and Collins, 2017). Reducing CH_4 emissions would therefore also reduce human mortality due to lower ozone concentrations, and Sarofim, Waldhoff and Anenberg (2017) calculated that this health benefit would exceed the climate change mitigation benefit of those emission reductions if they were valued at USD 46 per tonne CO_2eq. The UNEP CH_4 assessment (UNEP and CCAC, 2021) found that every Mt reduction in CH_4 emissions prevents approximately 1430 annual premature deaths in addition to 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 above 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 carries on for many decades after the emission occurs (Sterner, Johansson and Azar, 2014).

Summary points

A metric that establishes the equivalence regarding one key measure of the climate system's 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. what aspect of climate change does the policy focus on, and over which time horizon) {Section 9.1.1}.

The large difference in lifetimes for CO_2 and CH_4 means that the pulse-emission metrics strongly vary with the chosen time horizon {Section 9.1.2}. Step-pulse metrics for forcing and temperature (comparing a change in the rate of CH_4 emissions with a one-off emission of CO_2) show much less variation with the time horizon {Section 9.1.3}.

A step-pulse metric (GWP*) can be used to calculate an equivalent CO_2 emissions time series which gives a good approximation of the temperature time series that would result from the original CH_4 emissions time series {Section 9.1.3 and shown in Figure 9}.

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 that particular metrics and time-horizon ranges are more relevant than others {Section 9.1.6}.

Climate metrics for CH_4 include the radiative effects of the resulting increases in ozone (and stratospheric water vapour) but not the effects on human health and crop yields. These could double the social cost of CH_4 {Section 9.1.7}.

9.2 THE USE OF GHG METRICS IN IMPACT AND MITIGATION APPLICATIONS

Emission 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 climate-change impacts prevented 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 involved in specific decisions. Decision-makers may have to decide between different mitigation options with different costs and benefits, which may involve evaluating the impact of reducing CO_2 and CH_4 emissions. To make an objective choice between these options, decision-makers need to be able to quantify the effect of interest for 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 in 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, and scenarios targeting both CH_4 and CO_2 emission cuts lead to lower temperature outcomes (Sun *et al.*, 2021). Stakeholders may wish to set an individual reduction target for CH_4 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 compared with the level of ambition for other gases, by expressing their targets in terms of CO_2 equivalents.

9.2.1 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 impact of using or the benefit of avoiding a given product or service, or about the consequences of substituting one product or service for another. The ISO 14044 standard (ISO, 2006) not only specifies the requirements and provides guidelines for LCAs overall, but also for life cycle inventory (LCI) studies, which is the data collection portion of LCA. An LCI is accounting for all process inputs and outputs (including resource inputs and emissions to the environment) involved in the system of interest.

At 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 to fully aggregate them into a single score for typically 10 to 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 or removal of different greenhouse gas emissions 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 allocate emissions into processes/products), any LCA needs to choose appropriate impact assessment models. Available LCIA methods, including ReCiPe2016 (Huijbregts *et al.*, 2017) or LC-IMPACT (Verones *et al.*, 2020), bring together a number of environmental impact categories (e.g. carbon footprint or climate-change impacts, eutrophication, ecotoxicity and others) and propose characterization factors (CFs) so as to quantitatively link the elementary flows to the selected impact categories.

To provide guidance and standardize procedures, a United Nations Environment Programme (UNEP) working group gave recommendations for specific impact categories. The choices of impact categories and impact assessment methods need to be defined as part of a study's goals and the definition of its scope. This involves determining the temporal scope and selecting an appropriate metric for climate-change impact assessment or a simple climate model as used in LIME (Inaba and Itsubo, 2018; Tang, Tokimatsu and Itsubo, 2018). The 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 recommends that the selection of the impact categories should be based on the specific requirements of the LCA practitioner for meeting the objectives of a given 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 issued by 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 to estimate 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 Programme (UNEP, 2021) has recommended 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 exploring very short-term impacts (Cherubini *et al.*, 2016; Levasseur *et al.*, 2016; Jolliet *et al.*, 2018). These recommendations used metric values from IPCC (2013) that have subsequently been applied in various impact assessments (e.g. Reisinger, Ledgard and Falconer, 2017; Iordan, Verones and Cherubini, 2018; Tanaka *et al.*, 2019; Tibrewal and Venkataraman, 2021). It should be noted that the definition of very short, short, and long term is subjective. These considerations and wider points are also discussed in another recent report on the LCA for food items published by FAO (McLaren *et al.*, 2021).

Weighing up CH_4 reductions vs other factors is even more difficult. An LCA can provide the framework to ensure the analysis is comprehensive and that different ways of valuing CH_4 emissions (or reductions) can be used to assess the benefits of reducing emissions against potential negative trade-offs. These may be due to increased emissions of other greenhouse gases or other ecosystem services, such as food production or other environmental benefits. Some studies have also attempted comparing and aggregating various LCA impact-indicator categories to directly quantify the combined overall impact across all the different individual impacts considered. They include the so-called "endpoint" methods (e.g. ReCiPe and LC-IMPACT), which divide all the 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}. However, 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 final weighing up should reflect different values that need to be agreed on in an open discussion. The underlying issues are also further highlighted and discussed in the cross-cutting section of this report. Such an aggregation into a single-score LCA result needs to be critically examined since it rests on many normative choices and can disguise the complexity and trade-offs involved in LCA assessments. Modelling from impact-category results (e.g. CO_2 equivalents) to endpoint results (e.g. impacts on human health) leads to more uncertainty, as the 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. Needless to say, it is important to reflect on the choice of metrics as these can greatly affect the outcome of any assessment, but no general guidance can be given on the metrics to use as this will depend on the goals and objectives of the study.

9.2.2 Cost-benefit assessment of climate change mitigation

A cost-benefit analysis requires the benefits of reducing climate change-related damages by avoiding future emissions to be quantified. This would allow us to eva-luate the trade-offs between greenhouse gas mitigation choices and any detrimental effects from climate change (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, Luderer and Edenhofer, 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 the catastrophic impacts of climate change (Weitzman, 2012, 2013; Pindyck, 2013). Where included, the impact of catastrophic phenomena (for example, the dangerous rise in sea level or uncontrollable positive climate-forcer feedbacks such as a large and rapid release of CH_4 from permafrost) can drastically increase the estimated damage values (Weyant, 2017).

One emission metric that is consistent with the cost-benefit framework is the 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, Luderer and Edenhofer, 2013). The GDamP has been derived from an optimal pathway indicated by an integrated assessment model (IAM) within a cost-benefit framework. Under an optimal pathway, the GDamP is defined as the ratio of incremental damages avoided 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 the one hand, the GDamP is the most comprehensive available metric in the context of a 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 the 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, Fuglestvedt and Collins, 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 a 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 was 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 often applied metrics such as GWP_{100} from a cost-benefit perspective.

9.2.3 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, 2021). The GCP, which can be seen as a more general form of the GTP (Tol *et al.*, 2012), is defined as the ratio of the cost 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 9.2.2), a calculation of the GCP requires an IAM (but one that is 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 towards 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 that have 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.*, 2009a, 2013). A dynamic metric uses a time horizon with the end point typically being tied to the year of meeting a climate target (Berntsen, Tanaka and Fuglestvedt, 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 the ongoing use of GWP_{100} instead of that 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, Fuglestvedt and Godal, 2006; Johansson, Persson and Azar, 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), 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 making the GCP operational because the value of the GCP itself requires an assumption on a long-term future emission pathway towards a temperature goal. As a compromise, the use of the GCP has been recommended 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 CH_4 emissions to cost-effectively achieve some future temperature target, or simply quantify the marginal damages caused by CH_4 emissions, one arrives at CO_2-equivalent metrics for CH_4 somewhere between about 20 and 40. This is roughly consistent with GWP_{100}, but contrasts with values generated using other metrics such as GTP_{100} or GWP_{20}. When more complex net emission patterns are involved, however, then the use of different metrics applied to the same net emission patterns can result in very differently assessed mitigation outcomes (Brandão *et al.*, 2019). Therefore, even though GWP_{100} was not developed to derive cost-benefit or cost-effective outcomes, it may be adequate for those purposes and is not necessarily incompatible with cost-benefit and cost-effectiveness approaches to climate policy (e.g. cross-chapter Box 2 of IPCC, 2022).

9.2.4 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. Emission reduction policies usually involve trade-offs. To satisfy the 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 enable cross-sectoral comparisons as well. 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 lifetimes 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 other sectors' CO_2 in a meaningful way.

The use of metrics becomes necessary a) if one wants to compare the contributions of different emitters, sectors and so on that may be emitting different gases, or b) where there are potential policies' trade-offs that may differentially affect the emissions of different gases so that the value of CH_4 reductions must be weighed against possible increases in the emission of other gases. For a), it comes back to the issues addressed earlier in this chapter, and may be factored into 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 impacts for land use, as discussed in an earlier chapter (Section 8).

9.2.4.1 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 and as a result many actions aimed at reducing agricultural CH_4 emissions will have impacts on other GHGs; for example, some feed additives will lead to higher CO_2 emissions but reduce CH_4 emissions (as discussed in the subsections of Section 5 addressing mitigation). Therefore, policies developed to reduce emissions should consider the knock-on effects of mitigation actions on different GHGs, trade, food security, land use, water consumption, water and air pollution, among other factors.

Direct or indirect leakage of CH_4 emissions across national boundaries becomes an issue when evaluating any national mitigation strategies. Clearly, a reduction in ruminants in one country achieved through importing products derived from an increased ruminant population in another country (with a potentially higher emission intensity) would be an instance of direct leakage that could be reported as mitigating emissions in the importing country while effectively increasing them in the exporting country, with no beneficial effect for the world as a whole (depending on the respective emission intensities of the two countries). An example of indirect leakage, on the other hand, would be if one country reduced its CH_4 emissions by reducing the exports of ruminant-derived products and if that action resulted in increased ruminant numbers in the countries that previously imported ruminant products to replace the lack of availability of imports. This is an issue for all food-production systems since any change in food production in one region may have an effect either on the production level in other regions or 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, one that is deemed acceptable under the Paris Agreement, would be to jointly report on the mitigation activities of multiple countries that produce livestock products as well as trading a large number of other goods between themselves. This would be one possible way of ensuring that assessments factor in the 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 is used as the reference for emissions produced by components within the system, or inputs into and outputs from the system, it is essential to describe emissions of each GHG rather than using an available aggregate GHG-equivalent emission calculated with a single metric.

Because there has been no scientific consensus on how to separate 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.

That said, natural CH_4 emissions from wetlands, inland waters and wildlife (including insects) that occur on managed land could be included in scenarios of CH_4 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 in turn affect wild ruminant populations as well as termite abundance, which are both sources of CH_4 (Manzano and White, 2019).

For example, a comprehensive regional mitigation evaluation conducted in Sweden (Skytt, Nielsen and Jonsson, 2020) that included natural emissions, found that reducing CH_4 emissions from water bodies rather than CH_4 emissions from livestock was the preferred mitigation action. To gain greater clarity and allow better interpretation of the results, it could be useful to separately report indirect and direct emissions, where possible.

9.2.4.2 Designing a holistic emission reduction strategy

The successful mitigation 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, giving due consideration to the appropriate time horizons in order for the results to be achieved cost-effectively while minimizing any unintended adverse consequences.

Some policy decisions could end up temporarily delaying CO_2 reduction efforts in favour of action on other gases or vice versa, if this can achieve specific goals in a more cost-effective way, especially at subglobal levels. What is deemed the best approach calls for a public policy debate, addressing different socio-economic developments and how alternative mitigation strategies might contribute to the overall goal of sustainability. The do no significant harm (DNSH) principle invoked by the European Commission may also be helpful in this context. It states that a measure should not lead to significant harm to the environment or get in the way of any one of the six major environmental objectives.

Assessing a variety of potential scenarios and applying different metrics is necessary to have a better understanding of the outcomes of different policy designs. Any GHG emission metric simplifies the complexity of the climate system response to greenhouse gas emissions. Instead of describing future emission mitigation targets in terms of CO_2 equivalents, which are ambiguous, it would be clearer if these targets were also specified for individual gases – even if only indicative distinctions could be made, or if the emissions of the long-lived and short-lived GHGs were at least treated separately (Denison, Forster and Smith, 2019; Allen *et al.*, 2021).

The 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 the GWP_{100} or other single-pulse metrics. This matters especially if climate targets are not just addressing impacts at a certain point in time, but also assessing temporal developments and any trade-offs of warming impacts of different GHG policies before and after this point in time. For integrated policies, it is important that the goals not be independently defined for the different sectors of the economy, but that they also address the policies' effectiveness and any trade-offs between sectors that may be required to cost-effectively limit overall GHG emissions.

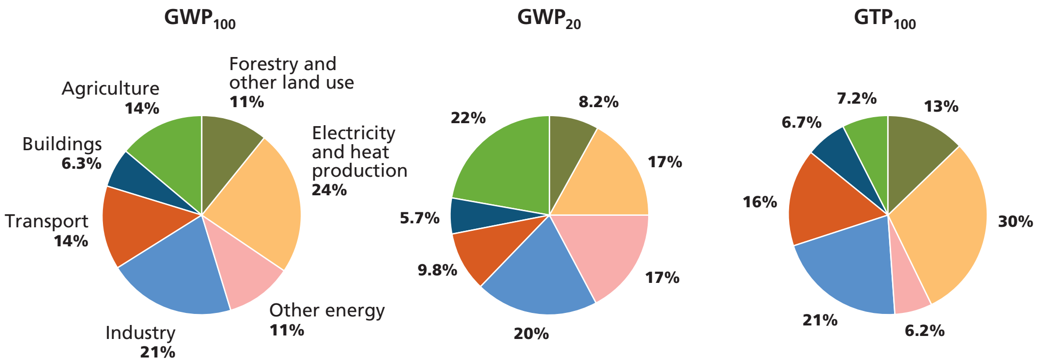
Additionally, reducing GHG emissions might involve agricultural intensification that could have negative impacts on animal welfare and biodiversity. In some sectors, GHG reductions might entail fewer trade-offs than in others, and even synergies. In order to assess the impact of changes on whole economies and to find the most effective GHG mitigation solutions, integrated assessment models should be used. Because these contain considerable uncertainties, it is best to apply several models and scenarios, which for the time being puts a great strain on resources. Improved scientific tools might be required for broader applications. The following sections provide some insights based on available studies.

9.2.5 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 is thus highly dependent on the specific metrics used for the integration. For instance, 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 11). The calculated contribution of agriculture to total greenhouse gas emissions ranges from 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.

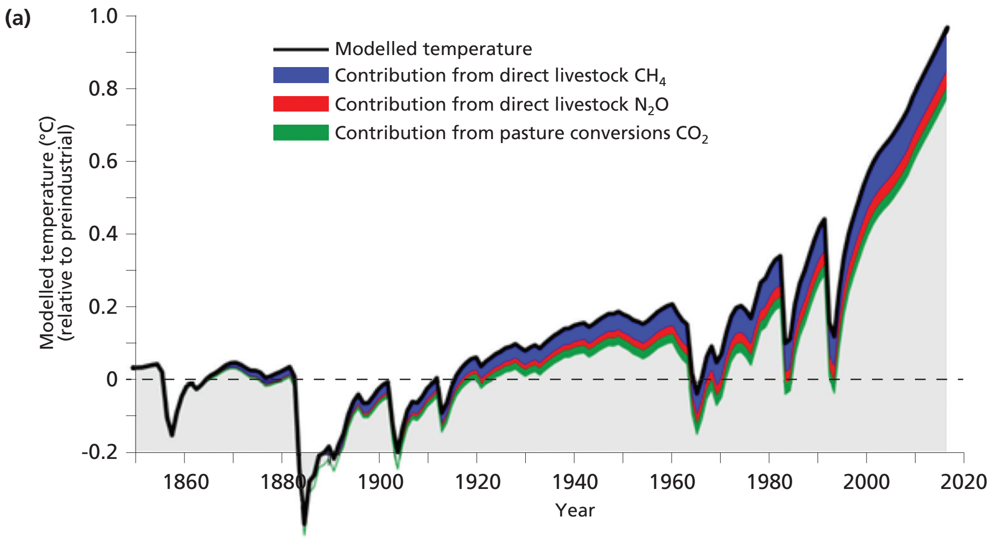
The climate impacts of different sectors can also be compared by exploring their contribution to global temperature increases from past emissions. This provides an alternative perspective and overcomes the problem of relying on different greenhouse gas metrics to make comparisons between the emissions 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 global livestock-based emissions to global temperature increases up to 2015 (Figure 12).

Figure 11
Sectoral contribution to annual total greenhouse gas emissions in 2010 weighted by three different greenhouse gas metrics, GWP_{100}, GWP_{20} and GTP_{100}



Source: Reproduced from IPCC. 2014. *Climate change 2014: Synthesis report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (R.K. Pachauri & L.A. Meyer, eds.), 151 pp. Geneva, Switzerland.

Figure 12
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).

Source: Reproduced fromReisinger, A. & Clark, H. 2018. How much do direct livestock emissions actually contribute to global warming? *Global Change Biology*, 24(4): 1749–1761. https://doi.org/10.1111/gcb.13975

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.

9.2.6 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 regarding the life cycle assessment (Section 9.2.1). The sensitivity to multiple emission metrics, such as (but not limited to) GWP_{20}, GWP_{100} and GTP_{100}, could be explored to highlight how different impacts vary over a range of time frames.

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 their short atmospheric lifetime. Methane emissions in later parts of this century can therefore have greater climate-change impacts than emissions during the previous century because the warming contribution of CH_4 in the late twenty-first century would be felt at a time of higher global background temperatures. This wider analysis could include a re-analysis under different metrics or aggregation methods, or employ more robust climate modelling approaches.

9.2.7 Biogenic methane – Implications for metrics

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

In the context of climate change, it is important to distinguish between CH_4 indirectly generated from carbon in recently grown biomass and that derived from old carbon sources, such as fossil deposits. If CH_4 is generated from newly grown biomass, for example by enteric fermentation, carbon is converted to CH_4 whereas it 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 oxidized, the carbon will transform back into CO_2, a process that is common to both biogenic and fossil CH_4. In the case of CH_4 from fossil origin, this oxidation leads to a net increase in atmospheric CO_2 concentrations by adding more 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, in contrast, 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 Muñoz 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 to CH_4 of 2.75. The default values for metrics in IPCC AR5 were for biogenic CH_4.

However, this does not account for the difference in temporal developments. IPCC AR6 assumes that only 75 percent of CH_4 oxidation leads to CO_2, while 25 percent of carbon is removed by deposition of reactive intermediates. Accounting for the time taken for CH_4 oxidation, this 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) suggested 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

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 mitigation efforts but also other relevant climate and non-climate factors.

Applying 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 CH_4 emissions, it is necessary to distinguish between CH_4 derived from fossil versus of recent biogenic origin, with global warming potentials of CH_4 of fossil origin assigned a warming potential higher by 2.75 units.

9.3 CLIMATE TARGETS AND RELATED ISSUES

The agriculture and livestock sector's use of climate metrics occurs within a larger policy context relating to climate action and sustainable development. This section discusses the Paris Agreement, the term "climate neutrality", sustainable agriculture and equity. Its aim is to provide additional information, so that users can make decisions informed by the wider context, including the global goals to which every sector contributes. The section does not formulate any specific recommendations.

9.3.1 The Paris Agreement

9.3.1.1 The goals of the Paris Agreement

The Paris Agreement provides the current 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 preindustrial levels and pursuing efforts to limit the temperature increase to 1.5 °C above preindustrial 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).

9.3.1.2 The Paris Agreement and methane emissions

It is important to note that the Paris Agreement does not specifically discuss CH_4. 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’s Sixth Assessment Report (IPCC, 2021a; IPCC, 2022) underscores the need for deep, rapid and sustained reductions in greenhouse gas emissions and reaching at least net-zero CO_2 emissions. These reports also highlight the importance of strongly reducing the emissions of other greenhouse gases and air pollutants, especially CH_4, 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 the trade-offs between reductions of different gases while working toward the goals of the Paris Agreement.

9.3.1.3 Use of climate metrics

The Paris Agreement does not specify metrics. Nor does it use terms such as “net zero” or “carbon neutral”, although Article 4.1 does refer to the goal of “achieving a balance of anthropogenic sources and sinks”. That said, the common metric used under the UNFCCC since the Kyoto Protocol has been GWP_{100} (UNFCCC, 1997). The use of GWP_{100} following the Paris Agreement has been assessed in the academic literature, with varying conclusions. In its summary for policymakers, the IPCC (2021) stated that emission pathways which reach and sustain the net-zero GHG emissions defined by GWP_{100} are projected to result in a decline in surface temperature after an earlier peak. Schleussner *et al.* (2019) concluded that interpreting the Paris Agreement climate objectives by using GWP_{100} is internally consistent. In contrast, Wigley (2021) concluded that temperature outcomes in a scenario using GWP_{100}-scaling for methane were erroneous, and therefore its use was not recommended.

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, parties decided to use GWP_{100} values without climate–carbon feedback to report aggregate emissions and removals as provided by the IPCC's Fifth Assessment Report (AR5) (or a subsequent IPCC report upon future agreement). Besides the mandatory reporting based on GWP_{100}, the COP24 decision also allows countries to report additional information on aggregated CO_2-equivalent emissions by using other metrics assessed in IPCC reports, such as the GTP, for example. In addition, parties to the Paris Agreement agreed to use the same emissions reporting framework to account for their nationally determined contributions beyond 2030 (Decision 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	27.0
CH_4 fossil origin				30	29.8
	20-year time period				
CH_4 non-fossil origin	56	62	72	84	79.7
CH_4 fossil origin				85	82.5

Source: Authors' own elaboration.

9.3.1.4 Discussion of long- and short-lived greenhouse gases in recent IPCC reports

Following the adoption of the Paris Agreement, the UNFCCC invited the IPCC to produce a Special Report on global warming of 1.5 °C (IPCC, 2018). The report 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)." It posits a distinction between net-zero CO_2 emissions and net-zero GHG emissions, which is further emphasized in the Sixth Assessment Report's Summary for Policymakers:

> D.1.8 Achieving global net zero CO_2 emissions, with anthropogenic CO_2 emissions balanced by anthropogenic removals of CO_2, is a requirement for stabilizing CO_2-induced global surface temperature increase. This is different from achieving net zero GHG emissions, where metric-weighted anthropogenic GHG emissions equal metric-weighted anthropogenic GHG removals. For a given GHG emissions pathway, the pathways of individual GHGs determine the resulting climate response, whereas the choice of emissions metric used to calculate aggregated emissions and removals of different GHGs affects what point in time the aggregated GHGs are calculated to be net zero. 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 after an earlier peak (*high confidence*) (IPCC, 2021c, p. 30).

How individual gases contribute to global temperature increases is thus differentiated based on the distinct dynamics between long- and short-lived gases.

For CH_4, 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 for atmospheric concentrations to be largely driven by emissions occurring only in recent decades. Thus, bringing emission rates down below levels experienced a few decades ago will lead to reduced anthropogenic CH_4 concentrations, implied forcing and a contribution to temperature change. The time-independent relationship between warming and total cumulative emissions, observed for CO_2, therefore does not apply to CH_4.

To limit further temperature increases, the requirement for "net-zero" emissions – where emissions must either be completely eliminated or offset with additional CO_2 removals – is only strictly necessary from a physical science perspective for CO_2, given that its cumulative impacts extend into the very long term. For short-lived gases, a climate impact equivalent 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 (IPCC, 2022), and it would in theory be possible to achieve this temperature goal without entirely eliminating or offsetting CH_4 emissions (Tanaka and O'Neill, 2018). It should be noted, however, that the Paris Agreement does not refer to stabilized temperatures, but rather sets upper limits for temperature increase (Mace, 2016). Schleussner *et al.* (2019) show that using a step-pulse metric such as GWP* in the context of the Paris Agreement goals could undermine the integrity of the Agreement's mitigation target by failing to deliver net-zero CO_2 emissions and ensuring that warming is halted.

The rationale for treating CH_4 differently to CO_2 is made more evident still in the IPCC's Special Report on global warming of 1.5 °C (IPCC, 2018). It states that the interquartile range of methane emissions from agriculture across pathways assessed, which limit global warming to 1.5 °C with no or limited overshoot, should fall globally by approximately 11 to 30 percent by 2030, and 24 to 47 percent by 2050, relative to 2010 levels.

The physical dynamics of how long- and short-lived gases contribute to overall temperature change are well understood, and the different gas-specific options for reaching any given climate target are widely recognized (Allen *et al.*, 2021). However, the fundamental physical requirements outlined above are not the only things that a multi-gas climate policy must consider. Cost-effectiveness, equity and technical feasibility are other important considerations. Indeed, all modelled global pathways assessed by the IPCC that limit warming to 1.5 or below 2 degrees show strong and sustained reductions of global CH_4 emissions, in addition to reaching at least net-zero CO_2 emissions (IPCC, 2022).

9.3.2 Climate neutrality

9.3.2.1 Different uses of the term

"Climate neutral" is a term that is being used with increasing frequency. However, since the concept of climate neutrality is not uniquely defined, it is used in a variety of ways and with a variety of meanings. It is thus important for the term to be clearly defined, whenever it is used, so as to avoid misunderstandings.

In many cases, to talk of climate neutrality is synonymous with achieving net-zero GHG emissions (consider, for instance, the United Nations' Climate Neutral Now initiative, https://unfccc.int/climate-action/climate-neutral-now). For the aggregation of different GHG emissions and removals, the GWP_{100} climate metric is typically used. Climate neutrality is sometimes used more or less synonymously with the term “carbon neutral”, for example in the draft ISO standard ISO/CD14068, where carbon neutral includes all GHGs. However, in its Sixth Assessment Report, the IPCC 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” (IPCC, 2021b, p. 2221). The IPCC uses the term GHG neutrality when other non-CO_2 greenhouse gases are included.

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. The IPCC’s Special Report on global warming of 1.5 °C describes climate neutrality in the following terms:

> 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 (IPCC, 2018, p. 545).

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, 2021a) or no net contribution to additional temperature increases (Costa *et al.*, 2021; Place and Mitloehner, 2021; Allen *et al.*, 2022b). These distinctions are made by analogy with the CO_2-specific outcomes of achieving carbon neutrality as defined above, and based on the understanding that to stabilize the climate at any level 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 CH_4, since an ongoing emission of CH_4 at a slowly reducing rate results in a contribution to warming that remains stable over time. The evaluation of targets such as no net additional change in temperature at the national or corporate level cannot be undertaken from a physical science perspective alone; it also depends on economic, social, equity and political considerations, including responsibility for past warming (Allen *et al.* 2022a). This concept of climate neutrality has been applied 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, Cain and Allen, 2021). It is important to note that the stabilization of temperature contributions from each individual gas does not tell us whether this is cost-effective or equitable, or whether the implied emission reductions are technically feasible.

Each of the above definitions of climate neutrality – net-zero GHG emissions using GWP_{100}, having no net effect on climate, or no net *additional* change in temperature or radiative forcing relative to a reference date – can have very different implications for the emission reductions that would need to be achieved if short-lived gases such as CH_4 play a significant role in the overall emissions of a sector, and for the total amount of global warming contributed by sectors that adopt “climate neutral” targets. For this reason, claims and targets of climate neutrality, and the mitigation ambitions implied by those targets, are easily misunderstood and misinterpreted unless their specific meaning and implications are clarified.

9.3.2.2 Climate metrics and climate neutrality

If a *carbon neutral* commitment is made, following the definition in the IPCC's Sixth Assessment Report, 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, the metric-weighted anthropogenic GHG emissions associated with a subject are balanced by metric-weighted anthropogenic GHG removals. Neutrality often includes Scope 3 emissions (which are indirect emissions either upstream or downstream of a business, and not directly within the business's control). Net-zero GHG emissions are also metric-weighted net anthropogenic GHG emissions, but often do not include Scope 3 emissions. Different organizations work with 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 metrics to reach and sustain net-zero GHG levels will affect their resulting temperature outcome (IPCC, 2021; Fuglestvedt *et al.*, 2018). 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 the GWP_{100} (IPCC, 2021). However, it is important to be mindful that when organizations make commitments to reduce and/or offset aggregated GHG emissions using the GWP_{100} climate metric, it is not immediately clear how much this will change 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 and O'Neill, 2018; Allen *et al.*, 2021, 2022c). 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 global warming evolution resulting from global 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 levels at stabilization will depend on cumulative CO_2 emissions over the entire historical period and the ongoing emission rates of short-lived gases. While this is a robust physical concept, the assessment of what constitutes an appropriate target for any subglobal entity also depends on economic, social, equity and political considerations, including responsibility for past warming (Allen *et al.*, 2022a).

The vast differences in the future climate impact of short- and long-lived climate forcers are a key issue. 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 time frame. Even N_2O has a lifetime and climate impact that exceeds the time frame by which climate stabilization needs to occur if the Paris Agreement temperature targets are to be met, if not far exceeded. Additionally, the emission of these GHGs will not only hamper the achievement of climate targets by 2100, but also lead to increased temperature over longer time frames. However, short-lived climate forcers like CH_4, with an atmospheric lifetime in the order of a decade or less, do not need to be reduced to net zero to achieve the Paris Agreement goals. With CH_4, a modestly reducing emissions profile over time, whereby new emissions are balanced by the decay of CH_4 from recent historical emissions, leads to CH_4-caused climate stabilization, at a level determined largely by the ongoing rate of CH_4 emissions. However, greenhouse gas emission-reduction pathways that limit global warming to below 2 °C, assessed by the IPCC's Sixth Assessment Report (IPCC, 2022), require stringent CH_4 emission reductions of between 45 and 50 percent by 2050 relative to 2019 levels, given the plausible rates of energy decarbonization and the anticipated evolution of other climate forcers. More substantial CH_4 emission reductions provide a mechanism to lower the temperature, and thus may be an important contributor towards achieving the Paris Agreement temperature goal, where the unmasking of aerosol warming will lead to additional challenges (Shindell and Smith, 2019), e.g. limiting peak temperatures (e.g. Smith *et al.*, 2012) or contributing to a decline in temperature after 1.5 °C is exceeded.

Using step-pulse metrics to define global 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. While this could be used to achieve climate stabilization globally, it does not follow that GHG neutrality defined in this way is consistent with the Paris Agreement goals (Mace, 2016). In addition, we know with high confidence that merely stabilizing global warming due to CH_4 would make achieving temperature goals much more difficult or impossible to achieve because stabilizing radiative forcing from CH_4 would add approximately 0.2 °C to global warming in 2100 compared to a typical 1.5 °C-compliant scenario (Cain *et al.*, 2022). Current CH_4 emissions contribute about half a degree to global warming (IPCC, 2021), which could be reduced by future emission reductions. The remaining carbon budget for limiting warming to 1.5 °C ranges between 600 and 300 Gt CO_2 depending on whether non-CO_2 climate forcings are strongly or weakly mitigated, further demonstrating the magnitude of their role (IPCC, 2023).

An approach using GWP* at a subglobal level has been used in case studies in the livestock sector (Ridoutt, 2021b; del Prado, Manzano and Pardo, 2021). A key challenge in applying the method has to do with 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 organization or sector can assess whether their total contribution to radiative forcing is increasing and take management action to stabilize or reduce it. A situation where an organization or industry is making no additional contribution to radiative forcing could be regarded as consistent with climate stabilization and described as climate neutral for this particular definition of the term, noting that such an interpretation can be contested on the grounds of economic, social, equity and political considerations, including responsibility for past warming, especially given the lack of accepted definitions of climate neutrality, as outlined above. This approach has been 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 not 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.

Neither of these approaches to neutrality alone can resolve the question of what an acceptable level of radiative forcing from an organization or industry could be. If an acceptable level of radiative forcing or global warming could be identified on the basis of economic, social, equity and political considerations, then any of the above definitions could then be applied in such a way as to remain within that acceptable limit.

9.3.3 Methane abatement and sustainable agriculture

Climate metrics can help to define and report climate goals and actions from a multigas perspective. This helps to assess the impacts of emissions and removals of different greenhouse gases and fosters an understanding of the trade-offs between near- and longer-term climate effects. However, in pursuing climate action, it is important to also consider wider sustainability goals, such as those outlined in the United Nations' Sustainable Development Goals, as well as sustainability priorities relevant to 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.

9.3.4 Equity considerations

A 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. There are differences across countries for both residual emissions and the removal potential, leading to scientific, political and equity issues related to global net-zero GHG emissions (Fuglestvedt *et al.*, 2018). Certain applications of emission metrics may raise equity concerns if relevant issues are not considered upfront, 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 non-linearities in the climate system (Trudinger and Enting, 2005; Höhne and Blok, 2005).

Pulse-emission metrics (such as GWP_{100}) 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 tracing the whole emission trajectory from a sufficiently early baseline (for example, preindustrial). 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, if they are not applied to the full historical emissions, need to be mindful of equity considerations. In this sense, Rogelj and Schleussner (2019) argued that, given the inequality in historic emissions, using GWP* with a present-day baseline could result in highly unequal and unfair outcomes benefitting historically high-emitting countries, sectors, even down to the individual company or farm level. This "grandfathering" can be avoided by taking a preindustrial baseline, as noted above, but this raises challenges for the equitable allocation of responsibility for the warming within countries. This underscores that reflections on equity and fairness are central to any application of step-pulse metrics on the national or corporate level since the choice of the baseline (which has a significant impact on the metric-reported "equivalent-emissions") is a normative decision and not a physical one. These specific equity considerations do not apply to the use of pulse metrics such as GWP_{100}, as they treat every unit of a given GHG equally and independently from the emitter and the point in time at which the emission occurred.

On the other hand, Lynch *et al.* (2020) also argue that GWP_{100} net-zero targets implicitly set a baseline target for CO_2-induced warming at whatever level was reached prior to arriving at net-zero, irrespective of how much warming an emitter may continue to cause through their past emissions, and hence of ongoing responsibility for climate damages. Similar concerns about continued warming from historical CO_2 emissions were the basis of the "Brazilian proposal" for the Kyoto Protocol to set emission reduction targets based on historical contributions to global warming. However, setting a net-zero emissions target based on GWP* baselined on present-day emissions would compound rather than resolve such inequities, since it would retain the imbalance caused by historical CO_2 emissions, and add additional but separate inequities by allowing emitters with currently high CH_4 to continue emitting CH_4 at a high rate into the indefinite future, whereas emitters with currently low CH_4 emissions would be forced to remain at those low levels.

Metric selection and appropriate deployment may be chosen to reflect certain equity considerations, but requires a user to recognize and choose a certain perspective on the fundamental concerns raised, such as whether and how to set effort-sharing expectations based on the mitigation potential of contemporary emission reductions or on 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. The assumptions behind the selection of metrics and the interpretations regarding matters of equity should be reported transparently.

9.4 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 result from emissions, and may report these effects covering various time frames or with respect to various 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 organization. We recommend that practitioners follow the guidance set out below and consider how each point is relevant to their particular needs. Two examples are provided to show that the way a question is framed influences the selection of appropriate metric.

9.4.1 Points to consider

Example boxes: Section 9.4.1.2 describes a case study (Example 1) assessing the impact of a dairy farm which could start using a feed additive to reduce CH_4 emissions. Throughout Section 9.4.1, in boxes such as this one, we have included the steps relevant for each "point to consider", which were taken in considering the first example. A full account of this case study can be found in Section 9.4.2 and in the Appendix, which also features detailed modelling work.

9.4.1.1 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 apparent. For example, practitioners may be asked to define an emissions reduction target. But what is their overarching goal?

These goals might be:

- to minimize emissions of specified GHGs;
- to achieve some externally-determined target for aggregated GHG emissions based on a predefined metric;
- to limit (at a chosen level) or undo the organization's overall contribution to global warming;
- to identify a target that includes budgetary considerations;

- all of the above; and
- 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 one to address a particular question. When there is 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 9.1.1.

Example 1: A dairy farmer wants to assess the benefits of using a particular feed additive on their herd. The question is defined as follows: If I start using the feed additive, what will be the impact on climate change compared to not using the additive? This action is motivated by the desire to reduce the farm's environmental impact in the coming decades. The current emissions are known, and the farmer can assume that these emissions will remain stable and compare this to the emissions that would be generated if the feed additive were to be introduced.

9.4.1.2 Existing requirements for metrics

This may already be included in the answer to the first question. However, if that is not the case, are there any regulations which require the use of a particular metric? Although a particular metric may be mandatory in some cases, it is worth considering whether it fully meets your needs. If it does not, another metric or modelling exercise may be needed to inform your plans or policies. For example, a sensitivity analysis using several metrics can guide you in setting your targets to ensure that the impacts at different timescales and on temperature outcomes are duly considered. This sensitivity analysis can be particularly useful if your overall aim is to gain a better understanding of the wider environmental impact of different strategies.

See Section 9.1.2 and Section 9.1.3.

Example 1: The farmer already uses GWP_{100} in an existing GHG footprint calculator, but turns to other greenhouse gas metrics to inform an internal strategic analysis.

9.4.1.3 Time frame

Alternative emission metrics can differ greatly in how they report greenhouse gases as "equivalent" to one another (see Section 9.1 for further details). These differences arise primarily because different greenhouse gases show a distinct time-dependence in their impacts. Emission metrics typically set a predefined time horizon to constrain comparisons and provide a single measure of equivalence, where different time horizons will result in different valuations. In order to make a judgement on a suitable metric, the time frame under consideration to answer your question or meet your goal must therefore be explicitly considered. Is your priority minimizing your operation's contribution to global warming in 2050, 2100 or at another specific time, at all of these times and in any of the intervening years, or over an indefinite period so as 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 years) time horizon and another one for a long (e.g. 100 years) time horizon will show the difference in the temporal impacts of climate pollutants. This improves transparency as no single-term metric can effectively capture the time-dependency of the impacts of short-lived climate pollutants and long-lived climate pollutants (SLCPs and LLCPs). Ocko *et al.* (2017) compares 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 Life Cycle Initiative, jointly hosted by the United Nations Environment Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC), recommends reporting both the GTP_{100}, to indicate longer-term climate impacts, and the GWP_{100} to indicate shorter-term climate impacts, and optionally the GWP_{20} for very near-term climate impacts (Jolliet *et al.*, 2018). Proposed long-term metrics for the metric pairing include GTP_{100} (Cherubini and Tanaka, 2016; Cherubini *et al.*, 2016; Levasseur *et al.*, 2016; Jolliet *et al.*, 2018) and GWP_{100} (Ocko *et al.*, 2017). Proposed short-term metrics for the pairing include GWP_{100}, GWP_{20} and GTP_{20} (Cherubini *et al.*, 2016; Cherubini and Tanaka, 2016; Levasseur *et al.*, 2016; Ocko *et al.*, 2017; Jolliet *et al.* 2018). Another option would be to use the long-term impact metrics CGTP or GWP* to evaluate endpoint temperatures 100 years from now. As commonly used, they would report additional warming relative to a chosen year. This information could still be compared with a short-term metric if applied to 20 years from now.

The use of two or more metrics with different time horizons or formulations can help understand how robust a given mitigation strategy is across a range of time horizons and given various underlying motivations. For example, if a given mitigation strategy results in climate benefits when both GWP_{20} and GTP_{100} are used as alternative metrics, then this would be regarded as a highly robust strategy; whereas if a given mitigation strategy would deliver climate benefits for one metric but would increase climate change under another metric, then additional thought may be warranted to determine whether the strategy should be adopted. Note that, even if it does not yield benefits according to all metrics, it may still make sense to adopt a strategy using whichever metric is most aligned with an organization's objectives and time horizons for an action.

A related concept to consider is that of "discounting", whereby future benefits or impacts are valued at a declining rate compared to the present (see Section 9.1.6). The time horizon can also be chosen in line with the 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 place less value on impacts further into the future, emphasizing instead the impact of shorter-lived pollutants.

Economic considerations can provide further insight into the choice of metric. As indicated in Chapter 2 and Annex II of Working Group III of the IPCC's Sixth Assessment Report (IPCC, 2022b; Dhakal, Minx and Toth, 2022), there is increasing evidence supporting the use of GWP_{100} under pathways toward the Paris Agreement goals as an approximation of economically optimal metrics at least until the mid-century (Tanaka *et al.*, 2021). Metrics for CH_4 derived from cost-benefit and cost-effectiveness frameworks have values that lie roughly between 20 and 40, which is more consistent with GWP_{100} than with GTP_{100} or GWP_{20}. While this supports the adoption of GWP_{100} in the Paris Rulebook and the use of GWP_{100} in this context, it should be noted that this was an inadvertent outcome because GWP_{100} 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 demonstrate 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 predefined time, but spanning any number of years of interest. This is similar to providing multiple metrics and/or alternative time horizons to give 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. In other words, using the terminology introduced in Section 9.1.4, these metrics would yield information about the additional impact of ongoing emissions relative to the baseline year, but not as to the marginal impact of ongoing emissions.

When appropriate and practical, the use of climate models to estimate the climate impact is a suitable alternative to metrics (Farquharson *et al.*, 2017), as Example 1 illustrates. This is because it offers a more comprehensive and transparent way of describing complex climate impact than a simple metric. It could be used either on its own or as a justification for selecting the assessment using a single metric that is most consistent with these more detailed analyses.

See Section 9.1.5 and Section 9.1.6, and Section 9.2.1. to Section 9.2.3.

Example 1: The farmer primarily wants to know the climate benefits (i.e. lower temperatures) of a given intervention in the span of a decade. They would also want to know what implications it might have (i.e. higher temperature) at any point before or after.

9.4.1.4 Context and counterfactual baseline

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 emission scenario you are considering, potentially combining the impacts of past emissions with those of current emissions, and in how these combined impacts might relate to an overall climate objective? Or do you only wish to assess the potentially avoidable future impacts that will occur due to present-day and immediate-future 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 a given emission contributes to global climate change (how much warmer the climate is because of this source), and conversely, how their specified climate impacts could be avoided if we didn't release any given emission (marginal warming, see Section 9.1.4). They can also be used to compare the climate impacts of alternative mitigation strategies by evaluating the CO_2-equivalent emissions based on two scenarios (e.g. with and without a particular mitigation strategy being implemented) and determining the difference between those emission scenarios. This is demonstrated through Example 1 (Section 9.4.2.1).

Step-pulse metrics like the GWP* capture the temperature impact of an emission relative to the temperature impact at a baseline year (additional warming; see Section 9.1.4). However, if you are calculating equivalent emissions from a baseline year, you should also give proper consideration to what that baseline is (see Section 9.4.1.5). As illustrated in Section 9.1.4, this consideration results in quite different perspectives for short- and long-lived GHGs, but also answers a fundamentally different question since 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; the only way to notably reduce temperatures below the baseline would be through 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; 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. CH_4) 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 the CO_2-equivalent emissions of a given mitigation scenario, 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 case for making that decision is more compelling. Should the use of different metrics lead to a different outcome, it would be worth considering again the context, the counterfactual scenario and the criteria used to make the decision.

See Section 9.1.4.

Example 1: We wish to compare a “business as usual” scenario with a “feed additive” scenario. We would also like to know the climate impact of these two scenarios, relative to a “no farm” scenario.

9.4.1.5 Comparability and transparency

The comparability of metrics and whether the assessment boundaries affecting the metrics are transparent is important for the selection process.

Since GWP_{100} is the most commonly used metric, including for reporting under the Paris Agreement, doing the impact assessment report with GWP_{100} is often perceived as a means of ensuring that it can easily be compared with various other assessments (Levasseur *et al.*, 2016). Should other metrics be selected, then also using GWP_{100} for the assessment can improve its comparability and transparency. If the assessment is greatly affected by the choice of metric, then explaining why the assessment results are different to those using GWP_{100} can improve users' understanding.

The boundaries or counterfactuals 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 by Forster *et al.* (2021) depends on the emissions in the present and 20 years ago. This means that it is dependent on the emission history of an individual emitter (see 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 present. In other words, using GWP* to calculate CO_2-warming equivalent emissions will indicate whether present-day CH_4 emissions are causing the temperature to rise or fall, but it will not tell you what the absolute level of warming caused by the CH_4 emissions is. It may therefore be incomplete or misleading to note only the direction of travel, and not the absolute level of warming.

For example, if CH_4 emissions declined 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 CH_4 emissions (the 10 or 1 million tonnes) determines how much a given source contributes to global warming, and is relevant for assessing whether that level of emission might be deemed acceptable. One straightforward option would be to use GWP* in conjunction with an absolute annual CH_4 emission expressed in terms of a metric reflecting marginal impacts, such as GWP_{100}.

Step-pulse metrics like GWP* depend not only on changes in emissions today, but also on 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 may be considerable year-to-year variability in CH_4 emissions, which can cause annual CO_2-warming equivalent emissions using GWP* to be more variable than those calculated with a pulse metric (Meinshausen and Nicholls, 2022). The cumulative impact calculated 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 a policy based on such emissions, which may need to be considered.

9.4.1.6 Other considerations

There may be other relevant considerations 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 effects on human health and food production (UNEP and CCAC, 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 whether there is a wider scope covering more than just the climate impacts of emissions.

See Section 9.1.7.

9.4.2 Examples

This section contains two examples designed to illustrate some of the concepts around metrics discussed above. Example 1 and Example 2 are explored quantitatively to give the reader insights into the implications of using the different metrics for analysing these case studies. Example 1 shows within how long a timescale 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 in the first example), which are different from pulse emissions used to define metrics such as GWP_{100}. Example 2 illustrates the importance of selecting an appropriate baseline, especially when step-pulse metrics such as GWP* are used.

9.4.2.1 Example 1: Evaluation of emission metrics in representing the benefits of using a feed additive

A dairy farmer wishes to use emission metrics to quantify the climate benefits that will result from using a certain feed additive on their herd. Their aim is to improve their environmental footprint over the next decade. This indicates that they want to compare the emissions when using the feed additive relative to what emissions would be without the feed additive (see Table 9). The farmer already uses GWP_{100} in a GHG calculator, so there is a precedent there.

Table 9 shows the emissions associated with the farm today ("control farm") and when the feed additive has been used. Methane emissions decrease with the introduction of the feed additive, but the CO_2 emissions increase (due to the production/distribution of the feed additive, based on current fossil fuel use in the energy supply). What is the climate impact of switching from the control farm to using the feed additive? Does the effect of the increase on CO_2 emissions outweigh the effect of the reduced CH_4 emissions? We will explore these questions next.

The farm's aggregated annual GHG emissions using GWP_{100} 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 CH_4 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. litres 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 the 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 a scenario in which those emissions were not made (the "marginal" impact of these emissions, as discussed in Section 9.1.4). Implementing the feed additive therefore reduces the marginal climate impacts of the farm, as assessed using the GWP_{100} (i.e. specifically, the total radiative forcing for one hundred years

Table 9. Annual emissions associated with the farm in Example 1

	CH_4 t	N_2O t	CO_2 t
Control farm's annual emissions	60	1.68	100
Feed additive farm's annual emissions	40	1.68	105

Source: Authors' own elaboration.

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
GWP_{100} CO_2eq tonnes saved per year	-540	0	5	-535
GWP_{20} CO_2eq tonnes saved per year	-1 594	0	5	-1 589
GTP_{100} CO_2eq tonnes saved per year	-94	0	5	-89
GTP_{20} CO_2eq tonnes saved per year	-1 040	0	5	-1 035
GWP* CO_2eq tonnes saved per year (for the first 20 years; 2020-2039)	-2 537	0	5	-2 532
GWP* CO_2eq tonnes saved per year (after 20 years of stabilized new emissions; 2040 onwards)	-157	0	5	-152

Source: Authors' own elaboration.

following each year's emissions is reduced by using the feed additive). Below, we consider why and how different metric approaches may provide different quantifications of these benefits.

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 13), 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 the IPCC's AR6 values for GWP and GTP (for example, 27 for GWP_{100} CH_4) – with the exception of CO_2eq emissions, based on GWP* – which use the IPCC's AR5 value of GWP_{100} (that is, 28 for GWP_{100} CH_4) as this is consistent with the GWP* formula used in AR6 (Smith, Cain and Allen, 2021; footnote in Section 7.6.1.4 of IPCC [2021]).

For GWP*, there is a greater value placed on the difference in the first 20 years after the feed additive is introduced (greater than the value placed by GWP_{20}),[4] and then a smaller value placed on the difference beyond that time (more similar to the value placed by GTP_{100}).[5] How can the same difference of 20 tonnes of CH_4 per year vary over time? If we were to tag each molecule of CH_4 from this farm in the atmosphere, when we reduce the CH_4 emission by 20 tonnes, the amount of tagged CH_4 left in the atmosphere would decline over a period of approximately 20 to 40 years. It would then stabilize at a new equilibrium. In other words, the impact of reducing CH_4 emissions on the atmosphere occurs in the few decades that immediately follow the change in emissions. Later than that, the annual changes in atmospheric methane levels 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. GWP_{100}), or only assess them at a particular time period (e.g. GTP_{100}).

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[4] The CO_2-equivalent emissions avoided annually in the first 20 years after the switch are calculated as follows, using GWP* based on the equation in Section 9.1.3: $28 \times (4.53 \times 40 - 4.25 \times 60) - 28 \times (4.53 \times 60 - 4.25 \times 60) = -2537$ tCO_2eq/year.

[5] The CO_2-equivalent emissions avoided annually more than 20 years after the switch are calculated using GWP* based on the same equation, but with 40 tonnes of methane every year: $28 \times (4.53 \times 40 - 4.25 \times 40) - 28 \times (4.53 \times 60 - 4.25 \times 60) = -157$ tCO_2eq/year.

To explore what that means in practice and to demonstrate how these metrics represent the different emissions, we have used a simple climate model, the aggregated carbon cycle, atmospheric chemistry and climate (ACC2) model (Tanaka and O'Neill, 2018 and Tanaka *et al.*, 2021; see the Appendix for details).

First, we have modelled a scenario where the control farm emissions occur between 2000 and 2100, and shown the impact of those emissions on global mean surface temperature (black line in Figure 13 I, b). We have then modelled the emissions for the scenario in which the farm starts using the feed additive in 2020 (black lines in Figure 13 II, d). In both cases, there are no emissions before 2000. The difference in temperature between these two scenarios (control emissions and feed additive emissions) is also shown by the black line in Figure 13 III, f. The feed additive lowers by about a quarter the level of global warming that this farm would cause by 2100. This clearly demonstrates that the level of global warming due to the increased CO_2 emissions (from producing the feed additive) is smaller than the amount by which the CH_4 reductions lower the temperature. With modelled temperature as the metric, there are clear benefits to using the feed additive compared to not using it.

The black lines show the temperature change from modelling these emission scenarios relative to the temperature in 2000, which will allow us to illustrate how different emission metrics represent the temperature change, using the black lines as a benchmark. The differences 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 panels b, d and f of Figure 13. 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 variations between the 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.

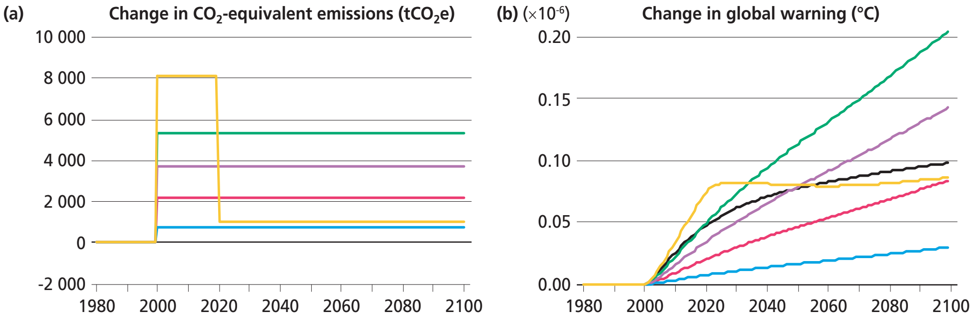
Panels b and d demonstrate the logic behind why GWP* places a large CO_2eq value on changes to CH_4 emissions for 20 years, and a small value thereafter. When you model the GWP* emissions (yellow), the temperature curve approximates the modelled temperature from the original emissions (black). Compared with the other emission metrics, GWP* is closest to a short-term metric like GWP_{20} for the first 20 years and GTP_{100} beyond that (Figure 13, panels a and c). It can be interpreted as GWP* approximating the modelled temperature outcomes by using these two values in one metric.

Just as CO_2 emissions act cumulatively, GWP* attempts to report CH_4 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 A2 c in the Appendix) could also be multiplied by a quantity known as the transient climate response to cumulative emissions (TCRE) – 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 using the pulse-emission metrics such as GWP_{100}.

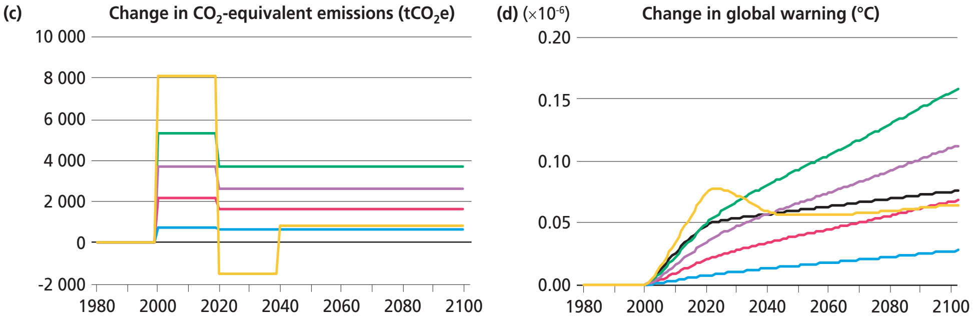
Figure 13

Scenarios computed using the aggregated carbon cycle, atmospheric chemistry and climate (ACC2) model.

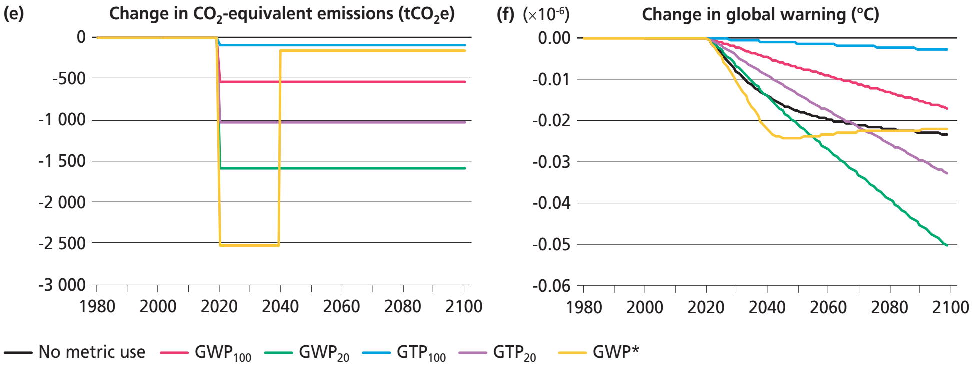
I. Control farm scenario compared to no farm scenario



II. Feed additive farm scenario compared to no farm scenario



III. Feed additive farm scenario compared to control farm scenario



Example 1: 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, CH_4 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. Nota bene, emissions associated with the land in the case of having no farm are not considered here (Manzano, P. & White, S. 2019. Intensifying pastoralism may not reduce greenhouse gas emissions: Wildlife-dominated landscape scenarios as a baseline in life-cycle analysis. Climate Research, *77: 91–97. https://doi.org/10.3354/cr01555 and Fløjgaard, C., Pedersen, P.B.M., Sandom, C.J., Svenning, J. & Ejrnæs, R. 2022. Exploring a natural baseline for large-herbivore biomass in ecological restoration.* Journal of Applied Ecology, 59*(1): 18–24. https://doi.org/10.1111/1365- 2664.14047).*

Source: Adapted from Tanaka, K. & O'Neill, B.C. 2018. The Paris Agreement zero-emissions goal is not always consistent with the 1.5 °C and 2 °C temperature targets. *Nature Climate Change*, 8(4): 319–324. https://doi.org/10.1038/s41558-018-0097-x and Tanaka, K., Boucher, O., Ciais, P., Johansson, D.J.A. & Morfeldt, J. 2021. Cost-effective implementation of the Paris Agreement using flexible greenhouse gas metrics. *Science Advances, 7*(22): eabf9020. https://doi.org/10.1126/sciadv. abf9020

When using GWP, the net avoided emission each year traps the same amount of additional energy in the climate system as would the equivalent amount of CO_2, integrated out to 100 or 20 years, for GWP_{100} or GWP_{20}, respectively. The difference in magnitude between GWP_{100} and GWP_{20} values has to do with the fact that they are averaged over the 100- or 20-year periods, and CH_4 has more radiative forcing impact in the first 20 years following the emission. Importantly, GWP_{100} and GWP_{20} do not reflect the varying warming effects within these time periods. Moreover, 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 timescale of the emissions in question. Panels b and d show that GWP_{20} (green), which is designed to focus on the warming potential over a 20-year period, approximates the additional warming over the first 20 years well, but after that overestimates the additional warming as well as the temperature change due to the feed additive (panel f). GWP_{100} (red), which is applied over a 100-year time horizon, underestimates the temperature reduction for the first hundred years. For GWP_{100}, the cumulative relative warming between the two scenarios (Figure A2 in the Appendix) is somewhat underestimated (red compared to black).

If using GTP_{100} or GTP_{20}, the net avoided emissions *each year* would yield the same change in temperature as from the equivalent amount of CO_2, at a time point 100 or 20 years following 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, panels 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 CH_4 emissions (see Section 9.1.2.2). Values of GTP_s are similar to those of GWP (Shine *et al.*, 2005). In this example, this is borne out by the GWP_{100} (red) temperature intersecting with the actual temperature (black) around 100 years after the emission change occurs. Similarly to GWP, there is a large variation between GTP values across different time horizons, because the climate impacts from CH_4 emissions decline rapidly after 20 years following the emission. The cumulative relative warming (Figure A2, panels c and f in the Appendix) from GTP_{20} shows good agreement (purple compared to black) for the first century approximately.

In Example 1, 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 when using the feed additive or not (Figure 13 f), GTP_{100} indicates a very much underestimated benefit, whereas GWP_{20} indicates a very much overestimated benefit beyond about 40 years. Meanwhile, GWP* overestimates the temperature benefit for the first 50 years, but shows an accurate agreement thereafter. The GWP* represents complex non-linear climate responses with only two timescales (Allen *et al.*, 2021), with coefficients based on the AR5 impulse-response model. The results in Figure 13 suggest the ACC2 model responds slightly differently to the model that GWP* approximates, which would explain some of the differences between the yellow and black lines in Figure 13, panels b, d and f. However, all metrics show that in this case there is a clear benefit to using the feed additive, which is borne out through modelling the actual changes in emissions (black). This conclusion will not necessarily hold for every scenario, for example in case the feed additive were 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 may be chosen. In the absence of a specific time horizon of interest, multiple time horizons could be considered using GWP_{20} (to approximate temperature impacts over the first 30 years) and GWP_{100} (to approximate temperature impacts over a 100-year horizon), as well as GTP_{100} (to approximate temperature impacts after one hundred years, for which the GTP_s version of the metric would be most suitable as it is a sustained change to emission rates) or even GWP* over the whole time series. Step-pulse metrics capture the time variations of impacts on temperature, but pulse metrics may yield an acceptable approximation of benefits (e.g. cumulative relative warming) over specific timescales.

Further details for this example may be found in the Appendix.

9.4.2.2 Example 2: Illustrating the path dependency of step-pulse metrics

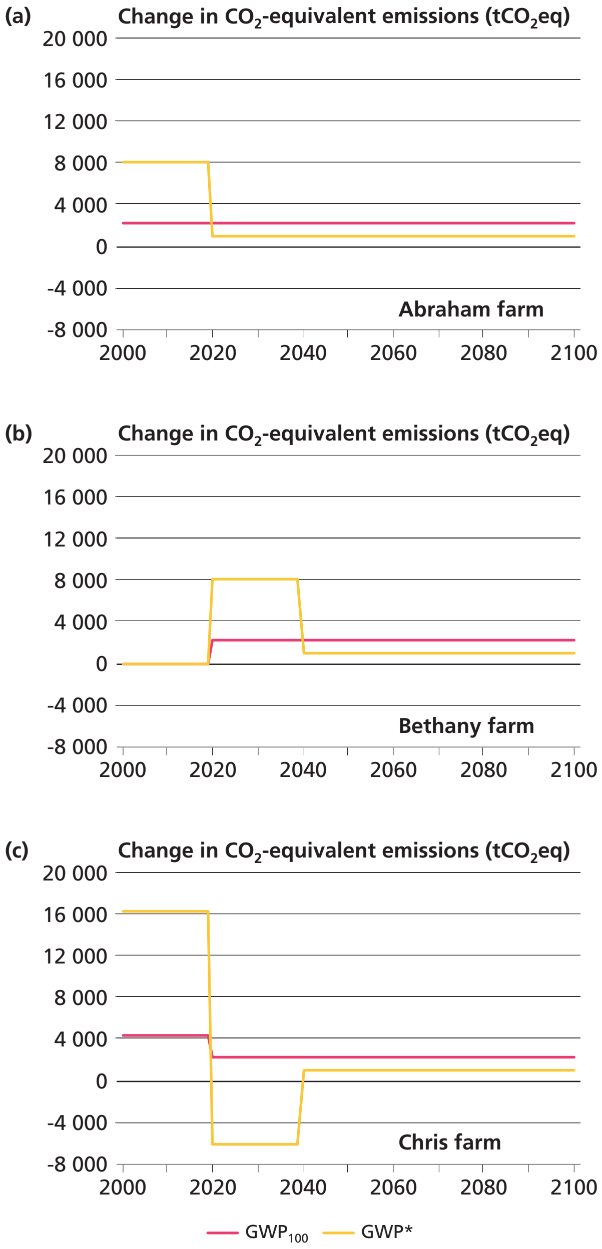
Consider that we have three farms which, at present, have the same number of cattle and emissions as the control farm in Example 1 (60 t CH_4, 1.68 t N_2O, 100 t CO_2). Despite having the same emissions in 2020, all farms have a different emissions history. Farm A (Abraham) had stable emissions for the whole time since being created in 2000. Farm B (Bethany) was only established in 2020, and thus had zero emissions before that year. Farm C (Chris) had twice the cattle/emissions when it started in 2000, but in 2020 it abruptly cut the herd/emissions in half.

In a pulse metric like GWP_{100}, 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 14. The yellow lines in Figure 14 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 depends on the emissions in 2000, which were then different for each farm. This reflects the “additional” nature of applying GWP* to a scenario, in 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 was 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 t CO_2-we), which is then reduced to around 1000 tonnes CO_2-we in the following years. As a result of the farms’ emissions being dominated by CH_4 the abrupt reduction in CH_4 emissions in CH_4 for farm C leads to highly negative CO_2-we values for the 20 years after the emissions are halved. This does not mean that the remaining emissions are no longer GHGs (those emissions do cause marginal warming), but rather that the temperature increase caused by emissions up to the year 2020 is partly reversed.

Without considering prior warming, applying GWP* could produce seemingly contradictory results for farms with the same emissions at present but with different histories. This is because GWP*, in this example, is showing the “additional” effects of the farms and not the “marginal” effects (see Section 9.1.4 where this terminology is explained). Example 2 goes to show that accounting for the full temperature increase caused by emissions should be acknowledged in order to avoid potentially misleading or inequitable outcomes. This could be done by applying GWP* to the full historical time series of emissions, or by applying it relative to the case where there was previously no farm (i.e. farm B) to show the maximum impact of the farm.

Figure 14
CO$_2$eq and CO$_2$-we emissions from the three farms calculated using GWP$_{100}$ and GWP*



The full results based on all the metrics considered here, including temperature calculations, can be found in Figure A3 of the Appendix.

Source: Authors' own elaboration.

Using a longer time series for step-pulse metrics, instead of a single year value taken in isolation, will result in a more complete assessment that does not lose sight of the wider context. Alternatively, CO_2-we emissions could be evaluated alongside absolute annual CH_4 emissions to indicate both the additional effects of these emissions and the level from which they are increasing or decreasing.

9.4.3 Summary of key features and limitations of GWP, GWP* and GTP

One aim of this section is to summarize some of the key features and limitations of GWP_{100}, GWP* and GTP. While by no means comprehensive, it provides an outline and a reminder for the agricultural practitioner of the issues explored in more detail throughout this report.

9.4.3.1 GWP_{100}

GWP_{100} is commonly used, including as the specified metric for reporting emissions to the United Nations. It provides an estimate of how much energy is accumulated over the 100-year time period relative to an absence of emissions. This type of metric is therefore useful if the impacts to the climate system as a whole over the coming century are compared to not emitting those GHGs (the marginal impacts in the above discussion). The same could be said of GWP_{20}, only for the 20 years following the emission. GWP_{100} and GWP_{20} are derived from pulse emissions, and when they are applied to sustained changes in the level of emissions over multiple years, the metric time horizons (100 and 20 years) are not indicative of the impact (e.g. Example 1 in Section 9.4.2.1). The disadvantage of GWP_{100} is that it does not relate directly to how much CH_4 emissions change the surface temperature over time (the additional impacts in the above discussion). In particular, it does not reflect the fact that reducing CH_4 emissions does not result in additional warming. It also under-represents the strong warming effect caused by introducing new CH_4 emissions. Consequently, the temperature outcomes over time from any trade-offs being considered would not be clear, because 1 tCO_2eq of CH_4 does not cause the same amount of warming at all times as 1 tCO_2eq of N_2O or CO_2.

See also Section 9.1.2.1.

9.4.3.2 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 CH_4, relative to the warming at the starting point of that time series (termed additional warming). This is a robust concept at the global level. However, any evaluation at the national or corporate level should also pay heed to economic, social, equity and political considerations (Allen *et al.*, 2022a).

The minimum time series required in the case of GWP* is two data points separated by 20 years, where GWP* can be used to evaluate the effect of those CH_4 emissions relative to the emissions 20 years prior. This may be a disadvantage for some applications when the data regarding emissions from 20 years prior is not available, or when assumptions as to what they might have been cannot be made; it also means that the valuation of present-day emissions depends strongly on the level of emissions from the same source produced 20 years earlier. This condition forces the user to specify the question being asked (i.e. do we care about the marginal contribution of an activity to global warming, or only about the additional contribution relative to a given reference date and, if so, why?), in order to ensure that it is correctly reflected in the assumptions made about past emissions, which relate to questions of responsibility, equity and fairness. If the present day is used to provide a baseline, then pathways are assessed relative to the present-day level of warming from CH_4. We would suggest that this level of warming be explicitly evaluated, as the omission of this information could lead to wrongly assuming that CH_4 emissions could cause cooling compared to no CH_4 emissions occurring. Instead, the baseline level of warming, which is reflected by CO_2-warming equivalent emissions calculated using GWP*, can be reversed through CH_4 emission reductions. Methane emissions reducing year on year will cause temperature to decrease 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 organization or farm was established), or from a future point in time that may be relevant for climate policy (e.g. since 1990 or at any future date when a proposed climate policy and associated accountability for emissions would come into effect). GWP* is a useful metric if a time series of emissions is being evaluated, or compared to another emission scenario, based on their respective impact on temperature (e.g. comparing the 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*.

See also Section 9.1.3.

9.4.3.3 GTP

GTP can be used to estimate the amount of warming that would arise from an emission at a specific time horizon, compared to an absence of that emission. It is therefore useful to use GTP if you want to compare the temperature change at a specified time with and without emitting the GHGs. The disadvantage of GTP 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 were related to the same end-point year that was of interest. For example, if offsetting a one-off 1 tCO_2eq emission of CH_4 with a CO_2 removal calculated using GTP, the temperature impact would be equivalent at that specific time horizon only.

When GTP_{100} is applied to emissions occurring over multiple years or decades, it does not represent the temperature impact 100 years from the start of the example. For this, the "sustained GTP" metric would give a better indication of temperature outcomes.

See also Section 9.1.2.2.

Conclusion

The report covers four main topics: the sources and sinks of methane emissions from food and agriculture, the quantification of methane emissions, the mitigation of methane emissions, and the metrics for quantifying the impact of methane emissions. The report highlights that:

- Microbial-mediated enteric fermentative processes in ruminant livestock contribute to around 30 percent of the total anthropogenic methane emissions, while the anaerobic digestion of animal manure and other organic wastes and rice paddies contribute to around 4.5 percent and 8 percent, respectively.
- The atmospheric sink through the chemical degradation of methane by hydroxyl and chlorine radicals in the troposphere and stratosphere is responsible for 90 to 96 percent of the global methane sink, the soil accounts for about 4 to 10 percent of the methane degraded, and the ocean acts as a small methane sink.
- Methane has a shorter lifetime than carbon dioxide, which affects the quantification of greenhouse gas emissions, particularly for methane.
- Various methods and methodologies are used to measure and estimate methane emissions from ruminant animals and the manure produced, including gas exchange techniques, head-stalls, the tracer gas technique, micrometeorological techniques, aircrafts, drones and satellites. In each of these methods, there is a trade-off between ease of use, repeatability and appropriateness for housed and grazing animals.
- The suitability of a given technique for determining methane emissions from rice paddies also depends on multiple factors.
- Several management practices that induce the increased redox potential of soil suppress methane production and hence the emissions from rice fields. The choice of options depends on the feasibility of the management and possible trade-offs.
- Decreasing enteric methane emissions from ruminant and rice paddies production is crucial to limiting the global temperature increase to 1.5 °C by 2050, and various strategies for enteric methane abatement are being investigated to that end.
- The report considers and analyses the state of play of the enteric methane mitigation strategies available at present (about 30 in total), namely their effectiveness, safety issues, the impact on other greenhouse gas emissions as well as economic, regulatory and societal aspects.
- Most research has been conducted on confined animals, and more research is needed to develop and evaluate anti-methanogenic strategies for grazing systems.
- Continuous research and development are needed to devise enteric methane mitigation strategies that are locally applicable, while more information is required to calculate the carbon footprint of interventions on a regional basis and to evaluate their impact on net greenhouse gas emissions.

- Greenhouse gas emission metrics are used to quantify the impact of emissions (and the mitigation thereof) on the climate system. Each GHG emission metric captures a specific climate impact over a specific time; the equivalence based on one metric does not imply equivalence based on other metrics.
- To provide the relevant information, the metric choice – including the time horizon – should take into account the specific matter being investigated and the relevant policy objectives.
- Pulse-emission metrics (e.g. GWP_{100}, GWP_{20}, GTP_{100}, GTP_{20}) provide information about future climate impacts of emission units, as opposed to the absence of those emissions, which are called the “marginal” impacts.
- Step-pulse metrics (e.g. GWP*, CGTP) provide information about “additional” impacts relative to a specified date.
- Depending on the question posed, either pulse-emission metrics or step-pulse metrics may be suitable. A range of metrics can be used to test whether results are consistent across different timescales or with respect to different impacts.
- Since metrics are used as tools by policy makers, it is important to consider them within the wider context of the Paris Agreement, definitions of climate neutrality, sustainable agriculture and equity considerations.

References

Aaheim, A., Fuglestvedt, J.S. & Godal, O. 2006. Costs savings of a flexible multi-gas climate policy. *The Energy Journal*, 27: 485–501. www.jstor.org/stable/23297097

Abbott, D.W., Aasen, I.M., Beauchemin, K.A., Grondahl, F., Gruninger, R., Hayes, M., Huws, S., Kenny, D.A., Krizsan, S.J., Kirwan, S., Lind, V., Meyer, U., Ramin, M., Theodoridou, K., von Soosten, D., Walsh, P., Waters, S. & Xing, X. 2020. Seaweed and seaweed bioactives for mitigation of enteric methane: Challenges and opportunities. *Animals*, 10(12): 2432. https://doi.org/10.3390/ani10122432

Abdalla, A.L., Louvandini, H., Sallam, S.M.A.H., Bueno, I.C.D.S., Tsai, S.M. & Figueira, A.V.D.O. 2012. In vitro evaluation, in vivo quantification, and microbial diversity studies of nutritional strategies for reducing enteric methane production. *Tropical Animal Health and Production*, 44(5): 953–964. https://doi.org/10.1007/s11250-011-9992-0

Abecia, L., Martínez-Fernandez, G., Waddams, K., Martín-García, A.I., Pinloche, E., Creevey, C.J., Denman, S.E., Newbold, C.J. & Yáñez-Ruiz, D.R. 2018. Analysis of the rumen microbiome and metabolome to study the effect of an antimethanogenic treatment applied in early life of kid goats. *Frontiers in Microbiology*, 9: 2227. https://doi.org/10.3389/fmicb.2018.02227

Abecia, L., Martín-García, A.I., Martínez, G., Newbold, C.J. & Yáñez-Ruiz, D.R. 2013. Nutritional intervention in early life to manipulate rumen microbial colonization and methane output by kid goats postweaning. *Journal of Animal Science*, 91(10): 4832–4840. https://doi.org/10.2527/jas.2012-6142

Abecia, L., Waddams, K.E., Martínez-Fernandez, G., Martín-García, A.I., Ramos-Morales, E., Newbold, C.J. & Yáñez-Ruiz, D.R. 2014. An antimethanogenic nutritional intervention in early life of ruminants modifies ruminal colonization by Archaea. *Archaea*, 2014: 1–12. https://doi.org/10.1155/2014/841463

Abernethy, S. & Jackson, R.B. 2022. Global temperature goals should determine the time horizons for greenhouse gas emission metrics. *Environmental Research Letters*, 17(2): 024019. https://doi.org/10.1088/1748-9326/ac4940

Aboagye, I.A. & Beauchemin, K.A. 2019. Potential of molecular weight and structure of tannins to reduce methane emissions from ruminants: A review. *Animals*, 9(11): 856. https://doi.org/10.3390/ani9110856

Aboagye, I.A., Oba, M., Castillo, A.R., Koenig, K.M., Iwaasa, A.D. & Beauchemin, K.A. 2018. Effects of hydrolyzable tannin with or without condensed tannin on methane emissions, nitrogen use, and performance of beef cattle fed a high-forage diet. *Journal of Animal Science*, 96(12): 5276–5286. https://doi.org/10.1093/jas/sky352

Achtnich, C., Bak, F. & Conrad, R. 1995. Competition for electron donors among nitrate reducers, ferric iron reducers, sulfate reducers, and methanogens in anoxic paddy soil. *Biology and Fertility of Soils*, 19(1): 65–72. https://doi.org/10.1007/BF00336349

Adegbeye, M.J., Elghandour, M.M., Monroy, J.C., Abegunde, T.O., Salem, A.Z., Barbabosa-Pliego, A. & Faniyi, T.O. 2019. Potential influence of Yucca extract as feed additive on greenhouse gases emission for a cleaner livestock and aquaculture farming – A review. *Journal of Cleaner Production*, 239: 118074. https://doi.org/10.1016/j.jclepro.2019.118074

Adejoro, F.A., Hassen, A., Akanmu, A.M. & Morgavi, D.P. 2020. Replacing urea with nitrate as a non-protein nitrogen source increases lambs' growth and reduces methane production, whereas acacia tannin has no effect. *Animal Feed Science and Technology*, 259: 114360. https://doi.org/10.1016/j.anifeedsci.2019.114360

Aguirre-Villegas, H.A., Larson, R.A. & Sharara, M.A. 2019. Anaerobic digestion, solid-liquid separation, and drying of dairy manure: Measuring constituents and modeling emission. *Science of the Total Environment*, 696(15): 134059. https://doi.org/10.1016/j.scitotenv.2019.134059

Alazzeh, A.Y., Sultana, H., Beauchemin, K.A., Wang, Y., Holo, H., Harstad, O.M. & McAllister, T.A. 2012. Using strains of Propionibacteria to mitigate methane emissions *in vitro*. *Acta Agriculturae Scandinavica, Section A – Animal Science*, 62(4): 263–272. https://doi.org/10.1080/09064702.2013.773056

Alberto, M.C.R., Hirano, T., Miyata, A., Wassmann, R., Kumar, A., Padre, A. & Amante, M. 2012. Influence of climate variability on seasonal and interannual variations of ecosystem CO_2 exchange in flooded and non-flooded rice fields in the Philippines. *Field Crops Research*, 134: 80–94. https://doi.org/10.1016/j.fcr.2012.05.002

Alberto, M.C.R., Wassmann, R., Buresh, R.J., Quilty, J.R., Correa, T.Q., Sandro, J.M. & Centeno, C.A.R. 2014. Measuring methane flux from irrigated rice fields by eddy covariance method using open-path gas analyzer. *Field Crops Research*, 160: 12–21. https://doi.org/10.1016/j.fcr.2014.02.008

Alberto, M.C.R., Wassmann, R., Hirano, T., Miyata, A., Kumar, A., Padre, A. & Amante, M. 2009. CO_2/heat fluxes in rice fields: Comparative assessment of flooded and non-flooded fields in the Philippines. *Agricultural and Forest Meteorology*, 149(10): 1737–1750. https://doi.org/10.1016/j.agrformet.2009.06.003

Alemu, A.W., Dijkstra, J., Bannink, A., France, J. & Kebreab, E. 2011. Rumen stoichiometric models and their contribution and challenges in predicting enteric methane production. *Animal Feed Science and Technology*, 166–167: 761–778. https://doi.org/10.1016/j.anifeedsci.2011.04.054

Alemu, A.W., Janzen, H., Little, S., Hao, X., Thompson, D.J., Baron, V., Iwaasa, A., Beauchemin, K.A. & Kröbel, R. 2017. Assessment of grazing management on farm greenhouse gas intensity of beef production systems in the Canadian Prairies using life cycle assessment. *Agricultural Systems*, 158: 1–13. https://doi.org/10.1016/j.agsy.2017.08.003

Alemu, A.W., Ominski, K.H. & Kebreab, E. 2011. Estimation of enteric methane emissions trends (1990–2008) from Manitoba beef cattle using empirical and mechanistic models. *Canadian Journal of Animal Science*, 91(2): 305–321. https://doi.org/10.4141/cjas2010-009

Alemu, A.W., Pekrul, L.K.D., Shreck, A.L., Booker, C.W., McGinn, S.M., Kindermann, M. & Beauchemin, K.A. 2021. 3-nitrooxypropanol decreased enteric methane production from growing beef cattle in a commercial feedlot: Implications for sustainable beef cattle production. *Frontiers in Animal Science*, 2: 641590. https://doi.org/10.3389/fanim.2021.641590

Alemu, A.W., Shreck, A.L., Booker, C.W., McGinn, S.M., Pekrul, L.K.D., Kindermann, M. & Beauchemin, K.A. 2020. Use of 3-nitrooxypropanol in a commercial feedlot to decrease enteric methane emissions from cattle fed a corn-based finishing diet. *Journal of Animal Science*, 99(1): skaa394. https://doi.org/10.1093/jas/skaa394

Allen, M.R., Friedlingstein, P., Girardin, C.A.J., Jenkins, S., Malhi, Y., Mitchell-Larson, E., Peters, G.P. & Rajamani, L. 2022c. Net zero: Science, origins, and implications. *Annual Review of Environment and Resources*, 47(1): 849–887. https://doi.org/10.1146/annurev-environ-112320-105050

Allen, M.R., Fuglestvedt, J.S., Shine, K.P., Reisinger, A., Pierrehumbert, R.T. & Forster, P.M. 2016. New use of global warming potentials to compare cumulative and short-lived climate pollutants. *Nature Climate Change*, 6(8): 773–776. https://doi.org/10.1038/nclimate2998

Allen, M.R., Peters, G.P., Shine, K.P., Azar, C., Balcombe, P., Boucher, O., Cain, M., Ciais, P., Collins, W., Frame, D.J., Friedlingstein, P., Forster, P.M., Fyson, C., Gasser, T., Hare, B., Jenkins, S., Hamburg, S., Johansson, D.J.A., Lynch, J., Macey, A., Morfeldt, J., Nauels, A., Ocko, I., Oppenheimer, M., Pacala, S.W., Pierrehumbert, R., Rogelj, J., Schaeffer, M., Schleussner, C.F., Shindell, D., Skeie, R.B., Smith, S.M. & Tanaka, K. 2022a. Indicate separate contributions of long-lived and short-lived greenhouse gases in emission targets. *npj Climate and Atmospheric Science*, 5(1): 5. https://doi.org/10.1038/s41612-021-00226-2

Allen, M.R., Shine, K.P., Fuglestvedt, J.S., Millar, R.J., Cain, M., Frame, D.J. & Macey, A.H. 2018. A solution to the misrepresentations of CO_2-equivalent emissions of short-lived climate pollutants under ambitious mitigation. *npj Climate and Atmospheric Science*, 1(1): 16. https://doi.org/10.1038/s41612-018-0026-8

Allen, M.R, Tanaka, K., Macey, A., Cain, M., Jenkins, S., Lynch, J. & Smith, M. 2021. Ensuring that offsets and other internationally transferred mitigation outcomes contribute effectively to limiting global warming. *Environmental Research Letters*, 16(7): 074009. https://doi.org/10.1088/1748-9326/abfcf9

Altermann, E., Schofield, L.R., Ronimus, R.S., Beattie, A.K. & Reilly, K. 2018. Inhibition of rumen methanogens by a novel archaeal lytic enzyme displayed on tailored bionanoparticles. *Frontiers in Microbiology*, 9: 2378. https://doi.org/10.3389/fmicb.2018.02378

Alvarado, A., Montañez-Hernández, L.E., Palacio-Molina, S.L., Oropeza-Navarro, R., Luévanos-Escareño, M.P. & Balagurusamy, N. 2014. Microbial trophic interactions and *mcr*A gene expression in monitoring of anaerobic digesters. *Frontiers in Microbiology*, 5: 1–14. https://doi.org/10.3389/fmicb.2014.00597

Alvarez-Hess, P.S., Little, S.M., Moate, P.J., Jacobs, J.L., Beauchemin, K.A. & Eckard, R.J. 2019a. A partial life cycle assessment of the greenhouse gas mitigation potential of feeding 3-nitrooxypropanol and nitrate to cattle. *Agricultural Systems*, 169: 14–23. https://doi.org/10.1016/j.agsy.2018.11.008

Alvarez-Hess, P.S., Moate, P.J., Williams, S.R.O., Jacobs, J.L., Beauchemin, K.A., Hannah, M.C., Durmic, Z. & Eckard, R.J. 2019b. Effect of combining wheat grain with nitrate, fat or 3-nitrooxypropanol on *in vitro* methane production. *Animal Feed Science and Technology*, 256: 114237. https://doi.org/10.1016/j.anifeedsci.2019.114237

Alves, T.P., Dall-Orsoletta, A.C. & Ribeiro-Filho, H.M.N. 2017. The effects of supplementing *Acacia mearnsii* tannin extract on dairy cow dry matter intake, milk production, and methane emission in a tropical pasture. *Tropical Animal Health and Production*, 49(8): 1663–1668. https://doi.org/10.1007/s11250-017-1374-9

Amadi, C.C., Van Rees, K.C.J. & Farrell, R.E. 2016. Soil–atmosphere exchange of carbon dioxide, methane and nitrous oxide in shelterbelts compared with adjacent cropped fields. *Agriculture, Ecosystems & Environment*, 223: 123–134. https://doi.org/10.1016/j.agee.2016.02.026

Amagase, H. 2006. Clarifying the real bioactive constituents of garlic. *The Journal of Nutrition*, 136(3): 716S-725S. https://doi.org/10.1093/jn/136.3.716S

Andersen, D.S., Van Weelden, M.B., Trabue, S.L. & Pepple, L.M. 2015. Lab-assay for estimating methane emissions from deep-pit swine manure storages. *Journal of Environmental Management*, 159: 18–26. https://doi.org/10.1016/j.jenvman.2015.05.003

Andersen, D.S., Yang, F., Trabue, S.L., Kerr, B.J. & Howe, A. 2018. Narasin as a manure additive to reduce methane production from swine manure. *Transactions of the American Society of Agricultural and Biological Engineers*, 61(3): 943–953. https://doi.org/10.13031/trans.12568

Antaya, N.T., Ghelichkhan, M., Pereira, A.B.D., Soder, K.J. & Brito, A.F. 2019. Production, milk iodine, and nutrient utilization in Jersey cows supplemented with the brown seaweed *Ascophyllum nodosum* (kelp meal) during the grazing season. *Journal of Dairy Science*, 102(9): 8040–8058. https://doi.org/10.3168/jds.2019-16478

Antaya, N.T., Soder, K.J., Kraft, J., Whitehouse, N.L., Guindon, N.E., Erickson, P.S., Conroy, A.B. & Brito, A.F. 2015. Incremental amounts of *Ascophyllum nodosum* meal do not improve animal performance but do increase milk iodine output in early lactation dairy cows fed high-forage diets. *Journal of Dairy Science*, 98(3): 1991–2004. https://doi.org/10.3168/jds.2014-8851

Antoni, D., Zverlov, V.V. & Schwarz, W.H. 2007. Biofuels from microbes. *Applied Microbiology and Biotechnology*, 77(1): 23–35. https://doi.org/10.1007/s00253-007-1163-x

Aoun, M., Amiand, G., Garres, P. & Boide, P. 2003. Food supplement used in feed formulations in ruminants. World Intellectual Property Organization. No. 056935, filed 14 January 2003, and issued 17 July 2003. https://patents.google.com/patent/WO2003056935A1/en

Appuhamy, J.A.D.R.N., France, J. & Kebreab, E. 2016. Models for predicting enteric methane emissions from dairy cows in North America, Europe, and Australia and New Zealand. *Global Change Biology*, 22(9): 3039–3056. https://doi.org/10.1111/gcb.13339

Appuhamy, J.A.D.R.N., Strathe, A.B., Jayasundara, S., Wagner-Riddle, C., Dijkstra, J., France, J. & Kebreab, E. 2013. Anti-methanogenic effects of monensin in dairy and beef cattle: A meta-analysis. *Journal of Dairy Science*, 96(8): 5161–5173. https://doi.org/10.3168/jds.2012-5923

Arbre, M., Rochette, Y., Guyader, J., Lascoux, C., Gómez, L.M., Eugène, M., Morgavi, D.P., Renand, G., Doreau, M. & Martin, C. 2016. Repeatability of enteric methane determinations from cattle using either the SF6 tracer technique or the GreenFeed system. *Animal Production Science*, 56(3): 238–243. https://doi.org/10.1071/AN15512

Archer, D., Eby, M., Brovkin, V., Ridgwell, A., Cao, L., Mikolajewicz, U., Caldeira, K., Matsumoto, K., Munhoven, G., Montenegro, A. & Tokos, K. 2009. Atmospheric lifetime of fossil fuel carbon dioxide. *Annual Review of Earth and Planetary Sciences*, 37(1): 117–134. https://doi.org/10.1146/annurev.earth.031208.100206

Archimède, H., Eugène, M., Marie-Magdeleine, C., Boval, M., Martin, C., Morgavi, D.P., Lecomte, P. & Doreau, M. 2011. Comparison of methane production between C3 and C4 grasses and legumes. *Animal Feed Science and Technology*, 166–167: 59–64. https://doi.org/10.1016/j.anifeedsci.2011.04.003

Argyle, J.L. & Baldwin, R.L. 1988. Modeling of rumen water kinetics and effects of rumen pH changes. *Journal of Dairy Science*, 71(5): 1178–1188. https://doi.org/10.3168/jds.S0022-0302(88)79672-1

Arndt, C., Hristov, A.N., Price, W.J., McClelland, S.C., Pelaez, A.M., Cueva, S.F., Oh, J., Bannink, A., Bayat, A., Crompton, L.A., Dijkstra, J., Eugène, M.A., Kebreab, E., Kreuzer, M., McGee, M., Martin, C, Newbold, C. J., Reynolds, C.K., Schwarm, A., Shingfield, K.J., Veneman, J.B., Yáñez-Ruiz, D.R. & Yu, Z. 2021. Strategies to mitigate enteric methane emissions by ruminants – A way to approach the 2.0°C target. *agriRxiv*, 2021: 20210085288. https://doi.org/10.31220/agriRxiv.2021.00040

Arndt, C., Leytem, A.B., Hristov, A.N., Zavala-Araiza, D., Cativiela, J.P., Conley, S., Daube, C., Faloona, I. & Herndon, S.C. 2018. Short-term methane emissions from 2 dairy farms in California estimated by different measurement techniques and US Environmental Protection Agency inventory methodology: A case study. *Journal of Dairy Science*, 101(12): 11461–11479. https://doi.org/10.3168/jds.2017-13881

Arndt, C., Powell, J.M., Aguerre, M.J., Crump, P.M. & Wattiaux, M.A. 2015b. Feed conversion efficiency in dairy cows: Repeatability, variation in digestion and metabolism of energy and nitrogen, and ruminal methanogens. *Journal of Dairy Sci*, 98(6): 3938–3950. https://doi.org/10.3168/jds.2014-8449

Arndt, C., Powell, J.M., Aguerre, M.J. & Wattiaux, M.A. 2015a. Performance, digestion, nitrogen balance, and emission of manure ammonia, enteric methane, and carbon dioxide in lactating cows fed diets with varying alfalfa silage-to-corn silage ratios. *Journal of Dairy Science*, 98(1): 418–430. https://doi.org/10.3168/jds.2014-8298

Arora, V.K., Katavouta, A., Williams, R.G., Jones, C.D., Brovkin, V., Friedlingstein, P., Schwinger, J., Bopp, L., Boucher, O., Cadule, P., Chamberlain, M.A., Christian, J.R., Delire, C., Fisher, R.A., Hajima, T., Ilyina, T., Joetzjer, E., Kawamiya, M., Koven, C.D., Krasting, J.P., Law, R.M., Lawrence, D.M., Lenton, A., Lindsay, K., Pongratz, J., Raddatz, T., Séférian, R., Tachiiri, K., Tjiputra, J.F., Wiltshire, A., Wu, T. & Ziehn, T. 2020. Carbon–concentration and carbon–climate feedbacks in CMIP6 models and their comparison to CMIP5 models. *Biogeosciences*, 17(16): 4173–4222. https://doi.org/10.5194/bg-17-4173-2020

Arrow, K.J., Cropper, M.L., Gollier, C., Groom, B., Heal, G.M., Newell, R.G., Nordhaus, W.D., Pindyck, R.S., Pizer, W.A., Portney, P.R., Sterner, T., Tol, R.S.J. & Weitzman, M.L. 2014. Should governments use a declining discount rate in project analysis? *Review of Environmental Economics and Policy*, 8(2): 145–163. https://doi.org/10.1093/reep/reu008

Asseng, S., Ewert, F., Rosenzweig, C., Jones, J.W., Hatfield, J.L., Ruane, A.C., Boote, K.J., Thorburn, P.J., Rötter, R.P., Cammarano, D. & Brisson, N. 2013. Uncertainty in simulating wheat yields under climate change. *Nature Climate Change*, 3(9): 827–832. https://doi.org/10.1038/nclimate1916

Aulakh, M.S., Wassmann, R., Bueno, C., Kreuzwieser, J. & Rennenberg, H. 2001. Characterization of root exudates at different growth stages of ten rice (*Oryza sativa* L.) cultivars. *Plant Biology*, 3(2): 139–148. https://doi.org/10.1055/s-2001-12905

Aulakh, M.S., Wassmann, R. & Rennenberg, H. 2001. Methane emissions from rice fields—quantification, mechanisms, role of management, and mitigation options. *Advances in Agronomy*, 70: 193–260. https://doi.org/10.1016/S0065-2113(01)70006-5

Aulakh, M.S., Wassmann, R. & Rennenberg, H. 2002. Methane transport capacity of twenty-two rice cultivars from five major Asian rice-growing countries. *Agriculture, Ecosystems & Environment*, 91(1): 59–71. https://doi.org/10.1016/S0167-8809(01)00260-2

Azzaz, H.H., Murad, H.A. & Morsy, T.A. 2015. Utility of ionophores for ruminant animals: A review. *Asian Journal of Animal Sciences*, 9(6): 254–265. https://doi.org/10.3923/ajas.2015.254.265

Baca-González, V., Asensio-Calavia, P., González-Acosta, S., Pérez de la Lastra, J.M. & Morales de la Nuez, A. 2020. Are vaccines the solution for methane emissions from ruminants? A systematic review. *Vaccines*, 8(3): 460. https://doi.org/10.3390/vaccines8030460

Baker, S.K. & Perth, W. 2000. Method for improving utilization of nutrients by ruminant or ruminant-like animals. United States of America. No. 6 036 950, filed 19 October 1994, and issued 14 March 2000. https://patents.google.com/patent/US6036950A/en

Bakkaloglu, S., Lowry, D., Fisher, R.E., France, J.L., Brunner, D., Chen, H. & Nisbet, E.G. 2021. Quantification of methane emissions from UK biogas plants. *Waste Management*, 124: 82-93. https://doi.org/10.1016/j.wasman.2021.01.011

Balakrishnan, D., Barbadikar, K., Latha, P.C. & Subrahmanyam, D. 2018. Crop improvement strategies for mitigation of methane emissions from rice. *Emirates Journal of Food and Agriculture*, 30: 451–462. https://doi.org/10.9755/ejfa.2018.v30.i6.1707

Balcombe, P., Speirs, J.F., Brandon, N.P. & Hawkes, A.D. 2018. Methane emissions: Choosing the right climate metric and time horizon. *Environmental Science: Processes & Impacts*, 20(10): 1323–1339. https://doi.org/10.1039/C8EM00414E

Baldocchi, D.D. 2003. Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: Past, present and future. *Global Change Biology*, 9(4): 479–492. https://onlinelibrary.wiley.com/doi/10.1046/j.1365-2486.2003.00629.x

Baldocchi, D.D., 2014. Measuring fluxes of trace gases and energy between ecosystems and the atmosphere – The state and future of the eddy covariance method. *Global Change Biology*, 20(12): 3600–3609. https://doi.org/10.1111/gcb.12649

Baldocchi, D.D., Detto, M., Sonnentag, O., Verfaillie, J., Teh, Y.A., Silver, W. & Kelly, N.M. 2012. The challenges of measuring methane fluxes and concentrations over a peatland pasture. *Agricultural and Forest Meteorology*, 153: 177–187. https://doi.org/10.1016/j.agrformet.2011.04.013

Baldocchi, D.D., Hincks, B.B. & Meyers, T.P. 1988. Measuring biosphere-atmosphere exchanges of biologically related gases with micrometeorological methods. *Ecology*, 69(5): 1331–1340. https://doi.org/10.2307/1941631

Baldwin, R.L. 1995. *Ruminant digestion and metabolism*. New York, USA, Chapman & Hall.

Baldwin, R.L., France, J., Beever, D.E., Gill, M. & Thornley, J.H.M. 1987. Metabolism of the lactating cow: III. Properties of mechanistic models suitable for evaluation of energetic relationships and factors involved in the partition of nutrients. *Journal of Dairy Research*, 54(1): 133–145. https://doi.org/10.1017/S0022029900025243

Baldwin, R.L., France, J. & Gill, M. 1987. Metabolism of the lactating cow: I. Animal elements of a mechanistic model. *Journal of Dairy Research*, 54(1): 77–105. https://doi.org/10.1017/S002202990002522X

Baldwin, R.L., Thornley, J.H.M. & Beever, D.E. 1987. Metabolism of the lactating cow: II. Digestive elements of a mechanistic model. *Journal of Dairy Research*, 54(1): 107–131. https://doi.org/10.1017/S0022029900025231

Ballou, M.A. 2012. Growth and Development Symposium: Inflammation: Role in the etiology and pathophysiology of clinical mastitis in dairy cows. *Journal of Animal Science*, 90(5): 1466–1478. https://doi.org/10.2527/jas.2011-4663

Bampidis, V., Azimonti, G., Bastos, M. de L., Christensen, H., Dusemund, B., Fašmon Durjava, M., Kouba, M., López-Alonso, M., López Puente, S., Marcon, F., Mayo, B., Pechova, A., Petkova, M., Ramos, F., Sanz, Y., Villa, R., Woutersen, R., Aquilina, G., Bories, G., Brantom, P.G., Gropp, J., Svensson, K., Tost, L., Anguita, M., Galobart, J., Manini, P., Tarres-Call, J. & Pizzo, F. 2021. Safety and efficacy of a feed additive consisting of 3-nitrooxypropanol (Bovaer® 10) for ruminants for milk production and reproduction (DSM Nutritional Products Ltd). *EFSA Journal*, 19(11): e06905. https://doi.org/10.2903/j.efsa.2021.6905

Bannink, A., Kogut, J., Dijkstra, J., France, J., Kebreab, E., Van Vuuren, A.M. & Tamminga, S. 2006. Estimation of the stoichiometry of volatile fatty acid production in the rumen of lactating cows. *Journal of Theoretical Biology*, 238(1): 36–51. https://doi.org/10.1016/j.jtbi.2005.05.026

Bannink, A., Reijs, J.W. & Dijkstra, J. 2008. Integrated approaches to evaluate nutritional strategies for dairy cows. In: J. France & E. Kebreab, eds. *Mathematical modelling in animal nutrition*, pp. 462–484. Wallingford, UK, CABI Publishing.

Bannink, A., van Schijndel, M.W. & Dijkstra, J. 2011. A model of enteric fermentation in dairy cows to estimate methane emission for the Dutch National Inventory Report using the IPCC Tier 3 approach. *Animal Feed Science and Technology*, 166–167: 603–618. https://doi.org/10.1016/j.anifeedsci.2011.04.043

Barchyn, T.E., Hugenholtz, C.H. & Fox, T.A. 2019. Plume detection modeling of a drone-based natural gas leak detection system. *Elementa: Science of the Anthropocene*, 7: 41. https://doi.org/10.1525/elementa.379

Baresi, L. & Bertani, G. 1984. Isolation of a bacteriophage for a methanogenic bacterium. In: *Abstracts of Annual Meeting of the American Society for Microbiology*, *New Orleans*, p. 133. Abstract l–74. Washington, DC, American Society for Microbiology.

Barreto-Mendes, L., Ferreira-Tinoco, I.D.F., Ogink, N., Osorio-Hernandez, R. & Osorio-Saraz, J.A. 2014. A refined protocol for calculating air flow rate of naturally-ventilated broiler barns based on CO_2 mass balance. *DYNA*, 81(185): 189. https://doi.org/10.15446/dyna.v81n185.38069

Bartlett, P.C., van Wijk, J., Wilson, D.J., Green, C.D., Miller, G.Y., Majewski, G.A. & Heider, L.E. 1991. Temporal patterns of lost milk production following clinical mastitis in a large Michigan Holstein herd. *Journal of Dairy Science*, 74(5): 1561–1572. https://doi.org/10.3168/jds.S0022-0302(91)78318-5

Batalla, I., Knudsen, M.T., Mogensen, L., Hierro, Ó.D., Pinto, M. & Hermansen, J.E. 2015. Carbon footprint of milk from sheep farming systems in Northern Spain including soil carbon sequestration in grasslands. *Journal of Cleaner Production*, 104: 121–129. https://doi.org/10.1016/j.jclepro.2015.05.043

Bauchop, T. 1967. Inhibition of rumen methanogenesis by methane analogues. *Journal of Bacteriology*, 94(1): 171–175. https://doi.org/10.1128/jb.94.1.171-175.1967

Bauchop, T. & Mountfort, D.O. 1981. Cellulose fermentation by a rumen anaerobic fungus in both the absence and the presence of rumen methanogens. *Applied and Environmental Microbiology*, 42(6): 1103–1110. https://doi.org/10.1128/aem.42.6.1103-1110.1981

Bayat, A.R., Kairenius, P., Stefański, T., Leskinen, H., Comtet-Marre, S., Forano, E., Chaucheyras-Durand, F. & Shingfield, K.J. 2015. Effect of camelina oil or live yeasts (*Saccharomyces cerevisiae*) on ruminal methane production, rumen fermentation, and milk fatty acid composition in lactating cows fed grass silage diets. *Journal of Dairy Science*, 98(5): 3166–3181. https://doi.org/10.3168/jds.2014-7976

Bayat, A.R., Ventto, L., Kairenius, P., Stefański, T., Leskinen, H., Tapio, I., Negussie, E., Vilkki, J. & Shingfield, K.J. 2017. Dietary forage to concentrate ratio and sunflower oil supplement alter rumen fermentation, ruminal methane emissions, and nutrient utilization in lactating cows. *Translational Animal Science*, 1(3): 277–286. https://doi.org/10.2527/tas2017.0032

Beauchemin, K.A., Kreuzer, M., O'Mara, F. & McAllister, T.A. 2008. Nutritional management for enteric methane abatement: A review. *Australian Journal of Experimental Agriculture*, 48(2): 21. https://doi.org/10.1071/EA07199

Beauchemin, K.A, McAllister, T. & McGinn, S.M. 2009. Dietary mitigation of enteric methane from cattle. CAB Reviews: Perspectives in agriculture, veterinary science. *Nutrition and Natural Resources*, 4(035): 1–18. https://doi.org/10.1079/PAVSNNR20094035

Beauchemin, K.A. & McGinn, S.M. 2005. Methane emissions from feedlot cattle fed barley or corn diets. *Journal of Animal Science*, 83(3): 653–661. https://doi.org/10.2527/2005.833653x

Beauchemin, K.A. & McGinn, S.M. 2006. Enteric methane emissions from growing beef cattle as affected by diet and level of intake. *Canadian Journal of Animal Science*, 86(3): 401–408. https://doi.org/10.4141/A06-021

Beauchemin, K.A., McGinn, S.M., Benchaar, C. & Holtshausen, L. 2009. Crushed sunflower, flax, or canola seeds in lactating dairy cow diets: Effects on methane production, rumen fermentation, and milk production. *Journal of Dairy Science*, 92(5): 2118–2127. https://doi.org/10.3168/jds.2008-1903

Beauchemin, K.A., McGinn, S.M., Martinez, T.F. & McAllister, T.A. 2007. Use of condensed tannin extract from quebracho trees to reduce methane emissions from cattle. *Journal of Animal Science*, 85(8): 1990–1996. https://doi.org/10.2527/jas.2006-686

Beauchemin, K.A., Ungerfeld, E.M., Abdalla, A.L., Alvarez, C., Arndt, C., Becquet, P., Benchaar, C., Berndt, A., Mauricio, R.M., McAllister, T.A., Oyhantcabal, W., Salami, S.A., Shalloo, L., Sun, Y., Tricarico, J., Uwizeye, A., De Camillis, C., Bernoux, M., Robinson, T. & Kebreab, E. 2022. Invited review: Current enteric methane mitigation options. *Journal of Dairy Science*, 105(12): 9297–9326. https://doi.org/10.3168/jds.2022-22091

Beauchemin, K.A., Ungerfeld, E.M., Eckard, R.J. & Wang, M. 2020. Review: Fifty years of research on rumen methanogenesis: Lessons learned and future challenges for mitigation. *Animal*, 14: s2–s16. https://doi.org/10.1017/S1751731119003100

Beddoes, J.C., Bracmort, K.S., Burns, R.T. & Lazarus, W.F. 2007. *An analysis of energy production costs from manure anaerobic digestion systems on U.S. livestock production facilities.* U.S. Department of Agriculture. Natural Resources Conservation Service. Technical note No. 1. https://directives.sc.egov.usda.gov/OpenNonWebContent.aspx?content=22533.wba

Belanche, A., Newbold, C.J., Morgavi, D.P., Bach, A., Zweifel, B. & Yáñez-Ruiz, D.R. 2020. A meta-analysis describing the effects of the essential oils blend Agolin Ruminant on performance, rumen fermentation and methane emissions in dairy cows. *Animals*, 10(4): 620. https://doi.org/10.3390/ani10040620

Bell, M.J., Potterton, S.L., Craigon, J., Saunders, N., Wilcox, R.H., Hunter, M., Goodman, J.R. & Garnsworthy, P.C. 2014. Variation in enteric methane emissions among cows on commercial dairy farms. *Animal*, 8(9): 1540–1546. https://doi.org/10.1017/S1751731114001530

Bellarby, J., Tirado, R., Leip, A., Weiss, F., Lesschen, J.P. & Smith, P. 2013. Livestock greenhouse gas emissions and mitigation potential in Europe. *Global Change Biology*, 19(1): 3–18. https://doi.org/10.1111/j.1365-2486.2012.02786.x

Beltran, I., van der Weerden, T.J., Alfaro, M.A., Amon, B., Klein, C.A.M., Grace, P., Hafner, S., Hassouna, M., Hutchings, N., Krol, D.J., Leytem, A.B., Noble, A., Salazar, F., Thorman, R.E. & Velthof, G.L. 2021. DATAMAN: A global database of nitrous oxide and ammonia emission factors for excreta deposited by livestock and land-applied manure. *Journal of Environmental Quality*, 50(2): 513–527. https://doi.org/10.1002/jeq2.20186

Benaouda, M., Martin, C., Li, X., Kebreab, E., Hristov, A.N., Yu, Z., Yáñez-Ruiz, D.R., Reynolds, Crompton, L.A., Dijkstra, J., Bannink, A., Schwarm, A., Kreuzer, M., McGee, M., Lund, P., Hellwing, A.L.P., Weisbjerg, M.R., Moate, P.J., Bayat, A.R., Shingfield, K.J., Peiren, N. & Eugène, M. 2019. Evaluation of the performance of existing mathematical models predicting enteric methane emissions from ruminants: Animal categories and dietary mitigation strategies. *Animal Feed Science and Technology*, 255: 114207. https://doi.org/10.1016/j.anifeedsci.2019.114207

Benchaar, C. 2020. Feeding oregano oil and its main component carvacrol does not affect ruminal fermentation, nutrient utilization, methane emissions, milk production, or milk fatty acid composition of dairy cows. *Journal of Dairy Science*, 103(2): 1516–1527. https://doi.org/10.3168/jds.2019-17230

Benchaar, C. 2021. Diet supplementation with thyme oil and its main component thymol failed to favorably alter rumen fermentation, improve nutrient utilization, or enhance milk production in dairy cows. *Journal of Dairy Science*, 104(1): 324–336. https://doi.org/10.3168/jds.2020-18401

Benchaar, C., Calsamiglia, S., Chaves, A.V., Fraser, G.R., Colombatto, D., McAllister, T.A. & Beauchemin, K.A. 2008. A review of plant-derived essential oils in ruminant nutrition and production. *Animal Feed Science and Technology*, 145(1): 209–228. https://doi.org/10.1016/j.anifeedsci.2007.04.014

Benchaar, C. & Greathead, H. 2011. Essential oils and opportunities to mitigate enteric methane emissions from ruminants. *Animal Feed Science and Technology*, 166–167: 338–355. https://doi.org/10.1016/j.anifeedsci.2011.04.024

Benchaar, C., Hassanat, F., Gervais, R., Chouinard, P.Y., Petit, H.V. & Massé, D.I. 2014. Methane production, digestion, ruminal fermentation, nitrogen balance, and milk production of cows fed corn silage- or barley silage-based diets. *Journal of Dairy Science*, 97(2): 961–974. https://doi.org/10.3168/jds.2013-7122

Benchaar, C., Hristov, A.N. & Greathead, H. 2009. Essential oils as feed additives in animal nutrition. In: T. Steiner, ed. *Phytogenics in animal nutrition*, pp. 111–146. Nottingham, UK, Nottingham University Press.

Benchaar, C., Pomar, C. & Chiquette, J. 2001. Evaluation of dietary strategies to reduce methane production in ruminants: A modelling approach. *Canadian Journal of Animal Science*, 81(4): 563–574. https://doi.org/10.4141/A00-119

Benchaar, C., Rivest, J., Pomar, C. & Chiquette, J. 1998. Prediction of methane production from dairy cows using existing mechanistic models and regression equations. *Journal of Animal Science*, 76(2): 617–627. https://doi.org/10.2527/1998.762617x

Bender, M. & Conrad, R. 1992. Kinetics of CH_4 oxidation in oxic soils exposed to ambient air or high CH_4 mixing ratios. *FEMS Microbiology Letters*, 101(4): 261–270. https://doi.org/10.1111/j.1574-6968.1992.tb05783.x

Benner, R., MacCubbin, A.E. & Hodson, R.E. 1984. Anaerobic biodegradation of the lignin and polysaccharide components of lignocellulose and synthetic lignin by sediment microflora. *Applied and Environmental Microbiology*, 47(5): 998–1004. https://doi.org/10.1128/aem.47.5.998-1004.1984

Benner, R., Newell, S.Y., MacCubbin, A.E. & Hodson, R.E. 1984. Relative contributions of bacteria and fungi to rates of degradation of lignocellulosic detritus in salt-marsh sediments. *Applied and Environmental Microbiology*, 48(1): 36–40. https://doi.org/10.1128/aem.48.1.36-40.1984

Bergamaschi, P., Frankenberg, C., Meirink, J.F., Krol, M., Dentener, F., Wagner, T., Platt, U., Kaplan, J.O., Körner, S., Heimann, M. & Dlugokencky, E.J. 2007. Satellite chartography of atmospheric methane from SCIAMACHY on board ENVISAT: 2. Evaluation based on inverse model simulations. *Journal of Geophysical Research*, 112(D2): D02304. https://doi.org/10.1029/2006JD007268

Bergen, W.G. & Bates, D.B. 1984. Ionophores: Their effect on production efficiency and mode of action. *Journal of Animal Science*, 58(6): 1465–1483. https://doi.org/10.2527/jas1984.5861465x

Berger, C., Lettat, A., Martin, C. & Noziere, P. 2014. Method for reducing methane production in a ruminant animal. United States of America. No. 0112889, filed 26 April 2012, and issued 30 January 2014. https://patents.google.com/patent/US20140112889A1/en

Berntsen, T., Tanaka, K. & Fuglestvedt, J.S. 2010. Does black carbon abatement hamper CO_2 abatement? *Climatic Change*, 103(3): 627–633. https://doi.org/10.1007/s10584-010-9941-3

Bird-Gardiner, T., Arthur, P.F., Barchia, I.M., Donoghue, K.A. & Herd, R.M. 2017. Phenotypic relationships among methane production traits assessed under ad libitum feeding of beef cattle. *Journal of Animal Science*, 95(10): 4391–4398. https://doi.org/10.2527/jas2017.1477

Bitsie, B., Osorio, A.M., Henry, D.D., Silva, B.C., Godoi, L.A., Supapong, C., Brand, T. & Schoonmaker, J.P. 2022. Enteric methane emissions, growth, and carcass characteristics of feedlot steers fed a garlic- and citrus-based feed additive in diets with three different forage concentrations. *Journal of Animal Science*, 100(5): skac139. https://doi.org/10.1093/jas/skac139

Blaxter, K.L. 1962. *The energy metabolism of ruminants*. London, Hutchinson.

Blaxter, K.L. & Clapperton, J.L. 1965. Prediction of the amount of methane produced by ruminants. *British Journal of Nutrition*, 19(1): 511–522. https://doi.org/10.1079/BJN19650046

Boadi, D.A., Wittenberg, K.M., Scott, S.L., Burton, D., Buckley, K., Small, J.A. & Ominski, K.H. 2004. Effect of low and high forage diet on enteric and manure pack greenhouse gas emissions from a feedlot. *Canadian Journal of Animal Science*, 84(3): 445–453. https://doi.org/10.4141/A03-079

Bodansky, D.M., Hoedl, S.A., Metcalf, G.E. & Stavins, R.N. 2016. Facilitating linkage of climate policies through the Paris outcome. *Climate Policy*, 16(8): 956–972. https://doi.org/10.1080/14693062.2015.1069175

Boeckx, P., Van Cleemput, O. & Meyer, T. 1998. The influence of land use and pesticides on methane oxidation in some Belgian soils. *Biology and Fertility of Soils*, 27(3): 293–298. https://doi.org/10.1007/s003740050436

Boeckx, P., Van Cleemput, O. & Villaralvo, I. 1997. Methane oxidation in soils with different textures and land use. *Nutrient Cycling in Agroecosystems*, 49(1): 91–95. https://doi.org/10.1023/A:1009706324386

Boeckx, P., Xu, X. & Van Cleemput, O. 2005. Mitigation of N_2O and CH_4 emission from rice and wheat cropping systems using dicyandiamide and hydroquinone. *Nutrient Cycling in Agroecosystems*, 72(1): 41–49. https://doi.org/10.1007/s10705-004-7352-4

Bogner, J., Pipatti, R. & Hashimoto, S. 2008. Mitigation of global greenhouse gas emissions from waste: Conclusions and strategies from the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report. Working Group III. *Waste Management & Research*, 26(1): 11–32.

Boone, L., Van linden, V., De Meester, S., Vandecasteele, B., Muylle, H., Roldán-Ruiz, I., Nemecek, T. & Dewulf, J. 2016. Environmental life cycle assessment of grain maize production: An analysis of factors causing variability. *Science of the Total Environment*, 553: 551–564. https://doi.org/10.1016/j.scitotenv.2016.02.089

Borken, W., Xu, Y.-J. & Beese, F. 2003. Conversion of hardwood forests to spruce and pine plantations strongly reduced soil methane sink in Germany. *Global Change Biology*, 9(6): 956–966. https://doi.org/10.1046/j.1365-2486.2003.00631.x

Born, M., Dörr, H. & Levin, I. 1990. Methane consumption in aerated soils of the temperate zone. *Tellus B: Chemical and Physical Meteorology*, 42(1): 2. https://doi.org/10.3402/tellusb.v42i1.15186

Boucher, O. 2012. Comparison of physically- and economically-based CO_2-equivalences for methane. *Earth System Dynamics*, 3(1): 49–61. https://doi.org/10.5194/esd-3-49-2012

Boucher, O., Friedlingstein, P., Collins, B. & Shine, K.P. 2009. The indirect global warming potential and global temperature change potential due to methane oxidation. *Environmental Research Letters*, 4(4): 044007. https://doi.org/10.1088/1748-9326/4/4/044007

Bouman, B., Barker, R., Humphreys, E., Tuong, T.P., Atlin, G.N., Bennett, J., Dawe, D., Dittert, K., Dobermann, A., Facon, T., Fujimoto, N., Gupta, R., Haefele, S., Hosen, Y., Ismail, A., Johnson, D., Johnson, S., Khan, S., Shan, L., Masih, I., Matsuno, Y., Pandey, S., Peng, S., Muthukumarisami, T. & Wassmann, R. 2007. Rice: Feeding the billions. In: D. Molden, ed. *Water for food, water for life: A comprehensive assessment of water management in agriculture*, pp. 515–549. London & Colombo, Sri Lanka, Earthscan and International Water Management Institute.

Bourdin, F., Sakrabani, R., Kibblewhite, M.G. & Lanigan, G.J. 2014. Effect of slurry dry matter content, application technique and timing on emissions of ammonia and greenhouse gas from cattle slurry applied to grassland soils in Ireland. *Agriculture, Ecosystems & Environment*, 188: 122–133. https://doi.org/10.1016/j.agee.2014.02.025

Brandão, M., Kirschbaum, M.U.F., Cowie, A.L. & Hjuler, S.V. 2019. Quantifying the climate change effects of bioenergy systems: Comparison of 15 impact assessment methods. *GCB Bioenergy*, 11(5): 727–743. https://doi.org/10.1111/gcbb.12593

Brannon, E.Q., Moseman-Valtierra, S.M., Rella, C.W., Martin, R.M., Chen, X. & Tang, J. 2016. Evaluation of laser-based spectrometers for greenhouse gas flux measurements in coastal marshes. *Limnology and Oceanography: Methods*, 14(7): 466–476. https://doi.org/10.1002/lom3.10105

Brede, J., Peukert, M., Egert, B., Breves, G. & Brede, M. 2021. Long-term Mootral application impacts methane production and the microbial community in the rumen simulation technique system. *Frontiers in Microbiology*, 12: 691502. https://doi.org/10.3389/fmicb.2021.691502

Breider, I.S., Wall, E. & Garnsworthy, P.C. 2019. Short communication: Heritability of methane production and genetic correlations with milk yield and body weight in Holstein-Friesian dairy cows. *Journal of Dairy Science*, 102(8): 7277–7281. https://doi.org/10.3168/jds.2018-15909

Breuninger, C., Oswald, R., Kesselmeier, J. & Meixner, F.X. 2012. The dynamic chamber method: Trace gas exchange fluxes (NO, NO_2, O_3) between plants and the atmosphere in the laboratory and in the field. *Atmospheric Measurement Techniques*, 5(5): 955–989. https://doi.org/10.5194/amt-5-955-2012

Bright, R.M. & Lund, M.T. 2021. CO_2-equivalence metrics for surface albedo change based on the radiative forcing concept: A critical review. *Atmospheric Chemistry and Physics*, 21(12): 9887–9907. https://doi.org/10.5194/acp-21-9887-2021

Brink, C., Kroeze, C. & Klimont, Z. 2001. Ammonia abatement and its impact on emissions of nitrous oxide and methane in Europe – Part 1: Method. *Atmospheric Environment*, 35(36): 6299–6312. https://doi.org/10.1016/S1352-2310(01)00434-4

Bruckner, T., Hooss, G., Füssel, H.-M. & Hasselmann, K. 2003. Climate system modeling in the framework of the tolerable windows approach: The ICLIPS climate model. *Climatic Change*, 56(1/2): 119–137. https://doi.org/10.1023/A:1021300924356

Buckel, W. & Thauer, R.K. 2013. Energy conservation via electron bifurcating ferredoxin reduction and proton/Na+ translocating ferredoxin oxidation. *Biochimica et Biophysica Acta (BBA) – Bioenergetics*, 1827(2): 94–113. https://doi.org/10.1016/j.bbabio.2012.07.002

Buckel, W. & Thauer, R.K. 2018a. Flavin-based electron bifurcation, a new mechanism of biological energy coupling. *Chemical Reviews*, 118(7): 3862–3886. https://doi.org/10.1021/acs.chemrev.7b00707

Buckel, W. & Thauer, R.K. 2018b. Flavin-based electron bifurcation, ferredoxin, flavodoxin, and anaerobic respiration with protons (Ech) or NAD+ (Rnf) as electron acceptors: A historical review. *Frontiers in Microbiology*, 9: 401. https://doi.org/10.3389/fmicb.2018.00401

Bühler, K., Wenk, C., Broz, J. & Gebert, S. 2006. Influence of benzoic acid and dietary protein level on performance, nitrogen metabolism and urinary pH in growing-finishing pigs. *Archives of Animal Nutrition*, 60(5): 382–389. https://doi.org/10.1080/17450390600884369

Burt, S. 2004. Essential oils: Their antibacterial properties and potential applications in foods – A review. *International Journal of Food Microbiology*, 94(3): 223–253. https://doi.org/10.1016/j.ijfoodmicro.2004.03.022

Butterbach-Bahl, K., Papen, H. & Rennenberg, H. 1997. Impact of gas transport through rice cultivars on methane emission from rice paddy fields. *Plant, Cell & Environment*, 20(9): 1175–1183. https://doi.org/10.1046/j.1365-3040.1997.d01-142.x

Cain, M., Jenkins, S., Allen, M.R., Lynch, J., Frame, D.J., Macey, A.H. & Peters, G.P. 2021. Methane and the Paris Agreement temperature goals. *Philosophical Transactions of the Royal Society. Series A – Mathematical, Physical and Engineering Sciences*, 380(2215): 20200456. https://doi.org/10.1098/rsta.2020.0456

Cain, M., Lynch, J., Allen, M.R., Fuglestvedt, J.S., Frame, D.J. & Macey, A.H. 2019. Improved calculation of warming-equivalent emissions for short-lived climate pollutants. *npj Climate and Atmospheric Science*, 2(1): 29. https://doi.org/10.1038/s41612-019-0086-4

Callaghan, M.J., Tomkins, N.W., Benu, I. & Parker, A.J. 2014. How feasible is it to replace urea with nitrates to mitigate greenhouse gas emissions from extensively managed beef cattle? *Animal Production Science*, 54(9): 1300. https://doi.org/10.1071/AN14270

Callaway, T.R., Martin, S.A., Wampler, J.L., Hill, N.S. & Hill, G.M. 1997. Malate content of forage varieties commonly fed to cattle. *Journal of Dairy Science*, 80(8): 1651–1655. https://doi.org/10.3168/jds.S0022-0302(97)76096-X

Calvet, S., Estellés, F., Cambra-López, M., Torres, A.G. & Van Den Weghe, H.F.A. 2011. The influence of broiler activity, growth rate, and litter on carbon dioxide balances for the determination of ventilation flow rates in broiler production. *Poultry Science*, 90(11): 2449–2458. https://doi.org/10.3382/ps.2011-01580

Canadell, J.G., Kirschbaum, M.U.F., Kurz, W.A., Sanz, M.-J., Schlamadinger, B. & Yamagata, Y. 2007. Factoring out natural and indirect human effects on terrestrial carbon sources and sinks. *Environmental Science & Policy*, 10(4): 370–384. https://doi.org/10.1016/j.envsci.2007.01.009

Cantalapiedra-Hijar, G., Abo-Ismail, M., Carstens, G.E., Guan, L.L., Hegarty, R., Kenny, D.A., Mcgee, M., Plastow, G., Relling, A. & Ortigues-Marty, I. 2018. Biological determinants of between-animal variation in feed efficiency of growing beef cattle. *Animal*, 12(s2): s321–s335. https://doi.org/10.1017/S1751731118001489

Canul Solis, J.R., Piñeiro Vázquez, A.T., Arceo Castillo, J.I., Alayón Gamboa, J.A., Ayala Burgos, A.J., Aguilar Pérez, C.F., Solorio Sánchez, F.J., Castelán Ortega, O.A., Lachica López, M., Quintana Owen, P. & Ku Vera, J.C. 2017. Design and construction of low-cost respiration chambers for ruminal methane measurements in ruminants. *Revista Mexicana de Ciencias Pecuarias*, 8(2): 185. https://doi.org/10.22319/rmcp.v8i2.4442

Capper, J.L. 2011. The environmental impact of beef production in the United States: 1977 compared with 2007. *Journal of Animal Science*, 89(12): 4249–4261. https://doi.org/10.2527/jas.2010-3784

Capper, J.L. & Bauman, D.E. 2013. The role of productivity in improving the environmental sustainability of ruminant production systems. *Annual Review of Animal Biosciences*, 1(1): 469–489. https://doi.org/10.1146/annurev-animal-031412-103727

Capper, J.L. & Cady, R.A. 2020. The effects of improved performance in the US dairy cattle industry on environmental impacts between 2007 and 2017. *Journal of Animal Science*, 98(1): 1–14. https://doi.org/10.1093/jas/skz291

Capper, J.L., Cady, R.A. & Bauman, D.E. 2009. The environmental impact of dairy production: 1944 compared with 2007. *Journal of Animal Science*, 87(6): 2160–2167. https://doi.org/10.2527/jas.2009-1781

Carbone, V., Schofield, L.R., Sutherland-Smith, A.J. & Ronimus, R.S. 2018. Discovering inhibitors of rumen methanogens using high-throughput X-ray crystallography and enzyme-screening techniques. *Acta Crystallographica Section A Foundations and Advances*, 74(a1): a48–a48. https://doi.org/10.1107/S0108767318099518

Cardoso, A.S., Berndt, A., Leytem, A., Alves, B.J.R., de Carvalho, I. das N.O., de Barros Soares, L.H., Urquiaga, S. & Boddey, R.M. 2016. Impact of the intensification of beef production in Brazil on greenhouse gas emissions and land use. *Agricultural Systems*, 143: 86–96. https://doi.org/10.1016/j.agsy.2015.12.007

Carlsen, H.N., Joergensen, L. & Degn, H. 1991. Inhibition by ammonia of methane utilization in *Methylococcus capsulatus* (Bath). *Applied Microbiology and Biotechnology*, 35(1): 124–127. https://doi.org/10.1007/BF00180649

Carlson, K.M., Gerber, J.S., Mueller, N.D., Herrero, M., MacDonald, G.K., Brauman, K.A., Havlik, P., O'Connell, C.S., Johnson, J.A., Saatchi, S. & West, P.C. 2017. Greenhouse gas emissions intensity of global croplands. *Nature Climate Change*, 7(1): 63–68.

Carpenter, L.J. & Liss, P.S. 2000. On temperate sources of bromoform and other reactive organic bromine gases. *Journal of Geophysical Research: Atmospheres*, 105(D16): 20539–20547. https://doi.org/10.1029/2000JD900242

Carro, M.D. & Ungerfeld, E.M. 2015. Utilization of organic acids to manipulate ruminal fermentation and improve ruminant productivity. In: A.K. Puniya, R. Singh & D.N. Kamra, eds. *Rumen microbiology: From evolution to revolution*, pp. 177–197. New Delhi, Springer India. https://doi.org/10.1007/978-81-322-2401-3_13

Carulla, J., Kreuzer, M., Machmueller, A. & Hess, H. 2005. Supplementation of *Acacia mearnsii* tannins decrease methanogenesis and urinary nitrogen in forage-fed sheep. *Australian Journal of Agricultural Research*, 56(9): 961–970. https://doi.org/10.1071/AR05022

Castillo-González, A., Burrola-Barraza, M., Domínguez-Viveros, J. & Chávez-Martínez, A. 2014. Rumen microorganisms and fermentation. *Archivos de medicina veterinaria*, 46(3): 349–361. https://doi.org/10.4067/S0301-732X2014000300003

Cha, E., Bar, D., Hertl, J.A., Tauer, L.W., Bennett, G., González, R.N., Schukken, Y.H., Welcome, F.L. & Gröhn, Y.T. 2011. The cost and management of different types of clinical mastitis in dairy cows estimated by dynamic programming. *Journal of Dairy Science*, 94(9): 4476–4487. https://doi.org/10.3168/jds.2010-4123

Cha, E., Hertl, J.A., Schukken, Y.H., Tauer, L.W., Welcome, F.L. & Gröhn, Y.T. 2013. The effect of repeated episodes of bacteria-specific clinical mastitis on mortality and culling in Holstein dairy cows. *Journal of Dairy Science*, 96(8): 4993–5007. https://doi.org/10.3168/jds.2012-6232

Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B. & Misselbrook, T. 2011. Manure management: Implications for greenhouse gas emissions. *Animal Feed Science and Technology*, 166–167: 514–531. https://doi.org/10.1016/j.anifeedsci.2011.04.036

Chagunda, M.G.G. 2013. Opportunities and challenges in the use of the Laser Methane Detector to monitor enteric methane emissions from ruminants. *Animal*, 7(s2): 394–400. https://doi.org/10.1017/S1751731113000724

Chagunda, M.G.G., Ross, D. & Roberts, D.J. 2009. On the use of a laser methane detector in dairy cows. *Computers and Electronics in Agriculture*, 68(2): 157–160. https://doi.org/10.1016/j.compag.2009.05.008

Chagunda, M.G.G. & Yan, T. 2011. Do methane measurements from a laser detector and an indirect open-circuit respiration calorimetric chamber agree sufficiently closely? *Animal Feed Science and Technology*, 165(1–2): 8–14. https://doi.org/10.1016/j.anifeedsci.2011.02.005

Chang, J., Peng, S., Ciais, P., Saunois, M., Dangal, S.R.S., Herrero, M., Havlík, P., Tian, H. & Bousquet, P. 2019. Revisiting enteric methane emissions from domestic ruminants and their δ13C CH_4 source signature. *Nature Communications*, 10(1): 3420. https://doi.org/10.1038/s41467-019-11066-3

Chao, S.C., Young, D.G. & Oberg, C.J. 2000. Screening for inhibitory activity of essential oils on selected bacteria, fungi and viruses. *Journal of Essential Oil Research*, 12(5): 639–649. https://doi.org/10.1080/10412905.2000.9712177

Chará, J., Rivera, J., Barahona, R., Murgueitio, E.R., Deblitz, C., Reyes, E., Mauricio, R.M., Molina, J.J., Flores, M. & Zuluaga, M. 2017. Intensive silvopastoral systems: Economics and contribution to climate change mitigation and public policies. In: F. Montagnini, ed. *Integrating landscapes: Agroforestry for biodiversity conservation and food sovereignty*. Advances in Agroforestry, 12. New York, USA, Springer Cham. https://doi.org/10.1007/978-3-319-69371-2

Chaves, A.V., Thompson, L.C., Iwaasa, A.D., Scott, S.L., Olson, M.E., Benchaar, C., Veira, D.M. & McAllister, T.A. 2006. Effect of pasture type (alfalfa vs. grass) on methane and carbon dioxide production by yearling beef heifers. *Canadian Journal of Animal Science*, 86(3): 409–418. https://doi.org/10.4141/A05-081

Chen, M. & Wolin, M.J. 1977. Influence of CH_4 production by *Methanobacterium ruminantium* on the fermentation of glucose and lactate by *Selenomonas ruminantium*. *Applied and Environmental Microbiology*, 34(6): 756–759. https://doi.org/10.1128/aem.34.6.756-759.1977

Chen, S., Rotaru, A.-E., Shrestha, P.M., Malvankar, N.S., Liu, F., Fan, W., Nevin, K.P. & Lovley, D.R. 2014. Promoting interspecies electron transfer with biochar. *Scientific Reports*, 4(1): 5019. https://doi.org/10.1038/srep05019

Chen, Y.-H. & Prinn, R.G. 2006. Estimation of atmospheric methane emissions between 1996 and 2001 using a three-dimensional global chemical transport model. *Journal of Geophysical Research: Atmospheres*, 111(D10): n/a-n/a. https://doi.org/10.1029/2005JD006058

Cheng, K., Ogle, S.M., Parton, W.J. & Pan, G. 2013. Predicting methanogenesis from rice paddies using the DAYCENT ecosystem model. *Ecological Modelling*, 261–262: 19–31. https://doi.org/10.1016/j.ecolmodel.2013.04.003

Cheng, K., Ogle, S.M., Parton, W.J. & Pan, G. 2014. Simulating greenhouse gas mitigation potentials for Chinese Croplands using the DAYCENT ecosystem model. *Global Change Biology*, 20(3): 948–962. https://doi.org/10.1111/gcb.12368

Cherubini, F., Fuglestvedt, J., Gasser, T., Reisinger, A., Cavalett, O., Huijbregts, M.A.J., Johansson, D.J.A., Jørgensen, S.V., Raugei, M., Schivley, G., Strømman, A.H., Tanaka, K. & Levasseur, A. 2016. Bridging the gap between impact assessment methods and climate science. *Environmental Science & Policy*, 64: 129–140. https://doi.org/10.1016/j.envsci.2016.06.019

Cherubini, F. & Tanaka, K. 2016. Amending the inadequacy of a single indicator for climate impact analyses. *Environmental Science & Technology*, 50(23): 12530–12531. https://doi.org/10.1021/acs.est.6b05343

Chieng, S. & Kuan, S.H. 2022. Harnessing bioenergy and high value–added products from rice residues: A review. *Biomass Conversion and Biorefinery*, 12. https://doi.org/10.1007/s13399-020-00891-y

Chianese, D.S., Rotz, C.A. & Richard, T.L. 2009. Whole-farm greenhouse gas emissions: A review with application to a Pennsylvania dairy farm. *Applied Engineering in Agriculture*, 25(3): 431–442. https://doi.org/10.13031/2013.26895

Chin, K.-J. & Conrad, R. 1995. Intermediary metabolism in methanogenic paddy soil and the influence of temperature. *FEMS Microbiology Ecology*, 18(2): 85–102. https://doi.org/10.1111/j.1574-6941.1995.tb00166.x

Chobtang, J., Ledgard, S.F., McLaren, S.J. & Donaghy, D.J. 2017. Life cycle environmental impacts of high and low intensification pasture-based milk production systems: A case study of the Waikato region, New Zealand. *Journal of Cleaner Production*, 140: 664–674. https://doi.org/10.1016/j.jclepro.2016.06.079

Christiansen, J.R., Romero, A.J.B., Jørgensen, N.O.G., Glaring, M.A., Jørgensen, C.J., Berg, L.K. & Elberling, B. 2015. Methane fluxes and the functional groups of methanotrophs and methanogens in a young Arctic landscape on Disko Island, West Greenland. *Biogeochemistry*, 122(1): 15–33. https://doi.org/10.1007/s10533-014-0026-7

CIGR (International Commission of Agricultural and Biosystems Engineering). 2002. 4th report of working group on climatization of animal houses heat and moisture production at animal and house levels. In: S. Pedersen & K. Sällvik, eds. *International Commission of Agricultural Engineering, Section II*, 46 pp. Research Centre Bygholm, Danish Institute of Agricultural Sciences. Horsens, Denmark. www.cigr.org/sites/default/files/documets/CIGR_4TH_WORK_GR.pdf

Clanton, C., Jacobson, L. & Schmidt, D. 2012. *Monensin addition to swine manure deep pits for foaming control*. University of Minnesota Extension Fact Sheet.

Clapperton, J.L. 1974. The effect of trichloroacetamide, chloroform and linseed oil given into the rumen of sheep on some of the end-products of rumen digestion. *British Journal of Nutrition*, 32(01): 155–161. https://doi.org/10.1079/BJN19740065

Clapperton, J.L. 1977. The effect of a methane-suppressing compound, trichloroethyl adipate, on rumen fermentation and the growth of sheep. *Animal Science*, 24(2): 169–181. https://doi.org/10.1017/S0003356100011636

Clauss, M., Dittmann, M.T., Vendl, C., Hagen, K.B., Frei, S., Ortmann, S., Müller, D.W.H., Hammer S., Munn A.J., Schwarm A. & Kreuzer M. 2020. Review: Comparative methane production in mammalian herbivores. *Animal*, 14: s113–s123. https://doi.org/10.1017/S1751731119003161

Clemens, J. & Ahlgrimm, H.-J. 2001. Greenhouse gases from animal husbandry: Mitigation options. *Nutrient Cycling in Agroecosystems*, 60(1): 287–300. https://doi.org/10.1023/A:1012712532720

Clemens, J., Trimborn, M., Weiland, P. & Amon, B. 2006. Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. *Agriculture, Ecosystems & Environment*, 112(2): 171–177. https://doi.org/10.1016/j.agee.2005.08.016

Climate Watch. 2019. Washington, DC, World Resources Institute. [Cited 5 June 2023]. www.climatewatchdata.org

Cluett, J., VanderZaag, A.C., Baldé, H., McGinn, S., Jenson, E., Hayes, A.C. & Ekwe, S. 2020. Effects of two manure additives on methane emissions from dairy manure. *Animals*, 10(5): 807. https://doi.org/10.3390/ani10050807

Cobellis, G., Trabalza-Marinucci, M. & Yu, Z. 2016. Critical evaluation of essential oils as rumen modifiers in ruminant nutrition: A review. *The Science of the Total Environment*, 545–546: 556–568. https://doi.org/10.1016/j.scitotenv.2015.12.103

Collier, S.M., Ruark, M.D., Oates, L.G., Jokela, W.E. & Dell, C.J. 2014. Measurement of greenhouse gas flux from agricultural soils using static chambers. *Journal of Visualized Experiments: JoVE*, 90: 52110. https://doi.org/10.3791/52110

Collins, W.J., Frame, D.J., Fuglestvedt, J.S. & Shine, K.P. 2020. Stable climate metrics for emissions of short and long-lived species – Combining steps and pulses. *Environmental Research Letters*, 15(2): 024018. https://doi.org/10.1088/1748-9326/ab6039

Collins, W.J., Fry, M.M., Yu, H., Fuglestvedt, J.S., Shindell, D.T. & West, J.J. 2013. Global and regional temperature-change potentials for near-term climate forcers. *Atmospheric Chemistry and Physics*, 13(5): 2471–2485. https://doi.org/10.5194/acp-13-2471-2013

Cong, W., Meng, J. & Ying, S.C. 2018. Impact of soil properties on the soil methane flux response to biochar addition: A meta-analysis. *Environmental Science: Processes & Impacts*, 20(9): 1202–1209. https://doi.org/10.1039/C8EM00278A

Congio, G.F.S., Batalha, C.D.A., Chiavegato, M.B., Berndt, A., Oliveira, P.P.A., Frighetto, R.T.S., Maxwell, T.M.R., Gregorini, P. & Da Silva, S.C. 2018. Strategic grazing management towards sustainable intensification at tropical pasture-based dairy systems. *Science of the Total Environment*, 636: 872–880. https://doi.org/10.1016/j.scitotenv.2018.04.301

Conley, S., Faloona, I., Mehrotra, S., Suard, M., Lenschow, D.H., Sweeney, C., Herndon, S., Schwietzke, S., Pétron, G., Pifer, J., Kort, E.A. & Schnell, R. 2017. Application of Gauss's theorem to quantify localized surface emissions from airborne measurements of wind and trace gases. *Atmospheric Measurement Techniques*, 10(9): 3345–3358. https://doi.org/10.5194/amt-10-3345-2017

Conrad, R. 1999. Contribution of hydrogen to methane production and control of hydrogen concentrations in methanogenic soils and sediments. *FEMS Microbiology Ecology*, 28(3): 193–202. https://doi.org/10.1111/j.1574-6941.1999.tb00575.x

Conrad, R. 2020a. Importance of hydrogenotrophic, aceticlastic and methylotrophic methanogenesis for methane production in terrestrial, aquatic and other anoxic environments: A mini review. *Pedosphere*, 30(1): 25–39. https://doi.org/10.1016/S1002-0160(18)60052-9

Conrad, R. 2020b. Methane production in soil environments – Anaerobic biogeochemistry and microbial life between flooding and desiccation. *Microorganisms*, 8(6): 881. https://doi.org/10.3390/microorganisms8060881

Cook, S.R., Maiti, P.K., Chaves, A.V., Benchaar, C., Beauchemin, K.A. & McAllister, T.A. 2008. Avian (IgY) anti-methanogen antibodies for reducing ruminal methane production: *In vitro* assessment of their effects. *Australian Journal of Experimental Agriculture*, 48(2): 260. https://doi.org/10.1071/EA07249

Coppa, M., Jurquet, J., Eugène, M., Dechaux, T., Rochette, Y., Lamy, J.-M., Ferlay, A. & Martin, C. 2021. Repeatability and ranking of long-term enteric methane emissions measurement on dairy cows across diets and time using GreenFeed system in farm-conditions. *Methods*, 186: 59–67. https://doi.org/10.1016/j.ymeth.2020.11.004

Cord-Ruwisch, R., Seitz, H.-J. & Conrad, R. 1988. The capacity of hydrogenotrophic anaerobic bacteria to compete for traces of hydrogen depends on the redox potential of the terminal electron acceptor. *Archives of Microbiology*, 149(4): 350–357. https://doi.org/10.1007/BF00411655

Costa, C., Wironen, M., Racette, K. & Wollenberg, E. 2021. *Global Warming Potential* (GWP*): Understanding the implications for mitigating methane emissions in agriculture*. Wageningen, Kingdom of the Netherlands, CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS). https://hdl.handle.net/10568/114632

Coutinho, F.H., Edwards, R.A. & Rodríguez-Valera, F. 2019. Charting the diversity of uncultured viruses of Archaea and Bacteria. *BMC Biology*, 17(1): 109. https://doi.org/10.1186/s12915-019-0723-8

Cowan, N., Maire, J., Krol, D., Cloy, J.M., Hargreaves, P., Murphy, R., Carswell, A., Jones, S.K., Hinton, N., Anderson, M., Famulari, D., Bell, M.J., Stack, P., Levy, P., Skiba, U.M. & Drewer, J. 2021. Agricultural soils: A sink or source of methane across the British Isles? *European Journal of Soil Science*, 72(4): 1842–1862. https://doi.org/10.1111/ejss.13075

Cowan, N.J., Famulari, D., Levy, P.E., Anderson, M., Bell, M.J., Rees, R.M., Reay, D.S. & Skiba, U.M. 2014. An improved method for measuring soil N_2O fluxes using a quantum cascade laser with a dynamic chamber. *European Journal of Soil Science*, 65(5): 643–652.

Crosson, P., Shalloo, L., O'Brien, D., Lanigan, G.J., Foley, P.A., Boland, T.M. & Kenny, D.A. 2011. A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. *Animal Feed Science and Technology*, 166–167: 29–45. https://doi.org/10.1016/j.anifeedsci.2011.04.001

CSIRO (Commonwealth Scientific and Industrial Research Organization). 2007. *Nutrient requirements of domesticated ruminants.* Collingwood, Australia, CSIRO Publishing.

Czerkawski, J.W. & Breckenridge, G. 1977. Design and development of a long-term rumen simulation technique (Rusitec). *British Journal of Nutrition*, 38(3): 371–384. https://doi.org/10.1079/BJN19770102

Danovaro, R., Molari, M., Corinaldesi, C. & Dell'Anno, A. 2016. Macroecological drivers of archaea and bacteria in benthic deep-sea ecosystems. *Science Advances*, 2(4): e1500961. https://doi.org/10.1126/sciadv.1500961

Daube, C., Conley, S., Faloona, I.C., Arndt, C., Yacovitch, T.I., Roscioli, J.R. & Herndon, S.C. 2019. Using the tracer flux ratio method with flight measurements to estimate dairy farm CH_4 emissions in central California. *Atmospheric Measurement Techniques*, 12(4): 2085–2095. https://doi.org/10.5194/amt-12-2085-2019

Davidson, E.A., Savage, K.V.L.V., Verchot, L.V. & Navarro, R. 2002. Minimizing artifacts and biases in chamber-based measurements of soil respiration. *Agricultural and Forest Meteorology*, 113(1–4): 21–37. https://doi.org/10.1016/S0168-1923(02)00100-4

Davies, A., Nwaonu, H.N., Stanier, G. & Boyle, F.T. 1982. Properties of a novel series of inhibitors of rumen methanogenesis; *in vitro* and *in vivo* experiments including growth trials on 2,4-bis (trichloromethyl)-benzo [1, 3]dioxin-6-carboxylic acid. *British Journal of Nutrition*, 47(3): 565–576. https://doi.org/10.1079/BJN19820068

de Haas, Y., Pszczola, M., Soyeurt, H., Wall, E. & Lassen, J. 2017. Invited review: Phenotypes to genetically reduce greenhouse gas emissions in dairying. *Journal of Dairy Science*, 100(2): 855-870. https://doi.org/10.3168/jds.2016-11246

de Haas, Y., Veerkamp, R.F., De Jong, G. & Aldridge, M.N. 2021. Selective breeding as a mitigation tool for methane emissions from dairy cattle. *Animal*, 15: 100294. https://doi.org/10.1016/j.animal.2021.100294

de Haas, Y., Windig, J.J., Calus, M.P.L., Dijkstra, J., de Haan, M., Bannink, A. & Veerkamp, R.F. 2011. Genetic parameters for predicted methane production and potential for reducing enteric emissions through genomic selection. *Journal of Dairy Science*, 94(12): 6122–6134. https://doi.org/10.3168/jds.2011-4439

de Mulder, T., Peiren, N., Vandaele, L., Ruttink, T., De Campeneere, S., Van de Wiele, T. & Goossens, K. 2018. Impact of breed on the rumen microbial community composition and methane emission of Holstein Friesian and Belgian Blue heifers. *Livestock Science*, 207: 38–44. https://doi.org/10.1016/j.livsci.2017.11.009

de Oliveira Monteschio, J., de Souza, K.A., Vital, A.C.P., Guerrero, A., Valero, M.V., Kempinski, E.M.B.C., Barcelos, V.C., Nascimento, K.F. & do Prado, I.N. 2017. Clove and rosemary essential oils and encapsuled active principles (eugenol, thymol and vanillin blend) on meat quality of feedlot-finished heifers. *Meat Science*, 130: 50–57. https://doi.org/10.1016/j.meatsci.2017.04.002

de Oliveira Silva, R., Barioni, L.G., Hall, J.A.J., Folegatti Matsuura, M., Zanett Albertini, T., Fernandes, F.A. & Moran, D. 2016. Increasing beef production could lower greenhouse gas emissions in Brazil if decoupled from deforestation. *Nature Climate Change*, 6(5): 493–497. https://doi.org/10.1038/nclimate2916

de Raphélis-Soissan, V., Li, L., Godwin, I.R., Barnett, M.C., Perdok, H.B. & Hegarty, R.S. 2014. Use of nitrate and *Propionibacterium acidipropionici* to reduce methane emissions and increase wool growth of Merino sheep. *Animal Production Science*, 54(10): 1860. https://doi.org/10.1071/AN14329

de Vries, M., van Middelaar, C.E. & de Boer, I.J.M. 2015. Comparing environmental impacts of beef production systems: A review of life cycle assessments. *Livestock Science*, 178: 279–288. https://doi.org/10.1016/j.livsci.2015.06.020

Dean, J.F., Middelburg, J.J., Röckmann, T., Aerts, R., Blauw, L.G., Egger, M., Jetten, M.S.M., de Jong, A.E.E, Meisel, O.H., Rasigraf, O., Slomp, C.P., in' t Zandt, M.H. & Dolman, A.J. 2018. Methane feedbacks to the global climate system in a warmer world. *Reviews of Geophysics*, 56(1): 207–250. https://doi.org/10.1002/2017RG000559

Deans, S.G. & Ritchie, G. 1987. Antibacterial properties of plant essential oils. *International Journal of Food Microbiology*, 5(2): 165–180. https://doi.org/10.1016/0168-1605(87)90034-1

Debruyne, S., Ruiz-González, A., Artiles-Ortega, E., Ampe, B., Van Den Broeck, W., De Keyser, E., Vandaele, L., Goossens, K. & Fievez, V. 2018. Supplementing goat kids with coconut medium chain fatty acids in early life influences growth and rumen papillae development until 4 months after supplementation but effects on *in vitro* methane emissions and the rumen microbiota are transient. *Journal of Animal Science*, 96(5): 1978–1995. https://doi.org/10.1093/jas/sky070

Deighton, M.H., Williams, S.R.O., Hannah, M.C., Eckard, R.J., Boland, T.M., Wales, W.J. & Moate, P.J. 2014. A modified sulphur hexafluoride tracer technique enables accurate determination of enteric methane emissions from ruminants. *Animal Feed Science and Technology*, 197: 47–63. https://doi.org/10.1016/j.anifeedsci.2014.08.003

Del Grosso, S.J., Parton, W.J., Mosier, A.R., Hartman, M.D., Brenner, J., Ojima, D.S. & Schimel, D.S. 2001. Simulated interaction of carbon dynamics and nitrogen trace gas fluxes using the DAYCENT model. In: M. Schaffer, L. Ma & S. Abernethy, eds. *Modeling carbon and nitrogen dynamics for soil management*, pp. 303–332. Boca Raton, USA, CRC Press.

del Prado, A., Chadwick, D., Cardenas, L., Misselbrook, T., Scholefield, D. & Merino, P. 2010. Exploring systems responses to mitigation of GHG in UK dairy farms. *Agriculture, Ecosystems & Environment*, 136(3): 318–332. https://doi.org/10.1016/j.agee.2009.09.015

del Prado, A., Crosson, P., Olesen, J.E. & Rotz, C.A. 2013. Whole-farm models to quantify greenhouse gas emissions and their potential use for linking climate change mitigation and adaptation in temperate grassland ruminant-based farming systems. *Animal*, 7: 373–385. https://doi.org/10.1017/S1751731113000748

del Prado, A., Manzano, P. & Pardo, G. 2021. The role of the European small ruminant dairy sector in stabilising global temperatures: Lessons from GWP* warming-equivalent emission metrics. *Journal of Dairy Research*, 88(1): 8–15. https://doi.org/10.1017/S0022029921000157

del Prado, A., Misselbrook, T., Chadwick, D., Hopkins, A., Dewhurst, R.J., Davison, P., Butler, A., Schröder, J. & Scholefield, D. 2011. SIMSDAIRY: A modelling framework to identify sustainable dairy farms in the UK. Framework description and test for organic systems and N fertiliser optimisation. *Science of the Total Environment*, 409(19): 3993–4009. https://doi.org/10.1016/j.scitotenv.2011.05.050

den Brok, G.M., Hendricks, J.G.L., Vrielink, M.G.M. & van der Peet-Schwering, C.M.C. 1999. *Urinary pH, ammonia emission and performance of growing/finishing pigs after the addition of a mixture of organic acids, mainly benzoic acid, to the feed*. Raalte, Kingdom of the Netherlands, Research Institute for Pig Husbandry. https://research.wur.nl/en/publications/urinary-ph-ammonia-emission-and-performance-of-growingfinishing-p

Deng, J., Guo, L., Salas, W., Ingraham, P., Charrier-Klobas, J.G., Frolking, S. & Li, C. 2018a. Changes in irrigation practices likely mitigate nitrous oxide emissions from California cropland. *Global Biogeochemical Cycles*, 32(10): 1514–1527. https://doi.org/10.1029/2018GB005961

Deng, J., Li, C., Burger, M., Horwath, W.R., Smart, D., Six, J., Guo, L., Salas, W. & Frolking, S. 2018b. Assessing short-term impacts of management practices on N_2O emissions from diverse Mediterranean agricultural ecosystems using a biogeochemical model. *Journal of Geophysical Research: Biogeosciences*, 123(5): 1557–1571. https://doi.org/10.1029/2017JG004260

Denier van der Gon, H.A.C., Kropff, M.J., van Breemen, N., Wassmann, R., Lantin, R.S., Aduna, E., Corton, T.M. & van Laar, H.H. 2002. Optimizing grain yields reduces CH_4 emissions from rice paddy fields. *Proceedings of the National Academy of Sciences of the United States of America*, 99(19): 12021–12024. https://doi.org/10.1073/pnas.192276599

Denier van der Gon, H.A.C., van Bodegom, P.M., Wassmann, R., Lantin, R.S. & Metra-Corton, T.M. 2001. Sulfate-containing amendments to reduce methane emissions from rice fields: Mechanisms, effectiveness and costs. *Mitigation and Adaptation Strategies for Global Change*, 6(1): 71–89. https://doi.org/10.1023/A:1011380916490

Denison, S., Forster, P.M. & Smith, C.J. 2019. Guidance on emissions metrics for nationally determined contributions under the Paris Agreement. *Environmental Research Letters*, 14(12): 124002. https://doi.org/10.1088/1748-9326/ab4df4

Denman, S.E., Martinez Fernandez, G., Shinkai, T., Mitsumori, M. & McSweeney, C.S. 2015. Metagenomic analysis of the rumen microbial community following inhibition of methane formation by a halogenated methane analog. *Frontiers in Microbiology*, 6. https://doi.org/10.3389/fmicb.2015.01087

Dennehy, C., Lawlor, P., Jiang, Y., Gardiner, G., Xie, S., Nghiem, L. & Zhan, X. 2017. Greenhouse gas emissions from different pig manure management techniques: A critical analysis. *Frontiers of Environmental Science & Engineering*, 11(3): 1–16. https://doi.org/10.1007/s11783-017-0942-6

Denninger, T.M., Schwarm, A., Birkinshaw, A., Terranova, M., Dohme-Meier, F., Münger, A., Eggerschwiler, L., Bapst, B., Wegmann, S., Clauss, M. & Kreuzer, M. 2020. Immediate effect of *Acacia mearnsii* tannins on methane emissions and milk fatty acid profiles of dairy cows. *Animal Feed Science and Technology*, 261: 114388. https://doi.org/10.1016/j.anifeedsci.2019.114388

Deuber, O., Luderer, G. & Edenhofer, O. 2013. Physico-economic evaluation of climate metrics: A conceptual framework. *Environmental Science & Policy*, 29: 37–45. https://doi.org/10.1016/j.envsci.2013.01.018

Dhakal, S., Minx, J.C. & Toth, F.L. 2022. Emissions trends and drivers. In: P.R. Shukla, J. Skea, R. Slade, A. Al Khourdajie, R. van Diemen, D. McCollum, M. Pathak, S. Some, P. Vyas, R. Fradera, M. Belkacemi, A. Hasija, G. Lisboa, S. Luz & J. Malley, eds. *Climate change 2022: Mitigation of climate change. Contribution of Working Group III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 215–294. Cambridge, UK & New York, USA, Cambridge University Press.

Difford, G.F., Plichta, D.R., Løvendahl, P., Lassen, J., Noel, S.J., Højberg, O., Wright, A.D.G., Zhu, Z., Kristensen, L., Nielsen, H.B. & Guldbrandtsen, B. 2018. Host genetics and the rumen microbiome jointly associate with methane emissions in dairy cows. *PLOS Genetics*, 14(10): e1007580. https://doi.org/10.1371/journal.pgen.1007580

Dijkstra, J., Bannink, A., France, J., Kebreab, E. & van Gastelen, S. 2018. Short communication: Antimethanogenic effects of 3-nitrooxypropanol depend on supplementation dose, dietary fiber content, and cattle type. *Journal of Dairy Science*, 101(10): 9041–9047. https://doi.org/10.3168/jds.2018-14456

Dijkstra, J., Neal, H.D.St.C., Beever, D.E. & France, J. 1992. Simulation of nutrient digestion, absorption and outflow in the rumen: Model description. *The Journal of Nutrition*, 122(11): 2239–2256. https://doi.org/10.1093/jn/122.11.2239

Dillon, J.A., Stackhouse-Lawson, K.R., Thoma, G.J., Gunter, S.A., Rotz, C.A., Kebreab, E., Riley, D.G., Tedeschi, L.O., Villalba, J., Mitloehner, F., Hristov, A.N. , Archibeque, S.L., Ritten, J.P. & Mueller, ND. 2021. Current state of enteric methane and the carbon footprint of beef and dairy cattle in the United States. *Animal Frontiers*, 11(4): 57–68. https://doi.org/10.1093/af/vfab043

Dini, Y., Gere, J., Briano, C., Manetti, M., Juliarena, P., Picasso, V., Gratton, R. & Astigarraga, L. 2012. Methane emission and milk production of dairy cows grazing pastures rich in legumes or rich in grasses in Uruguay. *Animals*, 2(2): 288–300. https://doi.org/10.3390/ani2020288

Donoghue, K.A., Bird-Gardiner, T., Arthur, P.F., Herd, R.M. & Hegarty, R.F. 2016. Genetic and phenotypic variance and covariance components for methane emission and postweaning traits in Angus cattle. *Journal of Animal Science*, 94(4): 1438–1445. https://doi.org/10.2527/jas.2015-0065

Doreau, M., Arbre, M., Popova, M., Rochette, Y. & Martin, C. 2018. Linseed plus nitrate in the diet for fattening bulls: Effects on methane emission, animal health and residues in offal. *Animal*, 12(3): 501–507. https://doi.org/10.1017/S1751731117002014

Dorman, H.J. & Deans, S.G. 2000. Antimicrobial agents from plants: Antibacterial activity of plant volatile oils. *Journal of Applied Microbiology*, 88(2): 308–316. https://doi.org/10.1046/j.1365-2672.2000.00969.x

Drewnoski, M.E., Pogge, D.J. & Hansen, S.L. 2014. High-sulfur in beef cattle diets: A review. *Journal of Animal Science*, 92(9): 3763–3780. https://doi.org/10.2527/jas.2013-7242

Du, C., Abdullah, J.J., Greetham, D., Fu, D., Yu, M., Ren, L., Li, S. & Lu, D. 2018. Valorization of food waste into biofertiliser and its field application. *Journal of Cleaner Production*, 187: 273–284. https://doi.org/10.1016/j.jclepro.2018.03.211

Dubois, B., Tomkins, N.W., Kinley, R.D., Bai, M., Seymour, S., Paul, N.A., & Nys, R.D. 2013. Effect of tropical algae as additives on rumen *in vitro* gas production and fermentation characteristics. *American Journal of Plant Sciences*, 04(12): 34–43. https://doi.org/10.4236/ajps.2013.412A2005

Dudley, Q.M., Liska, A.J., Watson, A.K. & Erickson, G.E. 2014. Uncertainties in life cycle greenhouse gas emissions from U.S. beef cattle. *Journal of Cleaner Production*, 75: 31–39. https://doi.org/10.1016/j.jclepro.2014.03.087

Duffield, T.F. & Bagg, R.N. 2000. Use of ionophores in lactating dairy cattle: A review. *The Canadian Veterinary Journal – La Revue vétérinaire canadienne*, 41(5): 388–394.

Duffield, T.F., Rabiee, A.R. & Lean, I.J. 2008a. A meta-analysis of the impact of monensin in lactating dairy cattle. Part 1. Metabolic effects. *Journal of Dairy Science*, 91(4): 1334–1346. https://doi.org/10.3168/jds.2007-0607

Duffield, T.F., Rabiee, A.R. & Lean, I.J. 2008b. A meta-analysis of the impact of monensin in lactating dairy cattle. Part 2. Production effects. *Journal of Dairy Science*, 91(4): 1347–1360. https://doi.org/10.3168/jds.2007-0608

Duin, E.C., Wagner, T., Shima, S., Prakash, D., Cronin, B., Yáñez-Ruiz, D.R., Duval, S. & Dunfield, P.F. 2007. The soil methane sink. In: D.S. Reay, C.N. Hewitt, K. Smith & J. Grace, eds. *Greenhouse gas sinks*, pp. 152–170. Wallingford, UK, CABI. https://doi.org/10.1079/9781845931896.0152

Duin, E.C., Wagner, T., Shima, S., Prakash, D., Cronin, B., Yáñez-Ruiz, D.R., Duval, S., Rümbeli, R., Stemmler, R.T., Thauer, R.K. & Kindermann, M. 2016. Mode of action uncovered for the specific reduction of methane emissions from ruminants by the small molecule 3-nitrooxypropanol. *Proceedings of the National Academy of Sciences*, 113(22): 6172–6177. https://doi.org/10.1073/pnas.1600298113

Dunfield, P.F., Yuryev, A., Senin, P., Smirnova, A.V., Stott, M.B., Hou, S., Ly, B., Saw, J.H., Zhou, Z., Ren, Y., Wang, J., Mountain, B.W., Crowe, M.A., Weatherby, T.M., Bodelier, P.L.E., Liesack, W., Feng, L., Wang, L. & Alam, M. 2007. Methane oxidation by an extremely acidophilic bacterium of the phylum Verrucomicrobia. *Nature*, 450(7171): 879–882. https://doi.org/10.1038/nature06411

Dürr, J.W., Cue, R.I., Monardes, H.G., Moro-Méndez, J. & Wade, K.M. 2008. Milk losses associated with somatic cell counts per breed, parity and stage of lactation in Canadian dairy cattle. *Livestock Science*, 117(2–3): 225–232. https://doi.org/10.1016/j.livsci.2007.12.004

Dutaur, L. & Verchot, L.V. 2007. A global inventory of the soil CH_4 sink. *Global Biogeochemical Cycles*, 21(4): n/a-n/a. https://doi.org/10.1029/2006GB002734

Duval, B.D., Aguerre, M., Wattiaux, M., Vadas, P.A. & Powell, J.M. 2016. Potential for reducing on-farm greenhouse gas and ammonia emissions from dairy cows with prolonged dietary tannin additions. *Water, Air, & Soil Pollution*, 227(9): 329. https://doi.org/10.1007/s11270-016-2997-6

Duval, S. & Kindermann, M. 2012. Use of nitrooxy molecules in feed for reducing methane emission in ruminants, and/or to improve ruminant performance. World Intellectual Property Organization. No. 084629, filed 20 December 2011, and issued 28 June 2012. https://patents.google.com/patent/WO2012084629A1/en

Duxbury, J.M. & Mosier, A.R. 1993. Status and issues concerning agricultural emissions of greenhouse gases. In: H.M. Kaiser & T.E. Drennen, eds. *Agricultural Dimensions of Global Climate Change*, pp. 229–258. Delray Beach, USA, St. Lucie Press.

Ebert, P.J., Bailey, E.A., Shreck, A.L., Jennings, J.S. & Cole, N.A. 2017. Effect of condensed tannin extract supplementation on growth performance, nitrogen balance, gas emissions, and energetic losses of beef steers. *Journal of Animal Science*, 95(3): 1345–1355. https://doi.org/10.2527/jas.2016.0341

Ebrahimi, S.H., Mohini, M., Singhal, K.K., Heidarian Miri, V. & Tyagi, A.K. 2011. Evaluation of complementary effects of 9,10-anthraquinone and fumaric acid on methanogenesis and ruminal fermentation *in vitro*. *Archives of Animal Nutrition*, 65(4): 267–277. https://doi.org/10.1080/1745039X.2011.594345

Edouard, N., Charpiot, A., Robin, P., Lorinquer, E., Dollé, J.-B. & Faverdin, P. 2019. Influence of diet and manure management on ammonia and greenhouse gas emissions from dairy barns. *Animal*, 13(12): 2903–2912. https://doi.org/10.1017/S1751731119001368

Edouard, N., Mosquera, J., van Dooren, H.J.C., Mendes, L.B. & Ogink, N.W.M. 2016. Comparison of CO_2- and SF_6-based tracer gas methods for the estimation of ventilation rates in a naturally ventilated dairy barn. *Biosystems Engineering*, 149: 11–23. https://doi.org/10.1016/j.biosystemseng.2016.06.001

Eger, M., Graz, M., Riede, S. & Breves, G. 2018. Application of Mootral^{TM} reduces methane production by altering the Archaea community in the rumen simulation technique. *Frontiers in Microbiology*, 9: 2094. https://doi.org/10.3389/fmicb.2018.02094

Ehrhardt, F., Soussana, J.-F., Bellocchi, G., Grace, P., McAuliffe, R., Recous, S., Sándor, R., Smith, P., Snow, V., de Antoni Migliorati, M. & Basso, B. 2018. Assessing uncertainties in crop and pasture ensemble model simulations of productivity and N_2O emissions. *Global Change Biology*, 24(2): e603–e616. https://doi.org/10.1111/gcb.13965

Ekeberg, D., Ogner, G., Fongen, M., Joner, E.J. & Wickstrøm, T. 2004. Determination of CH_4, CO_2 and N_2O in air samples and soil atmosphere by gas chromatography mass spectrometry, GC-MS. *Journal of Environmental Monitoring*, 6(7): 621–623. https://doi.org/10.1039/B401315H

Ekvall, T. & Weidema, B.P. 2004. System boundaries and input data in consequential life cycle inventory analysis. *The International Journal of Life Cycle Assessment*, 9(3): 161–171. https://doi.org/10.1007/BF02994190

Elghandour, M.M.Y., Salem, A.Z.M., Castañeda, J.S.M., Camacho, L.M., Kholif, A.E. & Chagoyán, J.C.V. 2015. Direct-fed microbes: A tool for improving the utilization of low quality roughages in ruminants. *Journal of Integrative Agriculture*, 14(3): 526–533. https://doi.org/10.1016/S2095-3119(14)60834-0

Ellis, J.L., Dijkstra, J., France, J., Parsons, A.J., Edwards, G.R., Rasmussen, S., Kebreab, E. & Bannink, A. 2012. Effect of high-sugar grasses on methane emissions simulated using a dynamic model. *Journal of Dairy Science*, 95(1): 272–285. https://doi.org/10.3168/jds.2011-4385

Ellis, J.L., Dijkstra, J., Kebreab, E., Archibeque, S., France, J. & Bannink, A. 2010. Prediction of methane production in beef cattle within a mechanistic digestion model. In: D. Sauvant, J. Milgen, P. Faverdin & N. Friggens, eds. *Modelling nutrient digestion and utilisation in farm animals*, pp. 181–188. Wageningen, Kingdom of the Netherlands, Wageningen Academic Publishers.

Ellis, K.J. & Morrison, J.F. 1975. A problem encountered in a study of the effects of lanthanide ions on enzyme-catalyzed reactions. *Analytical Biochemistry*, 68(2): 429–435. https://doi.org/10.1016/0003-2697(75)90638-7

EPA (US Environmental Protection Agency). 2006. *Global anthropogenic non-CO_2 greenhouse gas emissions: 1990-2020*. Washington, DC, EPA.

EPA. 2000. Bromoform. www.epa.gov/sites/default/files/2016-09/documents/bromoform.pdf

Errickson, F.C., Keller, K., Collins, W.D., Srikrishnan, V. & Anthoff, D. 2021. Equity is more important for the social cost of methane than climate uncertainty. *Nature*, 592(7855): 564–570. https://doi.org/10.1038/s41586-021-03386-6

Escobar-Bahamondes, P., Oba, M., Kröbel, R., McAllister, T.A., MacDonald, D. & Beauchemin, K.A. 2017. Estimating enteric methane production for beef cattle using empirical prediction models compared with IPCC Tier 2 methodology. *Canadian Journal of Animal Science*, 97(4): 599–612. https://doi.org/10.1139/cjas-2016-0163

Eugène, M., Archimède, H. & Sauvant, D. 2004. Quantitative meta-analysis on the effects of defaunation of the rumen on growth, intake and digestion in ruminants. *Livestock Production Science*, 85(1): 81–97. https://doi.org/10.1016/S0301-6226(03)00117-9

Eugène, M., Massé, D., Chiquette, J. & Benchaar, C. 2008. Meta-analysis on the effects of lipid supplementation on methane production in lactating dairy cows. *Canadian Journal of Animal Science*, 88(2): 331–337. https://doi.org/10.4141/CJAS07112

Eugène, M., Sauvant, D., Nozière, P., Viallard, D., Oueslati, K., Lherm, M., Mathias, E. & Doreau, M. 2019. A new Tier 3 method to calculate methane emission inventory for ruminants. *Journal of Environmental Management*, 231: 982–988. https://doi.org/10.1016/j.jenvman.2018.10.086

Eugster, W. & Merbold, L. 2015. Eddy covariance for quantifying trace gas fluxes from soils. *SOIL*, 1(1): 187–205. https://doi.org/10.5194/soil-1-187-2015

European Commission, Joint Research Centre & IES (Institute for Environment and Sustainability). 2010. International Reference Life Cycle Data System (ILCD) Handbook: General guide for Life Cycle Assessment – Detailed guidance. Luxembourg, Publications Office.

Eurostat. 2018. Archive: Agri-environmental indicator – Greenhouse gas emissions. Accessed from https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Archive:Agri-environmental_indicator_-_greenhouse_gas_emissions&oldid=374004#Methane_emissions_from_the_EU_agricultural_sector

Evans, B. 2018. The role ensiled forage has on methane production in the rumen. *Animal Husbandry, Dairy and Veterinary Science*, 2(4): n/a-n/a. https://doi.org/10.15761/AHDVS.1000143

Fangueiro, D., Pereira, J.L.S., Macedo, S., Trindade, H., Vasconcelos, E. & Coutinho, J. 2017. Surface application of acidified cattle slurry compared to slurry injection: Impact on NH_3, N_2O, CO_2 and CH_4 emissions and crop uptake. *Geoderma*, 306: 160–166. https://doi.org/10.1016/j.geoderma.2017.07.023

Fankhauser, S. 1994. The social costs of greenhouse gas emissions: An expected value approach. *The Energy Journal*, 15(2): 157–184. https://econpapers.repec.org/article/aenjournl/1994v15-02-a09.htm

FAO (Food and Agriculture Organization of the United Nations). 2016a. *Greenhouse gas emissions and fossil energy use from small ruminant supply chains: Guidelines for assessment*. Livestock Environmental Assessment and Performance Partnership. Rome. www.fao.org/3/a-i6434e.pdf

FAO. 2016b. *Environmental performance of large ruminant supply chains: Guidelines for assessment*. Livestock Environmental Assessment and Performance Partnership. Rome. www.fao.org/3/i6494e/i6494e.pdf

FAO. 2016c. *Environmental performance of animal feeds supply chains: Guidelines for assessment*. Livestock Environmental Assessment and Performance Partnership. Rome. www.fao.org/3/a-i6433e.pdf

FAO. 2016d. *Environmental performance of animal feeds supply chains: Guidelines for assessment*. Livestock Environmental Assessment and Performance Partnership. Rome. www.fao.org/3/i6433e/i6433e.pdf

FAO. 2018a. *Environmental performance of pig supply chains: Guidelines for assessment.* Livestock Environmental Assessment and Performance Partnership. Rome. www.fao.org/3/I8686EN/i8686en.pdf

FAO. 2018b. *Nutrient flows and associated environmental impacts in livestock supply chains. Guidelines for assessment*. Rome. www.fao.org/3/CA1328EN/ca1328en.pdf

FAO. 2019. *The State of Food and Agriculture 2019. Moving forward on food loss and waste reduction.* Rome. www.fao.org/3/ca6030en/ca6030en.pdf

FAO. 2020. *The State of Food and Agriculture 2020. Overcoming water challenges in agriculture*. Rome. https://doi.org/10.4060/cb1447en

FAO & IDF (International Dairy Federation). 2011. *Guide to good dairy farming practice*. Rome. www.fao.org/3/ba0027e/ba0027e00.pdf

FAOSTAT. 2017. *Food and agriculture data*. Statistics Division. Rome. [Cited 30 November 2019]. http://faostat.fao.org/site/291/default.aspx

Farquharson, D., Jaramillo, P., Schivley, G., Klima, K., Carlson, D. & Samaras, C. 2017. Beyond global warming potential: A comparative application of climate impact metrics for the Life Cycle Assessment of coal and natural gas based electricity. *Journal of Industrial Ecology*, 21(4): 857–873. https://doi.org/10.1111/jiec.12475

Feng, X.Y., Dijkstra, J., Bannink, A., van Gastelen, S., France, J. & Kebreab, E. 2020. Antimethanogenic effects of nitrate supplementation in cattle: A meta-analysis. *Journal of Dairy Science*, 103(12): 11375–11385. https://doi.org/10.3168/jds.2020-18541

Feng, X.Y. & Kebreab, E. 2020. Net reductions in greenhouse gas emissions from feed additive use in California dairy cattle. *PLoS ONE*, 15(9). https://doi.org/10.1371/journal.pone.0234289

Ferraretto, L.F. & Shaver, R.D. 2015. Effects of whole-plant corn silage hybrid type on intake, digestion, ruminal fermentation, and lactation performance by dairy cows through a meta-analysis. *Journal of Dairy Science*, 98(4): 2662–2675. https://doi.org/10.3168/jds.2014-9045

Ferry, J.G. 1999. Enzymology of one-carbon metabolism in methanogenic pathways. *FEMS Microbiology Reviews*, 23(1): 13–38. https://doi.org/10.1111/j.1574-6976.1999.tb00390.x

Ferry, J.G. 2011. Fundamentals of methanogenic pathways that are key to the biomethanation of complex biomass. *Current Opinion in Biotechnology*, 22(3): 351–357. https://doi.org/10.1016/j.copbio.2011.04.011

Ferry, J.G. 2015. Acetate metabolism in anaerobes from the domain *Archaea*. *Life*, 5(2): 1454–1471. https://doi.org/10.3390/life5021454

Fievez, V., Dohme, F., Danneels, M., Raes, K. & Demeyer, D. 2003. Fish oils as potent rumen methane inhibitors and associated effects on rumen fermentation *in vitro* and *in vivo*. *Animal Feed Science and Technology*, 104(1–4): 41–58. https://doi.org/10.1016/S0377-8401(02)00330-9

Finlay, B.J., Esteban, G., Clarke, K.J., Williams, A.G., Embley, T.M. & Hirt, R.P. 1994. Some rumen ciliates have endosymbiotic methanogens. *FEMS Microbiology Letters*, 117(2): 157–161. https://doi.org/10.1111/j.1574-6968.1994.tb06758.x

Firkins, J.L. & Mackie, R.I. 2020. Ruminal protein breakdown and ammonia assimilation. In: C.S. McSweeney and R.I. Mackie, eds. *Improving rumen function*, pp. 383–419. Cambridge, UK, Burleigh Dodds Science Publishing. https://doi.org/10.19103/AS.2020.0067.13

Firkins, J.L., Yu, Z., Park, T. & Plank, J.E. 2020. Extending Burk Dehority's perspectives on the role of ciliate protozoa in the rumen. *Frontiers in Microbiology*, 11: 123. https://doi.org/10.3389/fmicb.2020.00123

Flay, H.E., Kuhn-Sherlock, B., Macdonald, K.A., Camara, M., Lopez-Villalobos, N., Donaghy, D.J. & Roche, J.R. 2019. Hot topic: Selecting cattle for low residual feed intake did not affect daily methane production but increased methane yield. *Journal of Dairy Science*, 102(3): 2708–2713. https://doi.org/10.3168/jds.2018-15234

Flowers, G., Ibrahim, S.A. & AbuGhazaleh, A.A. 2008. Milk fatty acid composition of grazing dairy cows when supplemented with linseed oil. *Journal of Dairy Science*, 91(2): 722–730. https://doi.org/10.3168/jds.2007-0410

Flysjö, A., Henriksson, M., Cederberg, C., Ledgard, S. & Englund, J.-E. 2011. The impact of various parameters on the carbon footprint of milk production in New Zealand and Sweden. *Agricultural Systems*, 104(6): 459–469. https://doi.org/10.1016/j.agsy.2011.03.003

Fonty, G., Joblin, K., Chavarot, M., Roux, R., Naylor, G. & Michallon, F. 2007. Establishment and development of ruminal hydrogenotrophs in methanogen-free lambs. *Applied and Environmental Microbiology*, 73(20): 6391–6403. https://doi.org/10.1128/AEM.00181-07

Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M. & Zhang, H. 2021. The Earth's energy budget, climate feedbacks, and climate sensitivity. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu & B. Zhou, eds. *Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 923–1054. Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157896.001

Foskolos, A. & Moorby, Jonathan. 2017. The use of high sugar grasses as a strategy to improve nitrogen utilization efficiency: A meta-analysis. In: *Advances in Animal Biosciences*, 8(1): 1–131. https://linkinghub.elsevier.com/retrieve/pii/S2040470017001479

Fox, D.G., Tedeschi, L.O., Tylutki, T.P., Russell, J.B., Van Amburgh, M.E., Chase, L.E., Pell, A.N. & Overton, T.R. 2004. The Cornell Net Carbohydrate and Protein System model for evaluating herd nutrition and nutrient excretion. *Animal Feed Science and Technology*, 112(1): 29–78. https://doi.org/10.1016/j.anifeedsci.2003.10.006

Fraga-Corral, M., García-Oliveira, P., Pereira, A.G., Lourenço-Lopes, C., Jimenez-Lopez, C., Prieto, M.A. & Simal-Gandara, J. 2020. Technological application of tannin-based extracts. *Molecules*, 25(3): 614. https://doi.org/10.3390/molecules25030614

France, J. & Kebreab, E. 2008. *Mathematical modelling in animal nutrition*. Wallingford, UK, CABI Publishing.

Frankenberg, C., Meirink, J.F., Bergamaschi, P., Goede, A.P.H., Heimann, M., Körner, S., Platt, U., van Weele, M. & Wagner, T. 2006. Satellite chartography of atmospheric methane from SCIAMACHY on board ENVISAT: Analysis of the years 2003 and 2004. *Journal of Geophysical Research: Atmospheres*, 111(D7). https://doi.org/10.1029/2005JD006235

Franzolin, R. & Dehority, B.A. 2010. The role of pH on the survival of rumen protozoa in steers. *Revista Brasileira de Zootecnia*, 39: 2262–2267. https://doi.org/10.1590/S1516-35982010001000023

Fraser, M.D., Fleming, H.R., Theobald, V.J. & Moorby, J.M. 2015. Effect of breed and pasture type on methane emissions from weaned lambs offered fresh forage. *The Journal of Agricultural Science*, 153(6): 1128–1134. https://doi.org/10.1017/S0021859615000544

Freetly, H.C. & Brown-Brandl, T.M. 2013. Enteric methane production from beef cattle that vary in feed efficiency. *Journal of Animal Science*, 91(10): 4826–4831. https://doi.org/10.2527/jas.2011-4781

Frey, M. 2002. Hydrogenases: Hydrogen-activating enzymes. *ChemBioChem*, 3(2–3): 153–160. https://doi.org/10.1002/1439-7633(20020301)3:2/3<153::AID-CBIC153>3.0.CO;2-B

Frischknecht, R., Fantke, P., Tschümperlin, L., Niero, M., Antón, A., Bare, J., Boulay, A.-M., Cherubini, F., Hauschild, M.Z., Henderson, A., Levasseur, A., McKone, T.E., Michelsen, O., Canals, L.M., Pfister, S., Ridoutt, B., Rosenbaum, R.K., Verones, F., Vigon, B. & Jolliet, O. 2016. Global guidance on environmental life cycle impact assessment indicators: Progress and case study. *The International Journal of Life Cycle Assessment*, 21(3): 429–442. https://doi.org/10.1007/s11367-015-1025-1

Frith, O., Wassmann, R. & Sander, B.S. 2021. How Asia's rice producers can help limit global warming. *The Diplomat*, 13 October 2021. Washington, DC. [Cited 2 June 2023]. https://thediplomat.com/2021/10/how-asias-rice-producers-can-help-limit-global-warming/

Frutos, P., Hervás, G., Natalello, A., Luciano, G., Fondevila, M., Priolo, A. & Toral, P.G. 2020. Ability of tannins to modulate ruminal lipid metabolism and milk and meat fatty acid profiles. *Animal Feed Science and Technology*, 269: 114623. https://doi.org/10.1016/j.anifeedsci.2020.114623

Fuglestvedt, J., Rogelj, J., Millar, R.J., Allen, M., Boucher, O., Cain, M., Forster, P.M., Kriegler, E. & Shindell, D. 2018. Implications of possible interpretations of 'greenhouse gas balance' in the Paris Agreement. *Philosophical Transactions of the Royal Society. Series A – Mathematical, Physical and Engineering Sciences*, 376(2119): 20160445. https://doi.org/10.1098/rsta.2016.0445

Fuglestvedt, J.S., Shine, K.P., Berntsen, T., Cook, J., Lee, D.S., Stenke, A., Skeie, R.B., Velders, G.J.M. & Waitz, I.A. 2010. Transport impacts on atmosphere and climate: Metrics. *Atmospheric Environment*, 44(37): 4648–4677. https://doi.org/10.1016/j.atmosenv.2009.04.044

Furman, O., Shenhav, L., Sasson, G., Kokou, F., Honig, H., Jacoby, S., Hertz, T., Cordero, O.X., Halperin, E. & Mizrahi, I. 2020. Stochasticity constrained by deterministic effects of diet and age drive rumen microbiome assembly dynamics. *Nature Communications*, 11(1): 1904. https://doi.org/10.1038/s41467-020-15652-8

Gadde, B., Menke, C. & Wassmann, R. 2009. Rice straw as a renewable energy source in India, Thailand, and the Philippines: Overall potential and limitations for energy contribution and greenhouse gas mitigation. *Biomass and Bioenergy*, 33(11): 1532–1546. https://doi.org/10.1016/j.biombioe.2009.07.018

Galassi, G., Malagutti, L., Colombini, S., Rapetti, L. & Crovetto, G.M. 2011. Effects of benzoic acid on nitrogen, phosphorus and energy balance and on ammonia emission from slurries in the heavy pig. *Italian Journal of Animal Science*, 10(3): e38. https://doi.org/10.4081/ijas.2011.e38

Galyean, M.L. & Owens, F.N. 1991. Effects of diet composition and level of intake on site and extent of digestion in ruminants. In: T. Tsuda, Y. Sasaki & R. Kawashima, eds. *Physiological aspects of digestion and metabolism in ruminants*, pp. 483–514. New York, USA, Academic Press.

García-Chávez, I., Meraz-Romero, E., Castelán-Ortega, O., Zaragoza Esparza, J., Osorio Avalos, J., Robles Jimenez, L.E. & González-Ronquillo, M. 2020. *Corn silage, meta-analysis of the quality and yield of different regions in the world*. Preprints: 2020100094. https://doi.org/10.20944/preprints202010.0094.v1

Garcia-Lopez, P.M., Kung, L. & Odom, J.M. 1996. *In vitro* inhibition of microbial methane production by 9,10-anthraquinone. *Journal of Animal Science*, 74(9): 2276. https://doi.org/10.2527/1996.7492276x

Gasser, T., Peters, G.P., Fuglestvedt, J.S., Collins, W.J., Shindell, D.T. & Ciais, P. 2017. Accounting for the climate–carbon feedback in emission metrics. *Earth System Dynamics*, 8(2): 235–253. https://doi.org/10.5194/esd-8-235-2017

Gates, R.S., Casey, K.D., Xin, H. & Burns, R.T. 2009. Building emissions uncertainty estimates. *Transactions of the ASABE*, 52(4): 1345–1351. https://doi.org/10.13031/2013.27784

Gates, R.S., Casey, K.D., Xin, H., Wheeler, E.F. & Simmons, J.D. 2004. Fan assessment numeration system (FANS) design and calibration specifications. *Transactions of the ASAE*, 47(5): 1709–1715. https://doi.org/10.13031/2013.17613

Gates, R.S., Xin, H., Casey, K.D., Liang, Y. & Wheeler, E.F. 2005. Method for measuring ammonia emissions from poultry houses. *Journal of Applied Poultry Research*, 14(3): 622–634. https://doi.org/10.1093/japr/14.3.622

Gatica, G., Fernández, M.E., Juliarena, M.P. & Gyenge, J. 2020. Environmental and anthropogenic drivers of soil methane fluxes in forests: Global patterns and among-biomes differences. *Global Change Biology*, 26(11): 6604–6615.

Gaviria-Uribe, X., Bolivar, D.M., Rosenstock, T.S., Molina-Botero, I.C., Chirinda, N., Barahona, R. & Arango, J. 2020. Nutritional quality, voluntary intake and enteric methane emissions of diets based on novel Cayman grass and its associations with two *Leucaena* shrub legumes. *Frontiers in Veterinary Science*, 7: 579189. https://doi.org/10.3389/fvets.2020.579189

Gavrilova, O., Leip, A., Dong, H., Macdonald, J.D., Gomez Bravo, C.A., Amon, B., Barahona Rosales, R., del Prado, A., de Lima, M.A., Oyhantçabal, W., van der Weerden, T.J. & Widiawati, Y. 2019. Emissions from livestock and manure management. In: E. Calvo Buendia, K. Tanabe, A. Kranjc, J. Baasansuren, M. Fukuda, S. Ngarize, A. Osako, Y. Pyroshenko, P. Shermanau, S. Federici, eds. *2019 Refinement to the 2006 guidelines for National Greenhouse Gas Inventories. Agriculture, forestry and other land use*. Vol. 4, chap. 10. Geneva, Switzerland, IPCC. www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch10_Livestock.pdf

Ge, H.-X., Zhang, H.-S., Zhang, H., Cai, X.-H., Song, Y. & Kang, L. 2018. The characteristics of methane flux from an irrigated rice farm in East China measured using the eddy covariance method. *Agricultural and Forest Meteorology*, 249: 228–238. https://doi.org/10.1016/j.agrformet.2017.11.010

Gerber, P.J., Hristov, A.N., Henderson, B., Makkar, H., Oh, J., Lee, C., Meinen, R., Montes, F., Ott, T., Firkins, J., Rotz, A., Dell, C., Adesogan, A.T., Yang, W.Z., Tricarico, J.M., Kebreab, E., Waghorn, G., Dijkstra, J. & Oosting, S. 2013b. Technical options for the mitigation of direct methane and nitrous oxide emissions from livestock: A review. *Animal*, 7: 220–234. https://doi.org/10.1017/S1751731113000876

Gerber, P.J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A. & Tempio, G. 2013a. *Tackling climate change through livestock – A global assessment of emissions and mitigation opportunities.* Rome, FAO. www.fao.org/3/i3437e/i3437e.pdf

Gerber, P.J., Vellinga, T., Opio, C. & Steinfeld, H. 2011. Productivity gains and greenhouse gas emissions intensity in dairy systems. *Livestock Science*, 139(1–2): 100–108. https://doi.org/10.1016/j.livsci.2011.03.012

Gilbert, R.A., Ouwerkerk, D., Zhang, L.H. & Klieve, A.V. 2010. *In vitro* detection and primary cultivation of bacteria producing materials inhibitory to ruminal methanogens. *Journal of Microbiological Methods*, 80(2): 217–218. https://doi.org/10.1016/j.mimet.2009.12.004

Gilbert, R.A., Townsend, E.M., Crew, K.S., Hitch, T.C.A., Friedersdorff, J.C.A., Creevey, C.J., Pope, P.B., Ouwerkerk, D. & Jameson, E. 2020. Rumen virus populations: Technological advances enhancing current understanding. *Frontiers in Microbiology*, 11: 450. https://doi.org/10.3389/fmicb.2020.00450

Gilhespy, S.L., Anthony, S., Cardenas, L., Chadwick, D., del Prado, A., Li, C., Misselbrook, T., Rees, R.M., Salas, W., Sanz-Cobena, A. & Smith, P. 2014. First 20 years of DNDC (DeNitrification DeComposition): Model evolution. *Ecological Modelling*, 292: 51–62. https://doi.org/10.1016/j.ecolmodel.2014.09.004

Gillett, N.P. & Matthews, H.D. 2010. Accounting for carbon cycle feedbacks in a comparison of the global warming effects of greenhouse gases. *Environmental Research Letters*, 5(3): 034011. https://doi.org/10.1088/1748-9326/5/3/034011

Giltrap, D.L., Li, C. & Saggar, S. 2010. DNDC: A process-based model of greenhouse gas fluxes from agricultural soils. *Agriculture, Ecosystems & Environment*, 136(3): 292–300. https://doi.org/10.1016/j.agee.2009.06.014

Gislon, G., Colombini, S., Borreani, G., Crovetto, G.M., Sandrucci, A., Galassi, G., Tabacco, E. & Rapetti, L. 2020. Milk production, methane emissions, nitrogen, and energy balance of cows fed diets based on different forage systems. *Journal of Dairy Science*, 103(9): 8048–8061. https://doi.org/10.3168/jds.2019-18134

Giuntoli, J., Agostini, A., Edwards, R. & Marelli, L. 2017. *Solid and gaseous bioenergy pathways: Input values and GHG emissions. Calculated according to the methodology set in COM(2016) 767. Version 2*. Luxembourg, Publications Office. https://data.europa.eu/doi/10.2790/27486

Gleason, C.B., Beckett, L.M. & White, R.R. 2022. Rumen fermentation and epithelial gene expression responses to diet ingredients designed to differ in ruminally degradable protein and fiber supplies. *Scientific Reports*, 12(1): 2933. https://doi.org/10.1038/s41598-022-06890-5

Global Agenda for Sustainable Livestock (GASL). 2021. *Global network on silvopastoral systems (GNSPS)*. www.livestockdialogue.org/action-networks/action-networks/global-network-on-silvopastoral-systems-gnsps/en

Glumb, R., Davis, G. & Lietzke, C. 2014. *The TANSO-FTS-2 instrument for the GOSAT-2 greenhouse gas monitoring mission*. In: *2014 IEEE Geoscience and Remote Sensing Symposium*, pp. 1238–1240. Quebec City, Canada, IEEE (The Institute of Electrical and Electronics Engineers). https://doi.org/10.1109/IGARSS.2014.6946656

Goel, G. & Makkar, H.P.S. 2012. Methane mitigation from ruminants using tannins and saponins. *Tropical Animal Health and Production*, 44(4): 729–739. https://doi.org/10.1007/s11250-011-9966-2

Golston, L.M., Pan, D., Sun, K., Tao, L., Zondlo, M.A., Eilerman, S.J., Peischl, J., Neuman, J.A. & Floerchinger, C. 2020. Variability of ammonia and methane emissions from animal feeding operations in Northeastern Colorado. *Environmental Science & Technology*, 54(18): 11015–11024. https://doi.org/10.1021/acs.est.0c00301

Goopy, J. 2019. Creating a low enteric methane emission ruminant: What is the evidence of success to the present and prospects for developing economies? *Animal Production Science*, 59. https://doi.org/10.1071/AN18457

Goopy, J.P., Chang, C. & Tomkins, N. 2016. A comparison of methodologies for measuring methane emissions from ruminants. In: T.S. Rosenstock, M.C. Rufino, K. Butterbach-Bahl, L. Wollenberg & M. Richards, eds. *Methods for measuring greenhouse gas balances and evaluating mitigation options in smallholder agriculture*, pp. 97–117. New York, USA, Springer Cham. https://doi.org/10.1007/978-3-319-29794-1_5

Goopy, J.P., Onyango, A.A., Dickhoefer, U. & Butterbach-Bahl, K. 2018. A new approach for improving emission factors for enteric methane emissions of cattle in smallholder systems of East Africa – Results for Nyando, Western Kenya. *Agricultural Systems*, 161: 72–80. https://doi.org/10.1016/j.agsy.2017.12.004

Goopy, J.P., Robinson, D.L., Woodgate, R.T., Donaldson, A.J., Oddy, V.H., Vercoe, P.E. & Hegarty, R.S. 2015. Estimates of repeatability and heritability of methane production in sheep using portable accumulation chambers. *Animal Production Science*, 56(1): 116–122. https://doi.org/10.1071/AN13370

Goopy, J.P., Woodgate, R., Donaldson, A., Robinson, D.L. & Hegarty, R.S. 2011. Validation of a short-term methane measurement using portable static chambers to estimate daily methane production in sheep. *Animal Feed Science and Technology*, 166–167: 219–226. https://doi.org/10.1016/j.anifeedsci.2011.04.012

Gordon, R., Jamieson, R., Rodd, V., Patterson, G. & Harz, T. 2001. Effects of surface manure application timing on ammonia volatilization. *Canadian Journal of Soil Science*, 81(4): 525–533. https://doi.org/10.4141/S00-092

Grainger, C. & Beauchemin, K.A. 2011. Can enteric methane emissions from ruminants be lowered without lowering their production? *Animal Feed Science and Technology*, 166–167: 308–320. https://doi.org/10.1016/j.anifeedsci.2011.04.021

Grainger, C., Williams, S.R.O., Clarke, T., Wright, A.-D. & Eckard, R. 2010. Supplementation with whole cottonseed causes long-term reduction of methane emissions from lactating dairy cows offered a forage and cereal grain diet. *Journal of Dairy Science*, 93(6): 2612–9. https://doi.org/10.3168/jds.2009-2888

Granja-Salcedo, Y.T., Fernandes, R.M., Araujo, R.C.D., Kishi, L.T., Berchielli, T.T., Resende, F.D.D., Berndt, A. & Siqueira, G.R. 2019. Long-term encapsulated nitrate supplementation modulates rumen microbial diversity and rumen fermentation to reduce methane emission in grazing steers. *Frontiers in Microbiology*, 10: 614. https://doi.org/10.3389/fmicb.2019.00614

Greening, C., Geier, R., Wang, C., Woods, L.C., Morales, S.E., McDonald, M.J., Rushton-Green, R., Morgan, X.C., Koike, S., Leahy, S.C., Kelly, W.J., Cann, I., Attwood, G.T., Cook, G.M. & Mackie, R.I. 2019. Diverse hydrogen production and consumption pathways influence methane production in ruminants. *The ISME Journal*, 13(10): 2617–2632. https://doi.org/10.1038/s41396-019-0464-2

Gregorini, P., Villalba, J.J., Chilibroste, P. & Provenza, F.D. 2017. Grazing management: Setting the table, designing the menu and influencing the diner. *Animal Production Science*, 57(7): 1248–1268. https://doi.org/10.1071/AN16637

Gruninger, R.J., Zhang, X.M., Smith, M.L., Kung, L., Vyas, D., McGinn, S.M., Kindermann, M., Wang, M., Liang, Z. & Beauchemin, K.A. 2022. Application of 3-nitrooxypropanol and canola oil to mitigate enteric methane emissions of beef cattle results in distinctly different effects on the rumen microbial community. *Animal Microbiome*, 4(1): 35. https://doi.org/10.1186/s42523-022-00179-8

Guanter, L., Irakulis-Loitxate, I., Gorroño, J., Sánchez-García, E., Cusworth, D.H., Varon, D.J., Cogliati, S. & Colombo, R. 2021. Mapping methane point emissions with the PRISMA spaceborne imaging spectrometer. *Remote Sensing of Environment*, 265: 112671. https://doi.org/10.1016/j.rse.2021.112671

Güçlü-Üstündağ, Ö. & Mazza, G. 2007. Saponins: Properties, applications and processing. *Critical Reviews in Food Science and Nutrition*, 47(3): 231–258. https://doi.org/10.1080/10408390600698197

Guingand, N., Demerson, L. & Broz, J. 2005. Influence of adding 0.5 or 1% of benzoic acid to the feed of growing-finishing pigs on ammonia emission and performance. In: A. Krynski & R. Wrzesień, eds. *Animals and environment, Volume 1: Proceedings of the XIIth ISAH Congress on Animal Hygiene, Warsaw, Poland, 4-8 September 2005*, pp. 360–363. Warsaw, Poland, ISAH (International Society for Animal Hygiene).

Gulledge, J. & Schimel, J.P. 1998. Low-concentration kinetics of atmospheric CH_4 oxidation in soil and mechanism of NH_4+ inhibition. *Applied and Environmental Microbiology*, 64(11): 4291–4298. www.ncbi.nlm.nih.gov/pmc/articles/PMC106641/

Gunsalus, R.P., Romesser, J.A. & Wolfe, R.S. 1978. Preparation of coenzyme M analogs and their activity in the methyl coenzyme M reductase system of *Methanobacterium thermoautotrophicum*. *Biochemistry*, 17(12): 2374–2377. https://doi.org/10.1021/bi00605a019

Gunter, S.A. & Bradford, J.A. 2017. Technical note: Effect of bait delivery interval in an automated head-chamber system on respiration gas estimates when cattle are grazing rangeland. *The Professional Animal Scientist*, 33(4): 490–497. https://doi.org/10.15232/pas.2016-01593

Guo, Y., Wang, Y., Chen, S., Zheng, S., Guo, C., Xue, D., Kuzyakov, Y. & Wang, Z.-L. 2019. Inventory of spatio-temporal methane emissions from livestock and poultry farming in Beijing. *Sustainability*, 11(14): 3858. https://doi.org/10.3390/su11143858

Gurwick, N.P., Moore, L.A., Kelly, C. & Elias, P. 2013. A systematic review of biochar research, with a focus on its stability *in situ* and its promise as a climate mitigation strategy. *PLoS ONE*, 8(9): e75932. https://doi.org/10.1371/journal.pone.0075932

Guyader, J., Doreau, M., Morgavi, D.P., Gérard, C., Loncke, C. & Martin, C. 2016a. Long-term effect of linseed plus nitrate fed to dairy cows on enteric methane emission and nitrate and nitrite residuals in milk. *Animal*, 10(7): 1173–1181. https://doi.org/10.1017/S1751731115002852

Guyader, J., Eugène, M., Meunier, B., Doreau, M., Morgavi, D.P., Silberberg, M., Rochette, Y., Gerard, C., Loncke, C. & Martin, C. 2015. Additive methane-mitigating effect between linseed oil and nitrate fed to cattle. *Journal of Animal Science*, 93(7): 3564–3577. https://doi.org/10.2527/jas.2014-8196

Guyader, J., Eugène, M., Nozière, P., Morgavi, D.P., Doreau, M. & Martin, C. 2014. Influence of rumen protozoa on methane emission in ruminants: A meta-analysis approach. *Animal*, 8(11): 1816–1825. https://doi.org/10.1017/S1751731114001852

Guyader, J., Janzen, H.H., Kroebel, R. & Beauchemin, K.A. 2016b. Forage use to improve environmental sustainability of ruminant production. *Journal of Animal Science*, 94(8): 3147–3158. https://doi.org/10.2527/jas.2015-0141

Guyader, J., Little, S., Kröbel, R., Benchaar, C. & Beauchemin, K.A. 2017. Comparison of greenhouse gas emissions from corn- and barley-based dairy production systems in Eastern Canada. *Agricultural Systems*, 152: 38–46. https://doi.org/10.1016/j.agsy.2016.12.002

Haas, E., Klatt, S., Fröhlich, A., Kraft, P., Werner, C., Kiese, R., Grote, R., Breuer, L. & Butterbach-Bahl, K. 2013. LandscapeDNDC: A process model for simulation of biosphere–atmosphere–hydrosphere exchange processes at site and regional scale. *Landscape Ecology*, 28(4): 615–636. https://doi.org/10.1007/s10980-012-9772-x

Hacker, J.M., Chen, D., Bai, M., Ewenz, C., Junkermann, W., Lieff, W., McManus, B., Neininger, B., Sun, J., Coates, T., Denmead, T., Flesch, T., McGinn, S. & Hill, J. 2016. Using airborne technology to quantify and apportion emissions of CH_4 and NH_3 from feedlots. *Animal Production Science*, 56(3): 190. https://doi.org/10.1071/AN15513

Haisan, J., Sun, Y., Guan, L.L., Beauchemin, K.A., Iwaasa, A., Duval, S., Barreda, D.R. & Oba, M. 2014. The effects of feeding 3-nitrooxypropanol on methane emissions and productivity of Holstein cows in mid lactation. *Journal of Dairy Science*, 97(5): 3110–3119. https://doi.org/10.3168/jds.2013-7834

Halas, D., Hansen, C.F., Hampson, D.J., Kim, J.-C., Mullan, B.P., Wilson, R.H. & Pluske, J.R. 2010. Effects of benzoic acid and inulin on ammonia–nitrogen excretion, plasma urea levels, and the pH in faeces and urine of weaner pigs. *Livestock Science*, 134(1–3): 243–245. https://doi.org/10.1016/j.livsci.2010.06.153

Halasa, T., Nielen, M., De Roos, A.P.W., Van Hoorne, R., de Jong, G., Lam, T.J.G.M., van Werven, T. & Hogeveen, H. 2009. Production loss due to new subclinical mastitis in Dutch dairy cows estimated with a test-day model. *Journal of Dairy Science*, 92(2): 599–606. https://doi.org/10.3168/jds.2008-1564

Hales, K.E., Cole, N.A. & MacDonald, J.C. 2012. Effects of corn processing method and dietary inclusion of wet distillers grains with solubles on energy metabolism, carbon-nitrogen balance, and methane emissions of cattle. *Journal of Animal Science*, 90(9): 3174–3185. https://doi.org/10.2527/jas.2011-4441

Hall, J.O. 2000. Ionophore use and toxicosis in cattle. *Veterinary Clinics of North America: Food Animal Practice*, 16(3): 497–509. https://doi.org/10.1016/S0749-0720(15)30083-9

Hall, M.K., Winters, A.J. & Rogers, G.S. 2014. Variations in the diurnal flux of greenhouse gases from soil and optimizing the sampling protocol for closed static chambers. *Communications in Soil Science and Plant Analysis*, 45(22): 2970–2978. https://doi.org/10.1080/00103624.2014.956937

Hammitt, J.K., Jain, A.K., Adams, J.L. & Wuebbles, D.J. 1996. A welfare-based index for assessing environmental effects of greenhouse-gas emissions. *Nature*, 381(6580): 301–303. https://doi.org/10.1038/381301a0

Hammond, K.J., Burke, J.L., Koolaard, J.P., Muetzel, S., Pinares-Patiño, C.S. & Waghorn, G.C. 2013. Effects of feed intake on enteric methane emissions from sheep fed fresh white clover (*Trifolium repens*) and perennial ryegrass (*Lolium perenne*) forages. *Animal Feed Science and Technology*, 179(1–4): 121–132. https://doi.org/10.1016/j.anifeedsci.2012.11.004

Hammond, K.J., Crompton, L.A., Bannink, A., Dijkstra, J., Yáñez-Ruiz, D.R., O'Kiely, P., Kebreab, E., Eugène, M.A., Yu, Z., Shingfield, K.J., Schwarm, A., Hristov, A.N. & Reynolds, C.K. 2016. Review of current *in vivo* measurement techniques for quantifying enteric methane emission from ruminants. *Animal Feed Science and Technology*, 219: 13–30. https://doi.org/10.1016/j.anifeedsci.2016.05.018

Hammond, K.J., Humphries, D.J., Crompton, L.A., Green, C. & Reynolds, C.K. 2015. Methane emissions from cattle: Estimates from short-term measurements using a GreenFeed system compared with measurements obtained using respiration chambers or sulphur hexafluoride tracer. *Animal Feed Science and Technology*, 203: 41–52. https://doi.org/10.1016/j.anifeedsci.2015.02.008

Hand, K.J., Godkin, A. & Kelton, D.F. 2012. Milk production and somatic cell counts: A cow-level analysis. *Journal of Dairy Science*, 95(3): 1358–1362. https://doi.org/10.3168/jds.2011-4927

Hao, H.-T., Karthikeyan, O. & Heimann, K. 2015. Bio-refining of carbohydrate-rich food waste for biofuels. *Energies*, 8(7): 6350–6364. https://doi.org/10.3390/en8076350

Harms, U. & Thauer, R.K. 1996. Methylcobalamin: Coenzyme M methyltransferase Isoenzymes MtaA and MtbA from *Methanosarcina barkeri*. Cloning, sequencing and differential transcription of the encoding genes, and functional pverexpression of the MtaA gene in *Escherichia coli*. *European Journal of Biochemistry*, 235(3): 653–659. https://doi.org/10.1111/j.1432-1033.1996.00653.x

Harmsen, M., van den Berg, M., Krey, V., Luderer, G., Marcucci, A., Strefler, J. & Vuuren, D. 2016. How climate metrics affect global mitigation strategies and costs: A multi-model study. *Climatic Change*, 136: 1–14. https://doi.org/10.1007/s10584-016-1603-7

Harrison, M.T., Cullen, B.R., Mayberry, D.E., Cowie, A.L., Bilotto, F., Badgery, W.B., Liu, K., Davison, T., Christie, K.M., Muleke, A., & Eckard, R.J. 2021. Carbon myopia: The urgent need for integrated social, economic and environmental action in the livestock sector. *Global Change Biology*, 27(22): 5726–5761. https://doi.org/10.1111/gcb.15816

Harvey, M.J., Sperlich, P., Clough, T.J., Kelliher, F.M., McGeough, K.L., Martin, R.J. & Moss, R. 2020. Global Research Alliance N_2O chamber methodology guidelines: Recommendations for air sample collection, storage, and analysis. *Journal of Environmental Quality*, 49(5): 1110–1125. https://doi.org/10.1002/jeq2.20129

Hassanat, F. & Benchaar, C. 2019. Methane emissions of manure from dairy cows fed red clover- or corn silage-based diets supplemented with linseed oil. *Journal of Dairy Science*, 102(12): 11766–11776. https://doi.org/10.3168/jds.2018-16014

Hassanat, F., Gervais, R., Julien, C., Massé, D.I., Lettat, A., Chouinard, P.Y., Petit, H.V. & Benchaar, C. 2013. Replacing alfalfa silage with corn silage in dairy cow diets: Effects on enteric methane production, ruminal fermentation, digestion, N balance, and milk production. *Journal of Dairy Science*, 96(7): 4553–4567. https://doi.org/10.3168/jds.2012-6480

Hassanat, F., Gervais, R., Massé, D.I., Petit, H.V. & Benchaar, C. 2014. Methane production, nutrient digestion, ruminal fermentation, N balance, and milk production of cows fed timothy silage- or alfalfa silage-based diets. *Journal of Dairy Science*, 97(10): 6463–6474. https://doi.org/10.3168/jds.2014-8069

Hassouna, M., Calvet, S., Hayes, E., Gates, R.S. & Schrade, S. 2021. Measurement of gaseous emissions from animal housing. In: N.M. Holden, M.L. Wolfe, J.A. Ogejo & E.J. Cummins, eds. *Introduction to biosystems engineering*, pp. 1–21. Blacksburg, USA, Virginia Tech Publishing. https://doi.org/10.21061/IntroBiosystemsEngineering

Hassouna, M. & Eglin, T. 2016. Measuring emissions from livestock farming: Greenhouse gases, ammonia and nitrogen oxides. Paris, Ademe and INRA. www6.inrae.fr/animal_emissions_eng/News/Measuring-gaseous-emissions-from-animal-farms

Hassouna, M., Robin, P., Brachet, A., Paillat, J.M., Dollé, J.B. & Faverdin, P. 2010. Development and validation of a simplified method to quantify gaseous emissions from cattle buidings. In: *17th World Congress of the International Commission of Agriculture Engineering (CIGR). Symposium on Nanotechnologies Applied to Biosystems Engineering and the Environment, Quebec City, Canada, 13-17 June 2010.* [Cited 15 June 2023]. https://publications.cirad.fr/une_notice.php?dk=556757

Hassouna, M., Robin, P., Charpiot, A., Edouard, N. & Méda, B. 2013. Infrared photoacoustic spectroscopy in animal houses: Effect of non-compensated interferences on ammonia, nitrous oxide and methane air concentrations. *Biosystems Engineering*, 114(3): 318–326. https://doi.org/10.1016/j.biosystemseng.2012.12.011

Hawkins, J., Weersink, A., Wagner-Riddle, C. & Fox, G. 2015. Optimizing ration formulation as a strategy for greenhouse gas mitigation in intensive dairy production systems. *Agricultural Systems*, 137: 1–11. https://doi.org/10.1016/j.agsy.2015.03.007

Hayek, M.N., Harwatt, H., Ripple, W.J. & Mueller, N.D. 2021. The carbon opportunity cost of animal-sourced food production on land. *Nature Sustainability*, 4(1): 21–24. https://doi.org/10.1038/s41893-020-00603-4

Hayes, B.J., Lewin, H.A. & Goddard, M.E. 2013. The future of livestock breeding: Genomic selection for efficiency, reduced emissions intensity, and adaptation. *Trends in Genetics*, 29(4): 206–214. https://doi.org/10.1016/j.tig.2012.11.009

He, W., Dutta, B., Grant, B.B., Chantigny, M.H., Hunt, D., Bittman, S., Tenuta, M., Worth, D., VanderZaag, A., Desjardins, R.L. & Smith, W.N. 2020. Assessing the effects of manure application rate and timing on nitrous oxide emissions from managed grasslands under contrasting climate in Canada. *Science of the Total Environment*, 716: 135374. https://doi.org/10.1016/j.scitotenv.2019.135374

Hedley, C.B., Saggar, S. & Tate, K.R. 2006. Procedure for fast simultaneous analysis of the greenhouse gases: Methane, carbon dioxide, and nitrous oxide in air samples. *Communications in Soil Science and Plant Analysis*, 37(11–12): 1501–1510. https://doi.org/10.1080/00103620600709928

Hegarty, R. & Gerdes, R. 1998. Hydrogen production and transfer in the rumen. *Recent Advances in Animal Nutrition in Australia*, 12: 37–44.

Hegarty, R.S. 1999. Reducing rumen methane emissions through elimination of rumen protozoa. *Australian Journal of Agricultural Research*, 50(8): 1321. https://doi.org/10.1071/AR99008

Helander, I.M., Alakomi, H.-L., Latva-Kala, K., Mattila-Sandholm, T., Pol, I., Smid, E.J., Gorris, L.G.M. & von Wright, A. 1998. Characterization of the action of selected essential oil components on gram-negative bacteria. *Journal of Agricultural and Food Chemistry*, 46(9): 3590–3595. https://doi.org/10.1021/jf980154m

Hellwing, A.L.F., Lund, P., Weisbjerg, M.R., Brask, M. & Hvelplund, T. 2012. Technical note: Test of a low-cost and animal-friendly system for measuring methane emissions from dairy cows. *Journal of Dairy Science*, 95(10): 6077–6085. https://doi.org/10.3168/jds.2012-5505

Henckel, T., Jäckel, U., Schnell, S. & Conrad, R. 2000. Molecular analyses of novel methanotrophic communities in forest soil that oxidize atmospheric methane. *Applied and Environmental Microbiology*, 66(5): 1801–1808. https://doi.org/10.1128/AEM.66.5.1801-1808.2000

Herd, R.M., Arthur, P.F., Donoghue, K.A., Bird, S.H., Bird-Gardiner, T. & Hegarty, R.S. 2014. Measures of methane production and their phenotypic relationships with dry matter intake, growth, and body composition traits in beef cattle. *Journal of Animal Science*, 92(11): 5267–5274. https://doi.org/10.2527/jas.2014-8273

Herd, R.M., Velazco, J.I., Arthur, P.F. & Hegarty, R.F. 2016. Associations among methane emission traits measured in the feedlot and in respiration chambers in Angus cattle bred to vary in feed efficiency. *Journal of Animal Science*, 94(11): 4882–4891. https://doi.org/10.2527/jas.2016-0613

Herrero, M., Havlík, P., Valin, H., Notenbaert, A., Rufino, M.C., Thornton, P.K., Blümmel, M., Weiss, F., Grace, D. & Obersteiner, M. 2013. Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proceedings of the National Academy of Sciences of the United States of America*, 110(52): 20888–20893. https://doi.org/10.1073/pnas.1308149110

Herrero, M., Wirsenius, S., Henderson, B., Rigolot, C., Thornton, P., Havlík, P., de Boer, I. & Gerber, P.J. 2015. Livestock and the environment: What have we learned in the past decade? *Annual Review of Environment and Resources*, 40(1): 177–202. https://doi.org/10.1146/annurev-environ-031113-093503

Herron, J., Curran, T.P., Moloney, A.P. & O'Brien, D. 2019. Whole farm modelling the effect of grass silage harvest date and nitrogen fertiliser rate on nitrous oxide emissions from grass-based suckler to beef farming systems. *Agricultural Systems*, 175: 66–78. https://doi.org/10.1016/j.agsy.2019.05.013

Hersom, M. & Thrift, T. 2012. Application of ionophores in cattle diets. EDIS, vol. 12, Publication AN285. https://edis.ifas.ufl.edu/publication/AN285

Hilhorst, M.A., Melse, R.W., Willers, H.C., Groenestein, C.M. & Monteny, G.J. 2002. Reduction of methane emissions from manure. Wageningen, Kingdom of the Netherlands, IMAG. https://edepot.wur.nl/303610

Hill, J., McSweeney, C., Wright, A.-D.G., Bishop-Hurley, G. & Kalantarzadeh, K. 2016. Measuring methane production from ruminants. *Trends in Biotechnology*, 34(1): 26–35. https://doi.org/10.1016/j.tibtech.2015.10.004

Hironaka, R., Mathison, G.W., Kerrigan, B.K. & Vlach, I. 1996. The effect of pelleting of alfalfa hay on methane production and digestibility by steers. *Science of the Total Environment*, 180(3): 221–227. https://doi.org/10.1016/0048-9697(95)04948-7

Höhne, N. & Blok, K. 2005. Calculating historical contributions to climate change – Discussing the 'Brazilian Proposal'. *Climatic Change*, 71(1–2): 141–173. https://doi.org/10.1007/s10584-005-5929-9

Hollmann, M. & Beede, D.K. 2012. Comparison of effects of dietary coconut oil and animal fat blend on lactational performance of Holstein cows fed a high-starch diet. *Journal of Dairy Science*, 95(3): 1484–1499. https://doi.org/10.3168/jds.2011-4792

Hollmann, M., Powers, W.J., Fogiel, A.C., Liesman, J.S., Bello, N.M. & Beede, D.K. 2012. Enteric methane emissions and lactational performance of Holstein cows fed different concentrations of coconut oil. *Journal of Dairy Science*, 95(5): 2602–2615. https://doi.org/10.3168/jds.2011-4896

Honan, M., Feng, X., Tricarico, J.M. & Kebreab, E. 2021. Feed additives as a strategic approach to reduce enteric methane production in cattle: Modes of action, effectiveness and safety. *Animal Production Science*, 62(14): 1303–1317. https://doi.org/10.1071/AN20295

Hooss, G., Voss, R., Hasselmann, K., Maier-Reimer, E. & Joos, F. 2001. A nonlinear impulse response model of the coupled carbon cycle-climate system (NICCS). *Climate Dynamics*, 18(3–4): 189–202. https://doi.org/10.1007/s003820100170

Hoover, W.H. & Stokes, S.R. 1991. Balancing carbohydrates and proteins for optimum rumen microbial yield. *Journal of Dairy Science*, 74(10): 3630–3644. https://doi.org/10.3168/jds.S0022-0302(91)78553-6

Hou, Y., Velthof, G.L. & Oenema, O. 2015. Mitigation of ammonia, nitrous oxide and methane emissions from manure management chains: A meta-analysis and integrated assessment. *Global Change Biology*, 21(3): 1293–1312. https://doi.org/10.1111/gcb.12767

Houdijk, J.G.M., Tolkamp, B.J., Rooke, J.A. & Hutchings, M.R. 2017. Animal health and greenhouse gas intensity: The paradox of periparturient parasitism. *International Journal for Parasitology*, 47(10): 633–641. https://doi.org/10.1016/j.ijpara.2017.03.006

Houweling, S., Krol, M., Bergamaschi, P., Frankenberg, C., Dlugokencky, E.J., Morino, I., Notholt, J., Sherlock, V., Wunch, D., Beck, V. & Gerbig, C. 2014. A multi-year methane inversion using SCIAMACHY, accounting for systematic errors using TCCON measurements. *Atmospheric Chemistry and Physics*, 14(8): 3991–4012. https://doi.org/10.5194/acp-14-3991-2014

Hristov, A.N., Harper, M., Meinen, R., Day, R., Lopes, J., Ott, T., Venkatesh, A. & Randles, C.A. 2017. Discrepancies and uncertainties in bottom-up gridded inventories of livestock methane emissions for the contiguous United States. *Environmental Science & Technology*, 51(23): 13668–13677. https://doi.org/10.1021/acs.est.7b03332

Hristov, A.N., Johnson, K.A. & Kebreab, E. 2014. Livestock methane emissions in the United States. *Proceedings of the National Academy of Sciences of the United States of America*, 111(14): E1320-E1320. https://doi.org/10.1073/pnas.1401046111

Hristov, A.N., Kebreab, E., Niu, M., Oh, J., Bannink, A., Bayat, A.R., Boland, T.M., Brito, A.F., Casper, D.P., Crompton, L.A., Dijkstra, J., Eugène, M., Garnsworthy, P.C., Haque, N., Hellwing, A.L.F., Huhtanen, P., Kreuzer, M., Kuhla, B., Lund, P., Madsen, J., Martin, C., Moate, P.J., Muetzel, S., Muñoz, C., Peiren, N., Powell, J.M., Reynolds, C.K., Schwarm, A., Shingfield, K.J., Storlien, T.M., Weisbjerg, M.R., Yáñez-Ruiz, D.R. & Yu, Z. 2018. Symposium review: Uncertainties in enteric methane inventories, measurement techniques, and prediction models. *Journal of Dairy Science*, 101(7): 6655–6674. https://doi.org/10.3168/jds.2017-13536

Hristov, A.N., Oh, J., Firkins, J.L., Dijkstra, J., Kebreab, E., Waghorn, G., Makkar, H.P.S., Adesogan, A.T., Yang, W., Lee, C., Gerber, P.J., Henderson, B. & Tricarico, J.M. 2013a. Mitigation of methane and nitrous oxide emissions from animal operations: I. A review of enteric methane mitigation options. *Journal of Animal Science*, 91(11): 5045–5069. https://doi.org/10.2527/jas.2013-6583

Hristov, A.N., Oh, J., Giallongo, F., Frederick, T.W., Harper, M.T., Weeks, H.L., Branco, A.F., Price, W.J., Moate, P.J., Deighton, M.H., Williams, S.R.O, Kindermann, M. & Duval, S. 2016. Short communication: Comparison of the GreenFeed system with the sulfur hexafluoride tracer technique for measuring enteric methane emissions from dairy cows. *Journal of Dairy Science*, 99(7): 5461–5465. https://doi.org/10.3168/jds.2016-10897

Hristov, A.N., Oh, J., Giallongo, F., Frederick, T.W., Harper, M.T., Weeks, H.L., Branco, A.F., Moate, P.J., Deighton, M.H., Williams, S.R.O., Kindermann, M. & Duval, S. 2015b. An inhibitor persistently decreased enteric methane emission from dairy cows with no negative effect on milk production. *Proceedings of the National Academy of Sciences*, 112(34): 10663–10668. https://doi.org/10.1073/pnas.1504124112

Hristov, A.N., Oh, J., Giallongo, F., Frederick, T.W., Weeks, H.L., Zimmerman, P.R., Harper, M.T., Hristova, R.A., Zimmerman, R.S. & Branco, A.F. 2015a. The use of an automated system (GreenFeed) to monitor enteric methane and carbon dioxide emissions from ruminant animals. *Journal of Visualized Experiments*, (103): e52904. https://doi.org/10.3791/52904

Hristov, A.N., Ott, T., Tricarico, J., Rotz, A., Waghorn, G., Adesogan, A., Dijkstra, J., Montes, F., Oh, J., Kebreab, E. & Oosting, S.J. 2013b. Mitigation of methane and nitrous oxide emissions from animal operations: III. A review of animal management mitigation options. *Journal of Animal Science*, 91(11): 5095–5113. https://doi.org/10.2527/jas.2013-6585

Hristov, A.N., Vander Pol, M., Agle, M., Zaman, S., Schneider, C., Ndegwa, P., Vaddella, V.K., Johnson, K., Shingfield, K.J. & Karnati, S. 2009. Effect of lauric acid and coconut oil on ruminal fermentation, digestion, ammonia losses from manure, and milk fatty acid composition in lactating cows. *Journal of Dairy Science*, 92(11): 5561–5582. https://doi.org/10.3168/jds.2009-2383

Huang, Y., Zhang, W., Zheng, X., Li, J. & Yu, Y. 2004. Modeling methane emission from rice paddies with various agricultural practices. *Journal of Geophysical Research: Atmospheres*, 109(D8): D08113. https://doi.org/10.1029/2003JD004401

Huhtanen, P., Bayat, A.R., Lund, P., Hellwing, A.L.F. & Weisbjerg, M.R. 2020. Short communication: Variation in feed efficiency hampers use of carbon dioxide as a tracer gas in measuring methane emissions in on-farm conditions. *Journal of Dairy Science*, 103(10): 9090–9095. https://doi.org/10.3168/jds.2020-18559

Huhtanen, P., Cabezas-Garcia, E.H., Utsumi, S. & Zimmerman, S. 2015. Comparison of methods to determine methane emissions from dairy cows in farm conditions. *Journal of Dairy Science*, 98(5): 3394–3409. https://doi.org/10.3168/jds.2014-9118

Huhtanen, P. & Jaakkola, S. 1993. The effects of forage preservation method and proportion of concentrate on digestion of cell wall carbohydrates and rumen digesta pool size in cattle. *Grass and Forage Science*, 48(2): 155–165. https://doi.org/10.1111/j.1365-2494.1993.tb01848.x

Huhtanen, P., Ramin, M. & Hristov, A.N. 2019. Enteric methane emission can be reliably measured by the GreenFeed monitoring unit. *Livestock Science*, 222: 31–40. https://doi.org/10.1016/j.livsci.2019.01.017

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A. & van Zelm, R. 2017. ReCiPe2016: A harmonised life cycle impact assessment method at midpoint and endpoint level. *The International Journal of Life Cycle Assessment*, 22(2): 138–147. https://doi.org/10.1007/s11367-016-1246-y

Hulshof, R.B.A., Berndt, A., Gerrits, W.J.J., Dijkstra, J., van Zijderveld, S.M., Newbold, J.R. & Perdok, H.B. 2012. Dietary nitrate supplementation reduces methane emission in beef cattle fed sugarcane-based diets. *Journal of Animal Science*, 90(7): 2317–2323. https://doi.org/10.2527/jas.2011-4209

Hungate, R.E. 1967. Hydrogen as an intermediate in the rumen fermentation. *Archiv für Mikrobiologie*, 59(1–3): 158–164. https://doi.org/10.1007/BF00406327

Hungate, R.E., Smith, W., Bauchop, T., Yu, I. & Rabinowitz, J.C. 1970. Formate as an Intermediate in the bovine rumen fermentation. *Journal of Bacteriology*, 102(2): 389–397. https://doi.org/10.1128/jb.102.2.389-397.1970

Hussain, S., Peng, S., Fahad, S., Khaliq, A., Huang, J., Cui, K. & Nie, L. 2015. Rice management interventions to mitigate greenhouse gas emissions: A review. *Environmental Science and Pollution Research International*, 22(5): 3342–3360. https://doi.org/10.1007/s11356-014-3760-4

Husted, S. 1993. An open chamber technique for determination of methane emission from stored livestock manure. *Atmospheric Environment. Part A. General Topics*, 27(11): 1635–1642. https://doi.org/10.1016/0960-1686(93)90226-O

Hutchinson, I.A., Shalloo, L. & Butler, S.T. 2013. Expanding the dairy herd in pasture-based systems: The role of sexed semen use in virgin heifers and lactating cows. *Journal of Dairy Science*, 96(10): 6742–6752. https://doi.org/10.3168/jds.2012-6476

Huws, S.A., Creevey, C.J., Oyama, L.B., Mizrahi, I., Denman, S.E., Popova, M., Muñoz-Tamayo, R., Forano, E., Waters, S.M., Hess, M., Tapio, I., Smidt, H., Krizsan, S.J., Yáñez-Ruiz, D.R., Belanche, A., Guan, L., Gruninger, R.J., McAllister, T.A., Newbold, C.J., Roehe, R., Dewhurst, R.J., Snelling, T.J., Watson, M., Suen, G., Hart, E.H., Kingston-Smith, A.H., Scollan, N.D., do Prado, R.M., Pilau, E.J., Mantovani, H.C., Attwood, G.T., Edwards, J.E., McEwan, N.R., Morrisson, S., Mayorga, O.L., Elliott, C. & Morgavi, D.P. 2018. Addressing global ruminant agricultural challenges through understanding the rumen microbiome: Past, present, and future. *Frontiers in Microbiology*, 9: 2161. https://doi.org/10.3389/fmicb.2018.02161

Huws, S.A., Williams, C.L. & McEwan, N.R. 2020. Ruminal-ciliated protozoa. In: C.S. McSweeney & R.I. Mackie, eds. *Improving rumen function*, pp. 191–220. Cambridge, UK, Burleigh Dodds Science Publishing. https://doi.org/10.19103/AS.2020.0067.08

Immig, I., Fiedler, D., van Nevel, C. & Demeyer, D. 1995. Inhibition of methanogenesis in the rumen of a sheep with BES (bromoethanesulfonic acid). *Proceedings of the Society of Nutrition Physiology (Germany)*, p. 68. Frankfurt, Germany, DLG.

Inaba, A. & Itsubo, N. 2018. Preface. *The International Journal of Life Cycle Assessment*, 23(12): 2271–2275. https://doi.org/10.1007/s11367-018-1545-6

Institut National de la Recherche Agronomique (INRA). 2018. *INRA feeding system for ruminants*. Wageningen, Kingdom of the Netherlands, Wageningen Academic Publishers. https://doi.org/10.3920/978-90-8686-292-4

Inubushi, K. 2020. Sustainable soil management in East, South and Southeast Asia. *Soil Science and Plant Nutrition*, 67: 1–9. https://doi.org/10.1080/00380768.2020.1835431

Iordan, C.M., Verones, F. & Cherubini, F. 2018. Integrating impacts on climate change and biodiversity from forest harvest in Norway. *Ecological Indicators*, 89: 411–421. https://doi.org/10.1016/j.ecolind.2018.02.034

IPCC (Intergovernmental Panel on Climate Change). 1990. *Climate change: The IPCC scientific assessment*. (J.T. Houghton, G.J. Jenkins & J.J. Ephraums, eds.) 410 pp. Cambridge, UK & New York, USA, Cambridge University Press.

IPCC. 1996. Greenhouse gas inventory workbook. In: J.T. Houghton, L.G. Meira Filho, B. Lim, K. Treanton, I. Mamaty, Y. Bonduki, D.J. Griggs & B.A. Callender, eds. *Revised 1996 IPCC guidelines for national greenhouse gas inventories,* vol.2. Bracknell, UK.

IPCC. 2001. *Climate change 2001: The scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change* (J.T. Houghton, Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell & C.A. Johnson, eds.), 881 pp. Cambridge, UK & New York, USA, Cambridge University Press. www.ipcc.ch/site/assets/uploads/2018/03/WGI_TAR_full_report.pdf

IPCC. 2003. *Good practice guidance for land use, land-use change and forestry.* (J. Penman, M. Gytarski, T. Hiraishi, T. Krug, D. Kruger, R. Pipatti, L. Buendia & K. Miwa, T. Ngara, K. Tanabe & F. Wagner, eds.) IPCC National Greenhouse Gas Inventories Programme. Hayama, Japan, IGES.

IPCC. 2006. *2006 IPCC guidelines for national greenhouse gas inventories.* (S. Eggelston, L. Buendia, K. Miwa, T. Ngara & K. Tanabe, eds.) Hayama, Japan, IGES.

IPCC. 2007. *Climate change 2007: Synthesis report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (R.K. Pachauri & A. Reisinger, eds.), 104 pp. Geneva, Switzerland.

IPCC. 2013. *Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (T.F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P.M. Midgley, eds.), 1535 pp. Cambridge, UK & New York, USA, Cambridge University Press.

IPCC. 2014. *Climate change 2014: Synthesis report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (R.K. Pachauri & L.A. Meyer, eds.), 151 pp. Geneva, Switzerland.

IPCC. 2018. *Special report: Global warming of 1.5 °C. An IPCC Special Report on the impacts of global warming of 1.5 °C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty* (V. Masson-Delmotte, P. Zhai, H.-O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J.B.R. Matthews, Y. Chen, X. Zhou, M.I. Gomis, E. Lonnoy, T. Maycock, M. Tignor & T. Waterfield, eds.), 616 pp. Cambridge, UK & New York, USA, Cambridge University Press. www.ipcc.ch/site/assets/uploads/sites/2/2019/06/SR15_Full_Report_High_Res.pdf

IPCC. 2019. Agriculture, forestry and other land use. In: E. Calvo Buendia, K. Tanabe, A. Kranj, J. Baasansuren, M. Fukuda, S. Ngarize, A. Osako, Y. Pyrozhenko, P. Shermanau & S. Federici, eds. *2019 Refinement to the 2006 IPCC guidelines for national greenhouse gas inventories*. Geneva, Switzerland. [Cited 25 April 2021]. www.ipcc-nggip.iges.or.jp/public/2019rf/vol4.html

IPCC. 2021a. *Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu & B. Zhou, eds.). Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157896

IPCC. 2021b. Annex VII: Glossary (J.B.R Matthews, V. Möller, R. van Diemen, J.S. Fuglestvedt, V. Masson-Delmotte, C. Méndez, S. Semenov & A. Reisinger, eds.) In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu & B. Zhou, eds. *Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 2215–2256. Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157896.022

IPCC. 2021c. Summary for policymakers. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu & B. Zhou, eds. *Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 3–32. Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157896.001

IPCC. 2022a. *Climate change 2022: Mitigation of climate change. Contribution of Working Group III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (P.R. Shukla, J. Skea, R. Slade, A. Al Khourdajie, R. van Diemen, D. McCollum, M. Pathak, S. Some, P. Vyas, R. Fradera, M. Belkacemi, A. Hasija, G. Lisboa, S. Luz & J. Malley, eds.). Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157926

IPCC. 2022b. Annex II: Definitions, units and conventions (A. Al Khourdajie, R. van Diemen, W.F. Lamb, M. Pathak, A. Reisinger, S. de la Rue du Can, J. Skea, R. Slade, S. Some & L. Steg, eds.). In: P.R. Shukla, J. Skea, R. Slade, A. Al Khourdajie, R. van Diemen, D. McCollum, M. Pathak, S. Some, P. Vyas, R. Fradera, M. Belkacemi, A. Hasija, G. Lisboa, S. Luz & J. Malley, eds. *Climate change 2022: Mitigation of climate change. Contribution of Working Group III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 1823–1840. Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157926.021

IPCC. 2023. Summary for policymakers. In: H. Lee and J. Romero, eds. (forthcoming). *Climate change 2023: Synthesis report. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, 36 pp. Geneva, Switzerland, IPCC. www.ipcc.ch/report/ar6/syr/downloads/report/IPCC_AR6_SYR_SPM.pdf

ISO (International Organization for Standardization). 2006. Environmental management – Life cycle assessment – Principles and framework. ISO 14040: 2006. [Cited 15 June 2023]. www.iso.org/standard/37456.html

ISO. 2021. Radiative forcing management – Guidance for the quantification and reporting of radiative forcing-based climate footprints and mitigation efforts. ISO/WD TR 14082. [Cited 6 April 2006]. www.iso.org/standard/68505.html

Ito, A. & Inatomi, M. 2012. Use of a process-based model for assessing the methane budgets of global terrestrial ecosystems and evaluation of uncertainty. *Biogeosciences*, 9(2): 759–773.

Jacinthe, P.-A. & Lal, R. 2005. Labile carbon and methane uptake as affected by tillage intensity in a Mollisol. *Soil and Tillage Research*, 80(1): 35–45. https://doi.org/10.1016/j.still.2004.02.018

Jafari, S., Ebrahimi, M., Goh, Y.M., Rajion, M.A., Jahromi, M.F. & Al-Jumaili, W.S. 2019. Manipulation of rumen fermentation and methane gas production by plant secondary metabolites (saponin, tannin and essential oil) – A review of ten-year studies. *Annals of Animal Science*, 19(1): 3–29. https://doi.org/10.2478/aoas-2018-0037

Jain, N., Dubey, R., Dubey, D.S., Singh, J., Khanna, M., Pathak, H. & Bhatia, A. 2014. Mitigation of greenhouse gas emission with system of rice intensification in the Indo-Gangetic Plains. *Paddy and Water Environment,* 12: 355–363. https://doi.org/10.1007/s10333-013-0390-2

Jami, E. & Mizrahi, I. 2020. Host-rumen microbiome interactions and influences on feed conversion efficiency (FCE), methane production and other productivity traits. In: C.S. McSweeney & R.I. Mackie, eds. *Improving rumen function*, pp. 548–566. London, Burleigh Dodds Science Publishing. https://doi.org/10.19103/AS.2020.0067.18

Jang, I., Lee, S., Hong, J.-H. & Kang, H. 2006. Methane oxidation rates in forest soils and their controlling variables: A review and a case study in Korea. *Ecological Research*, 21(6): 849–854. https://doi.org/10.1007/s11284-006-0041-9

Janssen, P.H. 2010. Influence of hydrogen on rumen methane formation and fermentation balances through microbial growth kinetics and fermentation thermodynamics. *Animal Feed Science and Technology*, 160(1–2): 1–22. https://doi.org/10.1016/j.anifeedsci.2010.07.002

Janz, B., Weller, S., Kraus, D., Racela, H.S., Wassmann, R., Butterbach-Bahl, K. & Kiese, R. 2019. Greenhouse gas footprint of diversifying rice cropping systems: Impacts of water regime and organic amendments. *Agriculture, Ecosystems & Environment*, 270–271: 41–54. https://doi.org/10.1016/j.agee.2018.10.011

Jayanegara, A., Leiber, F. & Kreuzer, M. 2012. Meta-analysis of the relationship between dietary tannin level and methane formation in ruminants from *in vivo* and *in vitro* experiments. *Journal of Animal Physiology and Animal Nutrition*, 96(3): 365–375. https://doi.org/10.1111/j.1439-0396.2011.01172.x

Jayanegara, A., Sarwono, K.A., Kondo, M., Matsui, H., Ridla, M., Laconi, E.B. & Nahrowi. 2018. Use of 3-nitrooxypropanol as feed additive for mitigating enteric methane emissions from ruminants: A meta-analysis. *Italian Journal of Animal Science*, 17(3): 650–656. https://doi.org/10.1080/1828051X.2017.1404945

Jayanegara, A., Wina, E. & Takahashi, J. 2014. Meta-analysis on methane mitigating properties of saponin-rich sources in the rumen: Influence of addition levels and plant sources. *Asian-Australasian Journal of Animal Sciences*, 27(10): 1426–1435. https://doi.org/10.5713/ajas.2014.14086

Jayanegara, A., Yogianto, Y., Wina, E., Sudarman, A., Kondo, M., Obitsu, T. & Kreuzer, M. 2020. Combination effects of plant extracts rich in tannins and saponins as feed additives for mitigating *in vitro* ruminal methane and ammonia formation. *Animals*, 10(9): 1531. https://doi.org/10.3390/ani10091531

Jeffery, S., Verheijen, F.G.A., Kammann, C. & Abalos, D. 2016. Biochar effects on methane emissions from soils: A meta-analysis. *Soil Biology and Biochemistry*, 101: 251–258. https://doi.org/10.1016/j.soilbio.2016.07.021

Jeyanathan, J., Martin, C. & Morgavi, D.P. 2014. The use of direct-fed microbials for mitigation of ruminant methane emissions: A review. *Animal*, 8(2): 250–261. https://doi.org/10.1017/S1751731113002085

Jia, Y., Quack, B., Kinley, R.D., Pisso, I. & Tegtmeier, S. 2022. Potential environmental impact of bromoform from *Asparagopsis* farming in Australia. *Atmospheric Chemistry and Physics*, 22(11): 7631–7646. https://doi.org/10.5194/acp-22-7631-2022

Jiang, Y., Carrijo, D., Huang, S., Chen, J., Balaine, N., Zhang, W., van Groenigen, K.J. & Linquist, B. 2019. Water management to mitigate the global warming potential of rice systems: A global meta-analysis. *Field Crops Research*, 234: 47–54. https://doi.org/10.1016/j.fcr.2019.02.010

Jiao, H.P., Dale, A.J., Carson, A.F., Murray, S., Gordon, A.W. & Ferris, C.P. 2014. Effect of concentrate feed level on methane emissions from grazing dairy cows. *Journal of Dairy Science*, 97(11): 7043–7053. https://doi.org/10.3168/jds.2014-7979

Johansen, M., Lund, P. & Weisbjerg, M.R. 2018. Feed intake and milk production in dairy cows fed different grass and legume species: A meta-analysis. *Animal*, 12(1): 66–75. https://doi.org/10.1017/S1751731117001215.

Johansson, D.J.A. 2012. Economics- and physical-based metrics for comparing greenhouse gases. *Climatic Change*, 110(1): 123–141. https://doi.org/10.1007/s10584-011-0072-2

Johansson, D.J.A., Persson, U.M. & Azar, C. 2006. The cost of using global warming potentials: Analysing the trade-off between CO_2, CH_4 and N_2O. *Climatic Change*, 77(3): 291–309. https://doi.org/10.1007/s10584-006-9054-1

Johnson, D.E., Phetteplace, H.W. & Seidl, A.F. 2002. Methane, nitrous oxide and carbon dioxide emissions from ruminant livestock production systems. In: J. Takahashi & B.A. Young, eds. *Greenhouse Gases and Animal Agriculture*, pp. 77–85. Amsterdam, Elsevier.

Johnson, D.E., Ward, G.W. & Ramsey, J.J. 1996. Livestock methane: Current emissions and mitigation potential. In: E.T. Kornegay, ed. *Nutrient management of food animals to enhance and protect the environment*, pp. 219–234. New York, USA, Lewis Publishers.

Johnson, K.A., Huyler, M., Westberg, H., Lamb, B. & Zimmerman, P. 1994. Measurement of methane emissions from ruminant livestock using a sulfur hexafluoride tracer technique. *Environmental Science & Technology*, 28(2): 359–362. https://doi.org/10.1021/es00051a025

Johnson, K.A. & Johnson, D.E. 1995. Methane emissions from cattle. *Journal of Animal Science*, 73(8): 2483–2492. https://doi.org/10.2527/1995.7382483x

Jolliet, O., Antón, A., Boulay, A.-M., Cherubini, F., Fantke, P., Levasseur, A., McKone, T.E., Michelsen, O., Canals, L.M.I., Motoshita, M., Pfister, S., Verones, F., Vigon, B. & Frischknecht, R. 2018. Global guidance on environmental life cycle impact assessment indicators: Impacts of climate change, fine particulate matter formation, water consumption and land use. *The International Journal of Life Cycle Assessment*, 23(11): 2189–2207. https://doi.org/10.1007/s11367-018-1443-y

Jones, J.W., Hoogenboom, G., Porter, C.H., Boote, K.J., Batchelor, W.D., Hunt, L.A., Wilkens, P.W., Singh, U., Gijsman, A.J. & Ritchie, J.T. 2003. The DSSAT cropping system model. *European Journal of Agronomy*, 18(3–4): 235–265.

Jonker, A. & Waghorn, G.C., eds. 2020. Guidelines for use of sulphur hexafluoride (SF6) tracer technique to measure enteric methane emissions from ruminants (Second edition). MPI (Ministry for Primary Industries) Technical paper No. 2020/06. Palmerston North, New Zealand, New Zealand Agricultural Greenhouse Gas Research Centre. https://globalresearchalliance.org/wp-content/uploads/2018/02/SF6-Tracer-Technique-Guidelines_April-2014.pdf

Joo, H.S., Ndegwa, P.M., Heber, A.J., Bogan, B.W., Ni, J.-Q., Cortus, E.L. & Ramirez-Dorronsoro, J.C. 2014. A direct method of measuring gaseous emissions from naturally ventilated dairy barns. *Atmospheric Environment*, 86: 176–186. https://doi.org/10.1016/j.atmosenv.2013.12.030

Joos, F., Roth, R., Fuglestvedt, J.S., Peters, G.P., Enting, I.G., von Bloh, W., Brovkin, V., Burke, E.J., Eby, M., Edwards, N.R., Friedrich, T., Frölicher, T.L., Halloran, P.R., Holden, P.B., Jones, C., Kleinen, T., Mackenzie, F.T., Matsumoto, K., Meinshausen, M., Plattner, G.-K., Reisinger, A., Segschneider, J., Shaffer, G., Steinacher, M., Strassmann, K., Tanaka, K., Timmermann, A. & Weaver, A.J. 2013. Carbon dioxide and climate impulse response functions for the computation of greenhouse gas metrics: A multi-model analysis. *Atmospheric Chemistry and Physics*, 13(5): 2793–2825. https://doi.org/10.5194/acp-13-2793-2013

Jordan, E., Kenny, D., Hawkins, M., Malone, R., Lovett, D.K. & O'Mara, F.P. 2006a. Effect of refined soy oil or whole soybeans on intake, methane output, and performance of young bulls. *Journal of Animal Science*, 84(9): 2418–2425. https://doi.org/10.2527/jas.2005-354

Jordan, E., Lovett, D.K., Monahan, F.J., Callan, J., Flynn, B. & O'Mara, F.P. 2006b. Effect of refined coconut oil or copra meal on methane output and on intake and performance of beef heifers. *Journal of Animal Science*, 84(1): 162–170. https://doi.org/10.2527/2006.841162x

Kandlikar, M. 1995. The relative role of trace gas emissions in greenhouse abatement policies. *Energy Policy*, 23(10): 879–883. https://doi.org/10.1016/0301-4215(95)00108-U

Kanter, D.R., Wagner-Riddle, C., Groffman, P.M., Davidson, E.A., Galloway, J.N., Gourevitch, J.D., van Grinsven, H.J.M., Houlton, B.Z., Keeler, B.L., Ogle, S.M., Pearen, H., Rennert, K.J., Saifuddin, M., Sobota, D.J. & Wagner, G. 2021. Improving the social cost of nitrous oxide. *Nature Climate Change*, 11(12): 1008–1010. https://doi.org/10.1038/s41558-021-01226-z

Katayanagi, N., Fumoto, T., Hayano, M., Shirato, Y., Takata, Y., Leon, A. & Yagi, K. 2017. Estimation of total CH_4 emission from Japanese rice paddies using a new estimation method based on the DNDC-rice simulation model. *Science of the Total Environment*, 601–602: 346–355. https://doi.org/10.1016/j.scitotenv.2017.05.090

Kavanagh, I., Burchill, W., Healy, M.G., Fenton, O., Krol, D.J. & Lanigan, G.J. 2019. Mitigation of ammonia and greenhouse gas emissions from stored cattle slurry using acidifiers and chemical amendments. *Journal of Cleaner Production*, 237: 117822. https://doi.org/10.1016/j.jclepro.2019.117822

Kebreab, E., Bannink, A., Pressman, E.M., Walker, N., Karagiannis, A., van Gastelen, S. & Dijkstra, J. 2023. A meta-analysis of effects of 3-nitrooxypropanol on methane production, yield, and intensity in dairy cattle. *Journal of Dairy Science*, 106(2): 927–936. https://doi.org/10.3168/jds.2022-22211

Kebreab, E., France, J., McBride, B.W., Odongo, N.E., Bannink, A., Mills, J.A.N. & Dijkstra, J. 2006. Evaluation of models to predict methane emissions from enteric fermentation in North American dairy cattle. In: E. Kebreab, J. Dijkstra, A. Bannink, W.J.J. Gerrits & J. France, eds. *Nutrient digestion and utilization in farm animals: Modelling approaches*, pp. 299–313. Wallingford, UK, CABI Publishing.

Kebreab, E., Johnson, K.A., Archibeque, S.L., Pape, D. & Wirth, T. 2008. Model for estimating enteric methane emissions from United States dairy and feedlot cattle. *Journal of Animal Science*, 86(10): 2738–2748. https://doi.org/10.2527/jas.2008-0960

Kennedy, P.M. & Charmley, E. 2012. Methane yields from Brahman cattle fed tropical grasses and legumes. *Animal Production Science*, 52(4): 225. https://doi.org/10.1071/AN11103

Kessavalou, A., Mosier, A.R., Doran, J.W., Drijber, R.A., Lyon, D. & Heinemeyer, O. 1998. Fluxes of carbon dioxide, nitrous oxide, and methane in grass sod and winter wheat-fallow tillage management. *Journal of Environmental Quality*, 27(5): 1094–1104.

Kim, H., Lee, H.G., Baek, Y.-C., Lee, S. & Seo, J. 2020. The effects of dietary supplementation with 3-nitrooxypropanol on enteric methane emissions, rumen fermentation, and production performance in ruminants: A meta-analysis. *Journal of Animal Science and Technology*, 62(1): 31–42. https://doi.org/10.5187/jast.2020.62.1.31

Kim, S.-H., Mamuad, L.L., Kim, D.-W., Kim, S.-K. & Lee, S.-S. 2016. Fumarate reductase-producing Enterococci reduce methane production in rumen fermentation *in vitro*. *Journal of Microbiology and Biotechnology*, 26(3): 558–566. https://doi.org/10.4014/jmb.1512.12008

Kimura, M., Murase, J. & Lu, Y. 2004. Carbon cycling in rice field ecosystems in the context of input, decomposition and translocation of organic materials and the fates of their end products (CO_2 and CH_4). *Soil Biology and Biochemistry*, 36(9): 1399–1416. https://doi.org/10.1016/j.soilbio.2004.03.006

King, G.M. 1992. Ecological aspects of methane oxidation, a key determinant of global methane dynamics. In: K.C. Marshall, ed. *Advances in microbial ecology*, pp. 431–468. New York, USA, Plenum Press.

Kinley, R., de Nys, R., Vucko, M., Machado, L. & Tomkins, N. 2016. The red macroalgae *Asparagopsis taxiformis* is a potent natural antimethanogenic that reduces methane production during *in vitro* fermentation with rumen fluid. *Animal Production Science*, 56(3): 282–289. https://doi.org/10.1071/AN15576

Kinley, R.D., Martinez-Fernandez, G., Matthews, M.K., de Nys, R., Magnusson, M. & Tomkins, N.W. 2020. Mitigating the carbon footprint and improving productivity of ruminant livestock agriculture using a red seaweed. *Journal of Cleaner Production*, 259: 120836. https://doi.org/10.1016/j.jclepro.2020.120836

Kirschbaum, M.U.F. 2014. Climate-change impact potentials as an alternative to global warming potentials. *Environmental Research Letters*, 9(3): 034014. https://doi.org/10.1088/1748-9326/9/3/034014

Kirschbaum, M.U.F. 2017. Assessing the merits of bioenergy by estimating marginal climate-change impacts. *The International Journal of Life Cycle Assessment*, 22(6): 841–852. https://doi.org/10.1007/s11367-016-1196-4

Klevenhusen, F., Zeitz, J.O., Duval, S., Kreuzer, M. & Soliva, C.R. 2011. Garlic oil and its principal component diallyl disulfide fail to mitigate methane, but improve digestibility in sheep. *Animal Feed Science and Technology*, 166–167: 356–363. https://doi.org/10.1016/j.anifeedsci.2011.04.071

Klinsky, S. & Winkler, H. 2018. Building equity in: Strategies for integrating equity into modelling for a 1.5°C world. *Philosophical Transactions of the Royal Society. Series A – Mathematical, Physical and Engineering Sciences*, 376(2119): 20160461. https://doi.org/10.1098/rsta.2016.0461

Knapp, J.R., Laur, G.L., Vadas, P.A., Weiss, W.P. & Tricarico, J.M. 2014. Invited review: Enteric methane in dairy cattle production: Quantifying the opportunities and impact of reducing emissions. *Journal of Dairy Science*, 97(6): 3231–3261. https://doi.org/10.3168/jds.2013-7234

Knief, C. 2019. Diversity of methane cycling microorganisms in soils and their relation to oxygen. *Current Issues in Molecular Biology*, 33: 23–56. https://doi.org/10.21775/cimb.033.023

Knief, C., Lipski, A. & Dunfield, P.F. 2003. Diversity and activity of methanotrophic bacteria in different upland soils. *Applied and Environmental Microbiology*, 69(11): 6703–6714. https://doi.org/10.1128/AEM.69.11.6703-6714.2003

Kolling, G.J., Stivanin, S.C.B., Gabbi, A.M., Machado, F.S., Ferreira, A.L., Campos, M.M., Tomich, T.R., Cunha, C.S., Dill, S.W., Pereira, L.G.R. & Fischer, V. 2018. Performance and methane emissions in dairy cows fed oregano and green tea extracts as feed additives. *Journal of Dairy Science*, 101(5): 4221–4234. https://doi.org/10.3168/jds.2017-13841

Kolstad, C., Urama, K., Broome, J., Bruvoll, A., Olvera, M.C., Fullerton, D., Gollier, C., W. M. Hanemann, Hassan, R., Jotzo, F., Khan, M.R., Meyer, L. & Mundaca, L. 2014. Social, economic and ethical concepts and methods. In: O. Edenhofer, R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, T. Zwickel & J.C. Minx, eds. *Climate change 2014: Mitigation of climate change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 211–260. Cambridge, UK & New York, USA, Cambridge University Press.

Kouzuma, A., Kaku, N. & Watanabe, K. 2014. Microbial electricity generation in rice paddy fields: Recent advances and perspectives in rhizosphere microbial fuel cells. *Applied Microbiology and Biotechnology*, 98(23): 9521–9526. https://doi.org/10.1007/s00253-014-6138-0

Kozłowska, M., Cieślak, A., Jóźwik, A., El-Sherbiny, M., Stochmal, A., Oleszek, W., Kowalczyk, M., Filipiak, W. & Szumacher-Strabel, M. 2020. The effect of total and individual alfalfa saponins on rumen methane production. *Journal of the Science of Food and Agriculture*, 100(5): 1922–1930. https://doi.org/10.1002/jsfa.10204

Kraan, S. & Barrington, K.A. 2005. Commercial farming of *Asparagopsis armata* (Bonnemaisoniceae, Rhodophyta) in Ireland, maintenance of an introduced species? *Journal of Applied Phycology*, 17(2): 103–110. https://doi.org/10.1007/s10811-005-2799-5

Krattenmacher, N., Thaller, G. & Tetens, J. 2019. Analysis of the genetic architecture of energy balance and its major determinants dry matter intake and energy-corrected milk yield in primiparous Holstein cows. *Journal of Dairy Science*, 102(4): 3241–3253. https://doi.org/10.3168/jds.2018-15480

Kraus, D., Weller, S., Klatt, S., Santabárbara, I., Haas, E., Wassmann, R., Werner, C., Kiese, R. & Butterbach-Bahl, K. 2016. How well can we assess impacts of agricultural land management changes on the total greenhouse gas balance (CO_2, CH_4 and N_2O) of tropical rice-cropping systems with a biogeochemical model? *Agriculture, Ecosystems & Environment*, 224: 104–115. https://doi.org/10.1016/j.agee.2016.03.037

Kreidenweis, U., Breier, J., Herrmann, C., Libra, J. & Prochnow, A. 2021. Greenhouse gas emissions from broiler manure treatment options are lowest in well-managed biogas production. *Journal of Cleaner Production*, 280: 124969. https://doi.org/10.1016/j.jclepro.2020.124969

Kreuzer, M. & Hindrichsen, I.K. 2006. Methane mitigation in ruminants by dietary means: The role of their methane emission from manure. *International Congress Series*, 1293: 199–208. https://doi.org/10.1016/j.ics.2006.01.015

Kriegler, E. 2005. *Imprecise probability analysis for integrated assessment of climate change*. Potsdam, Germany, Universität Potsdam.

Kritee, K., Nair, D., Zavala-Araiza, D., Proville, J., Rudek, J., Adhya, T.K., Loecke, T., Esteves, T., Balireddygari, S., Dava, O., Ram, K., Abhilash, S.R., Madasamy, M., Dokka, R., Anandaraj, D., Athiyaman, D., Reddy, M., Ahuja, R. & Hamburg, S.P. 2018. High nitrous oxide fluxes from rice indicate the need to manage water for both long- and short-term climate impacts. *Proceedings of the National Academy of Sciences*, 115(39): 9720–9725. https://doi.org/10.1073/pnas.1809276115

Kumar, S.S., Kumar, A., Singh, S., Malyan, S.K., Baram, S., Sharma, J., Singh, R. & Pugazhendhi, A. 2020. Industrial wastes: Fly ash, steel slag and phosphogypsum-potential candidates to mitigate greenhouse gas emissions from paddy fields. *Chemosphere*, 241: 124824. https://doi.org/10.1016/j.chemosphere.2019.124824

Kumari, S., Hiloidhari, M., Naik, S.N. & Dahiya, R.P. 2019. Social cost of methane: Method and estimates for Indian livestock. *Environmental Development*, 32: 100462. https://doi.org/10.1016/j.envdev.2019.100462

Ku-Vera, J.C., Castelán-Ortega, O.A., Galindo-Maldonado, F.A., Arango, J., Chirinda, N., Jiménez-Ocampo, R., Valencia-Salazar, S.S., Flores-Santiago, E.J., Montoya-Flores, M.D., Molina-Botero, I.C., Piñeiro-Vázquez, A.T., Arceo-Castillo, J.I., Aguilar-Pérez, C.F., Ramírez-Avilés, L., Solorio-Sánchez, F.J. 2020. Review: Strategies for enteric methane mitigation in cattle fed tropical forages. *Animal*, 14(S3): s453–s463. https://doi.org/10.1017/S1751731120001780

Kuze, A., Suto, H., Shiomi, K., Kawakami, S., Tanaka, M., Ueda, Y., Deguchi, A., Yoshida, J., Yamamoto, Y., Kataoka, F. & Taylor, T.E. 2016. Update on GOSAT TANSO-FTS performance, operations, and data products after more than 6 years in space. *Atmospheric Measurement Techniques*, 9(6): 2445–2461.

Lahart, B., Shalloo, L., Herron, J., O'Brien, D., Fitzgerald, R., Boland, T.M. & Buckley, F. 2021. Greenhouse gas emissions and nitrogen efficiency of dairy cows of divergent economic breeding index under seasonal pasture-based management. *Journal of Dairy Science*, 104(7): 8039–8049. https://doi.org/10.3168/jds.2020-19618

Lamb, D.W., Schneider, D.A., Trotter, M.G., Schaefer, M.T. & Yule, I.J. 2011. Extended-altitude, aerial mapping of crop NDVI using an active optical sensor: A case study using a Raptor^{TM} sensor over wheat. *Computers and Electronics in Agriculture*, 77(1): 69–73. https://doi.org/10.1016/j.compag.2011.03.009

Lanigan, G. 1972. Metabolism of pyrrolizidine alkaloids in the ovine rumen. IV. Effects of chloral hydrate and halogenated methanes on rumen methanogenesis and alkaloid metabolism in fistulated sheep. *Australian Journal of Agricultural Research*, 23(6): 1085. https://doi.org/10.1071/AR9721085

Laradji, I., Rodriguez, P., Kalaitzis, F., Vazquez, D., Young, R., Davey, E. & Lacoste, A. 2020. Counting cows: Tracking illegal cattle ranching from high-resolution satellite imagery. *arXiv*. https://doi.org/10.48550/ARXIV.2011.07369

Lassen, J. & Difford, G.F. 2020. Review: Genetic and genomic selection as a methane mitigation strategy in dairy cattle. *Animal*, 14(53): s473–s483. https://doi.org/10.1017/S1751731120001561

Lassen, J. & Løvendahl, P. 2016. Heritability estimates for enteric methane emissions from Holstein cattle measured using noninvasive methods. *Journal of Dairy Science*, 99(3): 1959–1967. https://doi.org/10.3168/jds.2015-10012

Lassen, J., Løvendahl, P. & Madsen, J. 2012. Accuracy of noninvasive breath methane measurements using Fourier transform infrared methods on individual cows. *Journal of Dairy Science*, 95(2): 890–898. https://doi.org/10.3168/jds.2011-4544

Lassey, K.R. 2007. Livestock methane emission: From the individual grazing animal through national inventories to the global methane cycle. *Agricultural and Forest Meteorology*, 142(2–4): 120–132. https://doi.org/10.1016/j.agrformet.2006.03.028

Lassey, K.R. 2008. Livestock methane emission and its perspective in the global methane cycle. *Australian Journal of Experimental Agriculture*, 48(2): 114–118. https://doi.org/10.1071/EA07220

Latham, E.A., Anderson, R.C., Pinchak, W.E. & Nisbet, D.J. 2016. Insights on alterations to the rumen ecosystem by nitrate and nitrocompounds. *Frontiers in Microbiology*, 7:228. https://doi.org/10.3389/fmicb.2016.00228

Lauder, A.R., Enting, I.G., Carter, J.O., Clisby, N., Cowie, A.L., Henry, B.K. & Raupach, M.R. 2013. Offsetting methane emissions – An alternative to emission equivalence metrics. *International Journal of Greenhouse Gas Control*, 12: 419–429. https://doi.org/10.1016/j.ijggc.2012.11.028

Lauvaux, T., Giron, C., Mazzolini, M., d'Aspremont, A., Duren, R., Cusworth, D., Shindell, D. & Ciais, P. 2022. Global assessment of oil and gas methane ultra-emitters. *Science*, 375(6580): 557–561. https://doi.org/10.1126/science.abj4351

Le Liboux, S. & Peyraud, J.L. 1999. Effect of forage particle size and feeding frequency on fermentation patterns and sites and extent of digestion in dairy cows fed mixed diets. *Animal Feed Science and Technology*, 76(3–4): 297–319. https://doi.org/10.1016/S0377-8401(98)00220-X

Le Van, T.D., Robinson, J.A., Ralph, J., Greening, R.C., Smolenski, W.J., Leedle, J.A.Z. & Schaefer, D.M. 1998. Assessment of reductive acetogenesis with indigenous ruminal bacterium populations and *Acetitomaculum ruminis*. *Applied and Environmental Microbiology*, 64(9): 3429–3436. https://doi.org/10.1128/AEM.64.9.3429-3436.1998

Leahy, S.C., Kelly, W.J., Altermann, E., Ronimus, R.S., Yeoman, C.J., Pacheco, D.M., Li, D., Kong, Z., McTavish, S., Sang, C., Lambie, S.C., Janssen, P.H., Dey, D. & Atwood, G.T. 2010. The genome sequence of the rumen methanogen *Methanobrevibacter ruminantium* reveals new possibilities for controlling ruminant methane emissions. *PLoS ONE*, 5(1): e8926. https://doi.org/10.1371/journal.pone.0008926

Leahy, S.C., Kelly, W.J., Ronimus, R.S., Wedlock, N., Altermann, E. & Attwood, G.T. 2013. Genome sequencing of rumen bacteria and archaea and its application to methane mitigation strategies. *Animal*, 7: 235–243. https://doi.org/10.1017/S1751731113000700

Lee, C. & Beauchemin, K.A. 2014. A review of feeding supplementary nitrate to ruminant animals: Nitrate toxicity, methane emissions, and production performance. *Canadian Journal of Animal Science*, 94(4): 557–570. https://doi.org/10.4141/cjas-2014-069

Lejonklev, J., Løkke, M.M., Larsen, M.K., Mortensen, G., Petersen, M.A. & Weisbjerg, M.R. 2013. Transfer of terpenes from essential oils into cow milk. *Journal of Dairy Science*, 96(7): 4235–4241. https://doi.org/10.3168/jds.2012-6502

Leng, R.A. 2014. Interactions between microbial consortia in biofilms: A paradigm shift in rumen microbial ecology and enteric methane mitigation. *Animal Production Science*, 54(5): 519–543. https://doi.org/10.1071/AN13381

Leslie, M., Aspin, M. & Clark, H. 2008. Greenhouse gas emissions from New Zealand agriculture: Issues, perspectives and industry response. *Australian Journal of Experimental Agriculture*, 48(2): 1–5. https://doi.org/10.1071/EA07306

Levasseur, A., Cavalett, O., Fuglestvedt, J.S., Gasser, T., Johansson, D.J.A., Jørgensen, S.V., Raugei, M., Reisinger, A., Schivley, G., Strømman, A., Tanaka, K. &Cherubini, F. 2016. Enhancing life cycle impact assessment from climate science: Review of recent findings and recommendations for application to LCA. *Ecological Indicators*, 71: 163–174. https://doi.org/10.1016/j.ecolind.2016.06.049

Li, C. 2007. Quantifying greenhouse gas emissions from soils: Scientific basis and modeling approach. *Soil Science & Plant Nutrition*, 53(4): 344–352. https://doi.org/10.1111/j.1747-0765.2007.00133.x

Li, C., Frolking, S. & Frolking, T.A. 1992. A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. *Journal of Geophysical Research: Atmospheres*, 97(D9): 9759–9776. https://doi.org/10.1029/92JD00509

Li, C., Mosier, A., Wassmann, R., Cai, Z., Zheng, X., Huang, Y., Tsuruta, H., Boonjawat, J. & Lantin, R. 2004. Modeling greenhouse gas emissions from rice-based production systems: Sensitivity and upscaling. *Global Biogeochemical Cycles*, 18(1). https://doi.org/10.1029/2003GB002045

Li, C., Salas, W., Zhang, R., Krauter, C., Rotz, A. & Mitloehner, F. 2012. Manure-DNDC: A biogeochemical process model for quantifying greenhouse gas and ammonia emissions from livestock manure systems. *Nutrient Cycling in Agroecosystems*, 93(2): 163–200. https://doi.org/10.1007/s10705-012-9507-z

Li, H., Tan, Y., Ditaranto, M., Yan, J. & Yu, Z. 2017. Capturing CO_2 from biogas plants. *Energy Procedia*, 114: 6030–6035. https://doi.org/10.1016/j.egypro.2017.03.1738

Li, W. & Powers, W. 2012. Effects of saponin extracts on air emissions from steers. *Journal of Animal Science*, 90(11): 4001–4013. https://doi.org/10.2527/jas.2011-4888

Li, X., Norman, H.C., Kinley, R.D., Laurence, M., Wilmot, M., Bender, H., De Nys, R. & Tomkins, N. 2016. *Asparagopsis taxiformis* decreases enteric methane production from sheep. *Animal Production Science*, 58(4): 681. https://doi.org/10.1071/AN15883

Li, Z., Deng, Q., Liu, Y., Yan, T., Li, F., Cao, Y. & Yao, J. 2018. Dynamics of methanogenesis, ruminal fermentation and fiber digestibility in ruminants following elimination of protozoa: A meta-analysis. *Journal of Animal Science and Biotechnology*, 9(1): 89. https://doi.org/10.1186/s40104-018-0305-6

Liebetrau, J., Reinelt, T., Clemens, J., Hafermann, C., Friehe, J. & Weiland, P. 2013. Analysis of greenhouse gas emissions from 10 biogas plants within the agricultural sector. *Water Science and Technology: A Journal of the International Association on Water Pollution Research*, 67(6): 1370–1379. https://doi.org/10.2166/wst.2013.005

Liesack, W., Schnell, S. & Revsbech, N.P. 2000. Microbiology of flooded rice paddies. *FEMS Microbiology Reviews*, 24(5): 625–645. https://doi.org/10.1111/j.1574-6976.2000.tb00563.x

Lima, P.D.M.T., Filho, A.L.A., Issakowicz, J., Ieda, E.H., Corrêa, P.S., Teixeira de Mattos, W., Gerdes, L, McManus, C., Abdalla, A.L. & Louvandini, H. 2020. Methane emission, ruminal fermentation parameters and fatty acid profile of meat in Santa Inês lambs fed the legume macrotiloma. *Animal Production Science*, 60(5): 665. https://doi.org/10.1071/AN19127

Lima, P.R., Apdini, T., Freire, A.S., Santana, A.S., Moura, L.M.L., Nascimento, J.C.S., Rodrigues, R.T.S., Dijkstra, J., Neto, A.G. & Queiroz, M.Á. 2019. Dietary supplementation with tannin and soybean oil on intake, digestibility, feeding behavior, ruminal protozoa and methane emission in sheep. *Animal Feed Science and Technology*, 249: 10–17. https://doi.org/10.1016/j.anifeedsci.2019.01.017

Little, S., Benchaar, C., Janzen, H., Kröbel, R., Mcgeough, E. & Beauchemin, K. 2017. Demonstrating the effect of forage source on the carbon footprint of a Canadian dairy farm using whole-systems analysis and the Holos model: Alfalfa silage vs. corn silage. *Climate*, 5: 87. https://doi.org/10.3390/cli5040087

Liu, H., Puchala, R., LeShure, S., Gipson, T.A., Flythe, M.D. & Goetsch, A.L. 2019. Effects of lespedeza condensed tannins alone or with monensin, soybean oil, and coconut oil on feed intake, growth, digestion, ruminal methane emission, and heat energy by yearling Alpine doelings. *Journal of Animal Science*, 97(2): 885–899. https://doi.org/10.1093/jas/sky452

Liu, M., Liu, C., Liao, W., Xie, J., Zhang, X. & Gao, Z. 2021. Impact of biochar application on gas emissions from liquid pig manure storage. *Science of the Total Environment*, 771: 145454. https://doi.org/10.1016/j.scitotenv.2021.145454

Liu, Z., Powers, W. & Harmon, J. 2016. Estimating ventilation rates of animal houses through CO_2 balance. *Transactions of the ASABE (American Society of Agricultural and Biological Engineers*, 59(1): 321–328. https://doi.org/10.13031/trans.59.10235

Livingston, G.P. & Hutchinson, G.L. 1995. Enclosure-based measurement of trace gas exchange: Applications and sources of error. In: P.A. Matson & R.C. Harris, eds. *Biogenic trace gases: Measuring emissions from soil and water*, pp. 14–51. Oxford, UK, Blackwell Science Ltd.

Livsey, J., Kätterer, T., Vico, G., Lyon, S.W., Lindborg, R., Scaini, A., Da, C.T. & Manzoni, S. 2019. Do alternative irrigation strategies for rice cultivation decrease water footprints at the cost of long-term soil health? *Environmental Research Letters*, 14(7): 074011. https://doi.org/10.1088/1748-9326/ab2108

Llonch, P., Haskell, M.J., Dewhurst, R.J. & Turner, S.P. 2017. Current available strategies to mitigate greenhouse gas emissions in livestock systems: An animal welfare perspective. *Animal*, 11(2): 274–284. https://doi.org/10.1017/S1751731116001440

Lopez, S., McIntosh, F.M., Wallace, R.J. & Newbold, C.J. 1999. Effect of adding acetogenic bacteria on methane production by mixed rumen microorganisms. *Animal Feed Science and Technology*, 78(1–2): 1–9. https://doi.org/10.1016/S0377-8401(98)00273-9

López-Paredes, J., Goiri, I., Atxaerandio, R., García-Rodríguez, A., Ugarte, E., Jiménez-Montero, J.A., Alenda, R. & González-Recio, O. 2020. Mitigation of greenhouse gases in dairy cattle via genetic selection: 1. Genetic parameters of direct methane using noninvasive methods and proxies of methane. *Journal of Dairy Science*, 103(8): 7199–7209. https://doi.org/10.3168/jds.2019-17597

Lovanh, N.C., Warren, J.G. & Sistani, K.R. 2008. Ammonia and greenhouse gases emission from land application of swine slurry: A comparison of three application methods. In: Livestock Environment VIII, 31 August–4 September 2008. Iguassu Falls, Brazil, American Society of Agricultural and Biological Engineers. https://doi.org/10.13031/2013.25510

Løvendahl, P., Difford, G.F., Li, B., Chagunda, M.G.G., Huhtanen, P., Lidauer, M.H., Lassen, J. & Lund, P. 2018. Review: Selecting for improved feed efficiency and reduced methane emissions in dairy cattle. *Animal*, 12: s336–s349. https://doi.org/10.1017/S1751731118002276

Lovett, D.K, Lovell, S., Stack, L., Callan, J., Finlay, M., Conolly, J. & O'Mara, F.P. 2003. Effect of forage/concentrate ratio and dietary coconut oil level on methane output and performance of finishing beef heifers. *Livestock Production Science*, 84(2): 135–146. https://doi.org/10.1016/j.livprodsci.2003.09.010

Lovett, D., McGilloway, D., Bortolozzo, A., Hawkins, M., Callan, J., Flynn, B. & O'Mara, F. 2006b. *In vitro* fermentation patterns and methane production as influenced by cultivar and season of harvest of *Lolium perenne* L. *Grass and Forage Science*, 61(1): 9–21. https://doi.org/10.1111/j.1365-2494.2006.00500.x

Lovett, D.K., Shalloo, L., Dillon, P. & O'Mara, F.P. 2006a. A systems approach to quantify greenhouse gas fluxes from pastoral dairy production as affected by management regime. *Agricultural Systems*, 88(2): 156–179. https://doi.org/10.1016/j.agsy.2005.03.006

Lovett, D.K., Shalloo, L., Dillon, P. & O'Mara, F.P. 2008. Greenhouse gas emissions from pastoral based dairying systems: The effect of uncertainty and management change under two contrasting production systems. *Livestock Science*, 116(1): 260–274. https://doi.org/10.1016/j.livsci.2007.10.016

Lovett, D.K., Stack, L.J., Lovell, S., Callan, J., Flynn, B., Hawkins, M. & O'Mara, F.P. 2005. Manipulating enteric methane emissions and animal performance of late-lactation dairy cows through concentrate supplementation at pasture. *Journal of Dairy Science*, 88(8): 2836–2842. https://doi.org/10.3168/jds.S0022-0302(05)72964-7

Lu, Y., Wassmann, R., Neue, H.U. & Huang, C. 1999. Impact of phosphorus supply on root exudation, aerenchyma formation and methane emission of rice plants. *Biogeochemistry*, 47(2): 203–218. https://doi.org/10.1007/BF00994923

Lu, Y., Wassmann, R., Neue, H.-U. & Huang, C. 2000. Dissolved organic carbon and methane emissions from a rice paddy fertilized with ammonium and nitrate. *Journal of Environmental Quality*, 29(6): 1733–1740. https://doi.org/10.2134/jeq2000.00472425002900060002x

Lüscher, A., Mueller-Harvey, I., Soussana, J.F., Rees, R.M. & Peyraud, J.L. 2014. Potential of legume-based grassland–livestock systems in Europe: A review. *Grass and Forage Science*, 69(2): 206–228. https://doi.org/10.1111/gfs.12124

Ly, P., Jensen, L.S., Bruun, T.B. & de Neergaard, A. 2013. Methane (CH_4) and nitrous oxide (N_2O) emissions from the system of rice intensification (SRI) under a rain-fed lowland rice ecosystem in Cambodia. *Nutrient Cycling in Agroecosystems*, 97(1–3): 13–27. https://doi.org/10.1007/s10705-013-9588-3

Lynch, J., Cain, M., Pierrehumbert, R. & Allen, M. 2020. Demonstrating GWP*: A means of reporting warming-equivalent emissions that captures the contrasting impacts of short- and long-lived climate pollutants. *Environmental Research Letters*, 15(4): 044023. https://doi.org/10.1088/1748-9326/ab6d7e

MacAdam, J. & Villalba, J. 2015. Beneficial effects of temperate forage legumes that contain condensed tannins. *Agriculture*, 5(3): 475–491. https://doi.org/10.3390/agriculture5030475

MacDougall, A.H. 2016. The transient response to cumulative CO_2 emissions: A review. *Current Climate Change Reports*, 2(1): 39–47. https://doi.org/10.1007/s40641-015-0030-6

Mace, M.J. 2016. Mitigation commitments under the Paris Agreement and the way forward. *Climate Law*, 6(1–2): 21–39. https://doi.org/10.1163/18786561-00601002.

Machado, L., Magnusson, M., Paul, N.A., de Nys, R. & Tomkins, N. 2014. Effects of marine and freshwater macroalgae on *in vitro* total gas and methane production. *PLoS ONE*, 9(1): e85289. https://doi.org/10.1371/journal.pone.0085289

Machado, L., Magnusson, M., Paul, N.A., Kinley, R., de Nys, R. & Tomkins, N. 2016. Identification of bioactives from the red seaweed *Asparagopsis taxiformis* that promote antimethanogenic activity *in vitro*. *Journal of Applied Phycology*, 28(5): 3117–3126. https://doi.org/10.1007/s10811-016-0830-7

Machmüller, A. 2006. Medium-chain fatty acids and their potential to reduce methanogenesis in domestic ruminants. *Agriculture, Ecosystems & Environment*, 112(2): 107–114. https://doi.org/10.1016/j.agee.2005.08.010

Macome, F.M., Pellikaan, W.F., Hendriks, W.H., Warner, D., Schonewille, J.T. & Cone, J.W. 2018. *In vitro* gas and methane production in rumen fluid from dairy cows fed grass silages differing in plant maturity, compared to *in vivo* data. *Journal of Animal Physiology and Animal Nutrition*, 102(4): 843–852. https://doi.org/10.1111/jpn.12898

Macoon, B., Sollenberger, L.E., Moore, J.E., Staples, C.R., Fike, J.H. & Portier, K.M. 2003. Comparison of three techniques for estimating the forage intake of lactating dairy cows on pasture. *Journal of Animal Science*, 81(9): 2357–2366. https://doi.org/10.2527/2003.8192357x

Madsen, J., Bjerg, B.S., Hvelplund, T., Weisbjerg, M.R. & Lund, P. 2010. Methane and carbon dioxide ratio in excreted air for quantification of the methane production from ruminants. *Livestock Science*, 129(1–3): 223–227. https://doi.org/10.1016/j.livsci.2010.01.001

Maia, M.R.G., Fonseca, A.J.M., Oliveira, H.M., Mendonça, C. & Cabrita, A.R.J. 2016. The potential role of seaweeds in the natural manipulation of rumen fermentation and methane production. *Scientific Reports*, 6(1): 32321. https://doi.org/10.1038/srep32321

Majumdar, D. 2003. Methane and nitrous oxide emission from irrigated rice fields: Proposed mitigation strategies. *Current Science*, 84: 1317–1326.

Mak, T.M.W., Xiong, X., Tsang, D.C.W., Yu, I.K.M. & Poon, C.S. 2020. Sustainable food waste management towards circular bioeconomy: Policy review, limitations and opportunities. *Bioresource Technology*, 297: 122497. https://doi.org/10.1016/j.biortech.2019.122497

Makkar, H.P.S. 2018. Review: Feed demand landscape and implications of food-not feed strategy for food security and climate change. *Animal*, 12(8): 1744–1754. https://doi.org/10.1017/S175173111700324X

Makkar, H.P.S., Tran, G., Heuzé, V., Giger-Reverdin, S., Lessire, M., Lebas, F. & Ankers, P. 2016. Seaweeds for livestock diets: A review. *Animal Feed Science and Technology*, 212: 1–17. https://doi.org/10.1016/j.anifeedsci.2015.09.018

Mallapragada, D.S. & Mignone, B.K. 2020. A theoretical basis for the equivalence between physical and economic climate metrics and implications for the choice of Global Warming Potential time horizon. *Climatic Change*, 158(2): 107–124. https://doi.org/10.1007/s10584-019-02486-7

Malyan, S.K., Bhatia, A., Kumar, A., Gupta, D.K., Singh, R., Kumar, S.S., Tomer, R., Kumar, O. & Jain, N. 2016. Methane production, oxidation and mitigation: A mechanistic understanding and comprehensive evaluation of influencing factors. *Science of the Total Environment*, 572: 874–896. https://doi.org/10.1016/j.scitotenv.2016.07.182

Mamuad, L., Kim, S.H., Jeong, C.D., Choi, Y.J., Jeon, C.O. & Lee, S.-S. 2014. Effect of fumarate reducing bacteria on *in vitro* rumen fermentation, methane mitigation and microbial diversity. *Journal of Microbiology*, 52(2): 120–128. https://doi.org/10.1007/s12275-014-3518-1

Manne, A.S. & Richels, R.G. 2001. An alternative approach to establishing trade-offs among greenhouse gases. *Nature*, 410(6829): 675–677. https://doi.org/10.1038/35070541

Manzanilla-Pech, C.I.V., de Haas, Y., Hayes, B.J., Veerkamp, R.F., Khansefid, M., Donoghue, K.A., Arthur, P.F. & Pryce, J.E. 2016. Genomewide association study of methane emissions in Angus beef cattle with validation in dairy cattle. *Journal of Animal Science*, 94(10): 4151–4166. https://doi.org/10.2527/jas.2016-0431

Manzano, P. 2015. Pastoralist ownership of rural transformation: The adequate path to change. *Development*, 58(2–3): 326–332. https://doi.org/10.1057/s41301-016-0012-6

Manzano, P. & White, S. 2019. Intensifying pastoralism may not reduce greenhouse gas emissions: Wildlife-dominated landscape scenarios as a baseline in life-cycle analysis. *Climate Research*, 77: 91–97. https://doi.org/10.3354/cr01555

Mao, H.-L., Wang, J.-K., Zhou, Y.-Y. & Liu, J.-X. 2010. Effects of addition of tea saponins and soybean oil on methane production, fermentation and microbial population in the rumen of growing lambs. *Livestock Science*, 129(1): 56–62. https://doi.org/10.1016/j.livsci.2009.12.011

Markantonatos, X., Aharoni, Y., Richardson, L.F. & Varga, G.A. 2009. Effects of monensin on volatile fatty acid metabolism in periparturient dairy cows using compartmental analysis. *Animal Feed Science and Technology*, 153(1–2): 11–27. https://doi.org/10.1016/j.anifeedsci.2009.05.007

Markantonatos, X., Green, M.H. & Varga, G.A. 2008. Use of compartmental analysis to study ruminal volatile fatty acid metabolism under steady state conditions in Holstein heifers. *Animal Feed Science and Technology*, 143(1–4): 70–88. https://doi.org/10.1016/j.anifeedsci.2007.05.005

Marquardt, S., Ndung'u, P., Onyango, A.A. & Merbold, L. 2020. Protocol for a Tier 2 approach to generate region-specific enteric methane emission factors (EF) for cattle kept in smallholder systems. International Livestock Research Institute (ILRI) manual No. 39. Nairobi, ILRI. [Cited 20 January 2022]. https://cgspace.cgiar.org/handle/10568/109579

Marten, A.L. & Newbold, S.C. 2012. Estimating the social cost of non-CO_2 GHG emissions: Methane and nitrous oxide. *Energy Policy*, 51: 957–972. https://doi.org/10.1016/j.enpol.2012.09.073

Martin, C., Morgavi, D.P. & Doreau, M. 2010. Methane mitigation in ruminants: From microbe to the farm scale. *Animal*, 4(3): 351–365. https://doi.org/10.1017/S1751731109990620

Martin, C., Pomiès, D., Ferlay, A., Rochette, Y., Martin, B., Chilliard, Y., Morgavi, D.P. & Doreau, M. 2011. Methane output and rumen microbiota in dairy cows in response to long-term supplementation with linseed or rapeseed of grass silage- or pasture-based diets. *Proceedings of the New Zealand Society of Animal Production*, 71: 243–247. [Cited 20 January 2022]. www.nzsap.org/proceedings/2011/methane-output-and-rumen-microbiota-dairy-cows-response-long-term-supplementation

Martin, C., Rouel, J., Jouany, J.P., Doreau, M. & Chilliard, Y. 2008. Methane output and diet digestibility in response to feeding dairy cows crude linseed, extruded linseed, or linseed oil. *Journal of Animal Science*, 86(10): 2642–2650. https://doi.org/10.2527/jas.2007-0774

Martin, S.A. & Macy, J.M. 1985. Effects of monensin, pyromellitic diimide and 2-bromoethanesulfonic acid on rumen fermentation *in vitro*. *Journal of Animal Science*, 60(2): 544–550. https://doi.org/10.2527/jas1985.602544x

Marvin-Sikkema, F.D., Richardson, A.J., Stewart, C.S., Gottschal, J.C. & Prins, R.A. 1990. Influence of hydrogen-consuming bacteria on cellulose degradation by anaerobic fungi. *Applied and Environmental Microbiology*, 56(12): 3793–3797. https://doi.org/10.1128/aem.56.12.3793-3797.1990

Massé, D.I., Jarret, G., Hassanat, F., Benchaar, C. & Saady, N.M.C. 2016. Effect of increasing levels of corn silage in an alfalfa-based dairy cow diet and of manure management practices on manure fugitive methane emissions. *Agriculture, Ecosystems & Environment*, 221: 109–114. https://doi.org/10.1016/j.agee.2016.01.018

Mathers, J.C. & Miller, E.L. 1982. Some effects of chloral hydrate on rumen fermentation and digestion in sheep. *The Journal of Agricultural Science*, 99(1): 215–224. https://doi.org/10.1017/S0021859600055234

Mathison, G.W., Okine, E.K., McAllister, T.A., Dong, Y., Galbraith, J. & Dmytruk, O.I.N. 1998. Reducing methane emissions from ruminant animals. *Journal of Applied Animal Research*, 14(1): 1–28. https://doi.org/10.1080/09712119.1998.9706212

Mathot, M., Decruyenaere, V., Lambert, R. & Stilmant, D. 2016. Deep litter removal frequency rate influences on greenhouse gas emissions from barns for beef heifers and from manure stores. *Agriculture, Ecosystems & Environment*, 233: 94–105. https://doi.org/10.1016/j.agee.2016.08.022

Mathot, M., Decruyenaere, V., Stilmant, D. & Lambert, R. 2012. Effect of cattle diet and manure storage conditions on carbon dioxide, methane and nitrous oxide emissions from tie-stall barns and stored solid manure. *Agriculture, Ecosystems & Environment*, 148: 134–144. https://doi.org/10.1016/j.agee.2011.11.012

Maurer, D.L., Koziel, J.A., Harmon, J.D., Hoff, S.J., Rieck-Hinz, A.M. & Andersen, D.S. 2016. Summary of performance data for technologies to control gaseous, odor, and particulate emissions from livestock operations: Air management practices assessment tool (AMPAT). *Data in Brief*, 7: 1413–1429. https://doi.org/10.1016/j.dib.2016.03.070

Mauricio, R.M., Mould, F.L., Dhanoa, M.S., Owen, E., Channa, K.S. & Theodorou, M.K. 1999. A semi-automated *in vitro* gas production technique for ruminant feedstuff evaluation. *Animal Feed Science and Technology*, 79(4): 321–330. https://doi.org/10.1016/S0377-8401(99)00033-4

Mauricio, R.M., Ribeiro, R.S., Paciullo, D.S.C., Cangussú, M.A., Murgueitio, E., Chará, J. & Estrada, M.X.F. 2019. Silvopastoral systems in Latin America for biodiversity, environmental, and socioeconomic improvements. In: G. Lemaire, P.C.D.F. Carvalho, S. Kronberg & S. Recous, eds. *Agroecosystem diversity: Reconciling contemporary agriculture and environmental quality*, pp. 287–297. Waltham, USA, Elsevier. https://doi.org/10.1016/B978-0-12-811050-8.00018-2

McAllister, T.A., Cheng, K.-J., Okine, E.K. & Mathison, G.W. 1996. Dietary, environmental and microbiological aspects of methane production in ruminants. *Canadian Journal of Animal Science*, 76(2): 231–243. https://doi.org/10.4141/cjas96-035

McAllister, T.A., Meale, S.J., Valle, E., Guan, L.L., Zhou, M., Kelly, W.J., Henderson, G., Attwood, G.T. & Janssen, P.H. 2015. Ruminant nutrition symposium: Use of genomics and transcriptomics to identify strategies to lower ruminal methanogenesis. *Journal of Animal Science*, 93(4): 1431–1449. https://doi.org/10.2527/jas.2014-8329

McAllister, T.A. & Newbold, C.J. 2008. Redirecting rumen fermentation to reduce methanogenesis. *Australian Journal of Experimental Agriculture*, 48(2): 7–13. https://doi.org/10.1071/EA07218

McCauley, J., Labeeuw, L., Jaramillo Madrid, A.C., Nguyen, L., Nghiem, L., Chaves, A. & Ralph, P. 2020. Management of enteric methanogenesis in ruminants by algal-derived feed additives. *Current Pollution Reports*, 6: 188–205. https://doi.org/10.1007/s40726-020-00151-7

McCrabb, G.J., Berger, K.T., Magner, T., May, C. & Hunter, R.A. 1997. Inhibiting methane production in Brahman cattle by dietary supplementation with a novel compound and the effects on growth. *Australian Journal of Agricultural Research*, 48: 323–339. https://doi.org/10.1071/A96119

McDonald, P., Henderson, N. & Heron, S. 1991. *The biochemistry of silage*. Marlow, UK, Chalcombe Publications.

McDonnell, R.P., Hart, K.J., Boland, T.M., Kelly, A.K., McGee, M. & Kenny, D.A. 2016. Effect of divergence in phenotypic residual feed intake on methane emissions, ruminal fermentation, and apparent whole-tract digestibility of beef heifers across three contrasting diets1. *Journal of Animal Science*, 94(3): 1179–1193. https://doi.org/10.2527/jas.2015-0080

McGinn, S.M., Chen, D., Loh, Z., Hill, J., Beauchemin, K.A. & Denmead, O.T. 2008. Methane emissions from feedlot cattle in Australia and Canada. *Australian Journal of Experimental Agriculture*, 48(2): 183–185. https://doi.org/10.1071/EA07204

McGinn, S.M., Flesch, T.K., Crenna, B.P., Beauchemin, K.A. & Coates, T. 2007. Quantifying ammonia emissions from a cattle feedlot using a dispersion model. *Journal of Environmental Quality*, 36(6): 1585–1590. https://doi.org/10.2134/jeq2007.0167

McGinn, S.M., Flesch, T.K., Harper, L.A. & Beauchemin, K.A. 2006. An approach for measuring methane emissions from whole farms. *Journal of Environmental Quality*, 35(1): 14–20. https://doi.org/10.2134/jeq2005.0250

McKeough, P. 2022. A case for ensuring reductions in CO_2 emissions are given priority over reductions in CH_4 emissions in the near term. *Climatic Change*, 174(1–2): 4. https://doi.org/10.1007/s10584-022-03428-6

McLaren, S., Berardy, A., Henderson, A., Holden, N., Huppertz, T., Jolliet, O., De Camillis, C., Renouf, M., Rugani, B., Saarinen, M., van der Pols, J., Vázquez-Rowe, I., Antón Vallejo, A., Bianchi, M., Chaudhary, A., Chen, C., Cooreman-Algoed, M., Dong, H., Grant, T., Green, A., Hallström, E., Hoang, H., Leip, A., Lynch, J., McAuliffe, G., Ridoutt, B., Saget, S., Scherer, L., Tuomisto, H., Tyedmers, P. & van Zanten, H. 2021. *Integration of environment and nutrition in life cycle assessment of food Items: Opportunities and challenges*. Rome, FAO.

McManus, C.M., Rezende Paiva, S. & Faria, D. 2020. Genomics and climate change. *Revue Scientifique et Technique*, 39(2): 481–490. https://doi.org/10.20506/rst.39.2.3099

McSweeney, C.S., Palmer, B., McNeill, D.M. & Krause, D.O. 2001. Microbial interactions with tannins: Nutritional consequences for ruminants. *Animal Feed Science and Technology*, 91(1): 83–93. https://doi.org/10.1016/S0377-8401(01)00232-2

Meale, S.J., Popova, M., Saro, C., Martin, C., Bernard, A., Lagree, M., Yáñez-Ruiz, D.R., Boudra, H., Duval, S. & Morgavi, D.P. 2021. Early life dietary intervention in dairy calves results in a long-term reduction in methane emissions. *Scientific Reports*, 11(1): 3003. https://doi.org/10.1038/s41598-021-82084-9

Meinshausen, M. & Nicholls, Z. 2022. GWP* is a model, not a metric. *Environmental Research Letters*, 17(4): 041002. https://doi.org/10.1088/1748-9326/ac5930

Mengistu, G., McAllister, T., Tamayao, P., Ominski, K., Ribeiro, G., Okine, E. & McGeough, E. 2022. Evaluation of biochar products at two inclusion levels on ruminal *in vitro* methane production and fermentation parameters in a Timothy hay-based diet. *Canadian Journal of Animal Science*, 102: 1–5. https://doi.org/10.1139/cjas-2021-0069

Miller, S.M., Wofsy, S.C., Michalak, A.M., Kort, E.A., Andrews, A.E., Biraud, S.C., Dlugokencky, E.J., Eluszkiewicz, J., Fischer, M.L., Janssens-Maenhout, G., Miller, B.R., Miller, J.B., Montzka, S.A., Nehrkorn, T. & Sweeney, C. 2013. Anthropogenic emissions of methane in the United States. *Proceedings of the National Academy of Sciences*, 110(50): 20018–20022. https://doi.org/10.1073/pnas.1314392110

Miller, T.L. & Wolin, M.J. 2001. Inhibition of growth of methane-producing bacteria of the ruminant forestomach by hydroxymethylglutaryl~SCoA reductase inhibitors. *Journal of Dairy Science*, 84(6): 1445–1448. https://doi.org/10.3168/jds.S0022-0302(01)70177-4

Mills, J.A., Dijkstra, J., Bannink, A., Cammell, S.B., Kebreab, E. & France, J. 2001. A mechanistic model of whole-tract digestion and methanogenesis in the lactating dairy cow: Model development, evaluation, and application. *Journal of Animal Science*, 79(6): 1584. https://doi.org/10.2527/2001.7961584x

Min, B.R., Solaiman, S., Waldrip, H.M., Parker, D., Todd, R.W. & Brauer, D. 2020. Dietary mitigation of enteric methane emissions from ruminants: A review of plant tannin mitigation options. *Animal Nutrition*, 6(3): 231–246. https://doi.org/10.1016/j.aninu.2020.05.002

Mirheidari, A., Torbatinejad, N.M., Shakeri, P. & Mokhtarpour, A. 2020. Effects of biochar produced from different biomass sources on digestibility, ruminal fermentation, microbial protein synthesis and growth performance of male lambs. *Small Ruminant Research*, 183: 106042. https://doi.org/10.1016/j.smallrumres.2019.106042

Misiukiewicz, A., Gao, M., Filipiak, W., Cieslak, A., Patra, A.K. & Szumacher-Strabel, M. 2021. Review: Methanogens and methane production in the digestive systems of nonruminant farm animals. *Animal*, 15(1): 100060. https://doi.org/10.1016/j.animal.2020.100060

Mitsumori, M., Shinkai, T., Takenaka, A., Enishi, O., Higuchi, K., Kobayashi, Y., Nonaka, I., Asanuma, N., Denman, S.E. & McSweeney, C.S. 2012. Responses in digestion, rumen fermentation and microbial populations to inhibition of methane formation by a halogenated methane analogue. *British Journal of Nutrition*, 108(3): 482–491. https://doi.org/10.1017/S0007114511005794

Mizrahi, I., Wallace, R.J. & Moraïs, S. 2021. The rumen microbiome: Balancing food security and environmental impacts. *Nature Reviews Microbiology*, 19(9): 553–566. https://doi.org/10.1038/s41579-021-00543-6

Moate, P., Williams, S., Jacobs, J., Hannah, M., Beauchemin, K., Eckard, R. & Wales, W. 2017. Wheat is more potent than corn or barley for dietary mitigation of enteric methane emissions from dairy cows. *Journal of Dairy Science*, 100: 7139–7153.

Moate, P.J., Williams, S.R.O., Deighton, M.H., Hannah, M.C., Ribaux, B.E., Morris, G.L., Jacobs, J.L, Hill, J. & Wales, W. 2018. Effects of feeding wheat or corn and of rumen fistulation on milk production and methane emissions of dairy cows. *Animal Production Science*, 59(5): 891–905. https://doi.org/10.1071/AN17433

Mohammadi, A., Khoshnevisan, B., Venkatesh, G. & Eskandari, S. 2020. A critical review on advancement and challenges of biochar application in paddy fields: Environmental and life cycle cost analysis. *Processes*, 8(10): 1275. https://doi.org/10.3390/pr8101275

Mohammed, N., Lila, Z.A., Ajisaka, N., Hara, K., Mikuni, K., Hara, K., Kanda, S. & Itabashi, H. 2004. Inhibition of ruminal microbial methane production by beta-cyclodextrin iodopropane, malate and their combination *in vitro*. *Journal of Animal Physiology and Animal Nutrition*, 88(5–6): 188–195. https://doi.org/10.1111/j.1439-0396.2004.00456.x

Mohanty, S., Wassmann, R., Nelson, A., Moya, P. & Jagadish, S.V.K. 2013. *Rice and climate change: Significance for food security and vulnerability*. IRRI discussion paper series No. 49. Los Baños, Philippines, IRRI. https://assets.publishing.service.gov.uk/media/57a08a4440f0b652dd00064c/DPS49_content.pdf

Mohn, J., Zeyer, K., Keck, M., Keller, M., Zähner, M., Poteko, J., Emmenegger, L. & Schrade, S. 2018. A dual tracer ratio method for comparative emission measurements in an experimental dairy housing. *Atmospheric Environment*, 179: 12–22. https://doi.org/10.1016/j.atmosenv.2018.01.057

Molina, I.C., Angarita, E.A., Mayorga, O.L., Chará, J. & Barahona-Rosales, R. 2016. Effect of *Leucaena leucocephala* on methane production of Lucerna heifers fed a diet based on *Cynodon plectostachyus*. *Livestock Science*, 185: 24–29. https://doi.org/10.1016/j.livsci.2016.01.009

Møller, H.B., Moset, V., Brask, M., Weisbjerg, M.R. & Lund, P. 2014. Feces composition and manure derived methane yield from dairy cows: Influence of diet with focus on fat supplement and roughage type. *Atmospheric Environment*, 94: 36–43. https://doi.org/10.1016/j.atmosenv.2014.05.009

Møller, H.B., Sommer, S.G. & Ahring, B.K. 2004. Biological degradation and greenhouse gas emissions during pre-storage of liquid animal manure. *Journal of Environmental Quality*, 33(1): 27–36.

Monjardino, M., Revell, D. & Pannell, D.J. 2010. The potential contribution of forage shrubs to economic returns and environmental management in Australian dryland agricultural systems. *Agricultural Systems*, 103(4): 187–197. https://doi.org/10.1016/j.agsy.2009.12.007

Montes, F., Meinen, R., Dell, C., Rotz, A., Hristov, A.N., Oh, J., Waghorn, G., Gerber, P.J., Henderson, B., Makkar, H.P.S. & Dijkstra, J. 2013. Special topics – Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. *Journal of Animal Science*, 91(11): 5070–5094. https://doi.org/10.2527/jas.2013-6584

Moorby, J.M., Evans, R.T., Scollan, N.D., MacRae, J.C. & Theodorou, M.K. 2006. Increased concentration of water-soluble carbohydrate in perennial ryegrass (*Lolium perenne* L.). Evaluation in dairy cows in early lactation. *Grass and Forage Science*, 61(1): 52–59. https://doi.org/10.1111/j.1365-2494.2006.00507.x

Moraes, L.E., Strathe, A.B., Fadel, J.G., Casper, D.P. & Kebreab, E. 2014. Prediction of enteric methane emissions from cattle. *Global Change Biology*, 20(7): 2140–2148. https://doi.org/10.1111/gcb.12471

Morais, T., Inácio, A., Coutinho, T., Ministro, M., Cotas, J., Pereira, L. & Bahcevandziev, K. 2020. Seaweed potential in the animal feed: A review. *Journal of Marine Science and Engineering*, 8(8): 559. https://doi.org/10.3390/jmse8080559

Morgavi, D.P., Forano, E., Martin, C. & Newbold, C.J. 2010. Microbial ecosystem and methanogenesis in ruminants. *Animal*, 4(7): 1024–1036. https://doi.org/10.1017/S1751731110000546

Morgavi, D.P, Jouany, J.-P. & Martin, C. 2008. Changes in methane emission and rumen fermentation parameters induced by refaunation in sheep. *Australian Journal of Experimental Agriculture*, 48(2): 69-72. https://doi.org/10.1071/EA07236

Morgavi, D.P., Martin, C., Jouany, J.-P. & Ranilla, M.J. 2012. Rumen protozoa and methanogenesis: Not a simple cause-effect relationship. *The British Journal of Nutrition*, 107(3): 388–397. https://doi.org/10.1017/S0007114511002935

Morvay, Y., Bannink, A., France, J., Kebreab, E. & Dijkstra, J. 2011. Evaluation of models to predict the stoichiometry of volatile fatty acid profiles in rumen fluid of lactating Holstein cows. *Journal of Dairy Science*, 94(6): 3063–3080.

Moss, A.R., Givens, D.I. & Garnsworthy, P.C. 1994. The effect of alkali treatment of cereal straws on digestibility and methane production by sheep. *Animal Feed Science and Technology*, 49(3): 245–259. https://doi.org/10.1016/0377-8401(94)90050-7

Moss, R.H., Edmonds, J.A., Hibbard, K.A., Manning, M.R., Rose, S.K., van Vuuren, D.P., Carter, T.R., Emori, S., Kainuma, M., Kram, T., Meehl, G.A., Mitchell, J.F.B., Nakicenovic, N., Riahi, K., Smith, S.J., Stouffer, R.J., Thomson, A.M., Weyant, J.P. & Wilbanks, T.J. 2010. The next generation of scenarios for climate change research and assessment. *Nature*, 463(7282): 747–756. https://doi.org/10.1038/nature08823

Mottet, A., de Haan, C., Falcucci, A., Tempio, G., Opio, C. & Gerber, P. 2017. Livestock: On our plates or eating at our table? A new analysis of the feed/food debate. *Global Food Security*, 14: 1–8. https://doi.org/10.1016/j.gfs.2017.01.001

Mottet, A., Teillard, F., Boettcher, P., De' Besi, G. & Besbes, B. 2018. Review: Domestic herbivores and food security: Current contribution, trends and challenges for a sustainable development. *Animal*, 12: s188–s198. https://doi.org/10.1017/S1751731118002215

Mueller-Harvey, I. 2006. Unravelling the conundrum of tannins in animal nutrition and health. *Journal of the Science of Food and Agriculture*, 86(13): 2010–2037. https://doi.org/10.1002/jsfa.2577

Muetzel, S., Lowe, K., Janssen, P.H., Pacheco, D., Bird, N., Walker, N., Vidoni, O., Schweikert, L., Clasadonte, L. & Kindermann, M. 2019. Towards the application of 3-nitrooxypropanol in pastoral farming system. In: *Proceedings of the 7th Greenhouse Gas and Animal Agriculture (GGAA) Conference. 4–9 August 2019. Iguassu Falls, Brazil.* https://ainfo.cnptia.embrapa.br/digital/bitstream/item/206280/1/GreenhouseGasAnimal.pdf

Muizelaar, W., Groot, M., van Duinkerken, G., Peters, R. & Dijkstra, J. 2021. Safety and transfer study: Transfer of bromoform present in *Asparagopsis taxiformis* to milk and urine of lactating dairy cows. *Foods*, 10(3): 584. https://doi.org/10.3390/foods10030584

Munidasa, S., Eckard, R., Sun, X., Cullen, B., McGill, D., Chen, D. & Cheng, L. 2021. Challenges and opportunities for quantifying greenhouse gas emissions through dairy cattle research in developing countries. *Journal of Dairy Research*, 88(1): 3–7. https://doi.org/10.1017/S0022029921000182

Muñoz, C., Hube, S., Morales, J.M., Yan, T. & Ungerfeld, E.M. 2015. Effects of concentrate supplementation on enteric methane emissions and milk production of grazing dairy cows. *Livestock Science*, 175: 37–46. https://doi.org/10.1016/j.livsci.2015.02.001

Muñoz, C., Letelier, P.A., Ungerfeld, E.M., Morales, J.M., Hube, S. & Pérez-Prieto, L.A. 2016. Effects of pregrazing herbage mass in late spring on enteric methane emissions, dry matter intake, and milk production of dairy cows. *Journal of Dairy Science*, 99(10): 7945–7955. https://doi.org/10.3168/jds.2016-10919

Muñoz, C., Villalobos, R., Peralta, A.M.T., Morales, R., Urrutia, N.L. & Ungerfeld, E.M. 2021. Long-term and carryover effects of supplementation with whole oilseeds on methane emission, milk production and milk fatty acid profile of grazing dairy cows. *Animals*, 11(10): 2978. https://doi.org/10.3390/ani11102978

Muñoz, I. & Schmidt, J.H. 2016. Methane oxidation, biogenic carbon, and the IPCC's emission metrics. Proposal for a consistent greenhouse-gas accounting. *The International Journal of Life Cycle Assessment*, 21(8): 1069–1075. https://doi.org/10.1007/s11367-016-1091-z

Murgueitio, E., Calle, Z., Uribe, F., Calle, A. & Solorio, B. 2011. Native trees and shrubs for the productive rehabilitation of tropical cattle ranching lands. *Forest Ecology and Management*, 261(10): 1654–1663. https://doi.org/10.1016/j.foreco.2010.09.027

Murguia-Flores, F., Arndt, S., Ganesan, A.L., Murray-Tortarolo, G. & Hornibrook, E.R.C. 2018. Soil Methanotrophy Model (MeMo v1.0): A process-based model to quantify global uptake of atmospheric methane by soil. *Geoscientific Model Development*, 11(6): 2009–2032. https://doi.org/10.5194/gmd-11-2009-2018

Murphy, B., Crosson, P., Kelly, A.K. & Prendiville, R. 2017. An economic and greenhouse gas emissions evaluation of pasture-based dairy calf-to-beef production systems. *Agricultural Systems*, 154: 124–132. https://doi.org/10.1016/j.agsy.2017.03.007

Murphy, B., Crosson, P., Kelly, A.K. & Prendiville, R. 2018. Performance, profitability and greenhouse gas emissions of alternative finishing strategies for Holstein-Friesian bulls and steers. *Animal*, 12(11): 2391–2400. https://doi.org/10.1017/S1751731118000034

Murphy, M.R., Baldwin, R.L. & Koong, L.J. 1982. Estimation of stoichiometric parameters for rumen fermentation of roughage and concentrate diets. *Journal of Animal Science*, 55(2): 411–421. https://doi.org/10.2527/jas1982.552411x

Murray, P.J., Gill, E., Balsdon, S.L. & Jarvis, S.C. 2001. A comparison of methane emissions from sheep grazing pastures with differing management intensities. *Nutrient Cycling in Agroecosystems*, 60(1): 93–97. https://doi.org/10.1023/A:1012654928177

Murray, R.M., Bryant, A.M. & Leng, R.A. 1976. Rates of production of methane in the rumen and large intestine of sheep. *British Journal of Nutrition*, 36(1): 1–14. https://doi.org/10.1079/BJN19760053

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T. & Zhang, H. 2013. Anthropogenic and natural radiative forcing. In: T.F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P.M. Midgley, eds. *Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK & New York, USA, Cambridge University Press. www.ipcc.ch/site/assets/uploads/2018/02/WG1AR5_Chapter08_FINAL.pdf

Nagy, A., Fehér, J. & Tamás, J. 2018. Wheat and maize yield forecasting for the Tisza river catchment using MODIS NDVI time series and reported crop statistics. *Computers and Electronics in Agriculture*, 151: 41–49. https://doi.org/10.1016/j.compag.2018.05.035

National Academies of Sciences, Engineering, and Medicine (NASEM). 2016. *Nutrient requirements of beef cattle* (8th ed.). Animal nutrition series. Washington, DC, National Academies Press. https://doi.org/10.17226/19014

National Academies of Sciences, Engineering, and Medicine. 2018. *Improving characterization of anthropogenic methane emissions in the United States*. Washington, DC, National Academies Press. https://doi.org/10.17226/24987

National Research Council. 2000. *Nutrient requirements of beef cattle* (7th revised ed.). Washington, DC, National Academies Press. https://doi.org/10.17226/9791

National Research Council. 2001. *Nutrient requirements of dairy cattle* (7th revised ed.). Washington, DC, National Academies Press. https://doi.org/10.17226/9825

National Research Council. 2007. *Nutrient requirements of small ruminants: Sheep, goats, cervids, and New World camelids*. (7th ed.) Washington, DC, National Academies Press. https://doi.org/10.17226/11654

Nelson, A., Wassmann, R., Sander, B.O. & Palao, L.K. 2015. Climate-determined suitability of the water saving technology 'Alternate Wetting and Drying' in rice systems: A scalable methodology demonstrated for a province in the Philippines. *PLoS ONE*, 10(12): e0145268. https://doi.org/10.1371/journal.pone.0145268

Nelson, D.D., Shorter, J.H., McManus, J.B. & Zahniser, M.S. 2002. Sub-part-per-billion detection of nitric oxide in air using a thermoelectrically cooled mid-infrared quantum cascade laser spectrometer. *Applied Physics B*, 75(2): 343–350. https://doi.org/10.1007/s00340-002-0979-4

Nelson, K.M., Bui, T.Y. & Sander, B.O. 2021. Guide to supporting agricultural NDC implementation: GHG mitigation in rice-production in Vietnam. CGIAR Research Program on Climate Change, Agriculture and Food Security. Hanoi, IRRI. [Cited 1 February 2022]. https://ccafs.cgiar.org/resources/publications/guide-supporting-agricultural-ndc-implementation-ghg-mitigation-rice

Newbold, C.J., de La Fuente, G., Belanche, A., Ramos-Morales, E. & McEwan, N.R. 2015. The role of ciliate protozoa in the rumen. *Frontiers in Microbiology*, 6: 1313. https://doi.org/10.3389/fmicb.2015.01313

Newbold, C.J., El Hassan, S.M., Wang, J., Ortega, M.E. & Wallace, R.J. 1997. Influence of foliage from African multipurpose trees on activity of rumen protozoa and bacteria. *British Journal of Nutrition*, 78(2): 237–249. https://doi.org/10.1079/BJN19970143

Newbold, C.J., Lassalas, B. & Jouany, J.P. 1995. The importance of methanogens associated with ciliate protozoa in ruminal methane production *in vitro*. *Letters in Applied Microbiology*, 21(4): 230–234. https://doi.org/10.1111/j.1472-765X.1995.tb01048.x

Nguyen, S.H., Barnett, M.C. & Hegarty, R.S. 2016. Use of dietary nitrate to increase productivity and reduce methane production of defaunated and faunated lambs consuming protein-deficient chaff. *Animal Production Science*, 56(3): 290. https://doi.org/10.1071/AN15525

Nguyen, T.T.H. 2012. Life cycle assessment of cattle production: Exploring practices and system changes to reduce environmental impacts. Clermont-Ferrand, France, Université Blaise Pascal-Clermont-Ferrand II. PhD dissertation.

Nguyen-Van-Hung, N., Maguyon-Detras, M.C., Migo, M.V., Quilloy, R., Balingbing, C., Chivenge, P. & Gummert, M. 2020. Rice straw overview: Availability, properties, and management practices. In: M. Gummert, P.C. Nguyen-Van-Hung & B. Douthwaite, eds. *Sustainable rice straw management*, pp. 1–13. New York, USA, Springer Cham. https://doi.org/10.1007/978-3-030-32373-8_10

Ni, J.-Q., Heber, A.J., Lim, T.T., Tao, P.C. & Schmidt, A.M. 2008. Methane and carbon dioxide emission from two pig finishing barns. *Journal of Environmental Quality*, 37(6): 2001–2011. https://doi.org/10.2134/jeq2007.0386

Nicholls, Z.R.J., Meinshausen, M., Lewis, J., Gieseke, R., Dommenget, D., Dorheim, K., Fan, C.-S., Fuglestvedt, J.S., Gasser, T., Golüke, U., Goodwin, P., Hartin, C., Hope, A.P., Kriegler, E., Leach, N.J., Marchegiani, D., McBride, L.A., Quilcaille, Y., Rogelj, J., Salawitch, R.J., Samset, B.H., Sandstad, M., Shiklomanov, A.N., Skeie, R.B., Smith, C.J., Smith, S., Tanaka, K., Tsutsui, J. & Xie, Z 2020. Reduced complexity model intercomparison project phase 1: Introduction and evaluation of global-mean temperature response. *Geoscientific Model Development*, 13(11): 5175–5190. https://doi.org/10.5194/gmd-13-5175-2020

Nikaido, H. 1994. Prevention of drug access to bacterial targets: Permeability barriers and active efflux. *Science*, 264(5157): 382–388. https://doi.org/10.1126/science.8153625

Nisbet, E.G., Fisher, R.E., Lowry, D., France, J.L., Allen, G., Bakkaloglu, S., Broderick, T.J., Cain, M., Coleman, M., Fernandez, J., Forster, G., Griffiths, P.T., Iverach, C.P., Kelly, B.F.J., Manning, M.R., Nisbet-Jones, P.B.R., Pyle, J.A., Townsend-Small, A., al-Shalaan, A., Warwick, N. & Zazzeri, G. 2020. Methane mitigation: Methods to reduce emissions, on the path to the Paris Agreement. *Reviews of Geophysics*, 58(1): e2019RG000675. https://doi.org/10.1029/2019RG000675

Nisbet, E.G., Manning, M.R., Dlugokencky, E.J., Fisher, R.E., Lowry, D., Michel, S.E., Myhre, C.L, Platt, S.M., Allen, G., Bousquet, P., Brownlow, R., Cain, M., France, J.L., Hermansen, O., Hossaini, R., Jones, A.E., Levin, I., Manning, A.C., Myhre, G., Pyle, J.A., Vaughn, B.H., Warwick, N.J. & White, J.W.C. 2019. Very strong atmospheric methane growth in the 4 years 2014–2017: Implications for the Paris Agreement. *Global Biogeochemical Cycles*, 33(3): 318–342. https://doi.org/10.1029/2018GB006009

Niu, M., Kebreab, E., Hristov, A.N., Oh, J., Arndt, C., Bannink, A., Bayat, A.R, Brito, A.F., Boland, T., Casper, D., Crompton, L.A., Dijkstra, J., Eugène, M.A., Garnsworthy, P.C., Haque, M.N., Hellwing, A.L.F., Huhtanen, P., Kreuzer, M., Kuhla, B., Lund, P., Madsen, J., Martin, C., McClelland, C.S., McGee, M., Moate, P.J., Muetzel, S., Muñoz, C., O'Kiely, P., Peiren, N., Reynolds, C.K., Schwarm, A., Shingfield, K.J., Storlien, T.M., Weisbjerg, M.R., Yáñez-Ruiz, D.R. & Yu, Z. 2018. Prediction of enteric methane production, yield, and intensity in dairy cattle using an intercontinental database. *Global Change Biology*, 24(8): 3368–3389. https://doi.org/10.1111/gcb.14094

Nkemka, V.N., Beauchemin, K.A. & Hao, X. 2019. Treatment of feces from beef cattle fed the enteric methane inhibitor 3-nitrooxypropanol. *Water Science and Technology*, 80(3): 437–447. https://doi.org/10.2166/wst.2019.302

Nolan, J.V., Leng, R.A., Dobos, R.C. & Boston, R.C. 2014. The production of acetate, propionate and butyrate in the rumen of sheep: Fitting models to ^{14}C- or ^{13}C-labelled tracer data to determine synthesis rates and interconversions. *Animal Production Science*, 54(12): 2082. https://doi.org/10.1071/AN14539

Nollet, L., Demeyer, D. & Verstraete, W. 1997. Effect of 2-bromoethanesulfonic acid and *Peptostreptococcus productus* ATCC 35244 addition on stimulation of reductive acetogenesis in the ruminal ecosystem by selective inhibition of methanogenesis. *Applied and Environmental Microbiology*, 63(1): 194–200. https://doi.org/10.1128/aem.63.1.194-200.1997

Novilla, M.N. 1992. The veterinary importance of the toxic syndrome induced by ionophores. *Veterinary and Human Toxicology*, 34(1): 66–70. [Cited 15 June 2023]. https://pubmed.ncbi.nlm.nih.gov/1621367/

O'Brien, D. & Shalloo, L. 2016. *A review of livestock methane emission factors*. EPA Research Report. Teagasc and Environmental Protection Agency (EPA). [Cited 1 February 2022]. http://hdl.handle.net/11019/2656

O'Brien, D., Shalloo, L., Grainger, C., Buckley, F., Horan, B. & Wallace, M. 2010. The influence of strain of Holstein-Friesian cow and feeding system on greenhouse gas emissions from pastoral dairy farms. *Journal of Dairy Science*, 93(7): 3390–3402. https://doi.org/10.3168/jds.2009-2790

Ocko, I.B., Hamburg, S.P., Jacob, D.J., Keith, D.W., Keohane, N.O., Oppenheimer, M., Roy-Mayhew, J.D., Schrag, D.P. & Pacala, S.W. 2017. Unmask temporal trade-offs in climate policy debates. *Science*, 356(6337): 492–493. https://doi.org/10.1126/science.aaj2350

Odongo, N.E., Or-Rashid, M.M., Kebreab, E., France, J. & McBride, B.W. 2007. Effect of supplementing myristic acid in dairy cow rations on ruminal methanogenesis and fatty acid profile in milk. *Journal of Dairy Science*, 90(4): 1851–1858. https://doi.org/10.3168/jds.2006-541

Ogink, N.W.M., Mosquera, J., Calvet, S. & Zhang, G. 2013. Methods for measuring gas emissions from naturally ventilated livestock buildings: Developments over the last decade and perspectives for improvement. *Biosystems Engineering*, 116(3): 297–308. https://doi.org/10.1016/j.biosystemseng.2012.10.005

Ominski, K., McAllister, T., Stanford, K., Mengistu, G., Kebebe, E.G., Omonijo, F., Cordeiro, M., Legesse, G. & Wittenberg, K. 2021. Utilization of by-products and food waste in livestock production systems: A Canadian perspective. *Animal Frontiers*, 11(2): 55–63. https://doi.org/10.1093/af/vfab004

Omonode, R.A., Vyn, T.J., Smith, D.R., Hegymegi, P. & Gál, A. 2007. Soil carbon dioxide and methane fluxes from long-term tillage systems in continuous corn and corn–soybean rotations. *Soil and Tillage Research*, 95(1): 182–195. https://doi.org/10.1016/j.still.2006.12.004

O'Neill, B.C. 2003. Economics, natural science, and the costs of global warming potentials. *Climatic Change*, 58(3): 251–260. https://doi.org/10.1023/A:1023968127813

O'Neill, J.D. & Wilkinson, J.F. 1977. Oxidation of ammonia by methane oxidizing bacteria and the effects of ammonia on methane oxidation. *Journal of General Microbiology*, 100: 407–412. https://doi.org/10.1099/00221287-100-2-407

Oonk, H., Koopmans, J., Geck, C., Peters, B. & van Bergen, J. 2015. Methane emission reduction from storage of manure and digestate-slurry. *Journal of Integrative Environmental Sciences*, 12(sup1): 121–137. https://doi.org/10.1080/1943815X.2015.1096796

Opio, C., Gerber, P., Mottet, A., Falcucci, A., Tempio, G., MacLeod, M., Vellinga, T., Henderson, B. & Steinfeld, H. 2013. *Greenhouse gas emissions from ruminant supply chains – A global life cycle assessment*. Rome, FAO. www.fao.org/3/i3461e/i3461e.pdf

Osborne, V.R., Radhakrishnan, S., Odongo, N.E., Hill, A.R. & McBride, B.W. 2008. Effects of supplementing fish oil in the drinking water of dairy cows on production performance and milk fatty acid composition. *Journal of Animal Science*, 86(3): 720–729. https://doi.org/10.2527/jas.2007-0342

Owen, E., Smith, T. & Makkar, H. 2012. Successes and failures with animal nutrition practices and technologies in developing countries: A synthesis of an FAO e-conference. *Animal Feed Science and Technology*, 174(3–4): 211–226. https://doi.org/10.1016/j.anifeedsci.2012.03.010

Owen, J.J. & Silver, W.L. 2015. Greenhouse gas emissions from dairy manure management: A review of field-based studies. *Global Change Biology*, 21(2): 550–565. https://doi.org/10.1111/gcb.12687

Owens, F.N. & Goetsch, A.L. 1988. Ruminal fermentation. In: D.C. Church, ed. *The ruminant animal: Digestive physiology and nutrition*, pp. 145–171. Prospect Heights, USA, Waveland Press Inc.

Owens, J.L., Thomas, B.W., Stoeckli, J.L., Beauchemin, K.A., McAllister, T.A., Larney, F.J. & Hao, X. 2020. Greenhouse gas and ammonia emissions from stored manure from beef cattle supplemented 3-nitrooxypropanol and monensin to reduce enteric methane emissions. *Scientific Reports*, 10(1): 19310. https://doi.org/10.1038/s41598-020-75236-w

Özkan Gülzari, Ş., Vosough Ahmadi, B. & Stott, A.W. 2018. Impact of subclinical mastitis on greenhouse gas emissions intensity and profitability of dairy cows in Norway. *Preventive Veterinary Medicine*, 150: 19–29. https://doi.org/10.1016/j.prevetmed.2017.11.021

Pampolino, M.F., Laureles, E.V., Gines, H.C. & Buresh, R.J. 2008. Soil carbon and nitrogen changes in long-term continuous lowland rice cropping. *Soil Science Society of America Journal*, 72(3): 798–807. https://doi.org/10.2136/sssaj2006.0334

Papanastasiou, D.K., McKeen, S.A. & Burkholder, J.B. 2014. The very short-lived ozone depleting substance $CHBr_3$ (bromoform): Revised UV absorption spectrum, atmospheric lifetime and ozone depletion potential. *Atmospheric Chemistry and Physics*, 14(6): 3017–3025. https://doi.org/10.5194/acp-14-3017-2014

Pape, L., Ammann, C., Nyfeler-Brunner, A., Spirig, C., Hens, K. & Meixner, F.X. 2009. An automated dynamic chamber system for surface exchange measurement of non-reactive and reactive trace gases of grassland ecosystems. *Biogeosciences*, 6(3): 405–429. https://doi.org/10.5194/bg-6-405-2009

Parajuli, R., Dalgaard, T. & Birkved, M. 2018. Can farmers mitigate environmental impacts through combined production of food, fuel and feed? A consequential life cycle assessment of integrated mixed crop-livestock system with a green biorefinery. *Science of the Total Environment*, 619–620: 127–143. https://doi.org/10.1016/j.scitotenv.2017.11.082

Pardo, G., Martin-Garcia, I., Arco, A., Yañez-Ruiz, D.R., Moral, R. & del Prado, A. 2016. Greenhouse-gas mitigation potential of agro-industrial by-products in the diet of dairy goats in Spain: A life-cycle perspective. *Animal Production Science*, 56(3): 646. https://doi.org/10.1071/AN15620

Pardo, G., Moral, R., Aguilera, E. & del Prado, A. 2015. Gaseous emissions from management of solid waste: A systematic review. *Global Change Biology*, 21(3): 1313–1327. https://doi.org/10.1111/gcb.12806

Park, K.-H., Thompson, A.G., Marinier, M., Clark, K. & Wagner-Riddle, C. 2006. Greenhouse gas emissions from stored liquid swine manure in a cold climate. *Atmospheric Environment*, 40(4): 618–627. https://doi.org/10.1016/j.atmosenv.2005.09.075

Parker, W.S. 2013. Ensemble modeling, uncertainty and robust predictions. *WIREs Climate Change*, 4(3): 213–223. https://doi.org/10.1002/wcc.220

Parton, W.J., Hartman, M., Ojima, D. & Schimel, D. 1998. DAYCENT and its land surface submodel: Description and testing. *Global and Planetary Change*, 19(1): 35–48. https://doi.org/10.1016/S0921-8181(98)00040-X

Pathak, H., Li, C. & Wassmann, R. 2005. Greenhouse gas emissions from Indian rice fields: Calibration and upscaling using the DNDC model. *Biogeosciences*, 2(2): 113–123.

Patra, A.K. 2013. The effect of dietary fats on methane emissions, and its other effects on digestibility, rumen fermentation and lactation performance in cattle: A meta-analysis. *Livestock Science*, 155(2): 244–254. https://doi.org/10.1016/j.livsci.2013.05.023

Patra, A.K. 2014. A meta-analysis of the effect of dietary fat on enteric methane production, digestibility and rumen fermentation in sheep, and a comparison of these responses between cattle and sheep. *Livestock Science*, 162: 97–103. https://doi.org/10.1016/j.livsci.2014.01.007

Patra, A.K. & Saxena, J. 2009b. Dietary phytochemicals as rumen modifiers: A review of the effects on microbial populations. *Antonie van Leeuwenhoek*, 96(4): 363–375. https://doi.org/10.1007/s10482-009-9364-1

Patra, A.K. & Saxena, J. 2009a. The effect and mode of action of saponins on the microbial populations and fermentation in the rumen and ruminant production. *Nutrition Research Reviews*, 22(2): 204–219. https://doi.org/10.1017/S0954422409990163

Patra, A.K. & Saxena, J. 2011. Exploitation of dietary tannins to improve rumen metabolism and ruminant nutrition. *Journal of the Science of Food and Agriculture*, 91(1): 24–37. https://doi.org/10.1002/jsfa.4152

Patra, A.K. & Yu, Z. 2013. Effective reduction of enteric methane production by a combination of nitrate and saponin without adverse effect on feed degradability, fermentation, or bacterial and archaeal communities of the rumen. *Bioresource Technology*, 148: 352–360. https://doi.org/10.1016/j.biortech.2013.08.140

Patra, A.K. & Yu, Z. 2014. Combinations of nitrate, saponin, and sulfate additively reduce methane production by rumen cultures *in vitro* while not adversely affecting feed digestion, fermentation or microbial communities. *Bioresource Technology*, 155: 129–135. https://doi.org/10.1016/j.biortech.2013.12.099

Patra, A.K. & Yu, Z. 2015b. Effects of adaptation of *in vitro* rumen culture to garlic oil, nitrate, and saponin and their combinations on methanogenesis, fermentation, and abundances and diversity of microbial populations. *Frontiers in Microbiology*, 6: 1434. www.frontiersin.org/articles/10.3389/fmicb.2015.01434

Patra, A.K. & Yu, Z. 2015a. Effects of garlic oil, nitrate, saponin and their combinations supplemented to different substrates on *in vitro* fermentation, ruminal methanogenesis, and abundance and diversity of microbial populations. *Journal of Applied Microbiology*, 119(1): 127–138. https://doi.org/10.1111/jam.12819

Pattanaik, A.K., Sastry, V.R.B., Katiyar, R.C. & Lal, M. 2003. Influence of grain processing and dietary protein degradability on nitrogen metabolism, energy balance and methane production in young calves. *Asian-Australasian Journal of Animal Sciences*, 16(10): 1443–1450. https://doi.org/10.5713/ajas.2003.1443

Pavelka, M., Acosta, M., Kiese, R., Altimir, N., Brümmer, C., Crill, P., Darenova, E., Fuß, R., Gielen, B., Graf, A. & Klemedtsson, L. 2018. Standardisation of chamber technique for CO_2, N_2O and CH_4 fluxes measurements from terrestrial ecosystems. *International Agrophysics*, 32(4): 569–587. https://doi.org/10.1515/intag-2017-0045

Pearson, W., Boermans, H.J., Bettger, W.J., McBride, B.W. & Lindinger, M.I. 2005. Association of maximum voluntary dietary intake of freeze-dried garlic with Heinz body anemia in horses. *American Journal of Veterinary Research*, 66(3): 457–465. https://doi.org/10.2460/ajvr.2005.66.457

Pedersen, S., Blanes-Vidal, V., Jørgensen, H., Chwalibog, A., Haeussermann, A., Heetkamp, M.J.W. & Aarnink, A. 2008. Carbon dioxide production in animal houses: A literature review. *Agricultural Engineering International*, X: 1–9. [Cited 15 June 2023]. https://cigrjournal.org/index.php/Ejounral/article/view/1205

Peischl, J., Ryerson, T.B., Holloway, J.S., Trainer, M., Andrews, A.E., Atlas, E.L., Blake, D.R., Daube, B.C., Dlugokencky, E.J., Fischer, M.L. & Goldstein, A.H. 2012. Airborne observations of methane emissions from rice cultivation in the Sacramento Valley of California. *Journal of Geophysical Research: Atmospheres*, 117(D24): 1–13. https://doi.org/10.1029/2012JD017994

Pell, A.N. & Schofield, P. 1993. Computerized monitoring of gas production to measure forage digestion *in vitro*. *Journal of Dairy Science*, 76(4): 1063–1073. https://doi.org/10.3168/jds.S0022-0302(93)77435-4

Pellerin, S., Bamière, L., Angers, D., Béline, F., Benoit, M., Butault, J.-P., Chenu, C., Colnenne-David, C., De Cara, S., Delame, N., Doreau, M., Dupraz, P., Faverdin, P., Garcia-Launay, F., Hassouna, M., Hénault, C., Jeuffroy, M.-H., Klumpp, K., Metay, A., Moran, D., Recous, S., Samson, E., Savini, I., Pardon, L. & Chemineau, P. 2017. Identifying cost-competitive greenhouse gas mitigation potential of French agriculture. *Environmental Science & Policy*, 77: 130–139. https://doi.org/10.1016/j.envsci.2017.08.003

Pelletier, N., Pirog, R. & Rasmussen, R. 2010. Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. *Agricultural Systems*, 103(6): 380–389. https://doi.org/10.1016/j.agsy.2010.03.009

Peters, G.P., Aamaas, B., Berntsen, T. & Fuglestvedt, J.S. 2011. The integrated global temperature change potential (iGTP) and relationships between emission metrics. *Environmental Research Letters*, 6(4): 044021. https://doi.org/10.1088/1748-9326/6/4/044021

Petersen, S.O., Andersen, A.J. & Eriksen, J. 2012. Effects of cattle slurry acidification on ammonia and methane evolution during storage. *Journal of Environmental Quality*, 41(1): 88–94. https://doi.org/10.2134/jeq2011.0184

Petersen, S.O., Blanchard, M., Chadwick, D., del Prado, A., Edouard, N., Mosquera, J. & Sommer, S.G. 2013a. Manure management for greenhouse gas mitigation. *Animal*, 7: 266–282. https://doi.org/10.1017/S1751731113000736

Petersen, S.O., Dorno, N., Lindholst, S., Feilberg, A. & Eriksen, J. 2013b. Emissions of CH_2, N_2O, NH_3 and odorants from pig slurry during winter and summer storage. *Nutrient Cycling in Agroecosystems*, 95(1): 103–113. https://doi.org/10.1007/s10705-013-9551-3

Petersen, S.O., Hellwing, A.L.F., Brask, M., Højberg, O., Poulsen, M., Zhu, Z., Baral, K.R. & Lund, P. 2015. Dietary nitrate for methane mitigation leads to nitrous oxide emissions from dairy cows. *Journal of Environmental Quality*, 44(4): 1063–1070. https://doi.org/10.2134/jeq2015.02.0107

Pham, C.H., Saggar, S., Vu, C.C., Tate, K.R., Tran, T.T.T., Luu, T.T., Ha, H.T., Nguyen, H.L.T. & Sommer, S.G. 2017. Biogas production from steer manures in Vietnam: Effects of feed supplements and tannin contents. *Waste Management*, 69: 492–497. https://doi.org/10.1016/j.wasman.2017.08.002

Phesatcha, K., Phesatcha, B., Wanapat, M. & Cherdthong, A. 2020. Roughage to concentrate ratio and *Saccharomyces cerevisiae* inclusion could modulate feed digestion and *in vitro* ruminal fermentation. *Veterinary Sciences*, 7(4): 151. https://doi.org/10.3390/vetsci7040151

Philippe, F.-X. & Nicks, B. 2015. Review on greenhouse gas emissions from pig houses: Production of carbon dioxide, methane and nitrous oxide by animals and manure. *Agriculture, Ecosystems & Environment*, 199: 10–25. https://doi.org/10.1016/j.agee.2014.08.015

Pickering, N.K., Oddy, V.H., Basarab, J., Cammack, K., Hayes, B., Hegarty, R.S., Lassen, J., McEwan, J.C., Miller, S., Pinares-Patiño, C.S. & de Haas, Y. 2015. Animal board invited review: Genetic possibilities to reduce enteric methane emissions from ruminants. *Animal*, 9(9): 1431–1440. https://doi.org/10.1017/S1751731115000968

Pihlatie, M.K., Christiansen, J.R., Aaltonen, H., Korhonen, J.F.J., Nordbo, A., Rasilo, T., Benanti, G., Giebels, M., Helmy, M., Sheehy, J. & Jones, S. 2013. Comparison of static chambers to measure CH_4 emissions from soils. *Agricultural and Forest Meteorology*, 171–172: 124–136. https://doi.org/10.1016/j.agrformet.2012.11.008

Pinares-Patiño, C. & Waghorn, G. 2014. *Technical manual on respiration chamber designs*. Wellington, New Zealand, Ministry of Agriculture and Forestry. www.globalresearchalliance.org/wp-content/uploads/2012/03/GRA-MAN-Facility-BestPract-2012-FINAL.pdf

Pinares-Patiño, C.S., Hickey, S.M., Young, E.A., Dodds, K.G., MacLean, S., Molano, G., Sandoval, E., Kjestrup, H., Harland, R., Hunt, C. & Pickering, N.K. 2013. Heritability estimates of methane emissions from sheep. *Animal*, 7(s2): 316–321.

Pindyck, R.S. 2013. Climate change policy: What do the models tell us? *Journal of Economic Literature*, 51(3): 860–872. https://doi.org/10.1257/jel.51.3.860

Pitt, R.E. & Pell, A.N. 1997. Modeling ruminal pH fluctuations: Interactions between meal frequency and digestion rate. *Journal of Dairy Science*, 80(10): 2429–2441. https://doi.org/10.3168/jds.S0022-0302(97)76195-2

Pitt, R.E., van Kessel, J.S., Fox, D.G., Pell, A.N., Barry, M.C. & Van Soest, P.J. 1996. Prediction of ruminal volatile fatty acids and pH within the net carbohydrate and protein system. *Journal of Animal Science*, 74(1): 226. https://doi.org/10.2527/1996.741226x

Place, S.E. & Mitloehner, F.M. 2021. Pathway to climate neutrality for U.S. beef and dairy cattle production. Clear Center Climate Neutrality White Paper. Davis, USA, University of California Davis. https://clear.ucdavis.edu/sites/g/files/dgvnsk7876/files/inline-files/CLEAR%20Center%20Climate%20Neutrality%20White%20Paper_1.pdf

Polonsky, I.N., O'Brien, D.M., Kumer, J.B., O'Dell, C.W. & the geoCARB Team. 2014. Performance of a geostationary mission, geoCARB, to measure CO_2, CH_4 and CO column-averaged concentrations. *Atmospheric Measurement Techniques*, 7(4): 959–981. https://doi.org/10.5194/amt-7-959-2014

Potter, T.L., Arndt, C. & Hristov, A.N. 2018. Short communication: Increased somatic cell count is associated with milk loss and reduced feed efficiency in lactating dairy cows. *Journal of Dairy Science*, 101(10): 9510–9515. https://doi.org/10.3168/jds.2017-14062

Powell, J.M., Aguerre, M.J. & Wattiaux, M.A. 2011. Tannin extracts abate ammonia emissions from simulated dairy barn floors. *Journal of Environmental Quality*, 40(3): 907–914. https://doi.org/10.2134/jeq2010.0492

Powers, W. & Capelari, M. 2016. Analytical methods for quantifying greenhouse gas flux in animal production systems. *Journal of Animal Science*, 94(8): 3139–3146. https://doi.org/10.2527/jas.2015-0017

Pramanik, P. & Kim, P.J. 2014. Evaluating changes in cellulolytic bacterial population to explain methane emissions from air-dried and composted manure treated rice paddy soils. *The Science of the Total Environment*, 470–471: 1307–1312. https://doi.org/10.1016/j.scitotenv.2013.10.108

Pratt, C. & Tate, K. 2018. Mitigating methane: Emerging technologies to combat climate change's second leading contributor. *Environmental Science & Technology*, 52(11): 6084–6097. https://doi.org/10.1021/acs.est.7b04711

Priemé, A. & Ekelund, F. 2001. Five pesticides decreased oxidation of atmospheric methane in a forest soil. *Soil Biology and Biochemistry*, 33(6): 831–835. https://doi.org/10.1016/S0038-0717(00)00246-7

Pszczola, M., Calus, M.P.L. & Strabel, T. 2019. Genetic correlations between methane and milk production, conformation, and functional traits. *Journal of Dairy Science*, 102(6): 5342–5346. https://doi.org/10.3168/jds.2018-16066

Pszczola, M., Strabel, T., Mucha, S. & Sell-Kubiak, E. 2018. Genome-wide association identifies methane production level relation to genetic control of digestive tract development in dairy cows. *Scientific Reports*, 8(1): 15164. https://doi.org/10.1038/s41598-018-33327-9

Pumpanen, J., Kolari, P., Ilvesniemi, H., Minkkinen, K., Vesala, T., Niinistö, S., Lohila, A., Larmola, T., Morero, M., Pihlatie, M. & Janssens, I. 2004. Comparison of different chamber techniques for measuring soil CO_2 efflux. *Agricultural and Forest Meteorology*, 123(3–4): 159–176.

Qu, Q., Groot, J.C.J., Zhang, K. & Schulte, R.P.O. 2021. Effects of housing system, measurement methods and environmental factors on estimating ammonia and methane emission rates in dairy barns: A meta-analysis. *Biosystems Engineering*, 205: 64–75. https://doi.org/10.1016/j.biosystemseng.2021.02.012

Rae, H.A. 1999. Onion toxicosis in a herd of beef cows. *The Canadian Veterinary Journal*, 40(1): 55–57. https://clear.ucdavis.edu/sites/g/files/dgvnsk7876/files/inline-files/CLEAR%20Center%20Climate%20Neutrality%20White%20Paper_1.pdf

Raju, P. 2016. Homoacetogenesis as an alternative hydrogen sink in the rumen. Palmerston North, New Zealand, Massey University. PhD dissertation. https://mro.massey.ac.nz/handle/10179/10037

Ramin, M., Chagas, J.C., Smidt, H., Exposito, R.G. & Krizsan, S.J. 2021. Enteric and fecal methane emissions from dairy cows fed grass or corn silage diets supplemented with rapeseed oil. *Animals*, 11(5): 1322. https://doi.org/10.3390/ani11051322

Ramin, M., Fant, P. & Huhtanen, P. 2021. The effects of gradual replacement of barley with oats on enteric methane emissions, rumen fermentation, milk production, and energy utilization in dairy cows. *Journal of Dairy Science*, 104(5): 5617–5630. https://doi.org/10.3168/jds.2020-19644

Ramírez-Restrepo, C.A., Barry, T.N., Marriner, A., López-Villalobos, N., McWilliam, E.L., Lassey, K.R. & Clark, H. 2010. Effects of grazing willow fodder blocks upon methane production and blood composition in young sheep. *Animal Feed Science and Technology*, 155(1): 33–43. https://doi.org/10.1016/j.anifeedsci.2009.10.003

Ramírez-Restrepo, C.A., Clark, H. & Muetzel, S. 2016. Methane emissions from young and mature dairy cattle. *Animal Production Science*, 56(11): 1897. https://doi.org/10.1071/AN15102

Ramírez-Restrepo, C.A., Waghorn, G.C., Gillespie, H. & Clark, H. 2020. Partition of dietary energy by sheep fed fresh ryegrass (*Lolium perenne*) with a wide-ranging composition and quality. *Animal Production Science*, 60(8): 1008. https://doi.org/10.1071/AN19285

Ramos-Morales, E., de la Fuente, G., Duval, S., Wehrli, C., Bouillon, M., Lahmann, M., Preskett, D., Braganca, R. & Newbold, C.J. 2017a. Antiprotozoal effect of saponins in the rumen can be enhanced by chemical modifications in their structure. *Frontiers in Microbiology*, 8: 399. https://doi.org/10.3389/fmicb.2017.00399

Ramos-Morales, E., de la Fuente, G., Nash, R.J., Braganca, R., Duval, S., Bouillon, M.E., Lahmann, M. & Newbold, C.J. 2017b. Improving the antiprotozoal effect of saponins in the rumen by combination with glycosidase inhibiting iminosugars or by modification of their chemical structure. *PLoS ONE*, 12(9): e0184517. https://doi.org/10.1371/journal.pone.0184517

Rawnsley, R., Dynes, R.A., Christie, K.M., Harrison, M.T., Doran-Browne, N.A., Vibart, R. & Eckard, R. 2016. A review of whole farm-system analysis in evaluating greenhouse-gas mitigation strategies from livestock production systems. *Animal Production Science*, 58(6): 980–989. https://doi.org/10.1071/AN15632

Reba, M.L., Fong, B.N., Rijal, I., Adviento-Borbe, M.A., Chiu, Y.L. & Massey, J.H. 2020. Methane flux measurements in rice by static flux chamber and eddy covariance. *Agrosystems, Geosciences & Environment*, 3(1): 20119.

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.-P., Suh, S., Weidema, B.P. & Pennington, D.W. 2004. Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications. *Environment International*, 30(5): 701–720. https://doi.org/10.1016/j.envint.2003.11.005

Reed, J.D. 1995. Nutritional toxicology of tannins and related polyphenols in forage legumes. *Journal of Animal Science*, 73(5): 1516–1528. https://doi.org/10.2527/1995.7351516x

Reilly, J.M. & Richards, K.R. 1993. Climate change damage and the trace gas index issue. *Environmental and Resource Economics*, 3(1): 41–61. https://doi.org/10.1007/BF00338319

Reisinger, A. & Clark, H. 2018. How much do direct livestock emissions actually contribute to global warming? *Global Change Biology*, 24(4): 1749–1761. https://doi.org/10.1111/gcb.13975

Reisinger, A., Clark, H., Cowie, A.L., Emmet-Booth, J., Gonzalez Fischer, C., Herrero, M., Howden, M. & Leahy, S. 2021. How necessary and feasible are reductions of methane emissions from livestock to support stringent temperature goals? *Philosophical Transactions of the Royal Society. Series A – Mathematical, Physical and Engineering Sciences*, 379(2210): 20200452. https://doi.org/10.1098/rsta.2020.0452

Reisinger, A., Havlik, P., Riahi, K., van Vliet, O., Obersteiner, M. & Herrero, M. 2013. Implications of alternative metrics for global mitigation costs and greenhouse gas emissions from agriculture. *Climatic Change*, 117(4): 677–690. https://doi.org/10.1007/s10584-012-0593-3

Reisinger, A., Ledgard, S.F. & Falconer, S.J. 2017. Sensitivity of the carbon footprint of New Zealand milk to greenhouse gas metrics. *Ecological Indicators*, 81: 74–82. https://doi.org/10.1016/j.ecolind.2017.04.026

Renand, G., Vinet, A., Decruyenaere, V., Maupetit, D. & Dozias, D. 2019. Methane and carbon dioxide emission of beef heifers in relation with growth and feed efficiency. *Animals*, 9(12): 1136. https://doi.org/10.3390/ani9121136

Rennert, K., Errickson, F., Prest, B.C., Rennels, L., Newell, R.G., Pizer, W., Kingdon, C., Wingenroth, J., Cooke, R., Parthum, B., Smith, D., Cromar, K., Diaz, D., Moore, F.C., Müller, U.K., Plevin, R.J., Raftery, A.E., Ševčíková, H., Sheets, H., Stock, J.H., Tan, T., Watson, M., Wong, T.E. & Anthoff, D. 2022. Comprehensive evidence implies a higher social cost of CO_2. *Nature*, 610(7933): 687–692. https://doi.org/10.1038/s41586-022-05224-9

Richardson, C.M., Nguyen, T.T.T., Abdelsayed, M., Moate, P.J., Williams, S.R.O., Chud, T.C.S., Schenkel, F.S., Goddard, M.E., van den Berg, I., Cocks, B.G., Marett, L.C., Wales, W.J. & Pryce, J.E. 2021. Genetic parameters for methane emission traits in Australian dairy cows. *Journal of Dairy Science*, 104(1): 539–549. https://doi.org/10.3168/jds.2020-18565

Ridoutt, B. 2021b. Short communication: Climate impact of Australian livestock production assessed using the GWP* climate metric. *Livestock Science*, 246: 104459. https://doi.org/10.1016/j.livsci.2021.104459

Ridoutt, B. 2021a. Climate neutral livestock production – A radiative forcing-based climate footprint approach. *Journal of Cleaner Production*, 291: 125260. https://doi.org/10.1016/j.jclepro.2020.125260

Ridoutt, B. & Huang, J. 2019. When climate metrics and climate stabilization goals do not align. *Environmental Science & Technology*, 53(24): 14093–14094. https://doi.org/10.1021/acs.est.9b06593

Rivera, J.E., Chará, J. & Barahona, R. 2019. CH_4, CO_2 and N_2O emissions from grasslands and bovine excreta in two intensive tropical dairy production systems. *Agroforestry Systems*, 93(3): 915–928. https://doi.org/10.1007/s10457-018-0187-9

Rivero, M.J., Keim, J.P., Balocchi, O.A. & Lee, M.R.F. 2020. *In vitro* fermentation patterns and methane output of perennial ryegrass differing in water-soluble carbohydrate and nitrogen concentrations. *Animals*, 10(6): 1076. https://doi.org/10.3390/ani10061076

Robin, P., Amand, G., Aubert, C., Babela, N., Brachet, A., Berckmans, D., Burton, C., Canart, B., Cellier, P., Dollé, J.B., Dong, H.M., Durif, M., Ehrlacher, A., Eren Özcan, S., Espagnol, S., Gautier, F., Guingand, N., Guiziou, F., Hartung, E., Hassouna, M., Landrain, P., Lee, I.B., Leleu, C., Li, Y.S., Liao, X.D., Loubet, B., Loyon, L., Luth, Nicks, B., de Oliveira, P.A.V., Ponchant, P., Powers, W., Ramonet, Y., Sommer, S.G., Thiard, J., Wang, K.Y., Xin, H. & Youssef, A. 2010. *Reference procedures for the measurement of emissions from animal housing and storage of animal manure – Full report*. [Cited 30 January 2022]. www6.inrae.fr/animal_emissions_eng/ADEME-Metrology/Results/Full-Report

Robiou du Pont, Y., Jeffery, M.L., Gütschow, J., Rogelj, J., Christoff, P. & Meinshausen, M. 2017. Equitable mitigation to achieve the Paris Agreement goals. *Nature Climate Change*, 7(1): 38–43. https://doi.org/10.1038/nclimate3186

Rochette, P. & Hutchinson, G.L. 2005. Measurement of soil respiration in situ: Chamber techniques. *Micrometeorology in Agricultural Systems*, 47(12): 247–286. https://doi.org/10.2134/agronmonogr47.c12

Rochon, J.J., Doyle, C.J., Greef, J.M., Hopkins, A., Molle, G., Sitzia, M., Scholefield, D. & Smith, C.J. 2004. Grazing legumes in Europe: A review of their status, management, benefits, research needs and future prospects. *Grass and Forage Science*, 59(3): 197–214. https://doi.org/10.1111/j.1365-2494.2004.00423.x

Roehe, R., Dewhurst, R.J., Duthie, C.-A., Rooke, J.A., McKain, N., Ross, D.W., Hyslop, J.J., Waterhouse, A., Freeman, T.C., Watson, M. & Wallace, R.J. 2016. Bovine host genetic variation influences rumen microbial methane production with best selection criterion for low methane emitting and efficiently feed converting hosts based on metagenomic gene abundance. *PLoS Genetics*, 12(2): e1005846. https://doi.org/10.1371/journal.pgen.1005846

Rogelj, J. & Schleussner, C.-F. 2019. Unintentional unfairness when applying new greenhouse gas emissions metrics at country level. *Environmental Research Letters*, 14(11): 114039. https://doi.org/10.1088/1748-9326/ab4928

Romasanta, R.R., Sander, B.O., Gaihre, Y.K., Alberto, Ma.C., Gummert, M., Quilty, J., Nguyen, V.H., Castalone, A.G., Balingbing, C., Sandro, J. & Correa, T.Q. 2017. How does burning of rice straw affect CH_4 and N_2O emissions? A comparative experiment of different on-field straw management practices. *Agriculture, Ecosystems & Environment*, 239: 143–153. https://doi.org/10.1016/j.agee.2016.12.042

Romero, C.M., Redman, A.-A.P.H., Terry, S.A., Hazendonk, P., Hao, X., McAllister, T.A. & Okine, E. 2021. Molecular speciation and aromaticity of biochar-manure: Insights from elemental, stable isotope and solid-state DPMAS ^{13}C NMR analyses. *Journal of Environmental Management*, 280: 111705. https://doi.org/10.1016/j.jenvman.2020.111705

Romero-Perez, A., Okine, E.K., McGinn, S.M., Guan, L.L., Oba, M., Duval, S.M., Kindermann, M. & Beauchemin, K.A. 2014. The potential of 3-nitrooxypropanol to lower enteric methane emissions from beef cattle. *Journal of Animal Science*, 92(10): 4682–4693. https://doi.org/10.2527/jas.2014-7573

Roque, B.M., Salwen, J.K., Kinley, R. & Kebreab, E. 2019a. Inclusion of *Asparagopsis armata* in lactating dairy cows' diet reduces enteric methane emission by over 50 percent. *Journal of Cleaner Production*, 234: 132–138. https://doi.org/10.1016/j.jclepro.2019.06.193

Roque, B.M., Van Lingen, H.J., Vrancken, H. & Kebreab, E. 2019b. Effect of Mootral – a garlic- and citrus-extract-based feed additive – on enteric methane emissions in feedlot cattle. *Translational Animal Science*, 3(4): 1383–1388. https://doi.org/10.1093/tas/txz133

Roque, B.M., Venegas, M., Kinley, R.D., de Nys, R., Duarte, T.L., Yang, X. & Kebreab, E. 2021. Red seaweed (*Asparagopsis taxiformis*) supplementation reduces enteric methane by over 80 percent in beef steers. *PLoS ONE*, 16(3): e0247820. https://doi.org/10.1371/journal.pone.0247820

Rose, S., Khatri-Chhetri, A., Stier, M., Vu, T., Nelson, K. & Dittmer, K. 2021. Agricultural sub-sectors in new and updated NDCs: 2020-2021 dataset. CGIAR Research Program on Climate Change, Agriculture and Food Security. [Cited 5 June 2023]. https://cgspace.cgiar.org/handle/10568/115962

Roslev, P., Iversen, N. & Henriksen, K. 1997. Oxidation and assimilation of atmospheric methane by soil methane oxidizers. *Applied and Environmental Microbiology*, 63(3): 874–880. https://doi.org/10.1128/aem.63.3.874-880.1997

Rotz, C.A., Montes, F. & Chianese, D.S. 2010. The carbon footprint of dairy production systems through partial life cycle assessment. *Journal of Dairy Science*, 93(3): 1266–1282. https://doi.org/10.3168/jds.2009-2162

Rowe, S., Hickey, S., Jonker, A., Hess, M., Janssen, P., Johnson, T., Bryson, B., Knowler, K., Pinares-Patino, C.S., Bain, W., Elmes, S., Young, E., Wing, J., Waller, E., Pickering, N. & McEwan, J.C. 2019. Selection for divergent methane yield in New Zealand sheep – A ten year perspective. In: *Proceedings of the Association for the Advancement of Animal Breeding and Genetics*. Armidale, Australia, Association for the Advancement of Animal Breeding and Genetics (AAABG). [Cited 30 January 2022]. www.researchgate.net/publication/337570664_Selection_for_divergent_methane_yield_in_New_Zealand_sheep_-_A_ten_year_perspective

Ruan, L., Oikawa, P.Y., Géli, M., Verfaillie, J.G., Sturtevant, C.S., Knox, S.H., Nickerson, N., McArthur, G., Creelman, C., Saad, N., Alstad, K.P., Arata, C., Baldocchi, D.D. & Silver, W.L. 2014. Soil greenhouse gas flux measurements with automated and manual static chambers, forced diffusion chamber, and concentration profiles. In: *American Geophysical Union, Fall Meeting Abstracts*. Abstract No. B21F-0111. https://ui.adsabs.harvard.edu/abs/2014AGUFM.B21F0111R

Ruiz-González, A., Debruyne, S., Jeyanathan, J., Vandaele, L., De Campeneere, S. & Fievez, V. 2017. Polyunsaturated fatty acids are less effective to reduce methanogenesis in rumen inoculum from calves exposed to a similar treatment early in life. *Journal of Animal Science*, 95(10): 4677–4686. https://doi.org/10.2527/jas2017.1558

Russell, J.B. 1996. Mechanisms of ionophore action in ruminal bacteria. In: *Scientific update on Rumensin®/Tylan®/Micotil® for the professional feedlot consultant*, pp. E1–E18. Indianapolis, USA, Elenco Animal Health.

Russell, J.B. 2002. *Rumen microbiology and its role in ruminant nutrition*. Ithaca, USA, Agricultural Research Service, U.S. Department of Agriculture. [Cited 30 January 2022]. www.ars.usda.gov/research/software/download/?softwareid=409

Russell, J.B. & Strobel, H.J. 1989. Effect of ionophores on ruminal fermentation. *Applied and Environmental Microbiology*, 55(1): 1–6. https://doi.org/10.1128/aem.55.1.1-6.1989

Russell, J.B. & Wallace, R.J. 1997. Energy-yielding and energy-consuming reactions. In: P.N. Hobson & C.S. Stewart, eds. *The rumen microbial ecosystem*, pp. 246–282. London, Blackie Academic & Professional. https://doi.org/10.1007/978-94-009-1453-7_6

Saari, A., Rinnan, R. & Martikainen, P.J. 2004. Methane oxidation in boreal forest soils: Kinetics and sensitivity to pH and ammonium. *Soil Biology and Biochemistry*, 36(7): 1037–1046. https://doi.org/10.1016/j.soilbio.2004.01.018

Saenab, A., Wiryawan, K.G., Retnani, Y. & Wina, E. 2020. Synergistic effect of biofat and biochar of cashew nutshell on mitigate methane in the rumen. *Jurnal Ilmu Ternak dan Veteriner*, 25(3): 139. https://doi.org/10.14334/jitv.v25i3.2475

Salami, S.A., Luciano, G., O'Grady, M.N., Biondi, L., Newbold, C.J., Kerry, J.P. & Priolo, A. 2019. Sustainability of feeding plant by-products: A review of the implications for ruminant meat production. *Animal Feed Science and Technology*, 251: 37–55. https://doi.org/10.1016/j.anifeedsci.2019.02.006

Saleem, A.M., Ribeiro, G.O., Yang, W.Z., Ran, T., Beauchemin, K.A., McGeough, E.J., Ominski, K.H., Okine, E.K. & McAllister, T.A. 2018. Effect of engineered biocarbon on rumen fermentation, microbial protein synthesis, and methane production in an artificial rumen (RUSITEC) fed a high forage diet1. *Journal of Animal Science*. https://doi.org/10.1093/jas/sky204

Sánchez Zubieta, Á., Savian, J.V., de Souza Filho, W., Wallau, M.O., Gómez, A.M., Bindelle, J., Bonnet, O.J.F. & de Faccio Carvalho, P.C. 2021. Does grazing management provide opportunities to mitigate methane emissions by ruminants in pastoral ecosystems? *Science of the Total Environment*, 754: 142029. https://doi.org/10.1016/j.scitotenv.2020.142029

Sander, B.O., Wassmann, R. & Siopongco, J.D.L.C. 2016. Mitigating greenhouse gas emissions from rice production through water-saving techniques: Potential, adoption and empirical evidence. In: C.T. Hoanh, R. Johnston & V. Smakhtin, eds. *Climate change and agricultural water management in developing countries*, pp. 193–207. Wallingford, UK, CABI. https://doi.org/10.1079/9781780643663.0193

Sándor, R., Ehrhardt, F., Grace, P., Recous, S., Smith, P., Snow, V., Soussana, J.-F., Basso, B., Bhatia, A., Brilli, L. & Doltra, J. 2020. Ensemble modelling of carbon fluxes in grasslands and croplands. *Field Crops Research*, 252: 107791. https://doi.org/10.1016/j.fcr.2020.107791

Santonja, G.G., Georgitzikis, K., Scalet, B.M., Montobbio, P., Roudier, S. & Sancho, L.D. 2017. *Best available techniques (BAT) reference document for the intensive rearing of poultry or pig – Industrial emissions directive 2010/75/EU (Integrated pollution prevention and control)*. Luxembourg, Publications Office. https://doi.org/10.2760/020485

Sar, C., Mwenya, B., Santoso, B., Takaura, K., Morikawa, R., Isogai, N., Asakura, Y., Toride, Y. & Takahashi, J. 2005a. Effect of *Escherichia coli* W3110 on ruminal methanogenesis and nitrate/nitrite reduction *in vitro*. *Animal Feed Science and Technology*, 118(3–4): 295–306. https://doi.org/10.1016/j.anifeedsci.2004.10.004

Sar, C., Mwenya, B., Santoso, B., Takaura, K., Morikawa, R., Isogai, N., Asakura, Y., Toride, Y. & Takahashi, J. 2005b. Effect of *Escherichia coli* wild type or its derivative with high nitrite reductase activity on *in vitro* ruminal methanogenesis and nitrate/nitrite reduction. *Journal of Animal Science*, 83(3): 644–652. https://doi.org/10.2527/2005.833644x

Sarkr, R. 2012. Decision support systems for agrotechnology transfer. In: E. Lichtfouse, ed. *Organic fertilisation, soil quality and human health*, pp. 263–299. Dordrecht, Germany, Springer Dordrecht. https://doi.org/10.1007/978-94-007-4113-3

Saro, C., Hohenester, U.M., Bernard, M., Lagrée, M., Martin, C., Doreau, M., Boudra, H., Popova, M. & Morgavi, D.P. 2018. Effectiveness of interventions to modulate the rumen microbiota composition and function in pre-ruminant and ruminant lambs. *Frontiers in Microbiology*, 9: 1273. https://doi.org/10.3389/fmicb.2018.01273

Sarofim, M.C. & Giordano, M.R. 2018. A quantitative approach to evaluating the GWP timescale through implicit discount rates. *Earth System Dynamics*, 9(3): 1013–1024. https://doi.org/10.5194/esd-9-1013-2018

Sarofim, M.C., Waldhoff, S.T. & Anenberg, S.C. 2017. Valuing the ozone-related health benefits of methane emission controls. *Environmental and Resource Economics*, 66(1): 45–63. https://doi.org/10.1007/s10640-015-9937-6

Sass, R.L., Fisher, F.M., Harcombe, P.A. & Turner, F.T. 1990. Methane production and emission in a Texas rice field. *Global Biogeochemical Cycles*, 4(1): 47–68. https://doi.org/10.1029/GB004i001p00047

Sass, R.L., Fisher, F.M., Turner, F.T. & Jund, M.F. 1991. Methane emission from rice fields as influenced by solar radiation, temperature, and straw incorporation. *Global Biogeochemical Cycles*, 5(4): 335–350. https://doi.org/10.1029/91GB02586

Saunois, M., Bousquet, P., Poulter, B., Peregon, A., Ciais, P., Canadell, J.G., Dlugokencky, E.J., Etiope, G., Bastviken, D., Houweling, S., Janssens-Maenhout, G., Tubiello, F.N., Castaldi, S., Jackson, R.B., Alexe, M., Arora, V.K., Beerling, D.J., Bergamaschi, P., Blake, D.R., Brailsford, G., Brovkin, V., Bruhwiler, L., Crevoisier, C., Crill, P., Covey, K., Curry, C., Frankenberg, C., Gedney, N., Höglund-Isaksson, L., Ishizawa, M., Ito, A., Joos, F., Kim, H.-S., Kleinen, T., Krummel, P., Lamarque, J.-F., Langenfelds, R., Locatelli, R., Machida, T., Maksyutov, S., McDonald, K.C., Marshall, J., Melton, J.R., Morino, I., Naik, V., O'Doherty, S., Parmentier, F.-J.W., Patra, P.K., Peng, C., Peng, S., Peters, G.P., Pison, I., Prigent, C., Prinn, R., Ramonet, M., Riley, W.J., Saito, M., Santini, M., Schroeder, R., Simpson, I.J., Spahni, R., Steele, P., Takizawa, A., Thornton, B.F., Tian, H., Tohjima, Y., Viovy, N., Voulgarakis, A., van Weele, M., van der Werf, G.R., Weiss, R., Wiedinmyer, C., Wilton, D.J., Wiltshire, A., Worthy, D., Wunch, D., Xu, X., Yoshida, Y., Zhang, B., Zhang, Z. & Zhu, Q. 2016. The global methane budget 2000–2012. *Earth System Science Data*, 8(2): 697–751. https://doi.org/10.5194/essd-8-697-2016

Saunois, M., Stavert, A.R., Poulter, B., Bousquet, P., Canadell, J.G., Jackson, R.B., Raymond, P.A., Dlugokencky, E.J., Houweling, S., Patra, P. K., Ciais, P., Arora, V.K., Bastviken, D., Bergamaschi, P., Blake, D.R., Brailsford, G., Bruhwiler, L., Carlson, K. M., Carrol, M., Castaldi, S., Chandra, N., Crevoisier, C., Crill, P. M., Covey, K., Curry, C. L., Etiope, G., Frankenberg, C., Gedney, N., Hegglin, M.I., Höglund-Isakson, L., Hugelius, G., Ishizawa, M., Ito, A., Janssens-Maenhout, G., Jensen, K.M., Joos, F., Kleinen, T., Krummel, P.B., Langenfelds, R.L., Laruelle, G.G., Liu, L., Machida, T., Maksyutov, S., McDonald, K.C., McNorton, J., Miller, P.A., Melton, J.R., Morino, I., Müller, J., Murgia-Flores, F., Naik, V., Niwa, Y., Noce, S., O'Doherty, S., Parker, R.J.,

Peng, C., Peng, S., Peters, G.P., Prigent, C., Prinn, R., Ramonet, M., Regnier, P., Riley, W.J., Rosentreter, J.A., Segers, A., Simpson, I.J., Shi, H., Smith, S.J., Steele, P.L., Thornton, B.F., Tian, H., Tohjima, Y., Tubiello, F.N., Tsuruta, A., Viovy, N., Voulgarakis, A., Weber, T.S., van Weele, M., van der Werf, G.R., Weiss, R. ., Worthy, D., Wunch, D., Yin, Y., Yoshida, Y., Zhang, W., Zhang, Z., Zhao, Y., Zheng, B., Zhu, Q., Zhu, Q. & Zhuang, Q. 2019. The Global Methane Budget 2000–2017. *Earth System Science Data*, 12 (3): 1561–1623. https://doi.org/10.5194/essd-12-1561-2020

Sauvant, D. & Nozière, P. 2016. Quantification of the main digestive processes in ruminants: The equations involved in the renewed energy and protein feed evaluation systems. *Animal*, 10(5): 755–770. https://doi.org/10.1017/S1751731115002670

Savage, K., Phillips, R. & Davidson, E. 2014. High temporal frequency measurements of greenhouse gas emissions from soils. *Biogeosciences*, 11(10): 2709–2720. https://doi.org/10.5194/bg-11-2709-2014

Savian, J.V., Schons, R.M.T., Marchi, D.E., de Freitas, T.S., da Silva Neto, G.F., Mezzalira, J.C., Berndt, A., Bayer, C. & de Faccio Carvalho, P.C. 2018. Rotatinuous stocking: A grazing management innovation that has high potential to mitigate methane emissions by sheep. *Journal of Cleaner Production*, 186: 602–608. https://doi.org/10.1016/j.jclepro.2018.03.162

Scheutz, C. & Fredenslund, A.M. 2019. Total methane emission rates and losses from 23 biogas plants. *Waste Management*, 97: 38–46. https://doi.org/10.1016/j.wasman.2019.07.029

Schilde, M., von Soosten, D., Hüther, L., Meyer, U., Zeyner, A. & Dänicke, S. 2021. Effects of 3-nitrooxypropanol and varying concentrate feed proportions in the ration on methane emission, rumen fermentation and performance of periparturient dairy cows. *Archives of Animal Nutrition*, 75(2): 79–104. https://doi.org/10.1080/1745039X.2021.1877986

Schils, R.L.M., Olesen, J.E., del Prado, A. & Soussana, J.F. 2007. A review of farm level modelling approaches for mitigating greenhouse gas emissions from ruminant livestock systems. *Livestock Science*, 112(3): 240–251. https://doi.org/10.1016/j.livsci.2007.09.005

Schils, R.L.M., Verhagen, A., Aarts, H.F.M. & Šebek, L.B.J. 2005. A farm level approach to define successful mitigation strategies for GHG emissions from ruminant livestock systems. *Nutrient Cycling in Agroecosystems*, 71(2): 163–175. https://doi.org/10.1007/s10705-004-2212-9

Schink, B. 2002. Anaerobic digestion: Concepts, limits and perspectives. *Water Science and Technology*, 45(10): 1–8. https://doi.org/10.2166/wst.2002.0274

Schleussner, C.-F., Nauels, A., Schaeffer, M., Hare, W. & Rogelj, J. 2019. Inconsistencies when applying novel metrics for emissions accounting to the Paris Agreement. *Environmental Research Letters*, 14(12): 124055. https://doi.org/10.1088/1748-9326/ab56e7

Schmalensee, R. 1993. Comparing greenhouse gases for policy purposes. *The Energy Journal*, 14(1): 245–255. www.jstor.org/stable/41322490

Schofield, L.R., Beattie, A.K., Tootill, C.M., Dey, D. & Ronimus, R.S. 2015. Biochemical characterisation of phage pseudomurein endoisopeptidases PeiW and PeiP using synthetic peptides. *Archaea*, 2015: 1–12. https://doi.org/10.1155/2015/828693

Scholtz, M.M., Neser, F.W.C. & Makgahlela, M.L. 2020. A balanced perspective on the importance of extensive ruminant production for human nutrition and livelihoods and its contribution to greenhouse gas emissions. *South African Journal of Science*, 116(9/10). https://doi.org/10.17159/sajs.2020/8192

Schrade, S., Zeyer, K., Gygax, L., Emmenegger, L., Hartung, E. & Keck, M. 2012. Ammonia emissions and emission factors of naturally ventilated dairy housing with solid floors and an outdoor exercise area in Switzerland. *Atmospheric Environment*, 47: 183–194. https://doi.org/10.1016/j.atmosenv.2011.11.015

Schultze-Kraft, R., Rao, I.M., Peters, M., Clements, R.J., Bai, C. & Liu, G. 2018. Tropical forage legumes for environmental benefits: An overview. *Tropical Grasslands-Forrajes Tropicales*, 6(1): 1–14. https://doi.org/10.17138/tgft(6)1-14

Sejian, V., Lal, R., Lakritz, J. & Ezeji, T. 2011. Measurement and prediction of enteric methane emission. *International Journal of Biometeorology*, 55(1): 1–16. https://doi.org/10.1007/s00484-010-0356-7

Setyanto, P., Makarim, A.K., Fagi, A.M., Wassmann, R. & Buendia, L.V. 2000. Crop management affecting methane emissions from irrigated and rainfed rice in central Java (Indonesia). *Nutrient Cycling in Agroecosystems*, 58(1/3): 85–93. https://doi.org/10.1023/A:1009834300790

Shakoor, A., Shahzad, S.M., Chatterjee, N., Arif, M.S., Farooq, T.H., Altaf, M.M., Tufail, M.A., Dar, A.A. & Mehmood, T. 2021. Nitrous oxide emission from agricultural soils: Application of animal manure or biochar? A global meta-analysis. *Journal of Environmental Management*, 285: 112170. https://doi.org/10.1016/j.jenvman.2021.112170

Shalloo, L., Cromie, A. & McHugh, N. 2014. Effect of fertility on the economics of pasture-based dairy systems. *Animal*, 8 (Suppl. 1): 222–231. https://doi.org/10.1017/S1751731114000615

Shindell, D. & Smith, C.J. 2019. Climate and air-quality benefits of a realistic phase-out of fossil fuels. *Nature*, 573(7774): 408–411. https://doi.org/10.1038/s41586-019-1554-z

Shindell, D.T., Fuglestvedt, J.S. & Collins, W.J. 2017. The social cost of methane: Theory and applications. *Faraday Discussions*, 200: 429–451. https://doi.org/10.1039/C7FD00009J

Shine, K.P. 2009. The global warming potential – The need for an interdisciplinary retrial: An editorial comment. *Climatic Change*, 96(4): 467–472. https://doi.org/10.1007/s10584-009-9647-6

Shine, K.P., Allan, R.P., Collins, W.J. & Fuglestvedt, J.S. 2015. Metrics for linking emissions of gases and aerosols to global precipitation changes. *Earth System Dynamics*, 6(2): 525–540. https://doi.org/10.5194/esd-6-525-2015

Shine, K.P., Berntsen, T.K., Fuglestvedt, J.S., Skeie, R.B. & Stuber, N. 2007. Comparing the climate effect of emissions of short- and long-lived climate agents. *Philosophical Transactions of the Royal Society. Series A – Mathematical, Physical and Engineering Sciences*, 365(1856): 1903–1914. https://doi.org/10.1098/rsta.2007.2050

Shine, K.P., Fuglestvedt, J.S., Hailemariam, K. & Stuber, N. 2005. Alternatives to the global warming potential for comparing climate impacts of emissions of greenhouse gases. *Climatic Change*, 68(3): 281–302. https://doi.org/10.1007/s10584-005-1146-9

Shrestha, B.M., Bork, E.W., Chang, S.X., Carlyle, C.N., Ma, Z., Döbert, T.F., Kaliaskar, D. & Boyce, M.S. 2020. Adaptive multi-paddock grazing lowers soil greenhouse gas emission potential by altering extracellular enzyme activity. *Agronomy*, 10(11): 1781. https://doi.org/10.3390/agronomy10111781

Shukla, P.N., Pandey, K.D. & Mishra, V.K. 2013. Environmental determinants of soil methane oxidation and methanotrophs. *Critical Reviews in Environmental Science and Technology*, 43(18): 1945–2011. https://doi.org/10.1080/10643389.2012.672053

Singh, J.S., Singh, S., Raghubanshi, A.S., Singh, S., Kashyap, A.K. & Reddy, V.S. 1997. Effect of soil nitrogen, carbon and moisture on methane uptake by dry tropical forest soils. *Plant and Soil*, 196(1): 115–121. https://doi.org/10.1023/A:1004233208325

Singh, J.S. & Strong, P.J. 2016. Biologically derived fertilizer: A multifaceted bio-tool in methane mitigation. *Ecotoxicology and Environmental Safety*, 124: 267–276. https://doi.org/10.1016/j.ecoenv.2015.10.018

Singh, N., Abagandura, G.O. & Kumar, S. 2020. Short-term grazing of cover crops and maize residue impacts on soil greenhouse gas fluxes in two Mollisols. *Journal of Environmental Quality*, 49(3): 628–639. https://doi.org/10.1002/jeq2.20063

Sistani, K.R., Warren, J.G., Lovanh, N., Higgins, S. & Shearer, S. 2010. Greenhouse gas emissions from swine effluent applied to soil by different methods. *Soil Science Society of America Journal*, 74(2): 429–435. https://doi.org/10.2136/sssaj2009.0076

Sitaula, B.K., Luo, J. & Bakken, L.R. 1992. Rapid analysis of climate gases by wide bore capillary gas chromatography. *Journal of Environmental Quality*, 21(3): 493–496. https://doi.org/10.2134/jeq1992.00472425002100030030x

Sivropoulou, A., Papanikolaou, E., Nikolaou, C., Kokkini, S., Lanaras, T. & Arsenakis, M. 1996. Antimicrobial and cytotoxic activities of origanum essential oils. *Journal of Agricultural and Food Chemistry*, 44(5): 1202–1205. https://doi.org/10.1021/jf950540t

Skytt, T., Nielsen, S.N. & Jonsson, B.-G. 2020. Global warming potential and absolute global temperature change potential from carbon dioxide and methane fluxes as indicators of regional sustainability – A case study of Jämtland, Sweden. *Ecological Indicators*, 110: 105831. https://doi.org/10.1016/j.ecolind.2019.105831

Śliwiński, B.J., Kreuzer, M., Wettstein, H.R. & Machmüller, A. 2002. Rumen fermentation and nitrogen balance of lambs fed diets containing plant extracts rich in tannins and saponins, and associated emissions of nitrogen and methane. *Archiv für Tierernahrung*, 56(6): 379–392. https://doi.org/10.1080/00039420215633

Smith, L.G., Kirk, G.J.D., Jones, P.J. & Williams, A.G. 2019. The greenhouse gas impacts of converting food production in England and Wales to organic methods. *Nature Communications*, 10(1): 4641. https://doi.org/10.1038/s41467-019-12622-7

Smith, M.A., Cain, M. & Allen, M.R. 2021. Further improvement of warming-equivalent emissions calculation. *npj Climate and Atmospheric Science*, 4(1): 19. https://doi.org/10.1038/s41612-021-00169-8

Smith, S.M., Lowe, J.A., Bowerman, N.H.A., Gohar, L.K., Huntingford, C. & Allen, M.R. 2012. Equivalence of greenhouse-gas emissions for peak temperature limits. *Nature Climate Change*, 2(7): 535–538. https://doi.org/10.1038/nclimate1496

Sokolov, V., VanderZaag, A., Habtewold, J., Dunfield, K., Tambong, J.T., Wagner-Riddle, C., Venkiteswaran, J.J. & Gordon, R. 2020. Acidification of residual manure in liquid dairy manure storages and its effect on greenhouse gas emissions. *Frontiers in Sustainable Food Systems*, 4: 568648. https://doi.org/10.3389/fsufs.2020.568648

Söllinger, A., Tveit, A.T., Poulsen, M., Noel, S.J., Bengtsson, M., Bernhardt, J., Frydendahl Hellwing, A.L., Lund, P., Riedel, K., Schleper, C., Højberg, O. & Urich, T. 2018. Holistic assessment of rumen microbiome dynamics through quantitative metatranscriptomics reveals multifunctional redundancy during key steps of anaerobic feed degradation. *mSystems*, 3(4): e00038-18. https://doi.org/10.1128/mSystems.00038-18

Solomon, S., Daniel, J.S., Sanford, T.J., Murphy, D.M., Plattner, G.-K., Knutti, R. & Friedlingstein, P. 2010. Persistence of climate changes due to a range of greenhouse gases. *Proceedings of the National Academy of Sciences*, 107(43): 18354–18359. https://doi.org/10.1073/pnas.1006282107

Sommer, S.G. & Husted, S. 1995. A simple model of pH in slurry. *The Journal of Agricultural Science*, 124(3): 447–453. https://doi.org/10.1017/S0021859600073408

Sommer, S.G., Petersen, S.O. & Møller, H.B. 2004. Algorithms for calculating methane and nitrous oxide emissions from manure management. *Nutrient Cycling in Agroecosystems*, 69(2): 143–154. https://doi.org/10.1023/B:FRES.0000029678.25083.fa

Sommer, S.G., Sherlock, R.R. & Khan, R.Z. 1996. Nitrous oxide and methane emissions from pig slurry amended soils. *Soil Biology and Biochemistry*, 28(10): 1541–1544. https://doi.org/10.1016/S0038-0717(96)00146-0

Sorg, D. 2021. Measuring livestock CH_4 emissions with the laser methane detector: A review. *Methane*, 1(1): 38–57. https://doi.org/10.3390/methane1010004

Soteriades, A.D., Gonzalez-Mejia, A.M., Styles, D., Foskolos, A., Moorby, J.M. & Gibbons, J.M. 2018. Effects of high-sugar grasses and improved manure management on the environmental footprint of milk production at the farm level. *Journal of Cleaner Production*, 202: 1241–1252. https://doi.org/10.1016/j.jclepro.2018.08.206

Soussana, J.-F., Loiseau, P., Vuichard, N., Ceschia, E., Balesdent, J., Chevallier, T. & Arrouays, D. 2004. Carbon cycling and sequestration opportunities in temperate grasslands. *Soil Use and Management*, 20(2): 219–230. https://doi.org/10.1111/j.1475-2743.2004.tb00362.x

Soussana, J.F., Tallec, T. & Blanfort, V. 2010. Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. *Animal*, 4(3): 334–350. https://doi.org/10.1017/S1751731109990784

Spahni, R., Wania, R., Neef, L., van Weele, M., Pison, I., Bousquet, P., Frankenberg, C., Foster, P.N., Joos, F., Prentice, I.C. & van Velthoven, P. 2011. Constraining global methane emissions and uptake by ecosystems. *Biogeosciences*, 8(6): 1643–1665. https://doi.org/10.5194/bg-8-1643-2011

Sparrevik, M., Adam, C., Martinsen, V., Jubaedah & Cornelissen, G. 2015. Emissions of gases and particles from charcoal/biochar production in rural areas using medium-sized traditional and improved “retort” kilns. *Biomass and Bioenergy*, 72: 65–73. https://doi.org/10.1016/j.biombioe.2014.11.016

Sperber, J.L., Troyer, B., Norman, M., McPhillips, L.J., Watson, A.K. & Erickson, G.E. 2021. PSIV-7 effect of biochar supplementation in beef cattle growing diets on greenhouse gas emissions. *Journal of Animal Science*, 99(Suppl. 1): 211–212. https://doi.org/10.1093/jas/skab054.347

Sriphirom, P., Chidthaisong, A., Yagi, K., Tripetchkul, S. & Towprayoon, S. 2020. Evaluation of biochar applications combined with alternate wetting and drying (AWD) water management in rice field as a methane mitigation option for farmers' adoption. *Soil Science and Plant Nutrition*, 66(1): 235–246. https://doi.org/10.1080/00380768.2019.1706431

SRP (Sustainable Rice Platform). 2020. SRP performance indicators for sustainable rice cultivation (Version 2.1). Bangkok, Sustainable Rice Platform. https://sustainablerice.org/wp-content/uploads/2022/12/203-SRP-Performance-Indicators-Version-2.1.pdf

Staebell, C., Sun, K., Samra, J., Franklin, J., Chan Miller, C., Liu, X., Conway, E., Chance, K., Milligan, S. & Wofsy, S. 2021. Spectral calibration of the MethaneAIR instrument. *Atmospheric Measurement Techniques*, 14(5): 3737–3753. https://doi.org/10.5194/amt-14-3737-2021

Staerfl, S.M., Amelchanka, S.L., Kälber, T., Soliva, C.R., Kreuzer, M. & Zeitz, J.O. 2012b. Effect of feeding dried high-sugar ryegrass ('AberMagic') on methane and urinary nitrogen emissions of primiparous cows. *Livestock Science*, 150(1–3): 293–301. https://doi.org/10.1016/j.livsci.2012.09.019

Staerfl, S.M., Zeitz, J.O., Kreuzer, M. & Soliva, C.R. 2012a. Methane conversion rate of bulls fattened on grass or maize silage as compared with the IPCC default values, and the long-term methane mitigation efficiency of adding acacia tannin, garlic, maca and lupine. *Agriculture, Ecosystems & Environment*, 148: 111–120. https://doi.org/10.1016/j.agee.2011.11.003

Stavins, R., Zou, J., Brewer, T., Grand, M.C., Elzen, M., Finus, M. & Winkler, H. 2014. *Climate change 2014: Mitigation of climate change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 1001–1082. Cambridge, UK & New York, USA, Cambridge University Press.

Stefenoni, H.A., Räisänen, S.E., Cueva, S.F., Wasson, D.E., Lage, C.F.A., Melgar, A., Fetter, M.E., Smith, P., Hennessy, M., Vecchiarelli, B., Bender, J., Pitta, D., Cantrell, C.L., Yarish, C. & Hristov, A.N. 2021. Effects of the macroalga *Asparagopsis taxiformis* and oregano leaves on methane emission, rumen fermentation, and lactational performance of dairy cows. *Journal of Dairy Science*, 104(4): 4157–4173. https://doi.org/10.3168/jds.2020-19686

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M. & Haan, C. 2006. *Livestock's long shadow – Environmental issues and options*. Rome, FAO. https://www.fao.org/3/a0701e/a0701e.pdf

Steinkamp, R., Butterbach-Bahl, K. & Papen, H. 2001. Methane oxidation by soils of an N limited and N fertilized spruce forest in the Black Forest, Germany. *Soil Biology and Biochemistry*, 33(2): 145–153. https://doi.org/10.1016/S0038-0717(00)00124-3

Sterner, E., Johansson, D.J.A. & Azar, C. 2014. Emission metrics and sea level rise. *Climatic Change*, 127(2): 335–351. https://doi.org/10.1007/s10584-014-1258-1

Stevens, C.J. & Quinton, J.N. 2009. Diffuse pollution swapping in arable agricultural systems. *Critical Reviews in Environmental Science and Technology*, 39(6): 478–520. https://doi.org/10.1080/10643380801910017

Stewart, E.K., Beauchemin, K.A., Dai, X., MacAdam, J.W., Christensen, R.G. & Villalba, J.J. 2019. Effect of tannin-containing hays on enteric methane emissions and nitrogen partitioning in beef cattle. *Journal of Animal Science*, 97(8): 3286–3299. https://doi.org/10.1093/jas/skz206

Storm, I.M.L.D., Hellwing, A.L.F., Nielsen, N.I. & Madsen, J. 2012. Methods for measuring and estimating methane emission from ruminants. *Animals*, 2(2): 160–183. https://doi.org/10.3390/ani2020160

Strefler, J., Luderer, G., Aboumahboub, T. & Kriegler, E. 2014. Economic impacts of alternative greenhouse gas emission metrics: A model-based assessment. *Climatic Change*, 125(3): 319–331. https://doi.org/10.1007/s10584-014-1188-y

Styles, D., Dominguez, E.M. & Chadwick, D. 2016. Environmental balance of the UK biogas sector: An evaluation by consequential life cycle assessment. *Science of the Total Environment*, 560–561: 241–253. https://doi.org/10.1016/j.scitotenv.2016.03.236

Su, J., Hu, C., Yan, X., Jin, Y., Chen, Z., Guan, Q., Wang, Y., Schnürer, A. & Sun, C. 2015. Expression of barley SUSIBA2 transcription factor yields high-starch low-methane rice. *Nature*, 523(7562): 602–606. https://doi.org/10.1038/nature14673

Subharat, S., Shu, D., Zheng, T., Buddle, B.M., Janssen, P.H., Luo, D. & Wedlock, D.N. 2015. Vaccination of cattle with a methanogen protein produces specific antibodies in the saliva which are stable in the rumen. *Veterinary Immunology and Immunopathology*, 164(3–4): 201–207. https://doi.org/10.1016/j.vetimm.2015.02.008

Subharat, S., Shu, D., Zheng, T., Buddle, B.M., Kaneko, K., Hook, S., Janssen, P.H. & Wedlock, D.N. 2016. Vaccination of sheep with a methanogen protein provides insight into levels of antibody in saliva needed to target ruminal methanogens. *PLoS ONE*, 11: 0159861. https://doi.org/10.1371/journal.pone.0159861

Sun, D.-S., Jin, X., Shi, B., Xu, Y. & Yan, S. 2017. Effects of *Yucca schidigera* on gas mitigation in livestock production: A review. *Brazilian Archives of Biology and Technology*, 60: e160359. https://doi.org/10.1590/1678-4324-2017160359

Sun, T., Ocko, I.B., Sturcken, E. & Hamburg, S.P. 2021. Path to net zero is critical to climate outcome. *Scientific Reports*, 11(1): 22173. https://doi.org/10.1038/s41598-021-01639-y

Susilawati, H.L., Setyanto, P., Makarim, A.K., Ariani, M., Ito, K. & Inubushi, K. 2015. Effects of steel slag applications on CH_4, N_2O and the yields of Indonesian rice fields: A case study during two consecutive rice-growing seasons at two sites. *Soil Science and Plant Nutrition*, 61(4): 704–718. https://doi.org/10.1080/00380768.2015.1041861

Suybeng, B., Charmley, E., Gardiner, C.P., Malau-Aduli, B.S. & Malau-Aduli, A.E.O. 2019. Methane emissions and the use of *Desmanthus* in beef cattle production in northern Australia. *Animals*, 9(8): 542. https://doi.org/10.3390/ani9080542

Swain, C.K., Nayak, A.K., Bhattacharyya, P., Chatterjee, D., Chatterjee, S., Tripathi, R., Singh, N.R. & Dhal, B. 2018. Greenhouse gas emissions and energy exchange in wet and dry season rice: Eddy covariance-based approach. *Environmental Monitoring and Assessment*, 190(7): 423. https://doi.org/10.1007/s10661-018-6805-1

Szanto, G., Hamelers, H., Rulkens, W. & Veeken, A. 2007. NH_3, N_2O and CH_4 emissions during passively aerated composting of straw-rich pig manure. *Bioresource Technology*, 98(14): 2659–2670. https://doi.org/10.1016/j.biortech.2006.09.021

Szczechowiak-Piglas, J., Szumacher-Strabel, M., El-Sherbiny, M., Bryszak, M., Stochmal, A. & Adam, C. 2016. Effect of dietary supplementation with *Saponaria officinalis* root on rumen and milk fatty acid proportion in dairy cattle. *Animal Science Papers and Reports*, 34(3): 221–232.

Tamayao, P.J., Ribeiro, G.O., McAllister, T.A., Ominski, K.H., Saleem, A.M., Yang, H.E., Okine, E.K. & McGeough, E.J. 2021b. Effect of pine-based biochars with differing physiochemical properties on methane production, ruminal fermentation, and rumen microbiota in an artificial rumen (RUSITEC) fed barley silage. *Canadian Journal of Animal Science*, 101(3): 577–589. https://doi.org/10.1139/cjas-2020-0129

Tamayao, P.J., Ribeiro, G.O., McAllister, T.A., Yang, H.E., Saleem, A.M., Ominski, K.H., Okine, E.K. & McGeough, E.J. 2021a. Effects of post-pyrolysis treated biochars on methane production, ruminal fermentation, and rumen microbiota of a silage-based diet in an artificial rumen system (RUSITEC). *Animal Feed Science and Technology*, 273: 114802. https://doi.org/10.1016/j.anifeedsci.2020.114802

Tanaka, K., Boucher, O., Ciais, P., Johansson, D.J.A. & Morfeldt, J. 2021. Cost-effective implementation of the Paris Agreement using flexible greenhouse gas metrics. *Science Advances*, 7(22): eabf9020. https://doi.org/10.1126/sciadv.abf9020

Tanaka, K., Cavalett, O., Collins, W.J. & Cherubini, F. 2019. Asserting the climate benefits of the coal-to-gas shift across temporal and spatial scales. *Nature Climate Change*, 9(5): 389–396. https://doi.org/10.1038/s41558-019-0457-1

Tanaka, K., Johansson, D.J.A., O'Neill, B.C. & Fuglestvedt, J.S. 2013. Emission metrics under the 2 °C climate stabilization target. *Climatic Change*, 117(4): 933–941. https://doi.org/10.1007/s10584-013-0693-8

Tanaka, K., Kriegler, E., Bruckner, T., Hooss, G., Knorr, W. & Raddatz, T. 2007. Aggregated carbon cycle, atmospheric chemistry, and climate model (ACC2): Description of the forward and inverse modes. *Reports on Earth System Science*, vol. 40. Hambourg, Germany, Max Planck Institute for Meteorology. http://hdl.handle.net/11858/00-001M-0000-0011-FB8C-3

Tanaka, K. & O'Neill, B.C. 2018. The Paris Agreement zero-emissions goal is not always consistent with the 1.5 °C and 2 °C temperature targets. *Nature Climate Change*, 8(4): 319–324. https://doi.org/10.1038/s41558-018-0097-x

Tanaka, K., O'Neill, B.C., Rokityanskiy, D., Obersteiner, M. & Tol, R.S.J. 2009a. Evaluating global warming potentials with historical temperature. *Climatic Change*, 96(4): 443–466. https://doi.org/10.1007/s10584-009-9566-6

Tanaka, K., Peters, G.P. & Fuglestvedt, J.S. 2010. Policy update: Multicomponent climate policy: Why do emission metrics matter? *Carbon Management*, 1(2): 191–197. https://doi.org/10.4155/cmt.10.28

Tanaka, K., Raddatz, T., O'Neill, B.C. & Reick, C.H. 2009b. Insufficient forcing uncertainty underestimates the risk of high climate sensitivity. *Geophysical Research Letters*, 36(16): L16709. https://doi.org/10.1029/2009GL039642

Tang, L., Ii, R., Tokimatsu, K. & Itsubo, N. 2018. Development of human health damage factors related to CO_2 emissions by considering future socioeconomic scenarios. *The International Journal of Life Cycle Assessment*, 23(12): 2288–2299. https://doi.org/10.1007/s11367-015-0965-9

Tang, S., Ma, L., Wei, X., Tian, D., Wang, B., Li, Z., Zhang, Y. & Shao, X. 2019a. Methane emissions in grazing systems in grassland regions of China: A synthesis. *Science of the Total Environment*, 654: 662–670. https://doi.org/10.1016/j.scitotenv.2018.11.102

Tang, S., Wang, K., Xiang, Y., Tian, D., Wang, J., Liu, Y., Cao, B., Guo, D. & Niu, S. 2019b. Heavy grazing reduces grassland soil greenhouse gas fluxes: A global meta-analysis. *Science of the Total Environment*, 654: 1218–1224. https://doi.org/10.1016/j.scitotenv.2018.11.082

Tannant, D., Smith, K., Cahill, A., Hawthorne, I., Forde, O., Black, A. & Beckie, R. 2018. *Evaluation of a drone and laser-based methane sensor for detection of fugitive methane emissions: Draft submitted to BC Oil and Gas Research and Innovation Society*. Vancouver, Canada, University of British Columbia.

Tatsuoka, N., Hara, K., Mikuni, K., Hara, K., Hashimoto, H. & Itabashi, H. 2008. Effects of the essential oil cyclodextrin complexes on ruminal methane production *in vitro*. *Animal Science Journal*, 79(1): 68–75. https://doi.org/10.1111/j.1740-0929.2007.00499.x

Taylor, R.F., McGee, M., Kelly, A.K. & Crosson, P. 2020. Bioeconomic and greenhouse gas emissions modelling of the factors influencing technical efficiency of temperate grassland-based suckler calf-to-beef production systems. *Agricultural Systems*, 183: 102860. https://doi.org/10.1016/j.agsy.2020.102860

Tedeschi, L.O., Abdalla, A.L., Álvarez, C., Anuga, S.W., Arango, J., Beauchemin, K.A., Becquet, P., Berndt, A., Burns, R., De Camillis, C., Chará, J., Echazarreta, J.M., Hassouna, M., Kenny, D., Mathot, M., Mauricio, R.M., McClelland, S.C., Niu, M., Onyango, A.A., Parajuli, R., Pereira, L.G.R., del Prado, A., Tieri, M.P., Uwizeye, A. & Kebreab, E. 2022. Quantification of methane emitted by ruminants: A review of methods. *Journal of Animal Science*, 100(7): 1–22. https://doi.org/10.1093/jas/skac197

Tedeschi, L.O. & Fox, D.G., eds. 2020a. *The ruminant nutrition system: Volume I – An applied model for predicting nutrient requirements and feed utilization in ruminants*. Ann Arbor, USA, XanEdu.

Tedeschi, L.O. & Fox, D.G., eds. 2020b. *The ruminant nutrition system: Volume II – Tables of equations and coding.* Ann Arbor, USA, XanEdu.

Tedeschi, L.O., Molle, G., Menendez, H.M., Cannas, A. & Fonseca, M.A. 2019. The assessment of supplementation requirements of grazing ruminants using nutrition models. *Translational Animal Science*, 3(2): 811–828. https://doi.org/10.1093/tas/txy140

Tedeschi, L.O., Muir, J.P., Naumann, H.D., Norris, A.B., Ramírez-Restrepo, C.A. & Mertens-Talcott, S.U. 2021. Nutritional aspects of ecologically relevant phytochemicals in ruminant production. *Frontiers in Veterinary Science*, 8: 628445. https://doi.org/10.3389/fvets.2021.628445

Teferedegne, B., McIntosh, F., Osuji, P.O., Odenyo, A., Wallace, R.J. & Newbold, C.J. 1999. Influence of foliage from different accessions of the sub-tropical leguminous tree, *Sesbania sesban*, on ruminal protozoa in Ethiopian and Scottish sheep. *Animal Feed Science and Technology*, 78(1): 11–20. https://doi.org/10.1016/S0377-8401(98)00272-7

Terry, S.A., Redman, A.-A.P., Ribeiro, G.O., Chaves, A.V., Beauchemin, K.A., Okine, E. & McAllister, T.A. 2020. Effect of a pine enhanced biochar on growth performance, carcass quality, and feeding behavior of feedlot steers. *Translational Animal Science*, 4(2): 831–838. https://doi.org/10.1093/tas/txaa011

Terry, S.A., Ribeiro, G.O., Gruninger, R.J., Chaves, A.V., Beauchemin, K.A., Okine, E. & McAllister, T.A. 2019. A pine enhanced biochar does not decrease enteric CH_4 emissions, but alters the rumen microbiota. *Frontiers in Veterinary Science*, 6: 308. https://doi.org/10.3389/fvets.2019.00308

Thauer, R.K., Kaster, A.-K., Seedorf, H., Buckel, W. & Hedderich, R. 2008. Methanogenic archaea: Ecologically relevant differences in energy conservation. *Nature Reviews Microbiology*, 6(8): 579–591. https://doi.org/10.1038/nrmicro1931

Theodorou, M.K., Williams, B.A., Dhanoa, M.S., McAllan, A.B. & France, J. 1994. A simple gas production method using a pressure transducer to determine the fermentation kinetics of ruminant feeds. *Animal Feed Science and Technology*, 48(3–4): 185–197. https://doi.org/10.1016/0377-8401(94)90171-6

Theurer, C.B. 1986. Grain processing effects on starch utilization by ruminants. *Journal of Animal Science*, 63(5): 1649–1662. https://doi.org/10.2527/jas1986.6351649x

Thiel, A., Rümbeli, R., Mair, P., Yeman, H. & Beilstein, P. 2019a. 3-NOP: ADME studies in rats and ruminating animals. *Food and Chemical Toxicology*, 125: 528-539. https://doi.org/10.1016/j.fct.2019.02.002

Thiel, A., Schoenmakers, A.C.M., Verbaan, I.A.J., Chenal, E., Etheve, S. & Beilstein, P. 2019b. 3-NOP: Mutagenicity and genotoxicity assessment. *Food and Chemical Toxicology*, 123: 566–573. https://doi.org/10.1016/j.fct.2018.11.010

Thompson, L.R. & Rowntree, J.E. 2020. Invited review: Methane sources, quantification, and mitigation in grazing beef systems. *Applied Animal Science*, 36(4): 556–573. https://doi.org/10.15232/aas.2019-01951

Thomson, D.J. 1972. Physical form of the diet in relation to rumen fermentation. *Proceedings of the Nutrition Society*, 31(2): 127–134. https://doi.org/10.1079/PNS19720026

Thorman, R.E., Sagoo, E., Williams, J.R., Chambers, B.J., Chadwick, D.R., Laws, J.A. & Yamulki, S. 2007. The effect of slurry application timings on direct and indirect N_2O emissions from free draining grassland. In: A. Bosch, M.R. Teira & J.M. Villar, eds. *Proceedings of the 15th nitrogen workshop: Towards a better efficiency in N use*. Lleida, Spain, Editorial Milenio.

Thornley, J.H.M. & France, J. 2007. *Mathematical models in agriculture*. 2nd edition. Wallingford, UK, CABI Publishing.

Thornton, P.K. 2010. Livestock production: Recent trends, future prospects. *Philosophical Transactions of the Royal Society. Series B – Biological sciences*, 365(1554): 2853–2867. https://doi.org/10.1098/rstb.2010.0134

Tian, H., Lu, C., Ciais, P., Michalak, A.M., Canadell, J.G., Saikawa, E., Huntzinger, D.N., Gurney, K.R., Sitch, S., Zhang, B., Yang, J., Bousquet, P., Bruhwiler, L., Chen, G., Dlugokencky, E., Friedlingstein, P., Melillo, J., Pan, S., Poulter, B., Prinn, R., Saunois, M., Schwalm, C.R. & Wofsy, S.C.. 2016. The terrestrial biosphere as a net source of greenhouse gases to the atmosphere. *Nature*, 531: 225-228. https://doi.org/10.1038/nature16946

Tian, H., Yang, J., Xu, R., Lu, C., Canadell, J.G., Davidson, E.A., Jackson, R.B., Arneth, A., Chang, J., Ciais, P. & Gerber, S. 2019. Global soil nitrous oxide emissions since the preindustrial era estimated by an ensemble of terrestrial biosphere models: Magnitude, attribution, and uncertainty. *Global Change Biology*, 25(2): 640–659. https://doi.org/10.1111/gcb.14514

Tian, Z., Fan, Y., Wang, K., Zhong, H., Sun, L., Fan, D., Tubiello, F.N. & Liu, J. 2021. Searching for "Win-Win" solutions for food-water-GHG emissions tradeoffs across irrigation regimes of paddy rice in China. *Resources, Conservation and Recycling*, 166: 105360. https://doi.org/10.1016/j.resconrec.2020.105360

Tian, Z., Zhong, H., Sun, L., Fischer, G., van Velthuizen, H. & Liang, Z. 2014. Improving performance of Agro-Ecological Zone (AEZ) modeling by cross-scale model coupling: An application to japonica rice production in Northeast China. *Ecological Modelling*, 290: 155–164. https://doi.org/10.1016/j.ecolmodel.2013.11.020

Tibrewal, K. & Venkataraman, C. 2021. Climate co-benefits of air quality and clean energy policy in India. *Nature Sustainability*, 4(4): 305–313. https://doi.org/10.1038/s41893-020-00666-3

Tilley, J.M.A. & Terry, R.A. 1963. A two-stage technique for the *in vitro* digestion of forage crops. *Grass and Forage Science*, 18(2): 104–111. https://doi.org/10.1111/j.1365-2494.1963.tb00335.x

Timmer, C.P., Block, S. & Dawe, D. 2010. Long-run dynamics in rice consumption, 1960-2050. In: S. Pandey, D. Byerlee, D. Dawe, A. Dobermann, S. Mohanty, S. Rozelle & B. Hardy, eds. *Rice in the global economy: Strategic research and policy issues for food security*, pp. 139–174. Los Baños, Philippines, International Rice Research Institute (IRRI).

Tiwari, R., Kritee, K., Adhya, T.K., Loecke, T., Rudek, J., Nair, D., Ahuja, R., Balireddygari, S., Balakrishna, S., Ram, K., Venkataiah, L.C., Dava, O., Madasamy, M. & Salai, A. 2015. Sampling guidelines and analytical optimization for direct greenhouse gas emissions from tropical rice and upland cropping systems. *Carbon Management*, 6(3–4): 169–184. https://doi.org/10.1080/17583004.2015.1082233

Tol, R.S.J., Berntsen, T.K., O'Neill, B.C., Fuglestvedt, J.S. & Shine, K.P. 2012. A unifying framework for metrics for aggregating the climate effect of different emissions. *Environmental Research Letters*, 7(4): 044006. https://doi.org/10.1088/1748-9326/7/4/044006

Tomkins, N., Colegate, S. & Hunter, R. 2009. A bromochloromethane formulation reduces enteric methanogenesis in cattle fed grain-based diets. *Animal Production Science*, 49(12): 1053-1058. https://doi.org/10.1071/EA08223

Tomkins, N.W. & Charmley, E. 2015. Herd-scale measurements of methane emissions from cattle grazing extensive sub-tropical grasslands using the open-path laser technique. *Animal*, 9(12): 2029–2038. https://doi.org/10.1017/S1751731115001688

Tomkins, N.W., McGinn, S.M., Turner, D.A. & Charmley, E. 2011. Comparison of open-circuit respiration chambers with a micrometeorological method for determining methane emissions from beef cattle grazing a tropical pasture. *Animal Feed Science and Technology*, 166–167: 240–247. https://doi.org/10.1016/j.anifeedsci.2011.04.014

Topp, E. & Pattey, E. 1997. Soils as sources and sinks for atmospheric methane. *Canadian Journal of Soil Science*, 77(2): 167–177. https://doi.org/10.4141/S96-107

Toral, P.G., Monahan, F.J., Hervás, G., Frutos, P. & Moloney, A.P. 2018. Review: Modulating ruminal lipid metabolism to improve the fatty acid composition of meat and milk. Challenges and opportunities. *Animal*, 12(s2): s272–s281. https://doi.org/10.1017/S1751731118001994

Torres, C.M.M.E., Gonçalves Jacovine, L.A., Nolasco de Olivera Neto, S., Fraisse, C.W., Boecha Soares, C.P., de Castro Neto, F., Ferreira, L.R., Zanuncio, J.C. & Lemes, P.G. 2017. Greenhouse gas emissions and carbon sequestration by agroforestry systems in southeastern Brazil. *Scientific Reports*, 7(1): 16738. https://doi.org/10.1038/s41598-017-16821-4

Torrijos, M. 2016. State of development of biogas production in Europe. *Procedia Environmental Sciences*, 35: 881–889. https://doi.org/10.1016/j.proenv.2016.07.043

Trei, J.E., Parish, R.C., Singh, Y.K. & Scott, G.C. 1971. Effect of methane inhibitors on rumen metabolism and feedlot performance of sheep. *Journal of Dairy Science*, 54(4): 536–540. https://doi.org/10.3168/jds.S0022-0302(71)85882-4

Trei, J.E., Scott, G.C. & Parish, R.C. 1972. Influence of methane inhibition on energetic efficiency of lambs. *Journal of Animal Science*, 34(3): 510–515. https://doi.org/10.2527/jas1972.343510x

Tricarico, J.M., Kebreab, E. & Wattiaux, M.A. 2020. MILK Symposium review: Sustainability of dairy production and consumption in low-income countries with emphasis on productivity and environmental impact. *Journal of Dairy Science*, 103(11): 9791–9802. https://doi.org/10.3168/jds.2020-18269

Trudinger, C. & Enting, I. 2005. Comparison of formalisms for attributing responsibility for climate change: Non-linearities in the Brazilian Proposal approach. *Climatic Change*, 68(1–2): 67–99. https://doi.org/10.1007/s10584-005-6012-2

Uddin, M.E., Aguirre-Villegas, H.A., Larson, R.A. & Wattiaux, M.A. 2021. Carbon footprint of milk from Holstein and Jersey cows fed low or high forage diet with alfalfa silage or corn silage as the main forage source. *Journal of Cleaner Production*, 298: 126720. https://doi.org/10.1016/j.jclepro.2021.126720

Ueyama, M., Takeuchi, R., Takahashi, Y., Ide, R., Ataka, M., Kosugi, Y., Takahashi, K. & Saigusa, N. 2015. Methane uptake in a temperate forest soil using continuous closed-chamber measurements. *Agricultural and Forest Meteorology*, 213: 1–9. https://doi.org/10.1016/j.agrformet.2015.05.004

Ultee, A., Kets, E.P.W. & Smid, E.J. 1999. Mechanisms of action of carvacrol on the food-borne pathogen *Bacillus cereus*. *Applied and Environmental Microbiology*, 65(10): 4606–4610. https://doi.org/10.1128/aem.65.10.4606-4610.1999

Undi, M., Wilson, C., Ominski, K.H. & Wittenberg, K.M. 2008. Comparison of techniques for estimation offorage dry matter intake by grazing beef cattle. *Canadian Journal of Animal Science*, 88(4): 693–701. https://doi.org/10.4141/CJAS08041

UNEP (United Nations Environment Programme). 2021. *The Life Cycle Initiative*. [Cited 30 January 2022]. www.lifecycleinitiative.org/category/glam/

UNEP & CCAC (Climate and Clean Air Coalition). 2021. *Global methane assessment: Benefits and costs of mitigating methane emissions*. Nairobi, UNEP. [Cited 15 June 2023]. www.unep.org/resources/report/global-methane-assessment-benefits-and-costs-mitigating-methane-emissions

UNFCCC (United Nations Framework Convention on Climate Change). 1997. The Kyoto Protocol to the United Nations Framework Convention on Climate Change. [Cited 5 June 2023]. https://treaties.un.org/Pages/ViewDetails.aspx?src=IND&mtdsg_no=XXVII-7-a&chapter=27&clang=_en

Ungerfeld, E.M. 2015. Shifts in metabolic hydrogen sinks in the methanogenesis-inhibited ruminal fermentation: A meta-analysis. *Frontiers in Microbiology*, 6: 37. https://doi.org/10.3389/fmicb.2015.00037

Ungerfeld, E.M. 2018. Inhibition of rumen methanogenesis and ruminant productivity: A meta-analysis. *Frontiers in Veterinary Science*, 5: 113. https://doi.org/10.3389/fvets.2018.00113

Ungerfeld, E.M. 2020. Metabolic hydrogen flows in rumen fermentation: Principles and possibilities of interventions. *Frontiers in Microbiology*, 11: 589. https://doi.org/10.3389/fmicb.2020.00589

Ungerfeld, E.M., Beauchemin, K.A. & Muñoz, C. 2022. Current perspectives on achieving pronounced enteric methane mitigation from ruminant production. *Frontiers in Animal Science*, 2: 795200. https://doi.org/10.3389/fanim.2021.795200

Ungerfeld, E.M. & Forster, R.J. 2011. A meta-analysis of malate effects on methanogenesis in ruminal batch cultures. *Animal Feed Science and Technology*, 166–167: 282–290. https://doi.org/10.1016/j.anifeedsci.2011.04.018

Ungerfeld, E.M. & Hackmann, T.J. 2020. Factors influencing the efficiency of rumen energy metabolism. In: C.S. McSweeney & R.I. Mackie, eds. *Improving rumen function*, pp. 421–466. Cambridge, UK, Burleigh Dodds Science Publishing. https://doi.org/10.19103/AS.2020.0067.14

Ungerfeld, E.M. & Kohn, R. 2006. The role of thermodynamics in the control of ruminal fermentation. In: K. Sejrsen, T. Hvelplund & M.O. Nielsen, eds. *Ruminant physiology: Digestion, metabolism and impact of nutrition on gene expression, immunology and stress*, pp. 55–85. Wageningen, Kingdom of the Netherlands, Wageningen Academic Publishers.

Ungerfeld, E.M., Kohn, R.A., Wallace, R.J. & Newbold, C.J. 2007. A meta-analysis of fumarate effects on methane production in ruminal batch cultures. *Journal of Animal Science*, 85(10): 2556–2563. https://doi.org/10.2527/jas.2006-674

Ungerfeld, E.M., Rust, S.R., Boone, D.R. & Liu, Y. 2004. Effects of several inhibitors on pure cultures of ruminal methanogens. *Journal of Applied Microbiology*, 97(3): 520–526. https://doi.org/10.1111/j.1365-2672.2004.02330.x

Ungerfeld, E.M., Rust, S.R. & Burnett, R. 2006. Effects of butyrate precursors on electron relocation when methanogenesis is inhibited in ruminal mixed cultures. *Letters in Applied Microbiology*, 42(6): 567–572. https://doi.org/10.1111/j.1472-765X.2006.01890.x

United Nations. 1992. United Nations framework convention on climate change. FCC/INFORMAL/84/Rev.1. https://unfccc.int/sites/default/files/convention_text_with_annexes_english_for_posting.pdf

Vadenbo, C., Hellweg, S. & Guillén-Gosálbez, G. 2014. Multi-objective optimization of waste and resource management in industrial networks – Part I: Model description. *Resources, Conservation and Recycling*, 89: 52–63. https://doi.org/10.1016/j.resconrec.2014.05.010

Vallejo, A., García-Torres, L., Díez, J.A., Arce, A. & López-Fernández, S. 2005. Comparison of N losses (NO_{3^-}, N_2O, NO) from surface applied, injected or amended (DCD) pig slurry of an irrigated soil in a Mediterranean climate. *Plant and Soil*, 272(1–2): 313–325. https://doi.org/10.1007/s11104-004-5754-3

van den Berg, M., Hof, A.F., van Vliet, J. & van Vuuren, D.P. 2015. Impact of the choice of emission metric on greenhouse gas abatement and costs. *Environmental Research Letters*, 10(2): 024001. https://doi.org/10.1088/1748-9326/10/2/024001

van der Weerden, T., Beukes, P., de Klein, C., Hutchinson, K., Farrell, L., Stormink, T., Romera, A., Dalley, D., Monaghan, R., Chapman, D., Macdonald, K. & Dynes, R. 2018. The effects of system changes in grazed dairy farmlet trials on greenhouse gas emissions. *Animals*, 8(12): 234. https://doi.org/10.3390/ani8120234

van der Weerden, T., Noble, A.N., Luo, J., de Klein, C.A.M., Saggar, S., Giltrap, D., Gibbs, J. & Rys, G. 2020. Meta-analysis of New Zealand's nitrous oxide emission factors for ruminant excreta supports disaggregation based on excreta form, livestock type and slope class. *Science of the Total Environment*, 732: 139235. https://doi.org/10.1016/j.scitotenv.2020.139235

van der Werf, H.M.G., Knudsen, M.T. & Cederberg, C. 2020. Towards better representation of organic agriculture in life cycle assessment. *Nature Sustainability*, 3(6): 419–425. https://doi.org/10.1038/s41893-020-0489-6

van Gastelen, S., Dijkstra, J., Binnendijk, G., Duval, S.M., Heck, J.M.L., Kindermann, M., Zandstra, T. & Bannink, A. 2020. 3-nitrooxypropanol decreases methane emissions and increases hydrogen emissions of early lactation dairy cows, with associated changes in nutrient digestibility and energy metabolism. *Journal of Dairy Science*, 103(9): 8074-8093. https://doi.org/10.3168/jds.2019-17936

Van Hung, N., Migo, M.V., Quilloy, R., Chivenge, P. & Gummert, M. 2020. Life cycle assessment applied in rice production and residue management. In: M. Gummert, N.V. Hung, P. Chivenge & B. Douthwaite, eds. *Sustainable rice straw management*, pp. 161–174. New York, USA, Springer Cham. https://doi.org/10.1007/978-3-030-32373-8_10

van Kessel, J.A.S. & Russell, J.B. 1996. The effect of pH on ruminal methanogenesis. *FEMS Microbiology Ecology*, 20(4): 205–210. https://doi.org/10.1016/0168-6496(96)00030-X

van Lingen, H.J., Niu, M., Kebreab, E., Valadares Filho, S.C., Rooke, J.A., Duthie, C.-A., Schwarm, A., Kreuzer, M., Hynd, P.I., Caetano, M., Eugène, M., Martin, C., McGee, M., O'Kiely, P., Hünerberg, M., McAllister, T.A., Berchielli, T.T., Messana, J.D., Peiren, N., Chaves, A.V., Charmley, E., Cole, N.A., Hales, K.E., Lee, S.-S., Berndt, A., Reynolds, C.K., Crompton, L.A., Bayat, A.-R., Yáñez-Ruiz, D.R., Yu, Z., Bannink, A., Dijkstra, J., Casper, D.P. & Hristov, A.N. 2019. Prediction of enteric methane production, yield and intensity of beef cattle using an intercontinental database. *Agriculture, Ecosystems & Environment*, 283: 106575. https://doi.org/10.1016/j.agee.2019.106575

van Ouverkerk, E.N.J. & Pedersen, S. 1994. Application of the carbon dioxide mass balance method to evaluate ventilation rates in livestock buildings. In: *Proceedings of the XII World Congress on Agricultural Engineering, Milan, 29 August–1 September 1994*, pp. 516–529. CIGR.

van Well, B.V., Murray, S., Hodgkinson, J., Pride, R., Strzoda, R., Gibson, G. & Padgett, M. 2005. An open-path, hand-held laser system for the detection of methane gas. *Journal of Optics A: Pure and Applied Optics*, 7(6): S420–S424. https://doi.org/10.1088/1464-4258/7/6/025

van Zijderveld, S.M., Fonken, B., Dijkstra, J., Gerrits, W.J.J., Perdok, H.B., Fokkink, W. & Newbold, J.R. 2011. Effects of a combination of feed additives on methane production, diet digestibility, and animal performance in lactating dairy cows. *Journal of Dairy Science*, 94(3): 1445–1454. https://doi.org/10.3168/jds.2010-3635

van Zijderveld, S.M., Gerrits, W.J.J., Apajalahti, J.A., Newbold, J.R., Dijkstra, J., Leng, R.A. & Perdok, H.B. 2010. Nitrate and sulfate: Effective alternative hydrogen sinks for mitigation of ruminal methane production in sheep. *Journal of Dairy Science*, 93(12): 5856–5866. https://doi.org/10.3168/jds.2010-3281

VandeHaar, M.J., Armentano, L.E., Weigel, K., Spurlock, D.M., Tempelman, R.J. & Veerkamp, R. 2016. Harnessing the genetics of the modern dairy cow to continue improvements in feed efficiency. *Journal of Dairy Science*, 99(6): 4941–4954.

Vandermeulen, S., Ramírez-Restrepo, C.A., Beckers, Y., Claessens, H. & Bindelle, J. 2018. Agroforestry for ruminants: A review of trees and shrubs as fodder in silvopastoral temperate and tropical production systems. *Animal Production Science*, 58(5): 767. https://doi.org/10.1071/AN16434

Vanlierde, A., Dehareng, F., Gengler, N., Froidmont, E., McParland, S., Kreuzer, M., Bell, M., Lund, P., Martin, C. Kuhla, B. & Soyeurt, H. 2021. Improving robustness and accuracy of predicted daily methane emissions of dairy cows using milk mid-infrared spectra. *Journal of the Science of Food and Agriculture*, 101(8): 3394–3403. https://doi.org/10.1002/jsfa.10969

Varon, D.J., Jacob, D.J., McKeever, J., Jervis, D., Durak, B.O., Xia, Y. & Huang, Y. 2018. Quantifying methane point sources from fine-scale satellite observations of atmospheric methane plumes. *Atmospheric Measurement Techniques*, 11(10): 5673–5686.

Varon, D.J., Jervis, D.J, McKeever, J., Spence, I., Gains, D. & Jacob, D.J. 2021. High-frequency monitoring of anomalous methane point sources with multispectral Sentinel-2 satellite observations. *Atmospheric Measurement Techniques*, 14(4): 2771–2785. https://doi.org/10.5194/amt-14-2771-2021

Varshney, C.K. & Attri, A.K. 1999. Global warming potential of biogenic methane. *Tellus B*, 51(3): 612–615. https://doi.org/10.1034/j.1600-0889.1999.t01-2-00003.x

Vasta, V. & Luciano, G. 2011. The effects of dietary consumption of plants secondary compounds on small ruminants' products quality. *Small Ruminant Research*, 101(1): 150–159. https://doi.org/10.1016/j.smallrumres.2011.09.035

Vellinga, T.V. & Hoving, I.E. 2011. Maize silage for dairy cows: Mitigation of methane emissions can be offset by land use change. *Nutrient Cycling in Agroecosystems*, 89(3): 413–426. https://doi.org/10.1007/s10705-010-9405-1

Velthof, G.L., Kuikman, P.J. & Oenema, O. 2003. Nitrous oxide emission from animal manures applied to soil under controlled conditions. *Biology and Fertility of Soils*, 37(4): 221–230. https://doi.org/10.1007/s00374-003-0589-2

Veneman, J.B., Saetnan, E.R., Clare, A.J. & Newbold, C.J. 2016. MitiGate; an online meta-analysis database for quantification of mitigation strategies for enteric methane emissions. *Science of the Total Environment*, 572: 1166–1174. https://doi.org/10.1016/j.scitotenv.2016.08.029

Venterea, R.T., Burger, M. & Spokas, K.A. 2005. Nitrogen oxide and methane emissions under varying tillage and fertilizer management. *Journal of Environmental Quality*, 34(5): 1467–1477.

Vermorel, M., Bouvier, J.C. & Demarquilly, C. 1974. Influence du mode du conditionnement des fourrages deshydratés sur leur valeur énergétique nette pour le mouton en croissance. In: K.H. Menke & J.R. Reichl, eds. *Energy metabolism of farm animals.* European Association of Animal Production, publication No. 14. Hohenheim, Germany, Universität Hohenheim Dokumentationsstelle.

Verones, F., Hellweg, S., Antón, A., Azevedo, L.B., Chaudhary, A., Cosme, N., Cucurachi, S., de Baan, L., Dong, Y., Fantke, P., Golsteijn, L., Hauschild, M., Heijungs, R., Jolliet, O., Juraske, R., Larsen, H., Laurent, A., Mutel, C.L., Margni, M., Núñez, M., Owsianiak, M., Pfister, S., Ponsioen, T., Preiss, P., Rosenbaum, R.K., Roy, P.-O., Sala, S., Steinmann, Z., van Zelm, R., Van Dingenen, R., Vieira, M. & Huijbregts, M.A.J. 2020. LC-IMPACT: A regionalized life cycle damage assessment method. *Journal of Industrial Ecology*, 24(6): 1201–1219. https://doi.org/10.1111/jiec.13018

Vibart, R., de Klein, C., Jonker, A., van der Weerden, T., Bannink, A., Bayat, A.R., Crompton, L., Durand, A., Eugène, M., Klumpp, K., Kuhla, B., Lanigan, G., Lund, P., Ramin, M. & Salazar, F. 2021. Challenges and opportunities to capture dietary effects in on-farm greenhouse gas emissions models of ruminant systems. *Science of the Total Environment*, 769: 144989. https://doi.org/10.1016/j.scitotenv.2021.144989

Vigan, A., Hassouna, M., Guingand, N., Brame, C., Edouard, N., Eglin, T., Espagnol, S., Eugène, E., Génermont, S., Lagadec, S., Lorinquer, E., Loyon, L., Ponchant, P. & Robin, P. 2019. Development of a database to collect emission values for livestock systems. *Journal of Environmental Quality*, 48(6): 1899–1906. https://doi.org/10.2134/jeq2019.01.0007

Vijn, S., Compart, D.P., Dutta, N., Foukis, A., Hess, M., Hristov, A.N., Kalscheur, K.F., Kebreab, E., Nuzhdin, S.V., Price, N.N., Sun, Y., Tricarico, J.M., Turzillo, A., Weisbjerg, M.R., Yarish, C. & Kurt, T.D. 2020. Key considerations for the use of seaweed to reduce enteric methane emissions from cattle. *Frontiers in Veterinary Science*, 7: 597430. https://doi.org/10.3389/fvets.2020.597430

Villar, M.L., Hegarty, R.S., Nolan, J.V., Godwin, I.R. & McPhee, M. 2020. The effect of dietary nitrate and canola oil alone or in combination on fermentation, digesta kinetics and methane emissions from cattle. *Animal Feed Science and Technology*, 259: 114294. https://doi.org/10.1016/j.anifeedsci.2019.114294

Vogels, G.D., Hoppe, W.F. & Stumm, C.K. 1980. Association of methanogenic bacteria with rumen ciliates. *Applied and Environmental Microbiology*, 40(3): 608–612. https://doi.org/10.1128/aem.40.3.608-612.1980

von Soosten, D., Meyer, U., Flachowsky, G. & Dänicke, S. 2020. Dairy cow health and greenhouse gas emission intensity. *Dairy*, 1(1): 3. https://doi.org/10.3390/dairy1010003

Vyas, D., Alazzeh, A., McGinn, S.M., McAllister, T.A., Harstad, O.M., Holo, H. & Beauchemin, K.A. 2015. Enteric methane emissions in response to ruminal inoculation of *Propionibacterium* strains in beef cattle fed a mixed diet. *Animal Production Science*, 56(7): 1035. https://doi.org/10.1071/AN14801

Vyas, D., Alemu, A.W., McGinn, S.M., Duval, S.M., Kindermann, M. & Beauchemin, K.A. 2018. The combined effects of supplementing monensin and 3-nitrooxypropanol on methane emissions, growth rate, and feed conversion efficiency in beef cattle fed high-forage and high-grain diets. *Journal of Animal Science*, 96(7): 2923–2938. https://doi.org/10.1093/jas/sky174

Vyas, D., McGeough, E.J., McGinn, S.M., McAllister, T.A. & Beauchemin, K.A. 2014a. Effect of *Propionibacterium* spp. on ruminal fermentation, nutrient digestibility, and methane emissions in beef heifers fed a high-forage diet. *Journal of Animal Science*, 92(5): 2192–2201. https://doi.org/10.2527/jas.2013-7492

Vyas, D., McGeough, E.J., Mohammed, R., McGinn, S.M., McAllister, T.A. & Beauchemin, K.A. 2014b. Effects of *Propionibacterium* strains on ruminal fermentation, nutrient digestibility and methane emissions in beef cattle fed a corn grain finishing diet. *Animal*, 8(11): 1807–1815. https://doi.org/10.1017/S1751731114001657

Vyas, D., McGinn, S.M., Duval, S.M., Kindermann, M. & Beauchemin, K.A. 2016. Effects of sustained reduction of enteric methane emissions with dietary supplementation of 3-nitrooxypropanol on growth performance of growing and finishing beef cattle. *Journal of Animal Science*, 94(5): 2024–2034. https://doi.org/10.2527/jas.2015-0268

Waghorn, G. 2008. Beneficial and detrimental effects of dietary condensed tannins for sustainable sheep and goat production – Progress and challenges. *Animal Feed Science and Technology*, 147(1): 116–139. https://doi.org/10.1016/j.anifeedsci.2007.09.013

Wahab, I., Hall, O. & Jirström, M. 2018. Remote sensing of yields: Application of UAV imagery-derived NDVI for estimating maize vigor and yields in complex farming systems in sub-Saharan Africa. *Drones*, 2(3): 28. https://doi.org/10.3390/drones2030028

Waldhoff, S., Anthoff, D., Rose, S. & Tol, R. 2014. The marginal damage costs of different greenhouse gases: An application of FUND. Economics discussion paper No. 2011–43. https://doi.org/10.2139/ssrn.1974111

Waldo, S., Russell, E.S., Kostyanovsky, K., Pressley, S.N., O'Keeffe, P.T., Huggins, D.R., Stöckle, C.O., Pan, W.L. & Lamb, B.K. 2019. N_2O emissions from two agroecosystems: High spatial variability and long pulses observed using static chambers and the flux-gradient technique. *Journal of Geophysical Research: Biogeosciences*, 124(7): 1887–1904. https://doi.org/10.1029/2019JG005032

Wall, E., Simm, G. & Moran, D. 2010. Developing breeding schemes to assist mitigation of greenhouse gas emissions. *Animal*, 4(3): 366–376. https://doi.org/10.1017/S175173110999070X

Wang, B., Tu, Y., Zhao, S.P., Hao, Y.H., Liu, J.X., Liu, F.H., Xiong, B.H. & Jiang, L.S. 2017. Effect of tea saponins on milk performance, milk fatty acids, and immune function in dairy cow. *Journal of Dairy Science*, 100(10): 8043–8052. https://doi.org/10.3168/jds.2016-12425

Wang, C., Hou, F., Wanapat, M., Yan, T., Kim, E.J. & Scollan, N.D. 2020. Assessment of cutting time on nutrient values, *in vitro* fermentation and methane production among three ryegrass cultivars. *Asian-Australasian Journal of Animal Sciences*, 33(8): 1242–1251. https://doi.org/10.5713/ajas.19.0369

Wang, R., Wang, M., Ungerfeld, E.M., Zhang, X.M., Long, D.L., Mao, H.X., Deng, J.P., Bannink, A. & Tan, Z.L. 2018. Nitrate improves ammonia incorporation into rumen microbial protein in lactating dairy cows fed a low-protein diet. *Journal of Dairy Science*, 101(11): 9789–9799. https://doi.org/10.3168/jds.2018-14904

Warner, D., Bannink, A., Hatew, B., van Laar, H. & Dijkstra, J. 2017. Effects of grass silage quality and level of feed intake on enteric methane production in lactating dairy cows. *Journal of Animal Science*, 95(8): 3687–3699. https://doi.org/10.2527/jas.2017.1459

Warner, D., Hatew, B., Podesta, S.C., Klop, G., van Gastelen, S., van Laar, H., Dijkstra, J. & Bannink, A. 2016. Effects of nitrogen fertilisation rate and maturity of grass silage on methane emission by lactating dairy cows. *Animal*, 10(1): 34–43. https://doi.org/10.1017/S1751731115001640

Warner, D., Podesta, S.C., Hatew, B., Klop, G., van Laar, H., Bannink, A. & Dijkstra, J. 2015. Effect of nitrogen fertilization rate and regrowth interval of grass herbage on methane emission of zero-grazing lactating dairy cows. *Journal of Dairy Science*, 98(5): 3383–3393. https://doi.org/10.3168/jds.2014-9068

Wassmann, R. 2019. Environmental footprints of modernization trends in rice production systems of southeast Asia. In: *Oxford research encyclopedia of environmental science*. Oxford, UK, Oxford University Press. https://doi.org/10.1093/acrefore/9780199389414.013.230

Wassmann, R., Neue, H.-U., Lantin, R.S., Aduna, J.B., Alberto, M.C.R., Andales, M.J., Tan, M.J., Denier van der Gon, H.A.C., Hoffmann, H., Papen, H., Rennenberg, H. & Seiler, W. 1994. Temporal patterns of methane emissions from wetland rice fields treated by different modes of N application. *Journal of Geophysical Research*, 99(D8): 16457. https://doi.org/10.1029/94JD00017

Wassmann, R., Neue, H.-U., Lantin, R.S., Buendia, L.V. & Rennenberg, H. 2000. Characterization of methane emissions from rice fields in Asia. I. Comparison among field sites in five countries. *Nutrient Cycling in Agroecosystems*, 58(1): 1–12. https://doi.org/10.1023/A:1009848813994

Wassmann, R., Papen, H. & Rennenberg, H. 1993. Methane emission from rice paddies and possible mitigation strategies. *Chemosphere*, 26(1–4): 201–217. https://doi.org/10.1016/0045-6535(93)90422-2

Wassmann, R., Schütz, H., Papen, H., Rennenberg, H., Seiler, W., Aiguo, D., Renxing, S., Xingjian, S. & Mingxing, W. 1993. Quantification of methane emissions from Chinese rice fields (Zhejiang Province) as influenced by fertilizer treatment. *Biogeochemistry*, 20(2): 83–101. https://doi.org/10.1007/BF00004136

Weber, T.L., Hao, X., Gross, C.D., Beauchemin, K.A. & Chang, S.X. 2021. Effect of manure from cattle fed 3-nitrooxypropanol on anthropogenic greenhouse gas emissions depends on soil type. *Agronomy*, 11(2): 371. https://doi.org/10.3390/agronomy11020371

Wedlock, D.N., Janssen, P.H., Leahy, S.C., Shu, D. & Buddle, B.M. 2013. Progress in the development of vaccines against rumen methanogens. *Animal*, 7: 244–252. https://doi.org/10.1017/S1751731113000682

Wedlock, D.N, Pedersen, G., Denis, M., Dey, D., Janssen, P. & Buddle, B. 2010. Development of a vaccine to mitigate greenhouse gas emissions in agriculture: Vaccination of sheep with methanogen fractions induces antibodies that block methane production *in vitro*. *New Zealand Veterinary Journal*, 58(1): 29–36. https://doi.org/10.1080/00480169.2010.65058

Weiby, K.V., Krizsan, S.J., Eknæs, M., Schwarm, A., Whist, A.C., Schei, I., Steinshamn, H., Lund, P., Beauchemin, K. & Dønnem, I. 2022. Associations among nutrient concentration, silage fermentation products, *in vivo* organic matter digestibility, rumen fermentation and *in vitro* methane yield in 78 grass silages. *Animal Feed Science and Technology*, 285: 115249. https://doi.org/10.1016/j.anifeedsci.2022.115249

Weimer, P.J. 2015. Redundancy, resilience, and host specificity of the ruminal microbiota: Implications for engineering improved ruminal fermentations. *Frontiers in Microbiology*, 6: 269. https://doi.org/10.3389/fmicb.2015.00296

Weiss, R.F. 1981. Determinations of carbon dioxide and methane by dual catalyst flame ionization chromatography and nitrous oxide by electron capture chromatography. *Journal of Chromatographic Science*, 19(12): 611–616. https://doi.org/10.1093/chromsci/19.12.611

Weitzman, M.L. 2012. GHG targets as insurance against catastrophic climate damages. *Journal of Public Economic Theory*, 14(2): 221–244. https://doi.org/10.1111/j.1467-9779.2011.01539.x

Weitzman, M.L. 2013. A precautionary tale of uncertain tail fattening. *Environmental and Resource Economics*, 55(2): 159–173. https://doi.org/10.1007/s10640-013-9646-y

Weller, S., Janz, B., Jörg, L., Kraus, D., Racela, H.S.U., Wassmann, R., Butterbach-Bahl, K. & Kiese, R. 2016. Greenhouse gas emissions and global warming potential of traditional and diversified tropical rice rotation systems. *Global Change Biology*, 22(1): 432–448. https://doi.org/10.1111/gcb.13099

Weyant, J. 2017. Some contributions of integrated assessment models of global climate change. *Review of Environmental Economics and Policy*, 11(1): 115–137. https://doi.org/10.1093/reep/rew018

Whitehead, T.R., Spence, C. & Cotta, M.A. 2013. Inhibition of hydrogen sulfide, methane, and total gas production and sulfate-reducing bacteria in *in vitro* swine manure by tannins, with focus on condensed quebracho tannins. *Applied Microbiology and Biotechnology*, 97(18): 8403–8409. https://doi.org/10.1007/s00253-012-4562-6

Wigley, T.M.L. 1998. The Kyoto Protocol: CO_2, CH_4, and climate implications. *Geophysical Research Letters*, 25(13): 2285–2288. https://doi.org/10.1029/98GL01855

Wigley, T.M.L. 2021. The relationship between net GHG emissions and radiative forcing with an application to Article 4.1 of the Paris Agreement. *Climatic Change*, 169(1–2): 13. https://doi.org/10.1007/s10584-021-03249-z

Williams, S.R.O., Hannah, M.C., Eckard, R.J., Wales, W.J. & Moate, P.J. 2020. Supplementing the diet of dairy cows with fat or tannin reduces methane yield, and additively when fed in combination. *Animal*, 14(3): s464–s472. https://doi.org/10.1017/S1751731120001032

Williams, Y.J., Popovski, S., Rea, S.M., Skillman, L.C., Toovey, A.F., Northwood, K.S. & Wright, A.-D.G. 2009. A vaccine against rumen methanogens can alter the composition of archaeal populations. *Applied and Environmental Microbiology*, 75(7): 1860–1866. https://doi.org/10.1128/AEM.02453-08

Wiloso, E.I., Heijungs, R., Huppes, G. & Fang, K. 2016. Effect of biogenic carbon inventory on the life cycle assessment of bioenergy: Challenges to the neutrality assumption. *Journal of Cleaner Production*, 125: 78–85. https://doi.org/10.1016/j.jclepro.2016.03.096

Wina, E., Muetzel, S. & Becker, K. 2005. The impact of saponins or saponin-containing plant materials on ruminant production – A review. *Journal of Agricultural and Food Chemistry*, 53(21): 8093–8105. https://doi.org/10.1021/jf048053d

Winihayakul, S., Cookson, R., Scott, R., Zhou, J., Zou, X., Roldan, M., Richardson, K. & Roberts, N. 2008. Delivery of grasses with high levels of unsaturated, protected fatty acids. In: *Proceedings of the New Zealand Grassland Association*, pp. 211–216. Bleinheim, New Zealand. https://doi.org/10.33584/jnzg.2008.70.2721

Wirth, J. & Young, M. 2020. The intriguing world of archaeal viruses. *PLoS Pathogens*, 16(8): e1008574. https://doi.org/10.1371/journal.ppat.1008574

Wischer, G., Greiling, A.M., Boguhn, J., Steingass, H., Schollenberger, M., Hartung, K. & Rodehutscord, M. 2014. Effects of long-term supplementation of chestnut and valonea extracts on methane release, digestibility and nitrogen excretion in sheep. *Animal*, 8(6): 938–948. https://doi.org/10.1017/S1751731114000639

Wolin, M.J., Miller, T.L. & Stewart, C.S. 1997. Microbe-microbe interactions. In: P.N. Hobson & C.S. Stewart, eds. *The rumen microbial ecosystem*, pp. 467–491. London, Blackie Academic & Professional.

Wong, A. 2019. Unknown risk on the farm: Does agricultural use of ionophores contribute to the burden of antimicrobial resistance? *mSphere*, 4(5): e00433-19. https://doi.org/10.1128/mSphere.00433-19

Wood, J.M., Scott Kennedy, F. & Wolfe, R.S. 1968. Reaction of multihalogenated hydrocarbons with free and bound reduced vitamin B12. *Biochemistry*, 7(5): 1707–1713. https://doi.org/10.1021/bi00845a013

Wood, T.A., Wallace, R.J., Rowe, A., Price, J., Yáñez-Ruiz, D.R., Murray, P. & Newbold, C.J. 2009. Encapsulated fumaric acid as a feed ingredient to decrease ruminal methane emissions. *Animal Feed Science and Technology*, 152(1–2): 62–71. https://doi.org/10.1016/j.anifeedsci.2009.03.006

Wright, A.D.G., Kennedy, P., O'Neill, C.J., Toovey, A.F., Popovski, F., Rea, S.M., Pimm, C.L. & Klein, L. 2004. Reducing methane emissions in sheep by immunization against rumen methanogens. *Vaccine*, 22(29–30): 3976–3985. https://doi.org/10.1016/j.vaccine.2004.03.053

Wu, J., Chen, Q., Jia, W., Long, C., Liu, W., Liu, G. & Cheng, X. 2020. Asymmetric response of soil methane uptake rate to land degradation and restoration: Data synthesis. *Global Change Biology*, 26(11): 6581–6593. https://doi.org/10.1111/gcb.15315

Wuebbles, D.J. & Hayhoe, K. 2002. Atmospheric methane and global change. *Earth-Science Reviews*, 57(3): 177–210. https://doi.org/10.1016/S0012-8252(01)00062-9

Xu, F., Li, Y., Ge, X., Yang, L. & Li, Y. 2018. Anaerobic digestion of food waste – Challenges and opportunities. *Bioresource Technology*, 247: 1047–1058. https://doi.org/10.1016/j.biortech.2017.09.020

Yagi, K., Sriphirom, P., Cha-un, N., Fusuwankaya, K., Chidthaisong, A., Damen, B. & Towprayoon, S. 2020. Potential and promisingness of technical options for mitigating greenhouse gas emissions from rice cultivation in Southeast Asian countries. *Soil Science and Plant Nutrition*, 66(1): 37–49. https://doi.org/10.1080/00380768.2019.1683890

Yan, M.-J., Humphreys, J. & Holden, N.M. 2013. The carbon footprint of pasture-based milk production: Can white clover make a difference? *Journal of Dairy Science*, 96(2): 857–865. https://doi.org/10.3168/jds.2012-5904

Yan, T., Mayne, C.S., Gordon, F.G., Porter, M.G., Agnew, R.E., Patterson, D.C., Ferris, C.P. & Kilpatrick, D.J. 2010. Mitigation of enteric methane emissions through improving efficiency of energy utilization and productivity in lactating dairy cows. *Journal of Dairy Science*, 93(6): 2630–2638. https://doi.org/10.3168/jds.2009-2929

Yan, X., Akiyama, H., Yagi, K. & Akimoto, H. 2009. Global estimations of the inventory and mitigation potential of methane emissions from rice cultivation conducted using the 2006 Intergovernmental Panel on Climate Change Guidelines. *Global Biogeochemical Cycles*, 23(2): n/a-n/a. https://doi.org/10.1029/2008GB003299

Yáñez-Ruiz, D.R., Abecia, L. & Newbold, C.J. 2015. Manipulating rumen microbiome and fermentation through interventions during early life: A review. *Frontiers in Microbiology*, 6. https://doi.org/10.3389/fmicb.2015.01133

Yáñez-Ruiz, D.R., Bannink, A., Dijkstra, J., Kebreab, E., Morgavi, D.P., O'Kiely, P., Reynolds, C.K., A. Schwarm, A., Shingfield, K.J., Yu, Z. & Hristov, A.N. 2016. Design, implementation and interpretation of *in vitro* batch culture experiments to assess enteric methane mitigation in ruminants – A review. *Animal Feed Science and Technology*, 216: 1–18. https://doi.org/10.1016/j.anifeedsci.2016.03.016

Yang, C., Rooke, J.A., Cabeza, I. & Wallace, R.J. 2016. Nitrate and inhibition of ruminal methanogenesis: Microbial ecology, obstacles, and opportunities for lowering methane emissions from ruminant livestock. *Frontiers in Microbiology*, 7: 132. https://doi.org/10.3389/fmicb.2016.00132

Yang, C.J., Mao, S.Y., Long, L.M. & Zhu, W.Y. 2012. Effect of disodium fumarate on microbial abundance, ruminal fermentation and methane emission in goats under different forage: Concentrate ratios. *Animal*, 6(11): 1788–1794. https://doi.org/10.1017/S1751731112000857

Yang, W.Z., Beauchemin, K.A., Koenig, K.M. & Rode, L.M. 1997. Comparison of hull-less barley, barley, or corn for lactating cows: Effects on extent of digestion and milk production. *Journal of Dairy Science*, 80(10): 2475–2486. https://doi.org/10.3168/jds.S0022-0302(97)76200-3

Yang, Y. & Heijungs, R. 2018. On the use of different models for consequential life cycle assessment. *The International Journal of Life Cycle Assessment*, 23(4): 751–758. https://doi.org/10.1007/s11367-017-1337-4

Young, F. & Ferris, C.F. 2011. Effect of concentrate feed level on methane production by grazing dairy cows. In: *Proceedings of the Agricultural Research Forum*, p. 58. Dublin, Teagasc.

Yu, G., Beauchemin, K.A. & Dong, R. 2021. A review of 3-nitrooxypropanol for enteric methane mitigation from ruminant livestock. *Animals*, 11(12): 3540. https://doi.org/10.3390/ani11123540

Yu, L., Huang, Y., Zhang, W., Li, T. & Sun, W. 2017. Methane uptake in global forest and grassland soils from 1981 to 2010. *Science of the Total Environment*, 607–608: 1163–1172. https://doi.org/10.1016/j.scitotenv.2017.07.082

Yuan, Z.P., Zhang, C.M., Zhou, L., Zou, C.X., Guo, Y.Q., Li, W.T., Liu, J.X. & Wu, Y.M. 2007. Inhibition of methanogenesis by tea saponin and tea saponin plus disodium fumarate in sheep. *Journal of Animal and Feed Sciences*, 16 (Suppl. 2): 560–565. https://doi.org/10.22358/jafs/74607/2007

Yurtseven, S., Avci, M., Çetin, M., Öztürk, İ. & Boğa, M. 2018. Emissions of some greenhouse gases from the manure of ewes fed on pomegranate peel, yucca extract, and thyme oil. *Applied Ecology and Environmental Research*, 16(4): 4217–4228. https://doi.org/10.15666/aeer/1604_42174228

Zardin, P.B., Velho, J.P., Jobim, C.C., Alessio, D.R.M., Haygert-Velho, I.M.P., Conceição, G.M.D. & Almeida, P.S.G. 2017. Chemical composition of corn silage produced by scientific studies in Brazil – A meta-analysis. *Semina: Ciências Agrárias*, 38(1): 503. https://doi.org/10.5433/1679-0359.2017v38n1p503

Zeng, Z.-C., Byrne, B., Gong, F.-Y., He, Z. & Lei, L. 2021. Correlation between paddy rice growth and satellite-observed methane column abundance does not imply causation. *Nature Communications*, 12(1): 1163. https://doi.org/10.1038/s41467-021-21434-7

Zhang, B., Tian, H., Ren, W., Tao, B., Lu, C., Yang, J., Banger, K. & Pan, S. 2016. Methane emissions from global rice fields: Magnitude, spatiotemporal patterns, and environmental controls. *Global Biogeochemical Cycles*, 30(9): 1246–1263. https://doi.org/10.1002/2016GB005381

Zhang, D.-F. & Yang, H.-J. 2012. Combination effects of nitrocompounds, pyromellitic diimide, and 2-bromoethanesulfonate on *in vitro* ruminal methane production and fermentation of a grain-rich feed. *Journal of Agricultural and Food Chemistry*, 60(1): 364–371. https://doi.org/10.1021/jf203716v

Zhang, G., Pedersen, S. & Kai, P. 2010. *Uncertainty analysis of using CO_2 production models by cows to determine ventilation rate in naturally ventilated buildings*. XVIIth World Congress of the International Commission of Agricultural Engineering (CIGR), 13–17 June 2010. Quebec City, Canada, Canadian Society for Bioengineering. [Cited 8 May 2021]. https://library.csbe-scgab.ca/all-publications/category/view/135-cigr-and-agm-quebec-city-2010

Zhang, G., Xiao, X., Dong, J., Xin, F., Zhang, Y., Qin, Y., Doughty, R.B. & Moore, B. 2020. Fingerprint of rice paddies in spatial–temporal dynamics of atmospheric methane concentration in monsoon Asia. *Nature Communications*, 11(1): 1–11. https://doi.org/10.1038/s41467-019-14155-5

Zhang, L., Huang, X., Xue, B., Peng, Q., Wang, Z., Yan, T. & Wang, L. 2015. Immunization against rumen methanogenesis by vaccination with a new recombinant protein. *PLoS ONE*, 10(10): e0140086. https://doi.org/10.1371/journal.pone.0140086

Zhang, L., Yuan, F., Bai, J., Duan, H., Gu, X., Hou, L., Huang, Y., Yang, M., He, J.-S., Zhang, Z., Yu, L., Song, C., Lipson, D. A., Zona, D., Oechel, W., Janssens, I. A. & Xu, X. 2020. Phosphorus alleviation of nitrogen-suppressed methane sink in global grasslands. *Ecology Letters*, 23(5): 821–830. https://doi.org/10.1111/ele.13480

Zhang, R., Edalati, H., El-Mashad, H.M. & Chen, Y. 2019. *Effect of solid separation on mitigation of methane emission in dairy manure lagoons*. Project report No. 15-0610-SA. Davis, USA, University of California Davis.

Zhang, X.M., Smith, M.L., Gruninger, R.J., Kung, L., Vyas, D., McGinn, S.M., Kindermann, M., Wang, M., Tan, Z.L. & Beauchemin, K.A. 2021. Combined effects of 3-nitrooxypropanol and canola oil supplementation on methane emissions, rumen fermentation and biohydrogenation, and total tract digestibility in beef cattle. *Journal of Animal Science*, 99(4): skab081. https://doi.org/10.1093/jas/skab081

Zhang, Z.-W., Wang, Y.-L., Chen, Y.-Y., Wang, W.-K., Zhang, L.-T., Luo, H.-L. & Yang, H.-J. 2019a. Nitroethanol in comparison with monensin exhibits greater feed efficiency through inhibiting rumen methanogenesis more efficiently and persistently in feedlotting lambs. *Animals*, 9(10): 784. https://doi.org/10.3390/ani9100784

Zhang, Z.-W., Wang, Y.-L., Wang, W.-K., Chen, Y.-Y., Si, X.-M., Wang, Y.-J., Wang, W., Cao, Z.-J., Li, S.-L. & Yang, H.-J. 2019b. The antimethanogenic nitrocompounds can be cleaved into nitrite by rumen microorganisms: A comparison of nitroethane, 2-nitroethanol, and 2-nitro-1-propanol. *Metabolites*, 10(1): 15. https://doi.org/10.3390/metabo10010015

Zhao, Y., Nan, X., Yang, L., Zheng, S., Jiang, L. & Xiong, B. 2020. A review of enteric methane emission measurement techniques in ruminants. *Animals*, 10(6): 1004. https://doi.org/10.3390/ani10061004

Zhao, Y.G., O'Connell, N.E. & Yan, T. 2016. Prediction of enteric methane emissions from sheep offered fresh perennial ryegrass (*Lolium perenne*) using data measured in indirect open-circuit respiration chambers. *Journal of Animal Science*, 94(6): 2425–2435. https://doi.org/10.2527/jas.2016-0334

Zhou, C.S., Xiao, W.J., Tan, Z.L., Salem, A.Z.M., Geng, M.M., Tang, S.X., Wang, M., Han, X.F. & Kang, J.H. 2012. Effects of dietary supplementation of tea saponins (*Ilex kudingcha* C.J. Tseng) on ruminal fermentation, digestibility and plasma antioxidant parameters in goats. *Animal Feed Science and Technology*, 176(1): 163–169. https://doi.org/10.1016/j.anifeedsci.2012.07.019

Zhou, X., Zeitz, J.O., Meile, L., Kreuzer, M. & Schwarm, A. 2015. Influence of pH and the degree of protonation on the inhibitory effect of fatty acids in the ruminal methanogen *Methanobrevibacter ruminantium* strain M1. *Journal of Applied Microbiology*, 119(6): 1482–1493. https://doi.org/10.1111/jam.12955

SOURCES OF TABLES AND FIGURES

Allen, M.R., Lynch, J., Cain, M. & Frame, D. 2022. Climate metrics for ruminant livestock. Oxford, UK, Oxford Martin Programme on Climate Pollutants. https://www.oxfordmartin.ox.ac.uk/downloads/reports/ClimateMetricsforRuminentLivestock_Brief_July2022_FINAL.pdf

Feng, X.Y. & Kebreab, E. 2020. Net reductions in greenhouse gas emissions from feed additive use in California dairy cattle. *PLoS ONE*, 15(9). https://doi.org/10.1371/journal.pone.0234289

Ferry, J.G. 2015. Acetate metabolism in anaerobes from the domain *Archaea. Life*, 5(2): 1454–1471. https://doi.org/10.3390/life5021454

Fløjgaard, C., Pedersen, P.B.M., Sandom, C.J., Svenning, J. & Ejrnæs, R. 2022. Exploring a natural baseline for large-herbivore biomass in ecological restoration. *Journal of Applied Ecology*, 59(1): 18–24. https://doi.org/10.1111/1365-2664.14047

Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M. & Zhang, H. 2021. The Earth's energy budget, climate feedbacks, and climate sensitivity. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu & B. Zhou, eds. *Climate change 2021: The physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, pp. 923–1054. Cambridge, UK & New York, USA, Cambridge University Press. https://doi.org/10.1017/9781009157896.001

Fuglestvedt, J.S., Berntsen, T.K., Godal, O., Sausen, R., Shine, K.P. & Skodvin, T. 2003. Metrics of climate change: Assessing radiative forcing and emission indices. *Climatic Change*, 58: 267–331. https://doi.org/10.1023/A:1023905326842

Hill, J., McSweeney, C., Wright, A.-D.G., Bishop-Hurley, G. & Kalantarzadeh, K. 2016. Measuring methane production from ruminants. *Trends in Biotechnology*, 34(1): 26–35. https://doi.org/10.1016/j.tibtech.2015.10.004

IPCC. 2014. *Climate change 2014: Synthesis report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (R.K. Pachauri & L.A. Meyer, eds.), 151 pp. Geneva, Switzerland.

Manzano, P. & White, S. 2019. Intensifying pastoralism may not reduce greenhouse gas emissions: Wildlife-dominated landscape scenarios as a baseline in life-cycle analysis. *Climate Research*, 77: 91–97. https://doi.org/10.3354/cr01555

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T. & Zhang, H. 2013. Anthropogenic and natural radiative forcing. In: T.F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P.M. Midgley, eds. *Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK & New York, USA, Cambridge University Press. www.ipcc.ch/site/assets/uploads/2018/02/ WG1AR5_Chapter08_FINAL.pdf

Plattner G.-K., Stocker, T., Midgley, P. & Tigno, M. 2009. IPCC Expert meeting on the science of alternative metrics, Oslo, Norway, 18-20 March 2009. www.ipcc.ch/site/assets/uploads/2018/05/expert-meeting-metrics-oslo.pdf

Reisinger, A. & Clark, H. 2018. How much do direct livestock emissions actually contribute to global warming? *Global Change Biology*, 24(4): 1749–1761. https://doi.org/10.1111/gcb.13975

Reisinger, A., Clark, H., Cowie, A.L., Emmet-Booth, J., Gonzalez Fischer, C., Herrero, M., Howden, M. & Leahy, S. 2021. How necessary and feasible are reductions of methane emissions from livestock to support stringent temperature goals? *Philosophical Transactions of the Royal Society. Series A – Mathematical, Physical and Engineering Sciences*, 379(2210): 20200452. https://doi.org/10.1098/rsta.2020.0452

Russell, J.B. & Wallace, R.J. 1997. Energy-yielding and energy-consuming reactions. In: P.N. Hobson & C.S. Stewart, eds. *The rumen microbial ecosystem*, pp. 246–282. London, Blackie Academic & Professional. https://doi.org/10.1007/978-94-009-1453-7_6

Tanaka, K. & O'Neill, B.C. 2018. The Paris Agreement zero-emissions goal is not always consistent with the 1.5 °C and 2 °C temperature targets. *Nature Climate Change*, 8(4): 319–324. https://doi.org/10.1038/s41558-018-0097-x

Tanaka, K., Boucher, O., Ciais, P., Johansson, D.J.A. & Morfeldt, J. 2021. Cost-effective implementation of the Paris Agreement using flexible greenhouse gas metrics. *Science Advances*, 7(22): eabf9020. https://doi.org/10.1126/sciadv.abf9020

Tedeschi, L.O., Abdalla, A.L., Álvarez, C., Anuga, S.W., Arango, J., Beauchemin, K.A., Becquet, P., Berndt, A., Burns, R., De Camillis, C., Chará, J., Echazarreta, J.M., Hassouna, M., Kenny, D., Mathot, M., Mauricio, R.M., McClelland, S.C., Niu, M., Onyango, A.A., Parajuli, R., Pereira, L.G.R., del Prado, A., Tieri, M.P., Uwizeye, A. & Kebreab, E. 2022. Quantification of methane emitted by ruminants: A review of methods. *Journal of Animal Science*, 100(7): 1–22. https://doi.org/10.1093/jas/skac197

Ungerfeld, E.M. 2020. Metabolic hydrogen flows in rumen fermentation: Principles and possibilities of interventions. *Frontiers in Microbiology*, 11: 589. https://doi.org/10.3389/fmicb.2020.00589

Appendix

Details of case studies

This section provides technical details of the model-based temperature calculations in Section 9.4 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)[6] 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 concentrations) on annual, decadal and centennial timescales. Such models do not deal with interannual and decadal variability of the earth system, nor with the seasonal cycle within a year. They do not generally provide projections at a regional scale.

ACC2 consists of a carbon cycle, atmospheric chemistry, physical climate and economy modules. In the examples given here, ACC2 is used as a simple climate model, without the economy module, which is required when ACC2 is used as an integrated assessment model. The inputs into ACC2 are the emission scenarios of greenhouse gases and air pollutants. The outputs from the model are the projections of atmospheric concentrations and radiative forcing of CO_2, CH_4 and N_2O, among other, and global-annual-mean temperature changes relative to preindustrial levels.

The physical climate module of ACC2 is an energy balance model coupled with the 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 the key non-linearities of the global carbon cycle. 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 with rising atmospheric CO_2 concentration due to the CO_2 fertilization effect. The atmospheric chemistry module accounts for the tropospheric O_3 production from CH_4 emissions. The lifetime of CH_4 is related to the OH concentration, which itself depends on the CH_4 concentration and pollutant emissions, providing a positive feedback to the lifetime of CH_4. 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 CO_2, 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 °C, the best estimate of Working Group I for the IPCC's Sixth

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[6] 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's 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 (coloured lines) and the temperatures purely derived from the model (black lines).

Assessment Report (IPCC, 2021). Other uncertain parameters are optimized by using historical data and observations based on a Bayesian approach (Tanaka *et al.*, 2009b).

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/m^2, an emission scenario in which the radiative forcing is stabilized at 4.5 W/m^2 in the year 2100 (Moss *et al.*, 2010). Thus, in our examples, emissions from individual farms are modelled on top of the RCP 4.5 scenario. The emission data for RCP 4.5 used in our analysis is consistent with that used in the intercomparison project for simple climate models (Nicholls *et al.*, 2020). When we added farm emissions to the RCP 4.5 scenario, we assumed 1000 times larger farm emissions than the original magnitudes. Then the temperature difference due to farm emissions calculated from the model was divided by 1000. Table 11 shows the change in absolute CO_2eq emissions when using the feed additive for individual farms aggregated using GWP, GTP and GWP*. We checked the sensitivity of the results with respect to the scaling factor and confirmed that the results do not depend on the scaling factor within a large range including 1000.

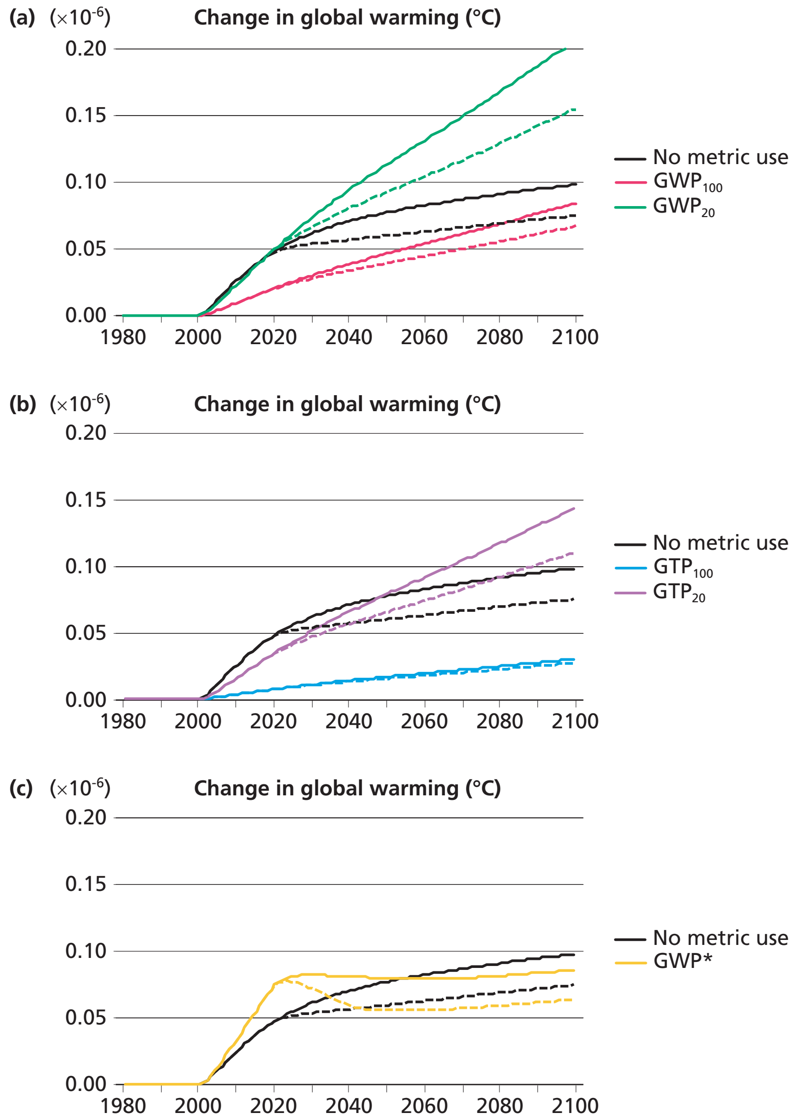
Table A1. Absolute emissions when using the feed additive, relative to no emissions, aggregated using GWP, GTP and GWP*

Unit	CH_4	N_2O	CO_2	Aggregated
Tonnes of each gas per year	40	1.68	105	N/A
GWP_{100} CO_2eq tonnes per year	1 080	458	105	1 644
GWP_{20} CO_2eq tonnes per year	3 188	458	105	3 751
GTP_{100} CO_2eq tonnes per year	188	391	105	684
GTP_{20} CO_2eq tonnes per year	2 080	498	105	2 683
GWP* CO_2eq tonnes per year (for the first 20 years)	5 074	458	105	5 637
GWP* CO_2eq tonnes per year (after 20 years of stabilized new emissions)	314	458	105	877

Note that CO_2eq emissions are calculated using the IPCC's AR6 values for GWP and GTP (for example, 27 for GWP_{100} CH_4), with the exception of CO_2eq emissions based on GWP, which use the IPCC's AR5 value of GWP_{100} (that is, 28 for GWP_{100} CH_4) as described in the GWP* formula (Smith, Cain and Allen, 2021; footnote of Section 7.6.1.4 in IPCC [2021]). Note that a change to using the AR6 value of GWP_{100} in the GWP* formula would remain well within the uncertainties and not affect the results in any meaningful way.*

Source: Authors' own elaboration.

Figure A1
Additional results for Example 1



(a) Change in global warming (°C)

(b) Change in global warming (°C)

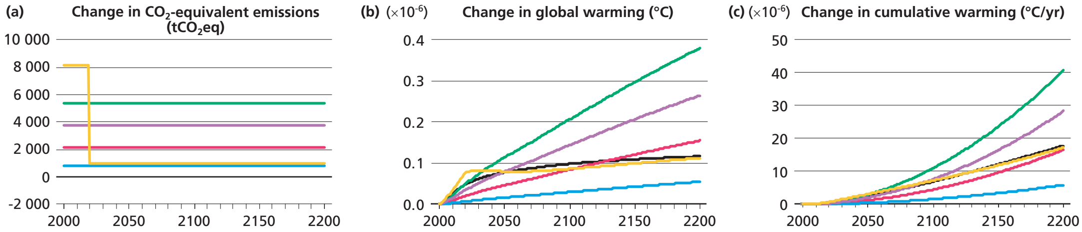
(c) Change in global warming (°C)

Modelled global warming from the control farm (solid) and the feed additive farm (dashed) scenarios (in black). Coloured lines show modelled global warming from CO_2 emissions derived using different metrics of equivalence. Panels a, b and c show GWP-based equivalence, GTP-based equivalence and GWP-based equivalence, respectively.*

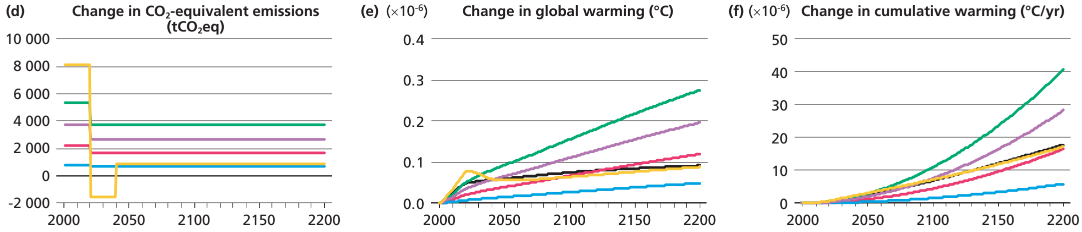
Source: Authors' own elaboration.

Figure A2
Detailed results for Example 1
(evaluation of emission metrics in representing the benefits of using a feed additive)

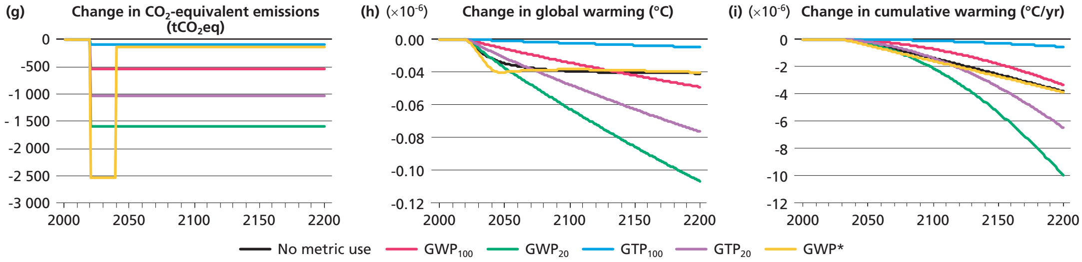
I. Control farm scenario compared to no farm scenario



II. Feed additive farm scenario compared to no farm scenario



III. Feed additive farm scenario compared to control farm scenario



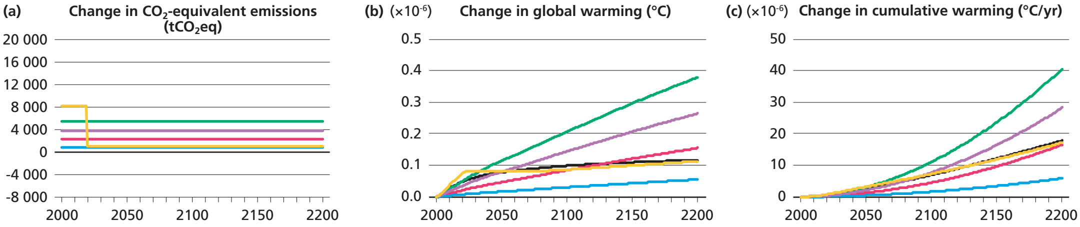
This figure shows the results for a longer timescale (until 2200), including cumulative warming, a proxy of climate damage. See the main text for further details.

Source: Authors' own elaboration.

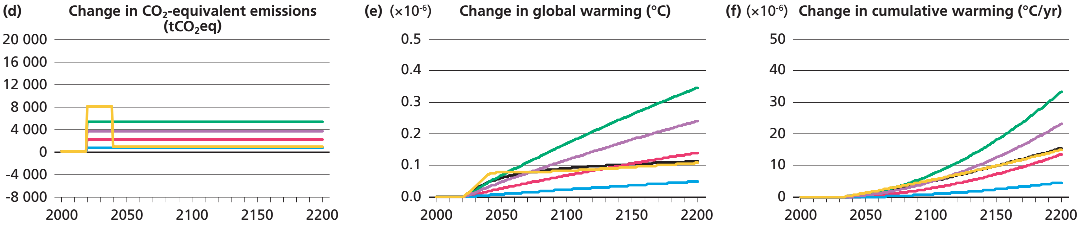
Figure A3

Detailed results for Example 2
(illustrating the path dependency of step-pulse metrics in representing the impact of three farmers with different historical emissions)

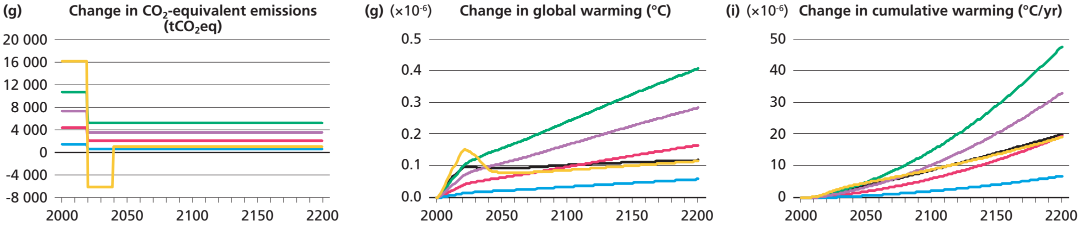
I. Abraham farm compared to no farm



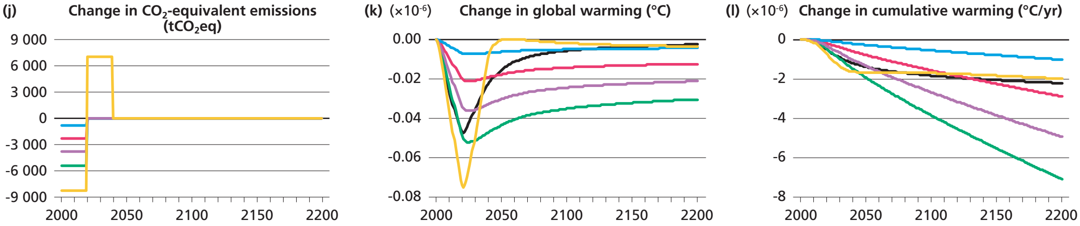
II. Bethany farm compared to no farm



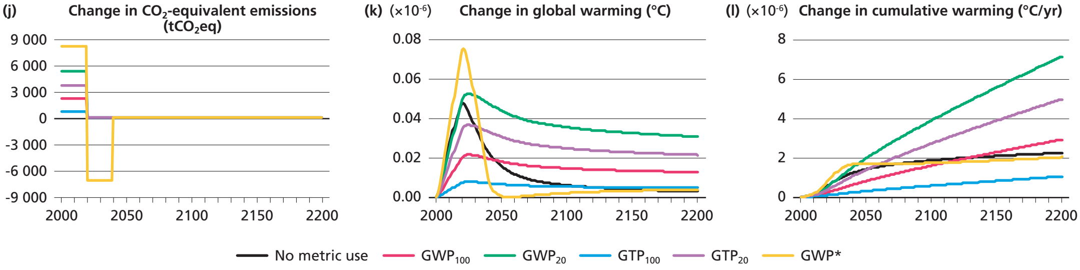
III. Chris farm compared to no farm



IV. Bethany farm compared to Abraham farm



V. Chris farm compared to Abraham farm



This figure shows the results for a longer timescale (until 2200), as well as the temperature outcome based on the same method as the one used in Example 1. See the main text for further details.

Source: Authors' own elaboration.

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