CH$_4$ and N$_2$O Emissions From Cattle Excreta: A Review of Main Drivers and Mitigation Strategies in Grazing Systems

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Cattle production systems are an important source of greenhouse gases (GHG) emitted to the atmosphere. Animal manure and managed soils are the most important sources of emissions from livestock after enteric methane. It is estimated that the N$_2$O and CH$_4$ produced in grasslands and manure management systems can contribute up to 25% of the emissions generated at the farm level, and therefore it is important to identify strategies to reduce the fluxes of these gases, especially in grazing systems where mitigation strategies have received less attention. This review describes the main factors that affect the emission of GHG from manure in bovine systems and the main strategies for their mitigation with emphasis on grazing production systems. The emissions of N$_2$O and CH$_4$ are highly variable and depend on multiple factors, which makes it difficult to use strategies that mitigate both gases simultaneously. We found that strategies such as the optimization of the diet, the implementation of silvopastoral systems and other practices with the capacity to improve soil quality and cover, and the use of nitrogen fixing plants are among the practices with more potential to reduce emissions from manure and at the same time contribute to increase carbon capture and improve food production. These strategies can be implemented to reduce the emissions of both gases and, depending on the method used and the production system, the reductions can reach up to 50% of CH$_4$ or N$_2$O emissions from manure according to different studies. However, many research gaps should be addressed in order to obtain such reductions at a larger scale.

Keywords: global warming, climate change, nutrient excretion, nitrification, mitigation, nitrogen losses

INTRODUCTION

Greenhouse gas (GHG) concentrations in the world have increased rapidly since pre-industrial times due to human activities, with negative effects on the climate (IPCC, Intergovernmental Panel on Climate Change, 2013). Methane (CH$_4$) concentrations have doubled while nitrous oxide (N$_2$O) concentrations in the atmosphere are 20% higher than pre-industrial levels (IPCC, Intergovernmental Panel on Climate Change, 2013). Agriculture is considered one of the main sources of CH$_4$ and N$_2$O, two high warming potential gases. Within the agricultural sector, animal production contributes 14.5% of human-induced emissions (Gerber et al., 2013) and produces ~37 and 65% of global emissions of CH$_4$ and N$_2$O, respectively (Steinfeld and Wassenaar, 2007).
Within livestock, cattle production systems can be broadly classified into confined, mixed在哪 in which cattle can be in-house during part of the day or the year, and grassland-based systems (Seré and Steinfeld, 1995). In confined and semiconfined systems, manure can be stored and processed to be disposed in the field, whereas in grazing systems, manure is deposited directly on pastures and is degraded under environmental and grazing conditions (Uchida et al., 2011). Manure (feces and urine) managed and deposited on grasslands and pastures is the second largest source of GHG emissions after enteric methane and is responsible for ~7% of agricultural emissions of CH4 and N2O worldwide (Aguirre-Villegas and Larson, 2017).

Nitrous oxide is the third most abundant GHG and accounts for 6% of all radiative forcing (Myhre et al., 2013). Despite its low concentration in the atmosphere compared to CH4 and CO2, N2O has a significant effect on global warming, as it has a lifespan of ~120 years and 265 times higher radiative potential than CO2 (IPCC (Intergovernmental Panel on Climate Change), 2014; U. S. E. P. A and United States Environmental Protection U. S. E. P. A. United States Environmental Protection Agency, 2021). In addition, it contributes significantly to the depletion of stratospheric ozone (Myhre et al., 2013).

Grasslands around the world emit about 2.2 Tg of N2O–N, 74% of which comes from anthropogenic sources (Dangal et al., 2019). Deposition of animal feces and urine is the biggest source of N2O emissions per year in grasslands (54%), followed by manure application (13%), and nitrogen fertilizers (7%) (Dangal et al., 2019). Nitrification and denitrification are the main responsible mechanisms for the production of N2O in soils, although nitrification-denitrification, codenitriﬁcation and chemodenitriﬁcation can also lead to the formation of N2O given a microbial community and suitable environmental conditions (Hallin et al., 2018). Regarding methane, ruminant manure is responsible for the emissions of 109 million tons of this GHG to the atmosphere per year, of which 86% comes from cattle. Three main factors affect the amount of CH4 emitted by manure: the type of storage, the climate, and the composition of manure (Opio et al., 2013). While most of CH4 emissions from manure occur during storage under anaerobic conditions, in tropical regions manure can also be a generator of a considerable amount of emissions of this gas at the grassland level (Montes et al., 2013; Cai et al., 2017).

However, it is important to mention that although these gases play an important role in global warming, as they are predominantly flow pollutant gasses, they differ in their impact from CO2 that is a stock pollutant with a very long-term persistence in the atmosphere and consequently with a greater cumulative effect on the climate (Lynch et al., 2021). In addition, grasslands around the world also hold a large mitigation potential for building and conserving soil carbon and could capture as much as 0.5 Pg C per year to 1 m depth, as they cover ~52.5 million km2 equivalent to 40.5% of the land area (Gerber et al., 2013; Lorenz and Lal, 2018).

This article reviews the magnitude of typical N2O and CH4 emissions from manure, analyses the factors affecting them, and discuss potential mitigation strategies in bovine production systems, with an emphasis on tropical and subtropical regions where grazing systems are predominant.

**NITROUS OXIDE AND METHANE EMISSIONS IN GRAZING SYSTEMS**

**Nitrous Oxide Emissions**

Ruminants are poor nitrogen converters, because only 5–30% of ingested nitrogen are up taken by the animal and the remaining 70–95% are excreted via feces and urine (Luo et al., 2010). Therefore, nitrogen loads in animal excreta, often exceed plant demands and are vulnerable to losses via gaseous emissions and leaching (Selbie et al., 2015). This is more critical as the proportion of nitrogen in animal urine has increased with increasing nitrogen intake; although it has remained relatively constant in feces (Jarvis et al., 1995). According to Jaimes and Chará (2016) the efficiency in the use of N by lactating cows varies between 8.96 and 27.82% in Colombia, which, according to the number of animals per unit area can generate the application of up to 374 kg of N/ha/yr from manure (Correa et al., 2012). Likewise, Rivera et al. (2018) found that cows excreted 72% of ingested nitrogen in tropical dairy systems, generating the deposition of 46.8 kg N/animal/yr from manure and 42.9 kg from urine when the diet had on average 14% crude protein (CP). For this reason, improving the efficiency in the use of this nutrient by ruminants may be a viable alternative not only to increase animal productivity but also to reduce GHG emissions by reducing N excretion. It must be noted however that in extensive grazing and pastoralist systems cattle can be undernourished, and the scarce nutrients can be used more effectively by animals (Manzano and White, 2019).

Grazed pastures are systems with a wide range of environmental and management conditions that can result in the emission of N2O (Wecking, 2021). A large proportion of total farm N2O emissions in grazing systems often occurs from relatively small areas (Luo et al., 2017). These sites can be located where animals congregate (feeding bins, water troughs and gateways), occur after additional irrigation or result from soil compaction due to trampling and in the soil underneath excreta patches (Roesch et al., 2019). The magnitude of N2O emissions depends on the interplay between prevailing soil microclimate, microbial activity, plant composition, biomass, and excreta composition, that in turn is defined by animal type and feed intake (Wecking, 2021). All these factors can alter the spatial heterogeneity of soil respiration and, hence, cause impact also on resulting N2O emissions (Shi et al., 2019). Nitrous oxide emissions from a single application of cattle urine and feces can be as high as 16.8 kg N2O-N/ha/yr and 5.57 kg N2O-N/ha/yr, respectively (Luo et al., 2018). A meta-analysis by López-Aizpún et al. (2019) showed that, when reporting urine derived N2O emissions, it was important to account for differences in animal...
diet, sex and breed, in addition to urine composition and nitrogen loads.

Factors Conditioning Nitrous Oxide Emissions From Manure

The two main processes that generate N$_2$O: nitrification and denitrification, are strongly influenced by climate and soil factors (Chen et al., 2008). The production of N$_2$O depends on the availability of substrates for both processes, i.e., NH$_4^+$ for nitrification and NO$_3^-$ for denitrification (Zaman et al., 2007). The most important factors are the presence of oxygen, temperature, pH, humidity, salinity, and soil management; in the case of denitrification, it also depends on the carbon available for heterotrophic processes (Dalal et al., 2003). In addition, these factors are regulated by climate, vegetation, chemical, and physical properties of soil (apparent density, organic C, pH, and clay content), and agricultural management practices (Uchida et al., 2011). Each of these factors is discussed in more detail below.

The process of nitrification was first described by Schloesing and Muentz in 1877. Nine years later, Gayon and Dupetit discovered denitrification (Elmerich and Newton, 2007). An overarching framework capturing the production and consumption of N$_2$O and NO by nitrification and denitrification within a conceptual model was published by Firestone and Davidson (1989) and has been acknowledged as the “hole in the pipe model” (Wecking, 2021). However, recent research suggests that a range of other biotic and abiotic pathways might also lead to the emission of N$_2$O e.g., heterotrophic nitrification, nitrifier denitrification, chemodenitrification, coupled nitrification-denitrification, co-denitrification, and anaerobic NH$_3$ oxidation—apart from potential other yet still undiscovered processes in the nitrogen-cycling network (Kuyper et al., 2018). Figure 1 shows the pathways of microbial driven nitrogen transformations in excreta patches in grassland ecosystems.

Irrespective of the underlying process, most nitrification and denitrification pathways in soil lead to the net emission of N$_2$O (Myrold, 2005). Only under certain conditions denitrifier activity, and to some extent that of nitrifiers, stimulate the uptake of atmospheric N$_2$O. Soils are more...
likely to act as a sink for N\textsubscript{2}O when the soil mineral nitrogen is low and when high soil moisture or other factors prevent microbial access to alternative oxygen (O\textsubscript{2}) sources (Philippot et al., 2009).

Among the factors affecting N\textsubscript{2}O emissions, the availability of C and N is critical, particularly when these elements are in labile organic form (van Groenigen et al., 2005). There is a linear relationship between the N input, either by fertilizer or by manure, and the emission of N\textsubscript{2}O in agricultural areas (Dobbie et al., 1999); although, according to Zebarth et al. (2008), this relationship can also be exponential. In relation to soil properties, the moisture content is perhaps the variable with greatest influence on N\textsubscript{2}O emissions (Saggar et al., 2004). Saturation values of 60–70% moisture promote the generation of N\textsubscript{2}O since they limit the O\textsubscript{2} diffusion, resulting in denitrification processes (Saggar et al., 2004). Nitrous oxide emissions are therefore higher in wet soils and after dry conditions, N\textsubscript{2}O production begins immediately after applying water to the soils (Chirinda et al., 2019).

Temperature is another factor influencing the level of emissions. When soil water content is close to the maximum retention capacity, N\textsubscript{2}O emissions respond to temperature changes (Machefert et al., 2002). However, some studies have shown no significant relationships between emission and temperature variations mainly due to the higher influence of moisture content in the flow of gases (Singurindty et al., 2009). Soil type can also influence N\textsubscript{2}O emissions, mainly through their effect on drainage level and moisture content (Luo et al., 2010). Poorly drained soils have higher N\textsubscript{2}O emissions than well-drained soils (Oenema et al., 2007). Poor drainage and changes in the physical properties of soil such as compaction can affect transformations from N to N\textsubscript{2}O because they affect the soil oxygen diffusion (Oenema et al., 1997) and can increase N gaseous losses (van der Meer, 2008).

Animal grazing with its trampling can favor this condition compared with grasslands without animal occupation. Compaction can also be increased when grazing takes place during winter and/or when there is inadequate management of animal stocking rates as this causes loss of structure and drastic decrease of porous space (Luo et al., 2010). pH can also affect the mechanisms that control N\textsubscript{2}O emissions. A study of denitrifying enzymes found a link between soil pH and soil emission rate (van der Weerden et al., 1999), as denitrification processes decrease as soil pH tends to acidity. According to Dalal et al. (2003), the optimal pH for nitrification activity is 7, while for denitrification the optimal pH is 7.0–8.0.

Nitrous Oxide Emissions in Feces and Urine
Most studies comparing N\textsubscript{2}O–N emission factors of feces and urine under grazing conditions suggest higher values for urine (van Groenigen et al., 2005; López-Aizpún et al., 2020). However, authors such as Sherlock et al. (2003) have reported similar values for both components, while Wachendorf et al. (2008) found higher emissions of N\textsubscript{2}O–N in livestock feces, suggesting the need of disaggregating emission factors based on the type of excreta (Luo et al., 2010; López-Aizpún et al., 2020). According to Meng et al. (2014) and Sordi et al. (2014), the fraction of N lost as N\textsubscript{2}O–N in the urine is greater than that of feces because only a relatively small fraction of N in manure is in an unstable condition and this depends on the diet of the animals.

According to the Intergovernmental Panel on Climate Change (IPCC) guidelines, it is estimated that the generation of N\textsubscript{2}O by manure deposited in the grasslands corresponds to 2% of the total excreted N (IPCC (Intergovernmental Panel on Climate Change), 2006). However, some studies have found that this value may be considerably lower, to such a point that by 2019 this same body changed that value to 0.4% to estimate direct emissions and another 0.27% for indirect emissions by leaching and volatilization (IPCC (Intergovernmental Panel on Climate Change), 2019). In a study carried out in New Zealand, Luo et al. (2008) reported emission factors (EF) for urine applications between 0.2 and 1.59%, depending on the season. This variability in EF highlights the importance of determining country-specific or climate region emissions. This variability can be caused by multiple factors such as: (i) moisture, carbon content, pH and soil structure; (ii) environmental conditions such as temperature and rain (due to its influence on soil moisture); (iii) quantity and availability of nutrients in soil and excreta; (iv) plant species present; and (v) soil management (Oertel et al., 2016; López-Aizpún et al., 2020).

Countries like New Zealand have advanced in disaggregating EF by type of livestock (bovines and sheep), type of excreta (feces and urine) and climate (wet and dry). In this country, EF for urine and feces are 1 and 0.25% respectively, and both have been implemented in the national inventory of agricultural greenhouse gases (van der Weerden et al., 2020). New Zealand values, applied to all major classes of livestock (sheep, cattle, and deer), are similar to those found by Chadwick et al. (2018) in studies in the United Kingdom (average urine and feces of 0.69 and 0.19%, respectively) and Ireland where EF of 1.18 and 0.31% have been found for these two emission sources, respectively (Krol et al., 2016). Similarly, Rivera et al. (2018) found EF between 1.37 and 1.77% for feces, and between 0.3 and 3.47% for urine in tropical conditions of Colombia, which are higher than those reported by Sordi et al. (2014) in Brazil who found values from 0.19 to 0.33% and 0.12 to 0.19% for urine and feces, respectively. Figure 2 represents the final distribution of N in urine and feces, showing higher amounts of N leached and higher amounts of NH\textsubscript{3} in urine than in feces, which could lead to higher emissions in urine patches.

Methane Emissions
The production of CH\textsubscript{4} occurs via the microbial degradation of the proteins, organic acids, carbohydrates, and soluble lipids present in excreta (Khan et al., 1997). According to the IPCC-Tier 1 (2006), 1 kg of CH\textsubscript{4} is emitted from dung annually per adult head of cattle in grazing systems, but according to others reports these values may be lower (0.45–0.67 kg/animal/day), and can be highly variable (IPCC (Intergovernmental Panel on Climate Change), 2019).

In the soil, CH\textsubscript{4} is produced under anaerobic conditions by methanogens and is converted to CO\textsubscript{2} by methanotrophs under
both aerobic and anaerobic conditions, and the net CH₄ flux in the soil-atmosphere system represents the balance between these two microbial processes (Le Mer and Roger, 2001). The impact of excreta deposition on CH₄ emission thus mainly depends on its relative impact on CH₄ production and CH₄ oxidation activities in the patch (Cai et al., 2017).

Around 15–30% of total global CH₄ emissions could be derived from soil source (Yanan et al., 2018). Ruminant manure is responsible for the emissions of 109 million tons of this GHG to the atmosphere per year, of which 86% comes from cattle. Three main factors affect the amount of CH₄ emitted by manure: the type of treatment, the climate, and the composition of manure (Opio et al., 2013). While most of CH₄ emissions from manure occur during storage under anaerobic conditions, in tropical regions manure can also be a generator of a considerable amount of emissions of this gas at the grassland level (Montes et al., 2013; Cai et al., 2017).

Even though pastures can emit CH₄, under certain conditions, upland soils including those covered by grasslands are also an important sink for atmospheric CH₄ as they can oxidize it at a faster rate than croplands (between 3 and 6 kg of CH₄/ha/yr), although at a slower rate than uncultivated soils (Boeckx and Van Cleemput, 2001). This is caused by the oxidative activity of methanotrophs and ammonium oxidizing bacteria (Shukla et al., 2013). The oxidative capacity is nevertheless affected, among other factors, by water content and inorganic N in the soil, by the use of inorganic fertilizers and by the NH₄⁺ released during urine urea hydrolysis (Le Mer and Roger, 2001; Saari et al., 2004). During the rainy season CH₄ emissions rise due to increased anaerobic conditions caused by water saturation but, when soil moisture is reduced due to decreased rainfall, these are replaced by oxidative processes with predominance of aerobic bacteria that generate negative methane flows and act as CH₄ sinks (Vischer et al., 2007).

**Factors Conditioning Methane Emissions**

As for N₂O, microbial processes that determine methane emissions into the atmosphere in grazing systems are conditioned by soil factors such as redox potential, pH, temperature, organic carbon and nitrogen content (Towprayoon et al., 2005). These factors can affect the proliferation of some soil microorganisms and, in turn, promote or limit bacterial metabolism through its impact on synthesis and enzymatic activity. The process of methanogenesis is regulated by the concentration of O₂, the content of organic matter as a substrate, and the factors that determine its redox potential (Conrad, 1996). Organic matter is the main input for triggering methane production processes. The increase of available organic matter, and its subsequent decomposition in soils under anaerobic conditions, stimulates methanogenesis by providing a substrate for the production of acetate and hydrogen and causing soil reducing conditions (Sass et al., 1991).

In pastures, most of the organic matter comes from plants through leaves senescence and root decomposition, and the transformation and deposit by animal excreta (Waschütz et al., 1992). For this reason, forage species and their physiological status can also influence methane emissions (Kerdchoechuen, 2005). The strictly anaerobic methanogen, mainly *Methanobacteria*, *Methanococci* and *Methanopyri*, are sensitive to changes in soil water content (Malyan et al., 2016).
Methanogenic activity may be stimulated by urine deposition that create anaerobic conditions. Furthermore, increased soil pH resulting from hydrolysis of urine urea and decreased redox potential may also favor methanogenic activities (Le Mer and Roger, 2001). Emissions of CH\textsubscript{4} from excreta deposited by animals on pastures range from 7 to 27% of total emissions by ruminants (Kreuzer and Hindrichsen, 2006). However, these emissions may become less significant depending on environmental conditions and manure management (Öenema et al., 2007). When anaerobic conditions occur, ruminant manure in pastures release significant amounts of CH\textsubscript{4} (Misselbrook et al., 2001).

**Methane Emissions in Feces and Urine**

In an evaluation of methane fluxes of an intensive silvopastoral system (iSPS) with high density of Leucaena, an intensive pasture monoculture system and a secondary dry forest, Rivera et al. (2018) found that both the forest and the iSPS had negative CH\textsubscript{4} flows of −0.56 and −0.02 kg of CH\textsubscript{4}/ha/yr respectively, probably due to the biodiversity of microorganisms found in their soils (Vallejo et al., 2010). Negative flows of CH\textsubscript{4} were also found during the dry season in three pasture systems in the north of Colombia (Espinosa-Carvajal et al., 2020). Methane flows are also influenced by the grass species and fertilization level. Pastrana et al. (2011) in a study evaluating three accessions of *Brachiaria humidicola* (Rendle) Schweickerdt found that methane emissions were increased from sink (~23.6 μg m\textsuperscript{-2} h\textsuperscript{-1}) when nitrogen fertilization was zero, to emit 107.9 μg m\textsuperscript{-2} h\textsuperscript{-1} with application of 150 kg of N/ha/yr and 59.7 μg m\textsuperscript{-2} h\textsuperscript{-1} with 300 kg of N/ha/yr. CH\textsubscript{4} emissions were also affected by the *B. humidicola* accession.

CH\textsubscript{4} emissions from excreta deposited by animals on pastures range from 7 to 27% of total emissions by ruminants (Kreuzer and Hindrichsen, 2006). However, these emissions may become less significant depending on environmental conditions and manure management (Öenema et al., 2007). According to recent IPCC estimates, 0.49 ± 0.43 g of CH\textsubscript{4} per kg of dry matter (DM) of manure can be generated on average (IPCC (Intergovernmental Panel on Climate Change), 2019). In addition, urine deposited directly in the grasslands increases emissions of this gas by up to 100 times compared to grasslands without urine patches (Oertel et al., 2016). The diversity of conditions that change CH\textsubscript{4} emissions has generated great dispersion in the results (Andueza et al., 2017), as there are reports of up to 1 g of CH\textsubscript{4} per kg of DM of manure generated, value that can be up to five times higher when manure is incorrectly stored under anaerobic conditions (Sneath et al., 2006). According to Chadwick (2005) manure emissions can range from 0.4 to 9.7% of the total C content deposited by manure.

Finally, Life Cycle Analysis (LCA) studies have identified that manure emissions (CH\textsubscript{4} and N\textsubscript{2}O) can be considerable under different production conditions. For example, Rivera et al. (2014), who evaluated the LCA in two dairy systems in Colombia, found that manure emissions accounted for 30% as CO\textsubscript{2}-eq of the total emitted on the farm and 22% of the total emitted throughout the LCA. In another study in dairy production systems, Rivera et al. (2016) found that manure emissions accounted for 6.5% of all farm-level emissions. According to this study, emission levels from manure depend mainly on the amount of excreted N given by the consumption of this nutrient in the diet, by the type of manure management, and by the rainfall regimen and use (or not) of irrigation in the grasslands.

**NITROUS OXIDE AND METHANE EMISSION MITIGATION STRATEGIES**

**Opportunities and Tradeoffs**

Opportunities to reduce N\textsubscript{2}O and CH\textsubscript{4} emissions from livestock manure are diverse and can be addressed to different parts of the animal production cycle to control the production and emission of these two gases (Montes et al., 2013). Given the different environmental and metabolic conditions that influence N\textsubscript{2}O and CH\textsubscript{4} flows from soil and manure, it is difficult to implement efficient mitigation alternatives that target both gases simultaneously (Montes et al., 2013). However, as presented in Table 1; Figure 3, measures related to improving production efficiency and improving soil protection with adequate cover and introduction of trees can contribute simultaneously to reduce emissions of both gases and, in addition, improve carbon sequestration.

The success of mitigation measures can be estimated based on the optimal conditions under which nitrification and denitrification processes occur, in the case of N\textsubscript{2}O and from the dry matter degradation processes of manure for CH\textsubscript{4}. Since CH\textsubscript{4} is produced under anaerobic conditions, while N\textsubscript{2}O production requires sufficient oxygen levels, some practices that reduce CH\textsubscript{4} production tend to increase N\textsubscript{2}O emissions. Table 1 presents a list of mitigation alternatives with their potential impact and limitations.

According to de Klein and Eckard (2008) and Wecking (2021), strategies to mitigate N\textsubscript{2}O emissions from grazing systems should include two complementary approaches: (1) to manage the sources of nitrogen uptake at the animal level, and (2) to control N\textsubscript{2}O production *in situ* through soil and pastoral management. Beukes et al. (2010) determined that measures such as improved animal genetics, reduction of nitrogen fertilization, and improved grazing management had the potential to decrease GHG emissions from pasture-based systems by 27–32%. Future mitigation should ideally be focused on finding combined strategies that avoid any offsetting effect of the desired mitigation benefits and also, help to improve the efficiency of nitrogen cycling through the soil-plant-animal system while, at the same time, reducing emissions (Cai et al., 2017).

The first approach includes improving management and feeding practices, supplying nutrients, in particular protein sources, according to animal requirements, and increasing animal productivity and nitrogen efficiency per kilogram of animal product through breeding and genetic manipulation (de Klein and Eckard, 2008; Wecking, 2021). For the second approach, mitigation strategies addressed...
at soil and grazing management are manifold and include managing the intensity and timing of grazing events (van der Weerden et al., 2018), increasing pasture productivity and soil carbon storage (Whitehead et al., 2018), and improving nutrient, fertilizer, and manure management (Kim and Giltrap, 2017). Removing animals from pasture areas can reduce treading damage, prevent leaching and gaseous losses of N, and thus preserve soil conditions (Luo et al., 2017). However, negative side effects of stand-off pads can be the accumulation of manure and reduced production that might outweigh the mitigation benefits. van der Weerden et al. (2018) showed that controlled grazing was beneficial on poorly drained soils where it contributed to reducing N₂O emissions, whereas the approach was not suitable on imperfectly-drained pasture.

Nitrous oxide and CH₄ emissions differ between intensive and non-intensive systems. Intensive systems can emit more GHG because they have a higher stocking rate, offer commercial feed, use fertilizers and irrigation. However, under tropical and subtropical conditions extensive systems are predominant; non-intensively managed pastures occupy 66% of the total grassland area around the world (Klein Goldewijk et al., 2017). Mitigation strategies must be in accordance with the type of system to achieve cost-effective reductions under grazing conditions. The main mitigation strategies are presented in more detail below:

### Dietary Manipulation

The most promising options for reducing GHG emissions at the livestock management level include improving animal production through dietary changes. Nitrogen (N) excretion rates, which affect N₂O emissions from manure, are based on dry matter consumption (DMC) and its N content (Vergé et al., 2012). Therefore, dietary manipulation to optimize protein consumption, and thus improve the efficiency of N utilization, is one of the most effective measures to reduce emissions from manure (Novak and Fiorelli, 2010). The more nitrogen used by an animal, the less will be excreted; it is recommended that an adjusted amount of nutrients be offered in the diet to meet the animal’s requirements, thus, avoiding increased excretion (Schils et al., 2008). This condition can occur in both high-supply N systems such as dairy production, as well as in tropical systems where the N supply in the feed is reduced. According to Klausner et al. (1998), in dairy systems, feeding of lactating cows with rations based on their production decreases the excretion of N by up to 34%. According to these authors, the reduction occurs by optimizing microbial fermentation in rumen which significantly improves the use of N.

Since urine is the main source of volatile N emissions, manipulating the N excretion pathway becomes an important N₂O and NH₃ mitigation tool. Urea is the main nitrogenous component of ruminant urine reaching 60–80% of total urinary

### Table 1: Mitigation alternatives and potential for reducing CH₄ and N₂O emissions in grazing systems.

| Strategy | Mitigation potential | Possible negative/Limiting aspect | GHG mitigated | References |
|----------|----------------------|-----------------------------------|---------------|------------|
| Urease and nitrification inhibitors | 15–45% | Labor cost, economic cost, difficult to use, and production of other GHGs. | N₂O | de Klein and Monaghan, 2011; Di and Cameron, 2012 |
| Time of application of manure in the field | 17% | Labor cost, economic cost. | N₂O | VanderZaag et al., 2011 |
| Use of plant species that can inhibit nitrification | 60% | Labor cost, economic cost. | N₂O | Byrnes et al., 2017 |
| Use of SPS as a strategy to improve soil conditions in terms of cover and biodiversity | 57% | Labor cost, economic cost. | N₂O | Chirinda et al., 2019 |
| Integrate manure with fertilizer and crop rotation | 60% | Labor cost, productivity decline. | N₂O | Nguyen et al., 2017 |
| Dietary manipulation | 35–55% | Economic cost, productivity decline, and difficult to use in non-intensive systems. | CH₄ and N₂O | Klausner et al., 1998; Hristov et al., 2011; Lee et al., 2012; Lombardi et al., 2021 |
| Legumes as an alternative to N fertilizer | 15–45% | Productivity decline, Limited forage species in some places or climates. | N₂O | Li et al., 2013 |
| Improved soil cover and biological integrity | 25–65% | Economic cost, labor cost. | CH₄ and N₂O | Chirinda et al., 2019 |
| Application of biochar and liming material | 54% | Economic cost, labor cost, economic cost, and difficult to use in non-intensive systems. | N₂O | Cayuela et al., 2014 |
N in high-production dairy cows (Montes et al., 2013) and decreasing proportionally as dietary CP decreases (Colmenero and Broderick, 2006). In low-protein diets, ureic N may drop to 46–53% of total urinary N (Hristov et al., 2011; Lee et al., 2012). For this reason, decreasing the concentration of CP in the diet is an effective method to mitigate N emissions from manure (Hristov et al., 2013). Similarly, emissions of manure deposited in the soil are reduced because low CP diets generate manure with a slower N mineralization rate (Powell et al., 2011). Optimizing N supply to animals can achieve between 12 and 21% less N excretion and 15–33% less N volatilization losses in livestock fed according to the physiological status of the animals (Erickson and Klopfenstein, 2010).

De Klein and Eckard (2008) concluded that N₂O reduction should be part of an integrated approach to improving efficiency in the use of N in animal production systems. According to these authors, current technologies could offer up to 50% reduction in N₂O emissions from a confinement system, but only up to 15% of a grazing system. In intensively operated pastoral systems, supplementation of cattle with N-low foods such as maize or silage, which generally reduce the concentration of N in the diet, can reduce urinary N losses and, consequently, NH₃ and N₂O emissions in manure and soil by 8–36% (de Klein and Monaghan, 2011).

Also, plants species such as Lolium perenne, Trifolium repens, and Plantago lanceolata, that may exhibit diuretic properties have the potential to reduce the urinary-N loading in individual urine patches by increasing the urination frequency of grazing animals (de Klein et al., 2019; des Roseaux et al., 2020). Although the increased urination frequency results in greater coverage of urine-affected soil, total paddock-scale N₂O emissions from urine patches are not likely to be higher if the total amount of
urinary-N excreted remains the same. In fact, they could be lower if the N₂O emission factor reduces with N loading rate (de Klein et al., 2019).

Diet manipulation can also reduce CH₄ emissions from manure. In a study by Lombardi et al. (2021) the supplementation of grazing beef steers with maize grain lowered CH₄ emissions of dung from 4.0 to 1.7 g CH₄-C/m². Dung from supplemented animals had higher N, starch, and DM content, which resulted in lower CH₄ emissions compared with dung from non-supplemented animals. Results from this study indicated that the initial water content may control the CH₄ emissions, since rainfall events after dung crusting did not increase the CH₄ fluxes from dung patches (Zhu et al., 2018). On the other hand, these authors state that supplementation with maize grain can thus have dual benefits in cattle production, through maximizing body weight gain in grazing steers (18% more) by improving the efficiency of utilization of nutrients (dietary N in particular) and through decreasing the total amount and/or the intensity of GHG emissions. In addition, the improvement in N utilization efficiency may reduce N₂O emissions from urine deposition (Cai et al., 2017).

This mitigation route can be used especially in grazing systems with daily rotation, where there is a greater control in feeding. In these systems animals are usually supplemented during milking or at certain times of the year (for example in summer or winter, according to the area). Also, the manipulation of the diet can be done to certain groups of animals that may demand more nutrients or simply by using pastures with various forage species that can be supplemented or whose nutrient supply is in accordance with the physiological state of the animal. This could be applied to both intensive and non-intensive systems (use of species that favor an adequate energy:protein balance).

Although diet supplementation might be difficult under very extensive systems, it is possible in more managed systems such as those under rotational grazing.

### Improved Soil Cover and Biological Integrity

Maintaining a more diverse environment with healthy soils and good pasture cover is another strategy that can help reduce emissions. Chirinda et al. (2019) found lower urine patch emission factors in seven locations in South America (0.42 vs. 0.18%) when the grasslands had greater plant cover compared to areas with poor cover or degraded pastures. The results indicate that, under rainy conditions, adequate plant cover, through good pasture management, helps reduce urine-induced N₂O emissions. According to these authors, higher emissions in low-covered soils are due to grass degradation that can stimulate or restrict N losses. For example, low plant cover can reduce N sinks for deposited excreta and therefore increase N vulnerability and loss through microbial soil and leaching processes (Chirinda et al., 2019). However, low plant cover may also be associated with fewer exuded plant roots that decrease microbial activity and N₂O emissions (Henry et al., 2008).

Improved soil cover and pasture management also contributes to maintain or increase soil organic matter (SOM) (Aryal et al., 2018), which plays a critical role in determining the N₂O emission response to urea deposition (Clough et al., 2020). Increasing SOM increases cation exchange capacity, reducing the soil solution NH₄⁺ concentration that in turn, reduces soil solution NH₃ and associated inhibition of NO₃⁻ oxidation, thus reducing urea-derived N₂O emissions (Breuillin-Sessoms et al., 2017). Soil buffer capacity increases with increasing soil organic matter alleviating increases in soil pH following urea deposition and associated solubilization of organic matter, and the formation of dissolved organic C and N (Breuillin-Sessoms et al., 2017).

On the other hand, excessive grazing without time for grass recovery increases the risk of soil compaction, an indicator of grass degradation. Compaction reduces soil porosity and pore continuity, decreases aeration, restricts plant growth, and increases soil N₂O emissions in urine patches (van Groenigen et al., 2005). In addition, soil acidification, which could also be an indicator of pasture degradation, has been shown to increase N₂O emissions as acidic conditions generally reduce plant growth and inhibit the activity of the enzyme N₂O reductase, which is responsible for transforming N₂O into dinitrogen (N₂) (Robinson et al., 2014).

### Implementation of Silvopastoral Systems

The incorporation of shrubs and trees in pastures in the called silvopastoral systems (SPS) can also contribute to reduce emissions by improving soil cover and health and by increasing the quality of the diet (Chará et al. 2019). These systems have demonstrated effects on the physical, chemical and microbiological properties of the soil both by the provision of shade and higher amount of heterogeneous biomass that is deposited on the soil in the form of leaves, branches, fruits, and exudates and by improving the root microbiome allowing the modification of microorganism populations in the soil that can regulate nitrification and oxidation processes (Vallejo et al., 2010, 2012). Silvopastoral systems can also contribute to increase SOM (Aryal et al., 2018) which contributes to reduce GHG flows as mentioned previously. Cubillos et al. (2016) found in a study including SPS of different ages, that these arrangements have significantly lower potential for ammonia nitrification compared to monoculture systems (between 15 and 20%), similar to those observed in forest patches, which is why N₂O flows are expected to be reduced under these systems. According to these authors, in SPSs ammonia oxidizing bacteria and archaea limit the rate in nitrification and the resulting N₂O emissions could be reduced.

On the other hand, such systems can modify emissions in feces by the presence of dung beetles that limit the interactions of manure with mineral soil, restricting substrates for nitrification and denitrification processes (Slade et al., 2016). Studies in different regions of Colombia have shown that SPSs have greater abundance, diversity, and activity of beetles than treeless pastures (Giraldo et al., 2011; Montoya-Molina et al., 2016). According to Slade et al. (2016) the presence of beetles in livestock systems reduces N₂O emissions by 14.7% and CH₄ emissions by 17%.

Another effect of these systems is the increased N partitioning into dung relative to urine as this has shown to reduce N₂O emissions from pastures, since the emission factor for dung is lower than that for urine (Luo et al. 2018). Feeding
animals condensed tannin (CT)-rich diets can also increase N partitioning into dung (Carulla et al., 2005). The inclusion of CT either as a dietary supplement or in forages fed to ruminants reduced urinary-N excretion, increased the amount of N excreted in the dung and improved N retention in the animal (Missetbrook et al., 2005). Since N₂O emissions are traditionally higher in urine, this may be a mitigation pathway. Silvopastoral systems use forage species such as *Leucaena leucocephala*, *Tithonia diversifolia*, and *Giriridia sepium*, which contain significant amounts of tannins in their leaves and stems (Barahona et al., 2006; Rivera et al., 2021). Rivera et al. (2018) compared an iSPS and a traditional system and found reductions in emission factors (N₂O) of 23% and 10 times less for feces and urine, respectively.

Finally, plant morphological factors that can affect soil N cycling and N₂O emissions include the effect of the root system on a plant’s ability to access water and nutrients, and plant canopy-effects on the dispersion of urine voided by grazing animals (de Klein et al., 2019). When roots are present, root morphology can affect soil structure and hydrology, both of which influence conditions that govern the reduction of N₂O to N₂ and diffusion of gases to the soil surface (Chapuis-Lardy et al., 2007). Root morphology can also affect plant N uptake and thus availability for soil nitrification and denitrification processes (Abalos et al., 2014). These authors found that combining two grasses, *L. perenne* and *Poa trivialis*, which is a high fertility responsive grass like *L. perenne*, produced the greatest amount of biomass and the lowest N₂O emissions. They suggested this was due to the complementarity of the root foraging strategies of these two species, where *P. trivialis* may access N hotspots not previously emptied by *L. perenne*. This combination, together with its very high total root biomass, is thought to increase mineral N uptake, thereby lowering soil nitrate content and subsequent N₂O emissions (Abalos et al., 2014).

Plant species diversity and interactions can also influence N uptake and N₂O emissions (Niklaus et al. 2016). Increasing plant species richness from 1 to 16 grassland species has been found to reduce N₂O emissions in the absence of N fertilizer (Niklaus et al., 2016), due to more efficient soil inorganic N uptake. However, this pattern was not observed when the diversity included a large proportion of legumes (de Klein et al., 2019).

Although pasture species with increasing root mass or rooting depth have greater ability to take up N, the winter-activity of pasture species such as *Lolium multiflorum* Lam.—i.e., the ability of roots to take up N under cooler conditions—appeared to be more important than specific root architecture (e.g., deep roots) for reducing N leaching losses (Woods et al., 2016). Woods et al. (2016) found that winter-active Italian ryegrass had the greatest N uptake and lowest N leaching, whereas the opposite was found for the tap-rooted *L. perenne*. The high N leaching losses from *M. sativa* in this study were attributed to poor winter herbage growth and the limited depth of the lysimeters (0.7 m) used for this deep rooting species.

**Legumes as an Alternative to N Fertilizer**

Biological nitrogen fixation (BNF) in association with forage legumes provides an alternative N source for grazing systems (Li et al., 2013). In tropical and subtropical conditions, legumes such as *Desmodium ovalifolium*, *Leucaena leucocephala*, *Centrosema pubescens*, *Stylosanthes guianensis*, *Cajanus cajan*, among others, can fix between 120 and 150 kg N/ha/yr. For SPS with *Leucaena leucocephala* planted in rows in Australia, nitrogen fixation rates range between 36 and 61.9 kg/ha/yr (Radrizzani et al., 2011; Conrad et al., 2018) while for an iSPS with high density of *L. leucocephala* in Mexico ranged between 77.1 and 80 kg N/ha over a period of 100 days (Sarabia-Salgado et al., 2020). This fixed N becomes available slowly over time to the grass in pastures after its release into soil via exudates from living legume roots, by mineralization of senesced legume tissues and in excreta after consumption by grazing animals (Ledgard et al., 2009), generating lower losses of N that can be converted into N₂O (Schmeer et al., 2014). Rising costs of fertilizer N and environmental regulations governing stocking densities and fertilizer N use on farms is increasing interest in the use of legumes in pastures. A review by Andrews et al. (2007) concluded that herbage and milk production from legumes-based pastures (perennial ryegrass with 20% white clover (*Trifolium pratense*) in herbage DM on an annual basis) are likely to be similar to that from a perennial ryegrass pasture receiving annual input of 200 kg/ha of N fertilizer and around 70% of that obtained with perennial ryegrass receiving an annual input of 350–400 kg/ha of fertilizer.

The N₂O emissions induced by the growth of legume crops/forages may be estimated solely as a function of the above-ground and below-ground N inputs from crop residues (Li et al., 2013). Accordingly, N₂O emissions from legume-based grasslands are much lower than fertilized grasslands. For example, Ruzjerez et al. (1994) reported up to five-fold more N₂O emission from heavily N fertilized grasslands than from their legume-based counterparts in New Zealand. A data synthesis indicates that the average soil N₂O emissions from field-grown legumes, N fertilized grass pastures and crops, and unfertilized soils are 1.29, 3.22, and 1.20 kg N/ha/yr, respectively (Li et al., 2013).

**Use of Forage Species With Potential for Biological Inhibition of Nitrification**

Within soil GHG emission mitigation strategies, especially for N₂O flows, the use of nitrification inhibition grass species (BNI) is an option to reduce gas production (Byrnes et al., 2017; Teutschlerova et al., 2019). According to Byrnes et al. (2017) and Beeckman et al. (2018) the use of nitrification inhibitors, whether synthetic or plant-based (biological nitrification inhibitors -BNI) reduce soil nitrification rates and therefore NO₃⁻ leachate and N₂O emissions. In addition, this reduction in the nitrification rate contributes to increase the efficiency in the use of N in the system (Yang et al., 2016).

Species such as *Brachiaria humidicola* (Byrnes et al., 2017), *Sorghum bicolor*, *Oryza sativa* and *Triticum aestivum* have demonstrated their ability to decrease N₂O flows (Subbarao et al., 2009; Zhu et al., 2012). Byrnes et al. (2017) reported that in urine patches with *B. humidicola* cv. Tully, N₂O emissions were 60% lower than in *B. hybrid* cv. Mulato (32 vs. 80 mg N₂O-N m⁻², respectively). These authors also found that the
high content of NO\textsubscript{3}\textsuperscript{-} had a positive relationship \((p < 0.05)\) with the number of copies of the \(amoA\) gene of ammonia oxidizing archaea and bacteria at sites with higher emissions. Studies regulating soil N dynamics associated with plant and microorganism interaction such as BNI and, more recently, inhibition of biological denitrification (BDI), have increased, as they represent an ecological, sustainable, and cost-effective strategy compared to the use of synthetic inhibitors (Subbarao et al., 2017).

On the other hand, root exudates can also affect the availability of soil mineral N, as C in these exudates may temporarily increase microbial immobilization of N (Fisk et al., 2015). Carbon from root exudates may thus reduce \(N_{2}O\) emissions derived from urine deposition due to immobilization of urine-N. Indeed, a number of studies have demonstrated that excess N in urine patches does not immediately become available for nitrification and denitrification (Bole et al., 2004). Instead, it can be immobilized into organic matter or fixed on clay particles.

### Application of Biochar and Liming Material

Biochar is a carbonaceous material produced during the thermal decomposition of different materials (wood, plant litter, crop residues, animal manure or waste products) under low-oxygen conditions (Cai et al., 2017). Biochar may inhibit nitrification by compounds such as \(\alpha\) -pinene and ethylene and constrain denitrification through physical and biochemical regulations, including improved soil aeration, increased sorption of substrates for denitrification, and increased soil pH to facilitate complete denitrification to \(N_{2}\) (Cayuela et al., 2014). A meta-analysis reports a 54% reduction of \(N\)\textsubscript{2}O emissions (with a confidence interval from \(-60\) to \(-48\%\)) following biochar addition to agricultural soils (Cayuela et al., 2014).

Biochar can also decrease CH\textsubscript{4} emission or increase CH\textsubscript{4} oxidation via increasing soil aeration and reducing soil bulk density, but some compounds contained in biochar may also inhibit the activity of methanotrophs and increase CH\textsubscript{4} emission (van Zwieten et al., 2010).

Since increased pH can enhance the activity of \(N\)\textsubscript{2}O reductase, lime application should be able to reduce \(N\)\textsubscript{2}O emissions. Liming has also been shown to enhance nitrification (Khan et al., 2011). Therefore, the effect of liming on \(N\)\textsubscript{2}O emission depends on its net effect on \(N\)\textsubscript{2}O reductase and nitrification. Generally, liming can increase \(N\)\textsubscript{2} emission and reduce the ratio of \(N\)\textsubscript{2}O to \(N\)\textsubscript{2} for emission from urine patches, but there are contradictory results about its effect on \(N\)\textsubscript{2}O emission (McMillan et al., 2016). In addition, owing to the decreased Al\textsuperscript{3+} toxicity as pH increases, liming has shown to increase soil CH\textsubscript{4} oxidation and reduce CH\textsubscript{4} emission; this effect might be more pronounced in acid soils (Kunhikrishnan et al., 2016). However, it should be noted that even though liming has merit for lowering \(N\)\textsubscript{2}O emissions, the increased nitrification after lime application may also potentially enhance a risk of \(N\)\textsubscript{2}O production from denitrification when the soils become anaerobic, due to the possible accumulation of the resultant NO\textsubscript{3}\textsuperscript{-} from nitrification (Barton et al., 2013).

Given the many uncertainties of the effect of liming on \(N\)\textsubscript{2}O emission, caution should be exercised in using liming as an option for mitigating \(N\)\textsubscript{2}O from excreta patches, and the effect of liming on the emission of CO\textsubscript{2}, CH\textsubscript{4}, and NH\textsubscript{3} from excreta patches should also be considered (Cai et al., 2017).

### Application of Nitrification and Urease Inhibitors

Since \(N\)\textsubscript{2}O emission from excreta patches mainly results from nitrification and denitrification, thus any inhibitor that can suppress these two processes could be used to mitigate \(N\)\textsubscript{2}O emissions from excreta patches Cai et al. (2017). Nitrification inhibitors (NIs) such as dicyandiamide (DCD), 3, 4-dimethylpyrazole phosphate (DMPP), nitrapyrin, and pyrazole derivatives (PD), can inhibit the conversion of \(NH\textsubscript{4}\textsuperscript{+} to \(N\)\textsubscript{2}O, decrease NO\textsubscript{3}\textsuperscript{-} production and the subsequent denitrification (Barneze et al., 2015). Additionally, Urease inhibitors (UIs) can slow down the conversion of urea to \(NH\textsubscript{4}\textsuperscript{+} and \(NH\textsubscript{3}\) and decrease the availability of \(NH\textsubscript{4}\textsuperscript{+} for nitrification (Cai et al., 2017). The efficacy of NIs in reducing \(N\)\textsubscript{2}O emission varies widely depending on the application rate, timing and method (Cai et al., 2017). Application of DCD significantly decreases \(N\)\textsubscript{2}O emissions from cattle urine (by 4.24 ± 1.10 kg N/ha) and cattle feces (by 0.66 ± 0.61 kg N/ha) patches (Cai et al., 2017). However, NIs may decrease CH\textsubscript{4} oxidation (or increase CH\textsubscript{4} emission) by the inhibitory effect of accumulating \(NH\textsubscript{4}\textsuperscript{+} and possibly directly affect methane monooxygenase (MMO), presumably due to the close structural relationship between ammonia monooxygenase (AMO) and MMO activities (Le Mer and Roger, 2001; Hatch et al., 2005). Therefore, the effect of NIs on CH\textsubscript{4} emission from both urine and feces patches needs to be further studied under a wide range of conditions, since uncertainties about the differential effect of urine and feces deposition on CH\textsubscript{4} emission are large (Cai et al., 2017).

Although this mitigation pathway is effective, its applicability is difficult under grazing conditions, since these are systems where the excretion of urine and feces is dispersed; also, the costs could be high and may not outweigh the benefits.

According to Adhikari et al. (2021) in intensively managed dairy pastures, urine deposited by cattle during grazing covers relatively small proportion of the total grazed area. Maire et al. (2018) reported that urine patches represent only about 6–12% of the grazed area in a single dairy cow grazing event. Therefore, there is a large potential for limiting the risk of NIs entry into these systems if these compounds can be targeted directly to urine patches, avoiding the need for their application across often large areas of pasture unaffected by urine deposition during grazing. Most \(N\)\textsubscript{2}O emissions from a urine patch occur within the first few days to weeks of its deposition, depending on soil and climatic conditions (Selbie et al., 2015). Technologies are therefore needed that can identify and treat urine patches shortly after deposition to inform optimum mitigation options for emission reductions (Marsden et al., 2017). Approaches to identifying urine patches include visually monitoring cows in the field, automated monitoring using electromagnetic induction, electrical conductivity measurements and optical sensing (Misselbrook et al., 2016), ground-based sensing and airborne technologies, such as remotely piloted aerial systems (RPAS), LiDAR and satellites using hyperspectral and...
near infra-red imaging, and temperature sensors (Dennis et al., 2013). The application rates for DCD, DMPP and nitrapyrin in previous field experiments involving urine ranged from 5 to 80, 1 to 5, and 1 to 10 kg/ha, respectively (Adhikari et al., 2021).

MANAGEMENT AND STORAGE OF MANURE AS A MEANS OF DECREASING CH₄ AND N₂O EMISSIONS

According to IPCC (Intergovernmental Panel on Climate Change). (2019), in bovine systems in tropical and subtropical areas only 4 to 15% of manure receives some type of management. Considering the manure handling chain, mitigation options involve adequate handling, storage, and application, especially in those systems where animals are kept in confinement permanently or during part of the day or the year. According to Montes et al. (2013) there are many options for mitigating emissions of N₂O and CH₄ during storage. For CH₄ mitigation, during solid manure storage, composting can be an efficient mitigation option, if it is properly managed. Samer (2015) found that adding straw to solid manure reduces CH₄ emissions, but under certain conditions, it could increase N₂O emissions.

In general, the most effective methods are anaerobic digestion and composting, which in turn have the advantage of generating products that replace fossil fuels and chemical fertilizers. Generally speaking, to mitigate N₂O emissions from manure deposits, the following are the best methods: (i) maintain anaerobic deposits (e.g., compact and covered); (ii) adopt a liquid manure system compared to a deep-bed system (although it has the drawback of increased water use); and (iii) add straw to immobilize ammonium. On the other hand, to mitigate CH₄ emissions, the following are the best methods: (i) anaerobic digestion (Chadwick et al., 2011); (ii) water removal in manure (opposite to N₂O mitigation); (iii) minimize the volume of liquid manure stored during the summer months; (iv) cooling; and (v) aeration of solid manure and composting heaps. According to these recommendations, apart from anaerobic digestion, there are no options to tackle both gases simultaneously, but there are some general strategies to reduce GHG emissions by manure management. Although some of the strategies proposed are efficient in reducing emissions, their actual mitigation potential must be evaluated against possible tradeoffs as they may rely on high use of electricity, fossil fuels, labor or water, or reduce productivity. For example, cooling, constant washing of excreta or the use of specialized structures, could increase costs or increase emissions of other gases such as CO₂, or generate other negative environmental impacts such as eutrophication and acidification.

FINAL REMARKS

Integrated production systems such as silvopastoral systems are strategic to reduce emissions of both CH₄ and N₂O through the reduction in the use of external inputs (i.e., fertilizer and feed supplements), soil protection and improvement of its structure and aeration, and efficient use of nutrients in the production process. An efficient production system that provides nutrients to animals according to their requirements not only contributes to reducing emissions but also allows for more efficient production. Although the reduction of emissions for integrated systems can be as high as 50%, the uptake of these alternatives is still very low and many research gaps remain to make these reductions more generalized.

With regard to mitigation practices, it is important to note that these may result in an “emission exchange” or increase in the flows of some GHGs. Therefore, due to numerous interactions, mitigation practices should not be evaluated in isolation but as a component of the bovine production system (Montes et al., 2013). Optimizing the animal diet to improve the efficiency of N use, balance N input with production level and maintaining fiber digestibility while reducing enteric fermentation of CH₄, are important steps to reduce N₂O and CH₄ emissions from manure. In addition, the use of BNI fodder, as well as providing adequate soil cover offered in well-managed systems, are strategies alternatives to reducing N₂O emissions (Figure 3).

Finally, it is important to mention that some strategies, if applied separately, have different limitations and drawbacks as mentioned in Table 1. For this reason, it is important to advance in studies focused on evaluating the economic impact of mitigation strategies, and to determine their impact on animal and food production. It is also important to work on the estimation of CH₄ emission factors and to evaluate mitigation strategies for this gas that has received less attention compared to N₂O.

AUTHOR CONTRIBUTIONS

JR and JC were involved in the conceptual development of the article. JR led the general construction of the texts and the literature review. JC contributed to manuscript revision and proposed some mitigation strategies. All authors contributed to the article and approved the submitted version.

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