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Chapter 12

Greenhouse Gas Emissions from Housing and Manure Management Systems at Confined Livestock Operations

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1. Introduction

As the name implies, the gases that assist in capturing heat in the atmosphere are termed as greenhouse gases (GHGs). The continuously rising concentrations of these gases are believed to work against nature’s natural process, trapping more heat than what is needed leading to an increase of earth’s climate temperature. Livestock production operations contribute both directly and indirectly to climate change through the emissions of greenhouse gases such as carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O). Generally, swine and ruminant livestock operations, especially dairy cows and beef cattle, contribute to the production of GHGs mainly CH₄, N₂O, and CO₂ in the environment. The CH₄, CO₂, and N₂O are considered as direct greenhouse gases. The indirect GHGs include carbon monoxide (CO), oxides of nitrogen (NOx), and non-methane volatile organic compound (NMVOCs). Characterization and quantification of N₂O and CH₄ emitted from livestock operations are important because these gases are believed to play a major role in the increase of Earth’s temperature. During the last two hundred and fifty years, anthropogenic activities, including demanding agricultural production, have increased the global atmospheric concentration of GHG, namely CO₂, CH₄, and N₂O by 36, 148, and 18%, respectively [1]. Total greenhouse gas (GHG) emissions in the US increased by 14.7% from 1990 to 2006. All agricultural sources combined were estimated to have generated 454 Tg (10¹²g) of CO₂ equivalents in the U.S. during 2006 [2]. The CH₄ emissions from enteric fermentation and manure management represent about 25 and 8% of the total CH₄ emissions from anthropogenic activities. The US Environmental Protection Agency (USEPA), Inventory of U.S. Greenhouse Gas Emissions, and Sinks identified manure management as generating 24 and 5% of CH₄ and N₂O emissions, respectively, from agricultural sources [2-3]. The USEPA has begun to consider regulating GHGs emitted by
the stationary sources, including manure management from CLOs. Thus, it is essential to obtain accurate estimates of GHG emissions from various ground level area sources (barns/housings, lagoons, pens, settling basins, silage piles, pasturelands, etc.) within CLOs to improve emissions inventories and to devise source-specific abatement strategies. In this chapter, GHG emission sources, emissions process, measurement methods and gas sampling protocol, and migration strategies including air scrubbing technology, biofilters, and best manure management practices in the context of livestock waste management were reviewed and discussed.

### 1.1. Sources of GHG in CLOs

Main sources of pollutant gases are broadly classified as natural (geogenic and biogenic) and anthropogenic. The anthropogenic sources again can be divided into mobile (vehicle, ships, trains, etc.) and stationary (power plants, chemical industries, refineries, intensive land uses, confined animal operations, etc.). Biogenic sources of GHGs, such as those contained in grass, hay, silage, and grains are a major part of bovine diets and are emitted from these biogenic sources during fermentation of starches, lipids, and proteins in the digestive system of cattle (enteric fermentation) and later in the feces and urine. Tables 1 and 2 describe the salient features of the characteristics of manure voided by the animal at CLOs. Ruminant livestock is the principal source of enteric methane emissions to the atmosphere, while manure management such as manure storage and treatment are the most important sources of CH₄ and N₂O emissions [4]. Globally, CH₄ is contributing 22%, and N₂O is contributing 6% of the total GHG. Enteric CH₄ is produced as a waste product of this fermentation process in the rumen. Figure 1 describes the number of factors affecting CH₄ production from rumen.

GHG emissions from livestock vary by animal type and growth stage due to different diets, feed conversion mechanisms, and the manure management [5]. Methane is produced by the microbes in the stomach of ruminants due to enteric fermentation, from freshly deposited manure due to bacterial degradation of organic matter, and from storage lagoons and settling basins due to anaerobic degradation of volatile solids by bacteria. Methane, with a global warming potential (GWP) of 21, can affect climate directly through its interaction with long-wave infrared energy and indirectly through atmospheric oxidation reactions. Methane is second in rank to CO₂ in importance and contributes around 18% of the overall greenhouse effect [6]. Table 3 describes the salient features of the three major GHGs. In addition to the anaerobic degradation of the organic materials, CO₂ is released from the use of fertilizers in crop/pasture production, fossil fuel used to run farm machinery (tractors, loaders, and irrigation pumps) and feed processing operations, the loss of tree for crop production on land adjacent to CLOs, and carbon loss from the soil for feed production.

Nitrous oxide is a GHG that contributes to stratospheric ozone depletion and is 310 times more potent as a GHG than CO₂. Nitrous oxide emissions are associated with manure management and the application and deposition of manure in crop/pasture land. Indirect N₂O emissions from livestock production include emissions from fertilizer use for feed
production, emissions from leguminous feed-crops, and emissions from aquatic sources following fertilizer application. Nitrous oxide is produced in soils through microbial processes of nitrification and denitrification and is released from manure and urine excreta, fertilizer and manure slurry applied for feed-crop production, dry manure piles and aerobic and anaerobic degradation of livestock manure/wastewater in lagoons. The amount of these gaseous emissions from livestock vary by animal type and growth stage due to different diets, daily feed intake, and quality of diet feed conversion mechanism, while GHG emissions from storage and treatment of manure depend on the type of storage, duration of storage, ambient temperature, and manure management practices.

**Figure 1.** Factors affecting methane production [7].
| Animal Type      | Average weight (pound) | Days on feed | Total solids (TS) | Volatile solids (VS) | N   | P   | P₂O₅ | K   | K₂O | Manure | Moisture |
|------------------|------------------------|--------------|-------------------|---------------------|-----|-----|------|-----|-----|--------|----------|
| Cattle           |                        |              |                   |                     |     |     |      |     |     |        |          |
| Cows/Heifers **  | 1000                   | 365          | 9.5               | 8.1                 | 0.36| 0.048| 0.11 | 0.23| 0.28| 82     | 88       |
| Finishing        | 1200                   | 153          | 5.1               | 4.2                 | 0.36| 0.048| 0.11 | 0.248| 0.30| 64     | 92       |
| Bulls **         | 1100                   | 365          | 6.2               | 5.7                 | 0.54| 0.092| 0.21 | 0.267| 0.32| 80     | 92       |
| Calves**         | 450                    | 210          | 3.4               | 2.9                 | 0.14| 0.044| 0.10 | 0.092| 0.11| 26     | 92       |
| Dairy-Milk cows¹ | 1300                   | 365          | 18                | 15.3                | 0.92| 0.16 | 0.36 | 0.44 | 0.53| 141    | 87       |
| Swine            |                        |              |                   |                     |     |     |      |     |     |        |          |
| Nursery          | 27.5                   | 36           | 0.3               | 0.2                 | 0.025| 0.0042| 0.01 | 0.01 | 0.01| 2.4    | 90       |
| Finishing        | 154                    | 120          | 1.0               | 0.8                 | 0.083| 0.0142| 0.03 | 0.037| 0.04| 10     | 90       |
| Gestating        | 440                    | 365          | 1.1               | 1.0                 | 0.071| 0.02 | 0.05 | 0.048| 0.06| 11     | 90       |
| Lactating        | 423                    | 365          | 2.5               | 2.3                 | 0.19 | 0.055| 0.13 | 0.12 | 0.14| 25     | 90       |
| Sheep**          | 100                    | 365          | 1.1               | 0.9                 | 0.04 | 0.009| 0.02 | 0.03 | 0.04| 4      | 75       |
| Poultry          |                        |              |                   |                     |     |     |      |     |     |        |          |
| Layers           | 3                      | 365          | 0.05              | 0.04                | 0.0035| 0.0011| 0.003 | 0.0013| 0.002| 0.19   | 75       |
| Broilers         | 2.8                    | 48           | 0.06              | 0.04                | 0.0025| 0.00073| 0.002 | 0.0014| 0.002| 0.23   | 74       |
| Turkeys²         | 25                     | 140          | 0.12              | 0.1                 | 0.0072| 0.00212| 0.005 | 0.0033| 0.004| 0.47   | 74       |
| Litter³          | 2                      | 1.6          | 0.089             | 0.038               | 0.086| 0.049| 0.059| 2.5  | 21   |        |          |
| Horse⁴          | 1100                   | 365          | 8.5               | 6.7                 | 0.27 | 0.05 | 0.12 | 0.14 | 0.16| 56.5   | 85       |

*ASAE standard D384.2 2005. Manure Production and Characteristics. ASABE, St. Joseph, MI 49085-9659
**Manure Characteristics, 2000. Mid West Plan Service, Ames, IA 50011-3080. MWPS-18, Section I.
¹Manure data on TS,N,P,K and manure provided by Dr. Tamilee Nenich, TCE Dairy Specialist. Volatile solids (VS) estimated to be 85% of TS.
²% wb = percent wt basis.
³Days on feed data from “economic Impact of the Texas Poultry Industry,” 2004, TCE publication, L-5214. Average weight, TS,VS,N,P,K and total manure averaged from data for female and male turkeys.
⁴Poultry Waste Management Handbook,” 1999. Natural Resource, Agriculture, and Engineering Service. Ithaca, NY 14853-5701. NARES-132. Pounds of whole poultry litter (as removed from production houses) per broiler sold. N,P and K values in pounds per 2.5 pounds of litter.
⁵Average weight, TS,VS,N,P,K and total manure averaged from data for sedentary and intense exercise horses.

Table 1. Animal manure production and characteristics [8].*
| Animal type | Year* | Animal numbers** (thousands) | TS | VS | N | P | P:O₃ | K | K:O | Total energy*** BTU x 10¹² Tera BTUs |
|-------------|-------|-------------------------------|----|----|---|---|-------|---|-----|-------------------------|
| Cattle      |       |                               |    |    |   |   |       |   |     | Thousands of tons per year on an as excreted" basis |
| Cows/Heifers| 2006  | 5780                          | 86498 | 10021 | 8544 | 380 | 51 | 116 | 243 | 291 | 145 |
| Finishing   |       | 5520                          | 27026 | 2149 | 1764 | 152 | 20 | 46 | 105 | 126 | 30 |
| Bulls       | 2006  | 370                           | 5402  | 419 | 385 | 36 | 6 | 14 | 18 | 22 | 6.43 |
| Calves      | 2006  | 2430                          | 6634  | 868 | 740 | 36 | 11 | 26 | 23 | 28 | 12.6 |
| Milk Cows   | 2006  | 334                           | 8581  | 1096 | 936 | 56 | 9.7 | 22.3 | 32 | 32 | 15.8 |
| Swine       |       |                               |      |    |    |   |   |     |    |     |                         |
| Nursery     | 2006  | 270                           | 12    | 1.36 | 1.16 | 0.12 | 0.02 | 0.05 | 0.05 | 0.06 | 0.02 |
| Finishing   | 2006  | 565                           | 339   | 34 | 28 | 2.81 | 0.48 | 1.10 | 1.24 | 1.49 | 0.48 |
| Other¹      | 2006  | 95                            | 312   | 31 | 29 | 2.26 | 0.65 | 1.49 | 1.46 | 1.75 | 0.48 |
| Sleep & Goats² | 2006 | 2140                          | 1759  | 484 | 400 | 18 | 4 | 9 | 13 | 16 | 6.8 |
| Poultry     |       |                               |      |    |    |   |   |     |    |     |                         |
| Layers³     | 2005  | 18688                         | 648   | 167 | 123 | 12 | 4 | 9 | 4 | 5 | 2.1 |
| Broilers    | 2005  | 627900                        | 3451  | 874 | 659 | 38 | 11 | 25 | 21 | 26 | 11.2 |
| Turkeys⁴    | 2004  | 14100                         | 468   | 120 | 96 | 7 | 2 | 5 | 3 | 4 | 1.63 |
| Litter      |       | 785                           | 622   | 494 | 28 | 12 | 27 | 15.4 | 18 | 8.4 |
| Horses⁵     | 1998  | 1067                          | 11000 | 1655 | 1304 | 53 | 10 | 23 | 26 | 32 | 22.2 |

*Year of estimated total population or production data from National Agricultural Statistical Services
** Animals finished or on feed per year.
*** Dry and ash free basis
¹Includes all hogs other than nursery and grow-finish. Estimates based on average nutrient data from gestating lactating sows in Table I.
²Includes sheep and goats. Manure and nutrient totals calculated using sheep data only.
³Include hens and pullets of egg-laying age.
⁴Animal numbers for turkey estimated from difference between total turkey and broiler population in Texas (615.6 million from TCE publication L-5214) and National Agricultural Statistical Service estimated number of broilers (601.5 million in 2004).
⁵Animal numbers for horses adopted from Texas Horse Industry Report, 1998, and from the Texas Horse Industry Quality Audit initiative, TCE, January 1998.

**Table 2.** Animal manure production and characteristics [8].

| GHG   | MW (g mol⁻¹) | Typical ambient concentration (ppm) | Life time (Yr) | Radiative efficiency (W m⁻² ppb⁻¹) | Global Warming Potential |
|-------|--------------|------------------------------------|----------------|-----------------------------------|--------------------------|
| CO₂   | 44.01        | 380                                | Up to 100      | 1.4×10⁻⁵                          | 1                        |
| CH₄   | 16.04        | 1.7                                | 12             | 3.7×10⁻⁴                          | 21                       |
| N₂O   | 44.01        | 320                                | 114            | 3.03×10⁻³                         | 310                      |

**Table 3.** Global warming potential of the GHGs [1]
1.2. Greenhouse gas inventory

The livestock industry is a significant contributor to the economy of any country. More than one billion ton of manure is produced annually by livestock in the United States. Animal manure is a valuable source of nutrients and renewable energy. However, most of the manure is collected in storage/treatment structures or left to decompose in the open, which poses a significant environmental hazard. Tables 4a, 4b, and 4c summarized the studies on GHG emission rates (ERs) from dairy, feed yard and swine operations. Based on a literature review using limited data for free-stall and naturally ventilated dairy operations, emissions of CO₂ from the dairy slurry manure storage facilities averaged 72 kg CO₂ m⁻³ yr⁻¹ (data ranged from 8.6 to 117 kg CO₂ m⁻³ yr⁻¹) [9-11]. Emissions of CO₂ from dairy housing averaged 1,989 kg CO₂ hd⁻¹ yr⁻¹ (data ranged from 1,697 to 2,281 kg CO₂ hd⁻¹ yr⁻¹, where hd⁻¹ is per head) [11]. Kinsman et al. [12] reported that the mean daily CH₄ emission per dairy cow (602 kg mean bodyweight) in a tie-stall barn ranged from 373 to 617 g CH₄ AU⁻¹d⁻¹ (436 to 721L), while the mean daily CO₂ emission per cow ranged from 11,900 to 17,500 g CO₂ AU⁻¹d⁻¹ (5,032 to 7,427 L). In a study by Amon et al. [13], CH₄ and N₂O emissions per livestock unit (LU=600 kg of body weight) or animal unit (AU=500 kg of body weight) from tie stalls for dairy cows were measured several times in the course of a year. Average emissions were 619.2 mg N₂O LU⁻¹d⁻¹ (516 mg N₂O AU⁻¹d⁻¹), and 194.4 g CH₄ LU⁻¹d⁻¹ (162 g CH₄ AU⁻¹d⁻¹). Emissions of CH₄ and N₂O from animal housing averaged 54 (1.0-100) kg CH₄ hd⁻¹ yr⁻¹ and 0.3 (0.0-0.6) kg N₂O hd⁻¹ yr⁻¹, respectively [11,13]. Ngwabie et al. [14] reported CH₄ emissions ranging from 25 to 312 g hd⁻¹ d⁻¹ (9 to 114 kg hd⁻¹ yr⁻¹) in a naturally ventilated dairy barn. Methane emission estimates for dairy cows have been reported to range from 230 g/cow/day [15] to 323 g cow⁻¹d⁻¹ [16]. Fiedler and Muller [17] show that CH₄ emissions from naturally ventilated dairy barns ranged from 672 to 528 g cow⁻¹d⁻¹. A study conducted in California, USA, indicated that CH₄ emissions of 296 and 438 g cow⁻¹d⁻¹ for dry and lactating cows, respectively, were mainly due to enteric fermentation and fresh manure produced negligible amount of CH₄ [18]. Most dairy facilities are naturally ventilated and do not have controlled air exchange. Therefore, in addition to the large variations in methane emission rates causing a range of methane concentrations in dairy facilities, CH₄ concentrations in dairy barns will also vary with geographical locations, weather conditions, and ventilation management practices. Only limited studies have been conducted on indoor air methane concentrations in dairy barns.

Most of the published literature reporting CH₄ emissions from feedlot manure systems used atmospheric dispersion modeling (inverse dispersion, backward Lagrangian stochastic model, IPCC tiers I and II algorithm, and Blaxter and Clapperton algorithm) to estimate emissions from a whole farm [19-21]. Zoe et al. [19] estimated summer CH₄ ER data for two Australian feedyards using an open-path tunable near infrared diode laser coupled with backward Lagrangian stochastic model of atmospheric dispersion. Methane ERs reported were 146 and 166 g hd⁻¹ d⁻¹ for Victoria and Queensland, respectively. Using the same techniques, the average CH₄ emissions were 166 and 214 g CH₄ hd⁻¹ d⁻¹ for feedlots in Queensland and Alberta, respectively [20]. Average daily CH₄ emissions were estimated to be 323 g hd⁻¹ d⁻¹ for a large beef feedlot in western Canada using the inverse dispersion
model [21]. Phetteplace et al. [22] determined GHG emissions from simulated beef and dairy livestock systems in the United States using a computer spreadsheet program. The methane N₂O and CO₂ ERs reported were 1.56, 11.4 and 3411 g hd⁻¹ d⁻¹ from manure management systems of a feedlot. Direct measurements using micrometeorological mass difference technique reported 70 g CH₄ hd⁻¹ d⁻¹ emissions from a confined beef feedyard in Australia where animals were fed a highly digestible high grain diet [23].

Emission rates of CO₂, CH₄ and N₂O from different pig housing systems are presented in Table 4c. CH₄ was observed to be emitted from all swine housing systems showing a large variation because of the different animal types, housing systems, and manure handling methods [24-27]. Methane emissions from fattening pigs range between 0.5 to 135 g pig⁻¹ d⁻¹, whereas emissions of 0.77 and 5.8 g pig⁻¹ d⁻¹ (Table 4c) were reported for sows and weaners, respectively. Similarly, CO₂, CH₄ and N₂O ERs in gestation pigs in North Dakota (USA) ranged from 5,350-15,830, 116-572, and 0.06-7.3 g d⁻¹ pig⁻¹., respectively [28]. Similarly, CO₂ ERs for different growing stages from swine operation ranged from 5,920 to 30,000 g pig⁻¹ d⁻¹. The highest N₂O ER was estimated from swine nursery in China [29]. Animal feces temporarily stored indoors deep pits are the principal source of CH₄ emissions in swine housings. The quantity of CH₄ emitted by the animal itself and the amount emitted from barns of fattening pigs is influenced by the diet and digestibility, daily weight gain of the pigs, and the temperature and type of housing system. Methane emissions are lower in summer when compared with autumn and winter due to higher air exchange rates. Also, the CH₄ generation might be influenced by the availability of oxygen over the emitting surfaces [30]. Significant amount N₂O emits from pig manure handling system is exclusively originated from deep litter and compost systems. The variation in the N₂O emissions mainly depends on the kind of housing system. Fattening pigs raised on partly or fully slatted floor emit very little N₂O while higher emissions reported for fatteners in deep litter and compost systems [30].

| Facility type and ground sources | Animal Number | CH₄ g/LU/d | CO₂ kg/LU/d | N₂O g/LU/d | Technology used | Remarks | References |
|---------------------------------|---------------|------------|-------------|------------|----------------|---------|------------|
| Free-stall dairy: Barn, settling basins, loafing pen, primary and secondary lagoon, walkway, silage pile. | 500 | 181 | 6.6 | 6 | Dynamic flux chamber coupled with GC and chromatograms acquired directly at the field | Five consecutive days in summer and winter; Reported values are annualized | [31] |
| Free-stall dairy: Barn, settling basins, loafing pen, primary and secondary lagoon, open-lot, compost piles. | 3500 | 836 | 5.5 | 3.4 | Dynamic flux chamber coupled with GC and chromatograms acquired directly at the field | Five consecutive days in summer | [32] |
| Open-lot dairy: Open-lot pen. | 700 | 0.20 to | OP-PATH FTIR | One to two days | [33] |
storage lagoon, and composting areas.  

| Facility type and ground sources | Animal Number | CH$_4$ | CO$_2$ | N$_2$O | Technology used | Remarks |
|----------------------------------|---------------|--------|--------|-------|----------------|---------|
| Tied-stall dairy:                | 16,995        | 323    | 495    | 28    | OP-TDLAS/ BLS (Wind-Trax 2.0) | 12 days, High grain diets and |
| Free-stall dairy:                | 10,800        | 490    | 952    | 6     | PAS Multi-gas Monitor (INNOVA 1412) | 2 or 3 days in each month over a year; BW 635 kg |
| Dairy with                       |               |        |        |       | TDLAS (DT: 100 ppb) coupled with Gaussian plume model | 7 conventional farms |
| Runoff pond | 6.2 | | | BW 185-635 kg 280-700 kg |
|---|---|---|---|---|
| **Feedyard:** Queensland Victoria | 13,800 16,500 | 146-166 | OP-TDLAS/ BLS (Wind-Trax 2.0) | High grain diets and animal weight 265-620 kg 280-700 kg [19] |
| **Pen surface** | 42,000 | 1.71 2.05 0.04 1.31 0.035 0.57 0.01 0.57 | Dynamic flux chamber coupled with gas chromatograph and chromatograms acquired directly at the field | Five consecutive days in summer [32] |
| **Run-off pond Composting areas** | | | | |
| **Open Feedlot:** Queensland (Australia) Alberta (Canada) | 13,800 22,500 | 166 214 | OP-TDLAS/ BLS (Wind-Trax 2.0) | High grain diets and animal weight 350-600 265-620 kg [20] |
| **Feedyard and Grazing:** Grazing Feedlot | A group of cattle 230 70 | | Microclimatological mass difference; same group of cattle tested at each the source. | High grain diets Grazed Animal weight 436 ± 21 kg [23] |
| **Simulated beef system** | 100 | 1.56 3.4 11 | Computer spreadsheet Program (Gibbs and Johnson, 1994) | Data collected from nine beef and dairies in US [22] |

| Country | Animal Stage | Housing and or manure Handling type | GHG emission factor | Reference |
|---|---|---|---|---|
| Germany | Fattening | Fully slatted floor Kennel housing | g d\(^{-1}\) AU\(^{-1}\) | N/A 17000 – 23000 11000 – 13000 69-135 18-36 N/A N/A [26] |
| Germany | Fattening | N/A | g d\(^{-1}\) AU\(^{-1}\) | N/A 0.5-1 N/A [5] |
| Holland | Sows Weaner Finisher | N/A N/A N/A | mg h\(^{-1}\) pig\(^{-1}\), mg h\(^{-1}\) pig\(^{-1}\) | N/A 2406 445 N/A N/A 1269 N/A N/A N/A [24] |
| Italy | Fattening | Fully slatted floor Vacuum system | g d\(^{-1}\) AU\(^{-1}\) | N/A 7.9±1.6 6.4±2.0 0.02±0.15 0.05±0.03 [38] |
| Belgium | Weaned Pigs | Straw litters Sawdust litters | g d\(^{-1}\) pig\(^{-1}\), g d\(^{-1}\) pig\(^{-1}\) | 463 481 1.58 0.77 0.35 1.4 [39] |
| Denmark | Finishing | Partly slatted floor | g fattening period\(^{-1}\) | 5540 302 9.1 [40] |
Table 4. a. Summary of GHG emission rates (ERs) estimated from dairy operations at different ground level area sources (GLAS) as reported by previous researchers (LU = live weight, BW = body weight). b. Summary of GHG emission rates (ERs) estimated from feedyard operations at different ground level area sources (GLAS) as reported by previous researchers. c. Summary of GHG ERs of swine operations with different housing and management schemes as reported in the literature (updated after Dong et al. [29]). AU = animal weight = 500 kg live weight.

2. Emission process

2.1. Methane emission process

During anaerobic fermentation, organic wastes are biologically degraded in the absence of oxygen to CH₄, CO₂, N₂, and H₂S. Methanogenic fermentation of organic materials occurs under strictly anaerobic and low redox potential (Eh < -200 mV) conditions where sulphate and nitrate concentrations are low [43]. Methanogens produce methane by breaking down organic matter in the absence of oxygen (anaerobically), releasing CO₂ and CH₄ according to the following equation:
This transformation requires the successive action of four different types of micro-organisms as shown in Figure 2 that degrade complex molecules to simpler compounds [44]:

a. **Hydrolytic microflora:** hydrolysis of longer chain carbohydrates, fats, and proteins are broken into shorter chain molecules. This can be aerobic, facultative, or strictly anaerobic.

b. **Fermentative microflora:** acidogenesis of shorter chain molecules produce carbon dioxide, hydrogen sulfide, alcohol, and more volatile fatty acids. This can be facultative or strictly anaerobic.

c. **Homoacetogenic or syntrophic microflora:** acetogenesis from previous metabolites. Simple molecules created through the first two steps are digested by specific bacteria to produce acetic acids as well as hydrogen and carbon dioxide.

d. **Methanogens:** methanogenesis of simple compounds such as H₂ + CO₂ and acetate. Products developed in stages 1-3 and convert (H₂ + CO₂ and acetate) them into methane, carbon dioxide and trace amount of other gases.

Numerous physical, chemical, and biological factors influence the physiology of methanogenic archaea and characteristics manure (organic waste) and the micro-environment of the anaerobic systems. One of the important obvious factors is temperature. Methane production increases with increasing temperature if other parameters are kept constant. The main factor determining the degree of CH₄ production is the amount of degradable organic matter contained in the effluent and organic animal waste. This fraction is commonly expressed in terms of biochemical or chemical oxygen demand (BOD or COD). The higher the BOD/COD value, the more CH₄ is produced. The potential amount of CH₄ formation from animal feces will depend on the amount of fecal matter excreted, the physical form of the deposit (shape, size), excretal form (solid, slurry, and effluent), climatic and soil conditions, and the length of time these deposits remain intact before being decomposed [43].

Methane production from manure when managed in a controlled setting will depend on the type of waste, temperature, and duration of storage, and the manner in which the waste is handled. On the other hand, emissions during composting of dung depend on aeration rate, water content, thermal insulation, weather conditions, and manure composition. Methane production during composting is related to the lack of oxygen in the decomposing biomass. A study reported that anaerobic digestion of the slurry reduced CH₄ emissions after field application, because the easily degradable organic compounds were already converted to CO₂ and CH₄ during digestion in the biogas plant [45]. The factors affecting CH₄ emission by soils are summarized as follows [43, 44]:

a. Gas diffusion in relation to oxydo-reduction level and CH₄ transfer, in particular the water content, the nature of clays, and the type of vegetation.
b. Microbial activities in general temperature, pH, Eh, substrate availability, physicochemical properties of soils.

c. Methanogenesis and, in particular, the competition with denitrification and sulphate reduction.

d. Methane-mono-oxygenazse activity—concentrations of H₂, CH₄, NH₄⁺, NO₃⁻, Cu.

![Figure 2](image-url). Schematic diagram showing anaerobic fermentation process.

### 2.2. Nitrous oxide emission process

Nitrous oxide (N₂O), nitrogen monoxide (NO), and nitrogen dioxide (NO₂) are the most plentiful nitrogen oxides in the atmosphere and being produced lavishly by biogenic sources such as plants and yeasts. Nitrous oxide is an ozone (O₃) depleting substance which reacts with O₃ in both the troposphere and in the stratosphere and has a long half-life (100 - 150 years). In livestock agriculture, N₂O emissions are associated with manure management and the application and deposition of manure in crop/pasture land. Indirect N₂O emissions from livestock production include emissions from fertilizer use for feed production, leguminous feed crops, and emissions from aquatic sources following fertilizer application. Thus, fertilized soils are important sources of N₂O. Soils contribute about 65% of the total
N\textsubscript{2}O produced by terrestrial ecosystems [46]. Nitrous oxide is formed in soils during the microbiological processes of nitrification and denitrification as shown in equations 2 and 3. Nitrous oxide production by nitrifying bacteria may arise either during NH\textsubscript{4}\textsuperscript{+} oxidation to NO\textsubscript{2}\textsuperscript{–} or during dissimilatory NO\textsubscript{2}\textsuperscript{–} reduction when O\textsubscript{2} supply is limited. During denitrification, N\textsubscript{2}O is an intermediate product in the dissimilatory reduction of NO\textsubscript{3}\textsuperscript{–} and NO\textsubscript{2}\textsuperscript{–} to N\textsubscript{2} under anaerobic conditions and may, therefore, be produced and consumed by denitrifying bacteria in soil [47].

Fertilizer and manure type may affect N\textsubscript{2}O emission in several ways [48] such as: (1) the type of N (NO\textsubscript{3}\textsuperscript{–}, NH\textsubscript{4}\textsuperscript{+}, and organic N) which affects N\textsubscript{2}O production during nitrification and denitrification, (2) the presence of freely available C, which stimulates denitrification activity and O\textsubscript{2} consumption in the soil following its application, and (3) effects on biological, chemical and physical soil processes because of changes in pH and the addition of other compounds (salt, water). The availability of N (NH\textsubscript{4}\textsuperscript{+} and NO\textsubscript{3}\textsuperscript{–}) [43, 49, 50], and the factors that alter the redox potential of the soil, such as changes in soil moisture [51-53], soil texture, and organic C, have major effects on the production of N\textsubscript{2}O in soils. In addition, several soil management practices such as tillage, soil compaction [54-55], irrigation, and drainage affect the production and transport of N\textsubscript{2}O release by influencing the physical condition of the soils such as aeration and soil water content.

\[ \text{NH}_4^+ + \text{O}_2 \rightarrow \text{NH}_2\text{OH} \quad \frac{1}{2} \text{O}_2 \rightarrow [\text{HNO}] \rightarrow \text{NO}_2^\text{–} \rightarrow \text{NO}_3^\text{–} (2) \]

\[ \text{NO}_3^\text{–} \rightarrow \text{N}_2\text{O}^\text{–} \rightarrow \text{N}_2 \quad \text{ATP} (3) \]

3. GHG measurement methods

3.1. Measurements of Greenhouse gas concentrations

Uncertainties in the accurate emission estimate are mainly dependent on the errors associated with sampling protocol and devices and gas analyzers. Thus, the validation of emission inventories using emission measurements is extremely important as well as source-related emission measurements that are feeding emission inventories. To develop GHG mitigating strategies, it is required to quantify GHG emissions from livestock operations under a wide range of productions and management circumstances. Gases at trace levels can be measured using different techniques. GHGs can be measured using infrared spectroscopy (IR), photoacoustic spectroscopy (PAS), gas chromatography (GC),
mass spectroscopy (MS), tunable diode laser absorption spectroscopy (TDLAS) technology, open path Fourier Transform Infrared Radiation (OP-FTIR) technologies, and solid-state electro-chemical technology. Instruments with mass spectrophotometers have very rapid response, can detect many gases at one time, and exhibit linear responses over a wide range of concentrations, while behaving very accurately and with stability. However, mass spectrophotometers, TDLs, and OP-FTIRs are expensive. Solid state electrochemical sensors are relatively inexpensive but they are unstable and require frequent calibration. The shelf lives of those sensors also vary from 12-18 months.

3.2. Flux measurement process at the ground level of the livestock facilities

The GHG emission estimation from different ground level area sources (GLAS) of manure managements in livestock operations such as lagoons (primary and secondary), barns, settling basins, silage piles, loafing pens, feedlots pens, compost windrows, and crop/pasture land is a very complicated process. Generally, two basic processes such as device independent and sampling device are widely used to estimate emission from emitting surfaces. In the device-independent techniques, the emission rate (amount (g or kg) of compound emitted per head per day or year is estimated from the concentrations of the measured emitted material using local micrometrical data, especially wind velocity profile data [56-57]. When using a sampling device, a chamber and winds tunnel is deployed on an emitting surface under some recommended operating conditions. Those devices may be static (sealed or vented) or flushed with zero grade air (containment free) at a known velocity or flow rate (known as dynamic). Generally, the emission rate is estimated as the product of concentration and air flow through the device [57-59]. There is debate about the suitability and accurateness to quantify pollutant emissions at CLOs and other area sources due to the creation of microenvironments in the chamber and the small measurement footprint relative to the size of the source [57-60]. Hudson et al. [61] compared and reported that odor emissions from a wind tunnel rates were 60 to 240 times higher than those in a flux chamber [62]. Parker et al [60,71] also demonstrated that water evaporation, wind speed, and temperature would be useful to standardize and compare emission rates from flux chamber and wind tunnels. They also suggested developing correction factors for each device, which depend on the geometry of the wind tunnel and chamber. Instruments and devices commonly used to measure gas emissions from CLOs were presented in the Table 1.

4. GHG scrubbing technology

In the livestock industries, End-of-Pipe technologies such as biofilters and wet scrubbers are commonly used in process-air applications i.e. potentially harmful particulates matter (PM) and pollutants in exhaust air of the housings/barns are treated. Generally, water with added active chemicals such acids and oxidizing agents (H₂O₂, H₂SO₄, O₃, kMnO₄, HOCl, etc) are tailored with the process to spray into the air stream coming out of the exhaust. This approach for reducing emission is basically a treatment of the exhaust air released from
mechanically ventilated animal housings. The main advantages for this approach are air can be treated without affecting the routine management operations and structural design inside the barn. Broadly, two types of air scrubbers are presently available: acid scrubbers and bio-trickling filters. The main purpose of these scrubbers is ammonia abatement; the scrubber systems are commercially available and considered as off-shelf techniques in such as the Netherlands, Germany and Denmark [64].

4.1. Scrubber descriptions

4.1.1. Spray type wet-scrubber

In an acid scrubber for ammonia, diluted acid mainly sulfuric acid (H₂SO₄) with pH of 2-4 is used to scrub ammonia from air and the ammonium salt is removed from the system with the discharge water. Spray scrubbers consist of empty cylindrical or rectangular chambers in which the gas stream is contacted with liquid droplets generated by spray nozzles. The spray nozzles (hydraulically or air or steam atomized), are used to extend the surface area of the scrubbing liquid and produce target droplets size that facilitates mass transfer of the contaminants gas(es) into liquid. They are mainly used for gas absorption. Particulate matters (PMs) and gaseous pollutants in the air stream are removed by either absorption or chemical reactions with the water solution. PM and pollutions from the scrubber process are removed periodically through the drain. Schematic of a typical spray nozzle scrubber configuration along with system components is shown in the Figure 3.

Figure 3. Schematic diagram spray type wet-scrubber system
4.1.2. Packed bed wet-scrubber

A packed tower air scrubber or bio-trickling filter is a reactor that is filled with an inert or inorganic packing material. The packing material usually has a large porosity, or void volume, and a large specific area [64]. Water with added chemicals is sprayed either continuously or intermittently from the top of the packed bed to keep it wet. The contact between the air and water, facilitates a mass transfer from soluble gases to a liquid phase when exhaust air is introduced wither horizontally (cross-current) or upwards (counter-current). A fraction of the trickling water is continuously recirculated, while another fraction is discharged and replaced by fresh water [65]. Schematics of typical spray nozzle scrubber and packed bed acid scrubbers configuration along with system components is shown in Figure 4.

![Schematic diagrams of packed bed trickling filters: top) counter current packed wet-scrubber, and bottom) cross current (adopted from [64-65).](image)

**Figure 4.** Schematic diagrams of packed bed trickling filters: top) counter current packed wet-scrubber, and bottom) cross current (adopted from [64-65).
4.1.3. Pollutants removal efficiency calculation

Gaseous pollutants removal efficiency is generally used as the criteria for determining the spray type wet scrubber performance can be defined as [66]:

\[ \gamma_{\text{total}} = \frac{c_{\text{in}} - c_{\text{out}}}{c_{\text{in}}} \times 100 \]  

Where:

- \( \gamma_{\text{total}} \) = Pollutant collection efficiency (%)
- \( c_{\text{in}} \) = Airborne Pollutant concentration before the scrubber (ppm)
- \( c_{\text{out}} \) = Airborne Pollutant concentration after the scrubber (ppm)

Similarly, the difference in the weight of the PM filters before and after scrubber sampled during 24 hours and the standardized airflow were used to calculate the average PM \(_{10}\) concentration. The details on the used method for PM\(_{10}\) determination can be found in [67]. For wet-scrubbers, the air flow rate is used to calculate average Empty Bed air Residence Time (EBRT). The air flow rate through the scrubbers can be determined either by measuring fans or by means of a CO\(_2\) balance method [68]. The EBRT can be defined as follows:

\[ EBRT = \frac{\text{Packing volume (m}^3\text{)}}{\text{Air flow rate (m}^3\text{s}^{-1}\text{)}} \times 100 \]  

4.2. Wet scrubber applications in AFOs

The development of wet air-scrubbers for mitigating air emissions from CLOs has started a longtime ago. To begin with, wet scrubbers were employed to reduce odor and particulates being discharged from livestock facilities [69-70]. Later on, scrubbing other airborne contaminants such as NH\(_3\) [66,71,65], H\(_2\)S [72], and pathogens [73] were also investigated to test their scrubbing efficacy. The collection efficiencies reported ranging minus to 100% for odor, 23% to 96% for NH\(_3\), and 36% to 96% for particulate matter. An acid scrubber and a bio-trickling filter (BTF) were developed to reduce ammonia and odor from swine and poultry houses in the Netherland. Melse and Ogink [71] reported an average ammonia removal efficiency of 96% in acid scrubbers (ranging from 40% to 100%). The average efficiency estimated using the air balance method was 71% (±4%). At least 24 measurement days are recommended to keep the relative error below 5% when using the air balance method in determining the NH\(_3\) removal efficiency of an acid packed bed scrubber [65].

Chemical scrubbers and bio-scrubbers have shown tremendous potentials in reducing high particulates and ammonia, however, are not very effective in removing typical odors [74-75]. The major limitations encountered in the development of wet scrubber technology for CLOs are low collection efficiency of the odorous compounds, high pressure drop, and high operating costs. An acid spray wet scrubber has the greatest potential for adaptation to existing swine facility ventilation fans because they do not cause excessive backpressure to the fans and do not significantly reduce building ventilation airflow [66].
Recent literature showed that the majority of scrubbers designed for CLO applications were employed for removing ammonia. In Europe, the most common commercial scrubbers are packed-type, which can be bought off the shelf for application at CLOs [76] and have been proven to effectively remove NH₃ by up to 96%, but their packing material resulted in large pressure drop [71]. Other types of wet scrubbers such as impingement plate, fiber bed [71], and rotating beds [77] have been used for NH₃ collection. These wet scrubbers also resulted high pressure drop and did not work well with the existing ventilation systems of AFOs because axial fans are typically used for movement of large volume of airflow under small differential static pressure conditions. Recently, a new generation multi-pollutants scrubbers have also been developed to address ammonia, odor, and particulates abatement released from livestock operations. This scrubber mainly consist of two or more scrubbing stages (combining the concepts of acid scrubbing, bio-scrubbing, and bio-filtration), each stage aims for the removal of one type of compound [78]. Three multi-stage scrubbers, one double-stage scrubber (acid stage+ bio-filter), one double-stage scrubber (acid stage + bio-scrubber), and one triple-stage scrubber (water stage + acid stage + bio-filter) were evaluated to test their effectiveness in reducing airborne dust, total bacteria, ammonia, and CO₂ emissions from swine houses in Netherlands. Those scrubbers reduced PM₁₀, PM₂.₅, total airborne bacteria, and ammonia emissions from 61 to 93%, 47 to 90%, 46 to 85%, and 70 to 100%, respectively [79]. Concentrations of CO₂ were not affected.

Most scrubbing technologies for CLOs are still in the developmental stages in the US. The major limitations encountered in the development of wet scrubber technology for CLOs are low efficiency, high pressure drop, and high operating costs. The spray-type wet scrubbers usually are generally shown low collection efficiency for NH₃ gas [80,70]. A spray scrubber with water has shown to be a collection efficacy of approximately 20%, although, ammonia is fairly water soluble 20% [81]. Higher NH₃ absorption can be achieved by spraying diluted acidic solution as a scrubbing liquid [73]. In addition, use of acidic substances for collecting ammonia is highly preferred because of its great potential to get the NH₃ recycled into liquid fertilizer. Ohio state university has developed spray wet scrubber and three of those were installed on a commercial deep-pit swine building, a poultry manure composting facility, and a covered swine manure storage, respectively. The field tests of the wet scrubber showed an ammonia collection efficiency up to 98% and 80% for exhaust air with low (5 ppmv) and high ammonia concentrations (200 ppm), respectively. However, it is not tested for GHG mitigation.

4.3. Biofilters

Biofiltration is an air-cleaning process which absorbs pollutant gases and particulates into a biofilm on the filter media. Microorganisms in the filter media degrade and break the volatile organic compounds (VOC) and oxidizable inorganic gases. Selection of a proper biofilter media is a critical factor for developing an efficient biofilter. These factors such as optimum environment for microorganisms (moisture, temperature, porosity, etc.), large surface area to maximize attachment area and sorption capacity, stable compaction properties, high moisture holding capacity, high pore space to maximize empty bed
residence time (EBRT), and minimize pressure drop [72]. Recently, Chen et al. [72] evaluated a pilot-scale wood chip-based (e.g., western cedar and hardwood) biofilter to reduce odor, H\textsubscript{2}S, and NH\textsubscript{3} from swine barn ventilation air for 13 weeks. They found that hardwood and western cedar biofilters can remove odor by 70.1 and 82.3%, respectively, and H\textsubscript{2}S by 81.8 and 88.6%, respectively. Biofilters saturate easily [82] and large pressure drops across them making it difficult for them to be adopted by animal facilities. Difficulty also rests with the stringent moisture and temperature requirements for the process and more frequent media replacement [83]. It reported that biofilters may not be suitable to reduce high odor concentrations due to nitrogen accumulation in the biofilter material that causes the release of other pollutants including nitrous oxide (N\textsubscript{2}O), a highly potent greenhouse gas [74]. The biofiltration is a simple technology but requires careful monitoring of operating parameters for treating contaminated air effectively. However, microbial process taking place in the filter beds are very complicated, which depends on few environmental and physical factors as summarized below [82,84]:

High ammonia loads generally trigger excessive nitrite/nitrate concentrations that inhibit a proper functioning of micro-organisms in the filter bed and leads to acidification which forms nitrite/nitrate salts. This in turn, declines the removal efficiency of the filter. Thus, by replacing saturated biofilter packing at regular interval this issue can be addressed. Maintaining adequate moisture in the filter bed is the critical factor for proper functioning by the filter because of drying out inlet side of the filter bed when relatively dry air is coming out of the exhaust housings. Thus, the biofilter bed has to be kept moist to ensure proper microbial functioning.

Biofilters are inherently prone to dust loads, thus, clogging the packing bed and increasing the pressure drop. The total pressure drop over the filter bed can be very high and in practice it is >200-300 Pa. This clogging when coupled with inadequate moisture in the filter bed may lead to reduced air flow which will decrease overall scrubbing performance of the filter bed. These in turn need an increase in energy input unit of air volume handled. The functional lifespan of the biofilter can be enhanced by pre-treating incoming air, routing it first through an acid scrubber and mist eliminator before entering the biofilters.

Thus, design and operational parameters such selection of packing material, maintaining optimum moisture content, weed control and assessing pressure drop are very critical for efficient operation of the biofilters. The functional lifespan of the biofilter can be enhanced by pre-treating incoming air, routing it first through an acid scrubber and mist eliminator before entering the biofilters.

4.4. GHG scrubbing technology

Existing GHG mitigation options related to manure management are focused on feed manipulation, animal management, and processes to treat and manage animal manure. GHG emitted from animal buildings, especially swine buildings with deep pits and dairy buildings, accounted for a large portion of GHG emission. However, mitigation technologies
for GHG emissions from these animal buildings are lacking. Recent adaptation of mechanical ventilation systems for these buildings made it possible for better maneuvering the air stream in routing it through a suitable scrubbing system. Gaseous emissions including methane must be reduced before it escapes into the atmosphere. Housings or sources with mechanical exhaust ventilation systems are advantageous for capturing methane emission before it enters the atmosphere. However, this process inherently needs to handle large quantities of exhaust/ventilation air at low cost. Thus, higher energy requirement to process huge amount of exhaust may not be economically feasible. Therefore, it is urgently needed to develop effective and economically feasible GHG mitigation technologies for the reduction of these emissions from animal barns to ensure sustainable and viable swine and dairy industries.

Nitrous oxide (N_2O) and NH_3 are fairly soluble in a wide variety of solvents including water, alcohols, sulfuric acid, etc.). Unlike ammonia and N_2O, CH_4 is highly insoluble in water, and thus, scrubbers developed for NH_3 cannot be used for mitigating CH_4. Methane is a very stable molecule at ambient temperatures so it cannot be removed by many of the scrubbing techniques that are used for other gases. It has been previously shown in rendering facilities that oxidants like chlorine dioxide are effective for the removal of VOCs and other organic compounds using exhaust wet scrubbers. However, these may be expensive when applied as scrubbing liquid for swine facilities. In swine pits and dairy buildings, CH_4 concentrations are too lean to burn; oxidation by scrubbing can be an effective and safe alternative for reducing CH_4 emissions. This situation warrant researching alternative scrubbing liquids to make the CH_4 scrubbing process economically feasible. Other possible oxidants such as hydrogen peroxides (H_2O_2), Ozone (O_3), potassium permanganate (KMnO_4), and hypochlorus acid (HOCl), etc) can be tested to reduce CH_4 in a suitable scrubber. Ozone, a strong oxidant, has been used extensively to improve air and water quality. It has also been used by agricultural engineers to control air quality in animal buildings [85,86]. Ozonating water has been proven to effectively oxidize organic compounds dissolved in water. Transfer of organic compounds from air to water by spray scrubber absorption is very promising for the capture of GHG, and deodorizing and sanitizing air without affecting the CLOs ventilation system. It is well known that ozone reacts with methane either in presence of ultraviolet (UV) light or high temperature. Therefore, to explore the methane reduction in a cost effective manner, a spray type wet scrubber can be tested with ozonated water (or other effective oxidants) and UV light. The scrubbing system can be consisted of a spray-wet-scrubber column made from a material (either glass or acrylic) with high UV light transmittance, flow control meters, an ozone generator, and oxidant liquid supply and collection systems. A dust cleaning system needs to be incorporated to allow UV light to always pass through the scrubber.

5. Management practices to reduce GHG emissions

Methane from enteric fermentation, manure storage and spreading, and nitrous oxide mainly from application of manure on land are the major sources of agricultural GHG emission. Prior to being applied to crop or pasture field, manure from CLOs is generally
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stored in a liquid or solid form. For most cases, in order to produce CH₄ gas to use as bio-fuel, manure is anaerobically digested or composted before land application. By adapting manure management and treatment practices to enable methane collection, methane emissions from anaerobic digestion can be recovered and used as energy. Anaerobic digestion provides an appropriate environment for the complete degradation of organic matter to low-odor end products [[87]]. It also produces methane (biogas), which can be used for the production of electricity and heat [10]. However, due to high content of ammonia, the digestion of only swine manure was not favorable [85,88]. Generally, CH₄ mitigation approaches can be broadly divided into four categories as follows:

1. Preventative or feed management
2. Manure managements including treatment process and land application methods
3. Adaptation of housing system design including inside manure storage
4. End of pipe air treatment.

5.1. Preventive method (feed management)

Preventative measures are the reduction in carbon/nitrogen inputs into the system of animal husbandry through a dietary manipulation to achieve reduced CH₄ production. An effective tool to reduce nutrient/mineral pollution and GHG emissions is proven to be dietary manipulation. Numerous studies have revealed that reducing crude protein in the diet could substantially lessen nitrogen excretion and ammonia volatilization without compromising productivity [89-93]. Hao et al. [94] studied the effects of DDGS on feces and manure composition in feedlot cattle and revealed that as the ratios of wheat DDGS (e.g., 0, 20, 40, and 60%) in animal diet increased (40 and 60% wheat DDGS), the likelihood of volatile fatty acids (VFAs) also increased. This led to the growth of odors produced from the breakdown of fiber and protein [89]. They suggested that it might be a practicable option to obtain 20% or less DDGS in animal diet to limit VFAs produced from the breakdown of fiber and protein [89]. Enteric fermentation produced approximately 80% of the CH₄ produced from ruminants. The chemical composition of diet is a vital feature, which affects rumen fermentation and methane emission by the animals. Furthermore, dietary manipulation also impacts the amount of GHG emissions, particularly from enteric fermentation. For example, feeding cattle with a high starch and low fiber diet reduces creation of acetate in the rumen and leads to lower methane production [93]. As a proportion of energy intake, a higher proportion of concentrate in the diet leads to a reduction in CH₄ emissions. Stored manure and high fiber fed animals tend to have higher emissions. Diet affecting emissions from manure applied soil has significant evidence. Replacing fibrous diets by starchy feedstuff has been shown to reduce methane from enteric fermentation and manure storage [95-96]. As the level of production is increased to meet global demand for ruminant meat and milk products, dietary manipulation will be useful in addressing environmental concerns. Sejian and Naqvi [7] described enteric methane reduction strategies under four categories as shown in Figure 5. Likewise, a detailed evaluation of mitigation options of methane emissions from enteric fermentation is presented in Table 5 [97]. Abatement of GHG emissions from ruminant animals has been focused on diet, rumen and animal
manipulations, such as improving forage quality, adding dietary supplement, reducing unproductive animals, and supplementing probiotics to change microbial population in rumen [96]. Dietary fat seems a promising alimentary alternative to depress ruminal methanogenesis without lessening ruminal pH as opposed to concentrates [98]. Beauchemin et al. [99] recently reviewed the effect of level of dietary lipid on CH$_4$ emissions over 17 studies and reported that with beef cattle, dairy cows and lambs, for every 1% (DMI basis) increase in fat in the diet, CH$_4$ (g/kg DMI) was reduced by 5.6%.

**Figure 5.** Different enteric methane mitigation strategies [7]
| Strategy                                      | Potential CH4 reduction | Technology/feasibility | Cost/production benefits                                                                 |
|----------------------------------------------|-------------------------|------------------------|------------------------------------------------------------------------------------------|
| Improving animal productivity               | 20-30%                  | Feasible and practical | Increased feed cost increased milk production use of fewer animals less feed per kg of milk |
| Increasing concentrate levels at high levels of intake | 25% and more            | Feasible, for high producing cows, but may increase N₂O and CO₂ emissions | Increased feed cost increased milk production use of fewer animals less feed per kg of milk production |
| Processing of forages, grinding/pelleting    | 20-40%                  | Feasible               | Increased cost of processing improved feed efficiency increased milk production             |
| Forage species and maturity                  | 20-25%                  | Feasible               | Increased feed efficiency increased milk production                                         |
| Rotational grazing of animals/early grazing  | 9% or more              | Feasible               | Increased cost of fencing increased management of animals increased feed intake increased milk production |
| Managed intensive grazing vs. confined feeding|                         | Feasible need more investigation | Cheaper feed cost may need supplements reduced milk fat/protein content higher net return   |
| Use of high quality forages/pastures         | 25% or more             | Feasible               | Increased feed intake increased milk production                                           |
| Preservation of forage as silage vs. hay/additives | up to 33% (model prediction) | Feasible       | Limited studies                                                                            |
| Addition of fats                             | up to 33%               | Feasible and practical, but usage limited to 5-6% in diet | Increased cost of diet increased or no effect on milk production may or may not affect milk fat |
| Strategy | Potential CH4 reduction | Technology/feasibility | Cost/production benefits |
|----------|-------------------------|------------------------|-------------------------|
| Use of ionophores, e.g., monensin, lasolocid | 11-30% | Feasible but long lasting public concern | Increased feed efficiency decreased feed intake increased milk production |
| Use probiotics | 10-50% (*in vitro*) | Feasible, needs more investigation | May increase feed intake may increase milk production or no change |
| Use of essential oils | 8-14% | Feasible, needs more investigation | Not quantified |
| Use of bovine somatotropin (bST) | 9-16% | Not approved for use in Canada | Reduced feed cost |
| Protozoa inhibitors | 20-50% | Not available for practical use | Practically and cost to be assessed |
| Propionate enhancers (fumarate, malate) | 5-11% (*in vitro*) up to 23% (*in vivo*) | Possible microbial adaptation to fumaric acid | Economic feasibility ruminal adaptation and level of inclusion need to be evaluated |
| Use of acetogens | not quantified | Not available, needs more investigation | Needs further investigation |
| Use of bacteriocins, e.g., Nisin, bovicin HC5 | up to 50% (*in vitro*) | May provide alternatives to ionophores needs more investigation | Production effects are to be evaluated |
| Use of methane inhibitors, e.g., BES, 9,10-anthraquinone | up to 71% (*in vitro*) | No compounds registered for use No long lasting effects identified | Increased cost of chemicals production effects not established |
| Immunization | 11-23% | Not available, needs more investigation | May increase cost of production increased gain |
| Genetic selection (use of high Net Feed Efficiency animals) | 21% | Long term feasibility | May increase cost of production increased gain |

*Table 5.* Summary of methane mitigation strategies for dairy cows [97]

### 5.2. Manure management

It is well known that GHG emissions (mainly CH4 and N2O) from manure differ significantly depending on the management system employed to process them. Therefore,
strategies for mitigating net GHG emissions should be aimed to manipulate manure properties or the conditions under which CH\textsubscript{4} and N\textsubscript{2}O are produced and utilized during manure storage and treatment. However, GHG mitigation options are critical and depend on several factors. These factors are economic, technical and material resources, climatic conditions, existing manure management practices, bio-energy sources, and a source of high quality fertilizer and soil amendments. One such approach is to manipulate livestock diet composition and/or include feed additives to alter manure pH, concentration and solubility of carbon and nitrogen, and other properties that are pertinent to CH\textsubscript{4} and N\textsubscript{2}O emissions [7]. Nitrogen excreted in urine is predominant in the form of urea that can easily be converted into ammonia and carbon dioxide by the enzyme urease (which is present in feces), thus resulting in emission of ammonia. Nitrogen excreted in feces is mainly present as protein, which is less susceptible to decomposition into ammonia [64]. Therefore, feed management aims at either reducing the nitrogen excretion in feces and urine by matching the amount and composition of feed more closely to animal requirements at various production stages, or shifting nitrogen excretion from urine to feces by increasing fibrous feedstuffs in the diet [64]. The use of these strategies can reduce the ammonia emission both for pigs [100-101], poultry [102-104] and dairy cattle [105-106]. About 50\% of ammonia emissions to the environment were reduced through feed management for pigs and poultry when compared to standard feed composition. However, feed manipulation for ammonia abatement may negatively affect the emission of methane and nitrous oxide during storage and after land application of the manure [107].

Another manure management option is to change the material used for bedding the animals, which could also affect manure pH and soluble C and N levels and thus, the emissions during manure storage and treatment. Composting technology, control of aeration, use of amendments, or co-composting livestock manure with other organic waste could also potentially modify conditions for GHG production and emission. The use of covers may also help retain N nutrients during storage. Floating covers of natural and synthetic, origin or composites of both have shown substantial reduction in NH\textsubscript{3} and H\textsubscript{2}S emissions when compared with uncover liquid manure. However, little is known about the effect of covers on GHG emissions. In a two week study, covers generally increased CO\textsubscript{2} and CH\textsubscript{4} emissions [108].

5.2.1. Animal population and low N grass

A large part of N from animal waste and farm effluents is lost to the environment as excess NH\textsubscript{3} or N\textsubscript{2}O from urine spots and animal manures instead not being recovered in livestock-production systems. There are various options for reducing NH\textsubscript{3} and N\textsubscript{2}O emissions from livestock facilities, but the most significant option is to improve overall N efficiency. By reducing the livestock numbers, the amount of excreta would be reduced, hence the amount of emissions. Another mitigation option would be to manipulate the N economy of the animal to reduce N excretion. A lower N content of pasture or silage would reduce N
excretion by animals and NH₃ volatilization loss [43]. Excretal N could be reduced by using grass grown with moderate fertilizer application [93].

5.2.2. Anaerobic digestion and gas capture

With the use of liquid-based livestock facilities, the primary method for reducing emissions is to recover the methane before it is emitted into the air. Methane recovery involves capturing and collecting the methane produced in the manure management system. This recovered methane can be flared or used to produce heat or electricity. Because most of the manure facility methane emissions occur at large confined animal operations (primarily dairies and hog farms), the most promising options for reducing these emissions involve recovering the methane at these facilities and using it for energy. Additionally, in the effluent management systems, where the animal waste is gathered and/or stored in a covered digester lagoon and permitted to decay anaerobically, farmers are allowed to collect the generated CH₄ for heating and bio-energy use. Once CH₄ is removed from the waste, the remaining digestate can be used as a fertilizer and soil conditioner [43]. This option saves farmers money for energy costs and reduces CH₄ emissions to the atmosphere [109]. Additionally, during anaerobic digestion of the waste/manure, N₂O emission is negligible since N₂O is formed during aerobic nitrification and anaerobic denitrification [40]. This is an important N₂O mitigation option which reduce N₂O emission in the farming system as follows [43]: (1) reduce the total amount of excreta N returned to pasture; (2) increase the efficiency of excreta and/or fertilizer N; and (3) avoid soil conditions that favor N₂O emissions.

5.2.3. Land application

GHG emissions from animal manure and wastewater management systems are influenced by different physicochemical and biological factors. The key factors responsible for CH₄, CO₂, and N₂O emissions are soil moisture, temperature, manure loading rates by the animal, depth of manure in the pen, redox potential, available C, diets, and microbial process. Limited studies on the overall impact of effluent application on the whole suite of gaseous emissions have indicated different effects on the emissions of greenhouse gasses. For example, injecting slurry into soil may reduce NH₃ emissions. In contrast, such slurry incorporation into the soil may trigger N₂O emissions. Similarly, anaerobic digestion of effluents and its subsequent land application can reduce N₂O emission. However, under anaerobic condition with substrate pH may increase higher NH₃ emissions and also higher emissions of CH₄. On the other hand, the direct applications of animal waste either solids or liquid form to pasture and/or crop land can result in CH₄ emissions. Additionally, this method is prone to N losses, and up to 90% of manure N losses occur in various forms, including N₂O. Ammonia (NH₃) also acts as a precursor for N₂O and NO production when emitted from animal excreta [110] and thus, any approach mitigating NH₃ will also reduce N₂O emissions. Brink et al. [111] reported that NH₃ abatement may have a contrasting effect on N₂O emissions, while abatement of N₂O results in a net decrease in NH₃ volatilization.
Application of swine slurry to crop/pasture field resulted in the high emissions of gaseous N, which also led to constraints on the amount of slurry N that can be applied per hectare of land. One potential option could be decreasing N content of the slurry by feeding low N diets. However, little work has been done on dietary manipulation as a means of decreasing N losses without compromising the animal production. It was reported that a greater decrease in N excretion can be achieved by decreasing the crude protein (CP) content of the diet. The denitification rate was lower from slurry collected from pigs on low CP diet (140 g kg⁻¹ CP) than a standard diet (205 g kg⁻¹ CP), showed similar N₂O emissions from both treatments [112]. Addition of available C to soil was previously found to increase denitrification and also the ratio of N₂:N₂O produced [113]. Therefore, higher C in the low-CP diet would have favored the production of N₂ rather than N₂O as the product of denitrification [43]. Therefore, the abatement strategies to reduce gaseous emissions of NH₃, N₂O, and CH₄ from animal waste and farm effluents would therefore require some trade-offs among these three gases.

5.3. Housing system design and management

The structure of a housing system, for example the combination of the floor-system, manure collection, and the manure removal system, largely determines the level of the emission of gaseous compounds, especially the emission of ammonia. Housing systems that reduce gaseous emissions basically comprise of at least one or more of the subsequent abatement principles [64]:

1. Reduction of emitting manure surface.
2. Fast and complete removal of the liquid manure from the pit to external slurry storage.
3. Applying an additional treatment, such as aeration, to obtain flushing liquid.
4. Cooling the manure surface.
5. Changing the chemical/physical properties of the manure, such as decreasing the pH.

The housing systems that have been developed to include the above principles are able to reduce their gaseous (ammonia) emissions to the atmosphere from approximately 30% to 80%. Brink et al [111] in Europe, estimated that while it may increase nitrous oxide emissions significantly the emission of methane was hardly affected by animal housing adaptations for ammonia abatement. Usually limited with mixed results, the effect of animal housing adaptations on odor emission was demonstrated. Furthermore, control of the indoor climate in terms of reducing air velocity at the manure surface, which decreases mass transfer at the manure-air interface [114-115], and has relatively low indoor temperatures, and results in less fouling of floors especially for pigs [116], can reduce ammonia and odor emissions to the atmosphere even further if emitting surface is reduced. The slurry-based manure management system methane emissions increase with the temperature of the stored slurry. The reduction of slurry storage temperature from 20 to 10°C resulted in a reduction in CH₄ emissions of 30 -50% [117]. In animal houses the volatilization of NH₃⁺ is linked to
the ammonium (NH\(_4^+\)) concentration, the pH and surface area of the manure stored in the house, the area contaminated by the animals, and the temperature and ventilation of the housing system. Decreasing the surface area soiled by manure has the potential of reducing NH\(_3\) emissions. In addition, the cattle housing ammonia and CH\(_4\) emissions can be reduced through a more regular removal of manure to a closed storage system and through the systematic, everyday scraping of the floor.

5.4. End-of-pipe air treatment
This approach for reducing emission is basically a treatment of the exhaust air released from mechanically ventilated animal housings. The main advantages for this approach are that air can be treated without affecting the routine management operations and structural design inside the barn. End-of-pipe air treatment techniques are applied mainly for treating ammonia released from the exhaust air of livestock facilities and are commercially available off-the-shelf in the Netherlands, Germany, and Denmark [64]. The state of art of End-of-Pipe techniques and their scrubbing performance are briefly discussed in the section 4. However, existing scrubber used for ammonia scrubbing does not scrub methane. Thus, an appropriate scrubber needs to be designed to scrub methane released from the exhausts of the animal housing or covered manure storage.

6. Conclusions
The livestock industry is a significant contributor to the economy of any country. More than one billion tons of manure is produced annually by livestock and poultry reared in the United States. Animal manure is a valuable source of nutrients and renewable energy in the country. On the other hand, livestock manure management is extremely challenging and resultant gaseous emissions may contribute to global warming. Livestock manure produces odor and emits GHGs such as carbon dioxide, methane and nitrous oxide that have prompted significant environmental quality degradation concerns. Major sources of agricultural GHG emissions include methane from enteric fermentation, manure storage and spreading, and nitrous oxide mainly from application of manure on land. GHG emissions from animal manure and wastewater management systems are influenced by soil/manure moisture, temperature, manure loading rate by the animal, depth of manure in the pen, redox potential, available carbon, diets, and microbial process. Mitigation options for GHG emissions are source and characteristics dependent. Mitigation of GHG emissions from animal waste must be addressed in the context of integrated waste management. Manure as a biomass goes through different chemical and biological processes for bio-energy recovery and thus, reduced methane emission. Anaerobic bio-digesters, covered lagoons or manure storages with methane flaring systems or small electricity generators are gaining popularity as viable technologies to abate GHG emissions from manure storages. In addition, since methane is generated under anaerobic conditions, switching manure management from liquid to dry manure, such as pack-bedded dairy option and hoop
structure swine buildings with bedding, are other possibly effective management strategies to reduce methane emission. Mitigation technologies for GHG emissions released from the animal housings are lacking. Therefore, it is urgently needed that other effective and economically feasible GHG mitigation technologies be developed for the reduction of GHG emissions from CLOs to ensure improved environmental quality and sustainable and livestock agriculture in order to meet the milk and protein demand of an ever increasing world population.

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