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

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Additional information is available at the end of the chapter

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 gasses 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 (CO_2), methane (CH_4) and nitrous oxide (N_2O). Generally, swine and ruminant livestock operations, especially dairy cows and beef cattle, contribute to the production of GHGs mainly CH_4, N_2O, and CO_2 in the environment. The CH_4, CO_2, and N_2O are considered as direct greenhouse gases. The indirect GHGs include carbon monoxide (CO), oxides of nitrogen (NO_x), and non-methane volatile organic compound (NMVOCs). Characterization and quantification of N_2O and CH_4 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 CO_2, CH_4, and N_2O 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 (10^{12}g) of CO_2 equivalents in the U.S. during 2006 [2]. The CH_4 emissions from enteric fermentation and manure management represent about 25 and 8% of the total CH_4 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 CH_4 and N_2O 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 CH_4 and N_2O emissions [4]. Globally, CH_4 is contributing 22%, and N_2O is contributing 6% of the total GHG. Enteric CH_4 is produced as a waste product of this fermentation process in the rumen. Figure 1 describes the number of factors affecting CH_4 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 CO_2 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, CO_2 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 CO_2. Nitrous oxide emissions are associated with manure management and the application and deposition of manure in crop/ pasture land. Indirect N_2O 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.
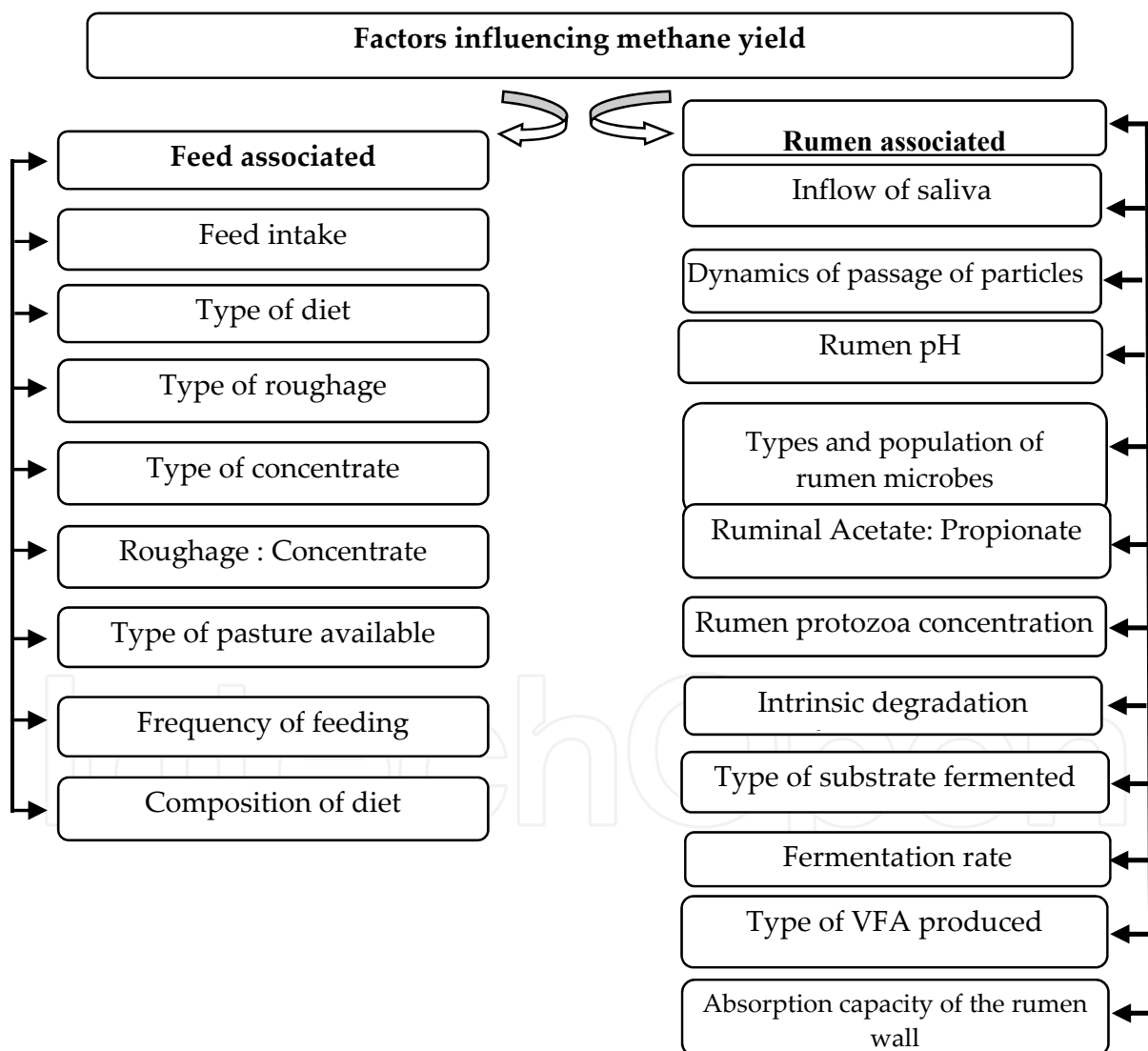
Figure 1. Factors affecting methane production [7].

Animal Type	Average weight (pound)	Days on feed	Total solids (TS)	Volatile solids (VS)	N	P	P₂O₅	K	K₂O	Manure	Moisture
				Pounds per day per animal on an "as excreted" basis							%, wb[1]
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 cows[1]	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
Turkeys[2]	25	140	0.12	0.1	0.0072	0.00212	0.005	0.0033	0.004	0.47	74
Litter[3]			2	1.6	0.089	0.038	0.086	0.049	0.059	2.5	21
Horse[4]	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.

[n]Milk vow 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.

[1]%, wb = percent wt basis.

[2]Days 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.

[3]Poultry 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.

[4]Average 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)	Total manure	TS	VS	N	P	P$_2$O$_5$	K	K$_2$O	Total energy***
			Thousands of tons per year on an as excreted" basis								BTU x 10^{12} Tera BTUs
Cattle											
Cows/Heifers		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		2430	6634	868	740	36	11	26	23	28	12.6
Milk Cows		334	8581	1096	936	56	9.7	22.3	32	32	15.8
Swine											
Nursery		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
Other[1]		95	312	31	29	2.26	0.65	1.49	1.46	1.75	0.48
Sleep & Goats[2]	**2006**	**2140**	**1759**	**484**	**400**	**18**	**4**	**9**	**13**	**16**	**6.8**
Poultry											
Layers[3]	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
Turkeys[4]	2004	14100	468	120	96	7	2	5	3	4	1.63
Litter			785	622	494	28	12	27	15.4	18	8.4
Horses[5]	**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

[1]Includes all hogs other than nursery and grow-finish. Estimates based on average nutrient data from gestating lactating sows in Table I.

[2]Includes sheep and goats. Manure and nutrient totals calculated using sheep data only.

[3]Include hens and pullets of egg-laying age.

[4]animal 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).

[5] Animal numbers for horses adopted form 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 Worming Potential
CO$_2$	44.01	380	Up to 100	1.4×10^{-5}	1
CH$_4$	16.04	1.7	12	3.7×10^{-4}	21
N$_2$O	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 CO_2 from the dairy slurry manure storage facilities averaged 72 kg CO_2 m^{-3} yr^{-1} (data ranged from 8.6 to 117 kg CO_2 m^{-3} yr^{-1}) [9-11]. Emissions of CO_2 from dairy housing averaged 1,989 kg CO_2 hd^{-1} yr^{-1} (data ranged from 1,697 to 2,281 kg CO_2 hd^{-1} yr^{-1}, where hd^{-1} is per head) [11]. Kinsman et al. [12] reported that the mean daily CH_4 emission per dairy cow (602 kg mean bodyweight) in a tie-stall barn ranged from 373 to 617 g CH_4 $AU^{-1}d^{-1}$(436 to 721L) ,while the mean daily CO_2 emission per cow ranged from 11,900 to 17,500 g CO_2 $AU^{-1}d^{-1}$(5,032 to 7,427 L). In a study by Amon et al. [13], CH_4 and N_2O 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 N_2O $LU^{-1}d^{-1}$ (516 mg N_2O $AU^{-1}d^{-1}$), and 194.4 g CH_4 $LU^{-1}d^{-1}$(162 g CH_4 $AU^{-1}d^{-1}$). Emissions of CH_4 and N_2O from animal housing averaged 54 (1.0-100) kg CH_4 hd^{-1} yr^{-1} and 0.3 (0.0-0.6) kg N_2O hd^{-1} yr^{-1}, respectively [11,13]. Ngwabie et al. [14] reported CH_4 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 $cow^{-1}d^{-1}$ [16]. Fiedler and Muller [17] show that CH4 emissions from naturally ventilated dairy barns ranged from 672 to 528 g $cow^{-1}d^{-1}$. A study conducted in California, USA, indicated that CH_4 emissions of 296 and 438 g $cow^{-1}d^{-1}$ for dry and lactating cows, respectively, were mainly due to enteric fermentation and fresh manure produced negligible amount of CH_4 [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, CH_4 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 CH_4 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 CH_4 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 CH_4 emissions were 166 and 214 g CH_4 hd^{-1} d^{-1} for feedlots in Queensland and Alberta, respectively [20]. Average daily CH_4 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 N_2O and CO_2 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 CH_4 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 CO_2, CH_4 and N_2O from different pig housing systems are presented in Table 4c. CH_4 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, CO_2, CH_4, and N_2O 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, CO_2 ERs for different growing stages from swine operation ranged from 5,920 to 30,000 g pig^{-1} d^{-1}. The highest N_2O ER was estimated from swine nursery in China [29]. Animal feces temporarily stored indoors deep pits are the principal source of CH_4 emissions in swine housings. The quantity of CH_4 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 CH_4 generation might be influenced by the availability of oxygen over the emitting surfaces [30]. Significant amount N_2O emits from pig manure handling system is exclusively originated from deep litter and compost systems. The variation in the N_2O emissions mainly depends on the kind of housing system. Fattening pigs raised on partly or fully slatted floor emit very little N_2O while higher emissions reported for fatteners in deep litter and compost systems [30].

Facility type and ground sources	Animal Number	CH_4 g/LU/d	CO_2 kg/LU/d	N_2O 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]

		0.55			(MDA Atlanta) /BLS (Wind-Trax 2.0)	in January, March and June, and September.	
storage lagoon, and composting areas.							
Tied-stall dairy: Housing, Storage, and spreading	12 cows	194.4		0.619	Mobile dynamic chamber coupled with FTIR and GC. Sampled alternately incoming- outgoing air of the chamber.	Several times in course of a year, and 24 hours a day during each experiment. LU =600 kg and AU=500kg	[13]
Tied-stall dairy: Housing (enteric), Manure storage,	118	373 24	14 – 20		DTs: 0-5000 ppm for CO_2 and 0-1000 ppm for CH_4.	112 days BW 602±62	[12]
Naturally ventilated dairy building with daily manure collection	720	305	8.9	NR	Ventilation by CO_2 balance and concentrations measured by INNOVA 1312	5 consecutive days in spring, summer, autumn, and winter	[34]
Free-stall dairy: Dry cows (Holstein) Lactating cows	9 9	296 438	NR NR	ND ND	PAS Multi-gas Monitor (INNOVA 1412)	Manure and Enteric CH_4 in environmental chambers. BWs 770 and 565 kg	[18]
Open-lot Dairy: Total emissions rates from open-lot, runoff pond, and compost area	10,000	1390	NR	20	INNOVA 1412 OP-FTIR coupled with BLS Wind-Trax2.0	Four months (January, March, June, September) and 2-3 days in each month	[35]
Free-stall dairy: Open lot Run-off pond Compost area	10,800	490 952 125	28 6 12	10 4.5 8.3	PAS Multi-gas Monitor (INNOVA 1412) coupled with BLS (Wind-trax 2.0)	2 or 3 days in each month over a year; BW 635 kg	[36]
Dairy with Conventional bedding Straw bedding		700 ± 400 1400 ± 200			TDLAS (DT: 100 ppb) coupled with Gaussian plume model	7 conventional farms 3 straw farms	[37]
ND = not detected by sensor; NR= not reported							
				a.			
Facility type and ground sources	Animal Number	CH_4	CO_2	N_2O	Technology used	Remarks	References
Feedyard: Feedlot	16,995	323			OP-TDLAS/ BLS (Wind-Trax 2.0)	12 days, High grain diets and	[21]

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]
Feedyard: Pen surface Run-off pond Composting areas	42,000	1.71 2.05 0.04	1.31 0.035 0.055	0.57 0.01 0.10	Dynamic flux chamber coupled with gas chromatograph and chromatograms acquired directly at the field	Five consecutive days in summer	[32]
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]
Feedlot 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]

b.

Country	Animal Stage	Housing and or manure Handing type	GHG emission factor				Reference
			Unit	CO_2	CH_4	N_2O	
Germany	Fattening	Fully slatted floor Kennel housing	g d^{-1} AU^{-1} g d^{-1} AU^{-1}	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} mg h^{-1} pig^{-1}	N/A N/A N/A	2406 445 1269	N/A N/A N/A	[24]
Italy	Fattening	Fully slatted floor Vacuum system	g d^{-1} AU^{-1} g d^{-1} AU^{-1}	N/A 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]

USA	Farrow-to-finish Farrow-to-weaner	N/A N/A	g d^{-1} pig^{-1} g d^{-1} pig^{-1}	N/A N/A	6.9-29.2 46.2	N/A N/A	[27]
USA	Gestation	Deep pit (3m) manure storage emptied twice a year	g d^{-1} AU^{-1}	5350-15830	116-572	0.06-7.3	[28]
United Kingdom	Fattening	Slurry	g d^{-1} AU^{-1}	N/A	85	0.4	[41]
China	Fattening Nursery Gestation Farrowing	Manure scrapped twice a day Flushed twice a day Manure removed twice a day Removed as produced	g d^{-1} AU^{-1} g d^{-1} AU^{-1} g d^{-1} AU^{-1} g d^{-1} AU^{-1}	16730 ±1060 29670±1090 5920±440 7490±110	32.1± 11.7 58.4±21.8 9.6±1.9 9.6±3.6	0.86±0.75 1.29±0.37 0.75±0.56 0.54±0.15	[29]
Taiwan	NA	3 stages (solid separation, anaerobic and aerobic treatment) plug-flow type wastewater treatment plant.	g hd-1 d^{-1}	11.5	13.3	0.03	[42]
c.							

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 CH_4, CO_2, N_2, and H_2S. 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 CO_2 and CH_4 according to the following equation:

$$\text{C}_6\text{H}_{12}\text{O}_6 \xrightarrow{\text{Microbial Action}} 3\text{CO}_2 + 3\text{CH}_4 \tag{1}$$

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 $H_2 + CO_2$ and acetate. Products developed in stages 1-3 and convert ($H_2 + CO_2$ 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 CH_4 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 CH_4 is produced. The potential amount of CH_4 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 CH_4 emissions after field application, because the easily degradable organic compounds were already converted to CO_2 and CH_4 during digestion in the biogas plant [45]. The factors affecting CH_4 emission by soils are summarized as follows [43, 44]:

a. Gas diffusion in relation to oxydo-reduction level and CH_4 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.
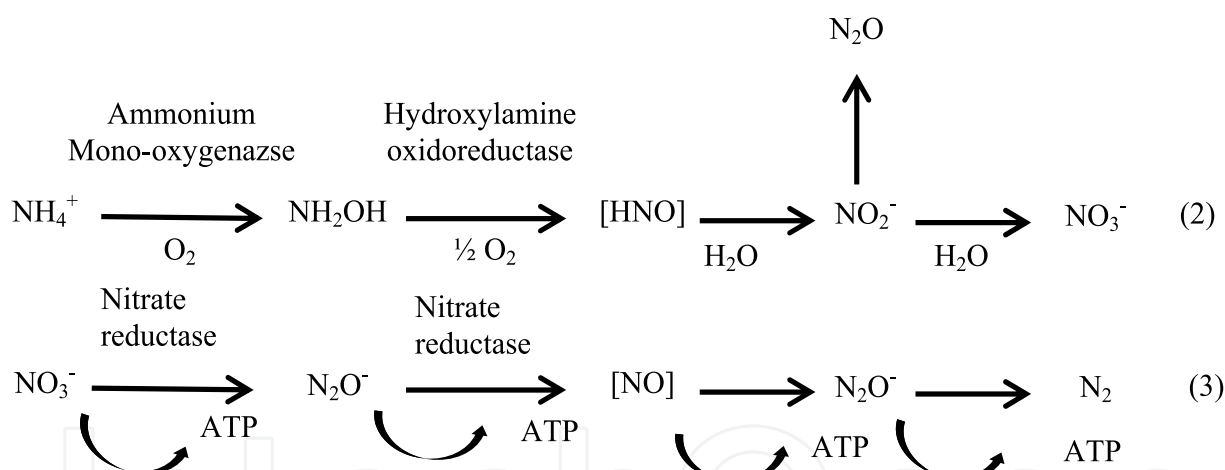d. Methane-mono-oxygenazse activity—concentrations of H_2, CH_4, $NH_4 +$, $NO_3 -$, Cu.
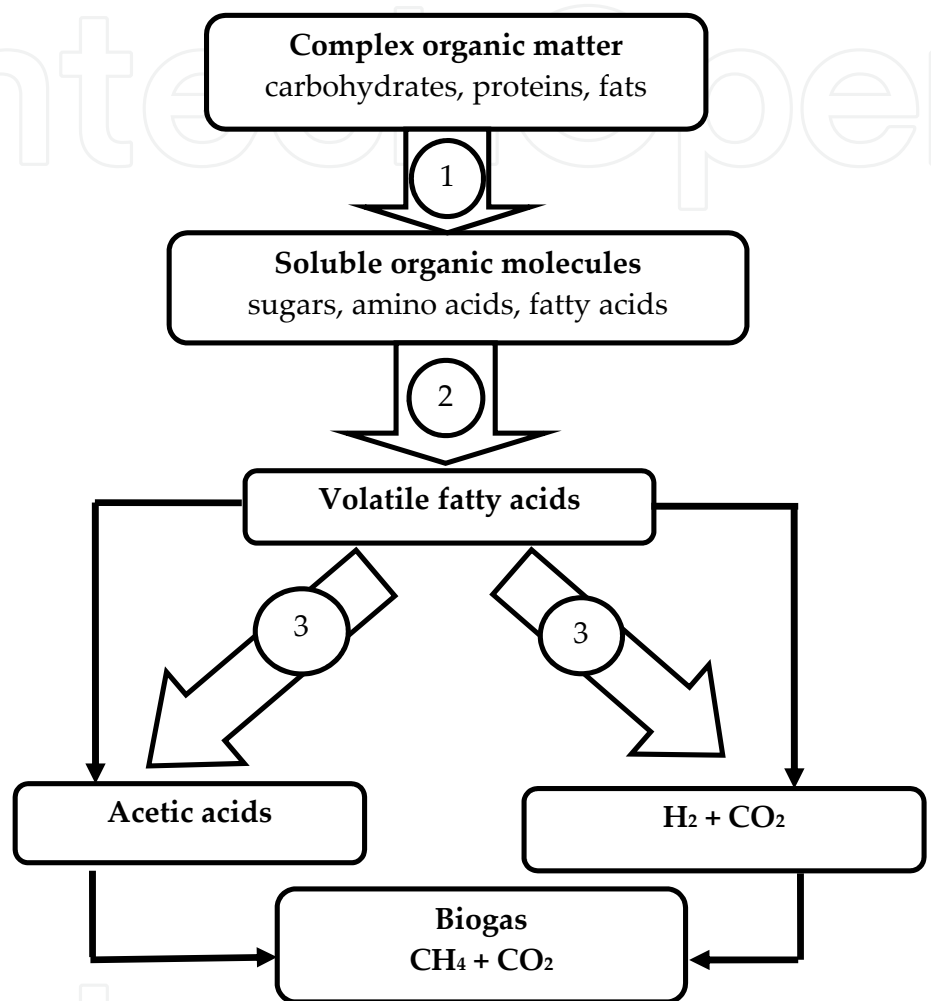
Figure 2. Schematic diagram showing anaerobic fermentation process.

2.2. Nitrous oxide emission process

Nitrous oxide (N_2O), nitrogen monoxide (NO), and nitrogen dioxide (NO_2) 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 (O_3) depleting substance which reacts with O_3 in both the troposphere and in the stratosphere and has a long half-life (100 - 150 years). In livestock agriculture, N_2O emissions are associated with manure management and the application and deposition of manure in crop/ pasture land. Indirect N_2O 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 N_2O. Soils contribute about 65% of the total

N_2O produced by terrestrial ecosystems [46]. Nitrous oxide gas 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_4^+ oxidation to NO_2^- or during dissimilatory NO_2^- reduction when O_2 supply is limited. During denitrification, N_2O is an intermediate product in the dissimilatory reduction of NO_3^- and NO_2^- to N_2 under anaerobic conditions and may, therefore, be produced and consumed by denitrifying bacteria in soil [47].

Fertilizer and manure type may affect N_2O emission in several ways [48] such as: (1) the type of N (NO_3^-, NH_4^+, and organic N) which affects N_2O production during nitrification and denitrification, (2) the presence of freely available C, which stimulates denitrification activity and O_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 ($NH4^+$ and NO_3^-) [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_2O 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_2O release by influencing the physical condition of the soils such as aeration and soil water content.

$$NH_4^+ \xrightarrow[O_2]{\text{Ammonium Mono-oxygenazse}} NH_2OH \xrightarrow[½ O_2]{\text{Hydroxylamine oxidoreductase}} [HNO] \xrightarrow[H_2O]{} NO_2^- \xrightarrow[H_2O]{} NO_3^- \quad (2)$$

$$NO_3^- \xrightarrow[ATP]{\text{Nitrate reductase}} N_2O^- \xrightarrow[ATP]{\text{Nitrate reductase}} [NO] \xrightarrow[ATP]{} N_2O^- \xrightarrow[ATP]{} N_2 \quad (3)$$

With N_2O released from NO_2^- in equation (2).

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 emission across the plume of 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 (H_2O_2, H_2SO_4, O_3, $kMnO_4$, $HOCl$, 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 (H_2SO_4) 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.
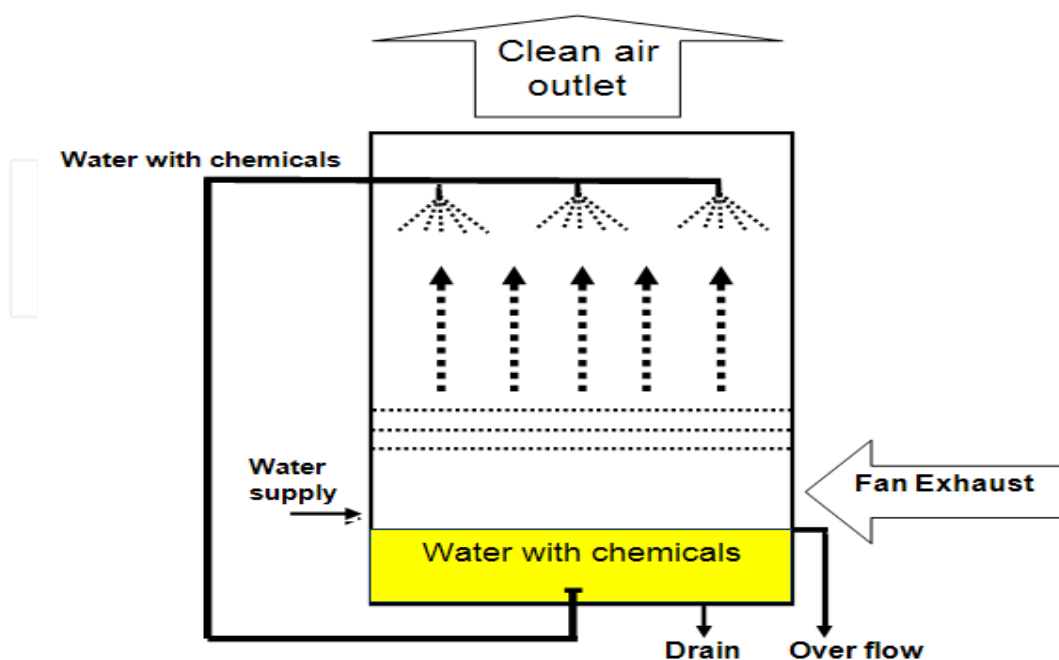
Figure 3. Schematic diagram spray type wet-scrubber system

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 a 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.
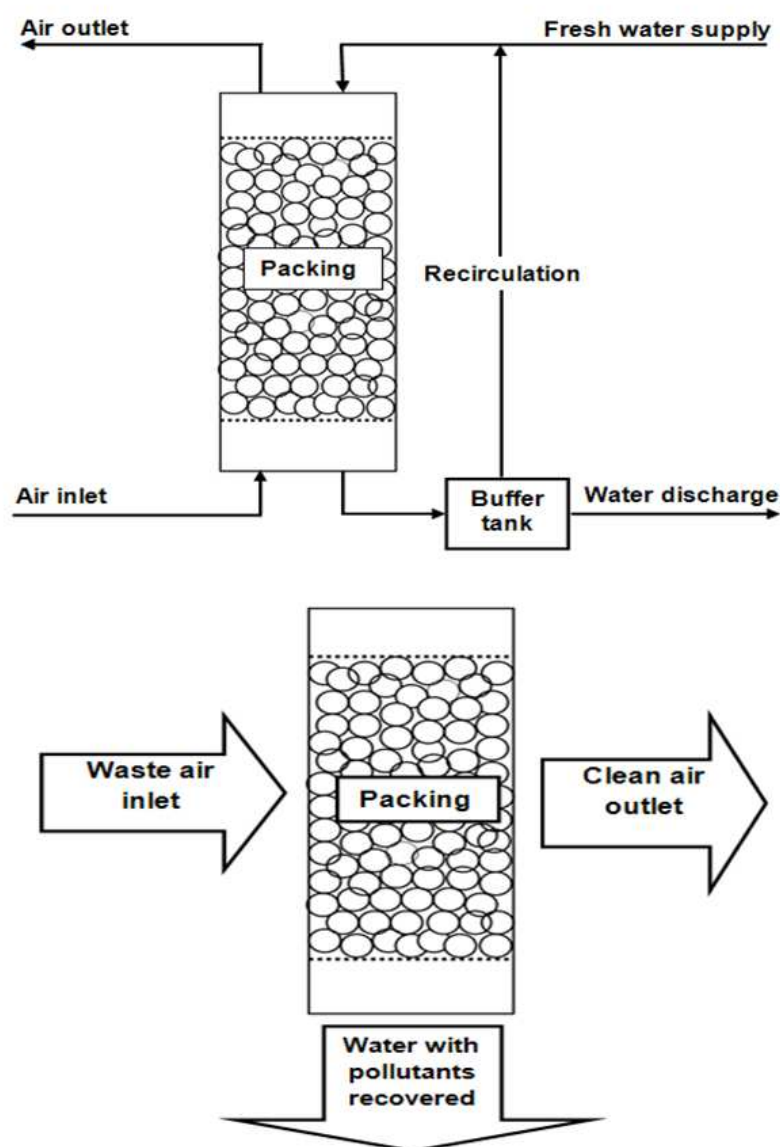
Figure 4. Schematic diagrams of packed bed trickling filters: top) counter current packed wet-scrubber, and bottom) cross current (adopted from [64-65]).

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]:

$$\gamma_{total} = \frac{C_{in} - C_{out}}{C_{in}} \times 100 \qquad (4)$$

Where:

γ_{total} = Pollutant collection efficiency (%)
C_{in} = Airborne Pollutant concentration before the scrubber (ppm)
C_{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:

$$EBRT = \frac{Packing\ volume\ (m^3)}{Air\ flow\ rate\ (m^3 s^{-1})} \times 100 \qquad (5)$$

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_2S [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 NH3, 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 are for application at CLOs [76] and have been proven to effectively remove NH_3 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 NH_3 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 CO_2 emissions from swine houses in Netherlands. Those scrubbers reduced PM_{10}, $PM_{2.5}$, total airborne bacteria, and ammonia emissions from 61 to 93%, 47 to 90%, 46 to 85%, and 70 to 100%, respectively [79]. Concentrations of CO_2 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 NH_3 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 NH_3 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 NH_3 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_2S, and NH_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_2S 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_2O), 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

stored in a liquid or solid form. For most cases, in order to produce CH_4 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, CH_4 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 CH_4 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 CH_4 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 CH_4 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 %.
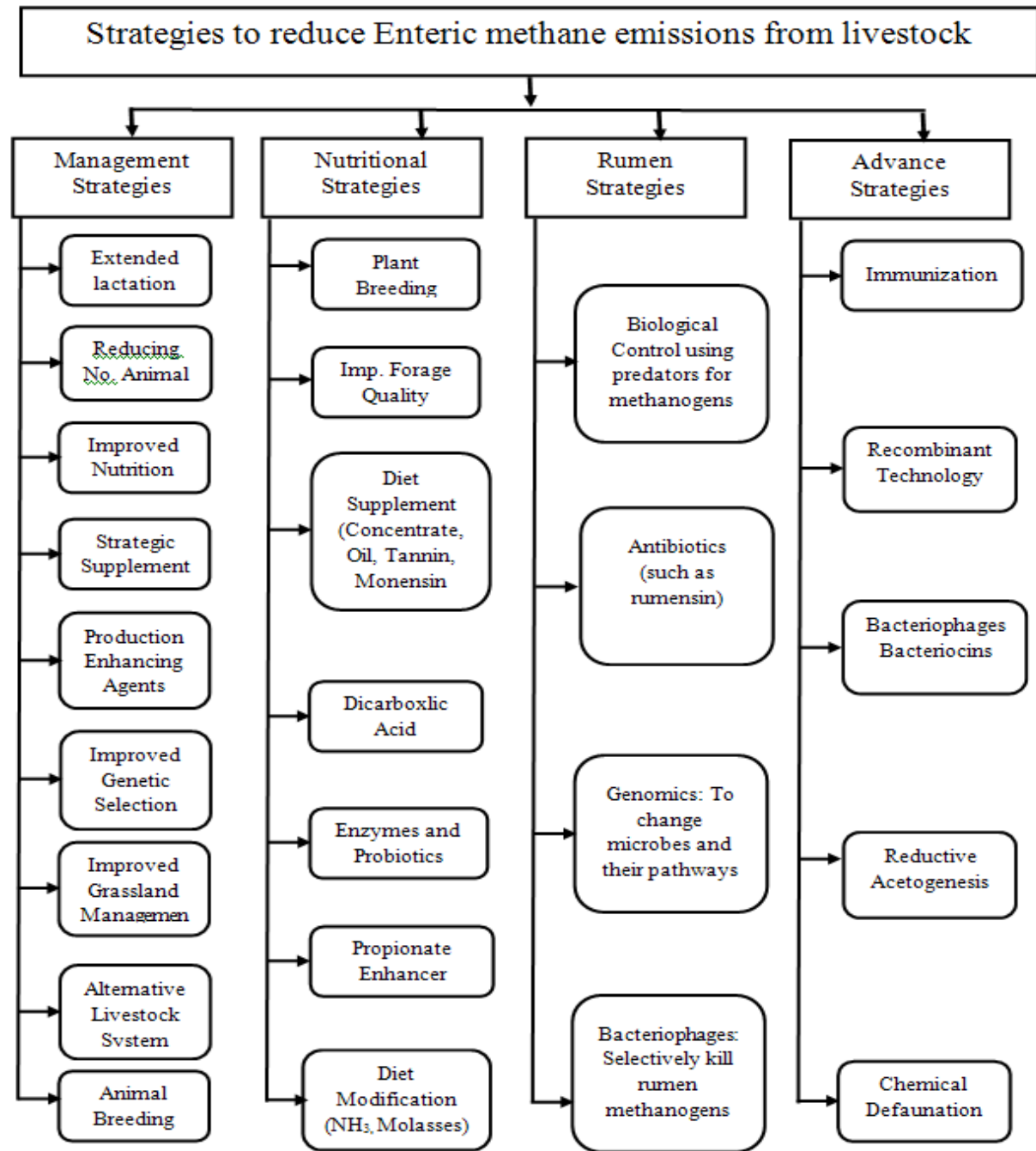
Figure 5. Different enteric methane mitigation strategies [7]

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 N_2O and CO_2 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 assed
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 CH_4 and N_2O) 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_4 and N_2O 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_4 and N_2O 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_3 and H_2S 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_2 and CH_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_3 or N_2O from urine spots and animal manures instead not being recovered in livestock-production systems. There are various options for reducing NH_3 and N_2O 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 NH_3 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 CH_4 for heating and bio-energy use. Once CH_4 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 CH_4 emissions to the atmosphere [109]. Additionally, during anaerobic digestion of the waste/manure, N_2O emission is negligible since N_2O is formed during aerobic nitrification and anaerobic denitrification [40]. This is an important N_2O mitigation option which reduce N_2O 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 N_2O 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 CH_4, CO_2, and N_2O 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 NH_3 emissions. In contrast, such slurry incorporation into the soil may trigger N_2O emissions. Similarly, anaerobic digestion of effluents and its subsequent land application can reduce N_2O emission. However, under anaerobic condition with substrate pH may increase higher NH_3 emissions and also higher emissions of CH_4. On the other hand, the direct applications of animal waste either solids or liquid form to pasture and/or crop land can result in CH_4 emissions. Additionally, this method is prone to N losses, and up to 90% of manure N losses occur in various forms, including N_2O. Ammonia (NH_3) also acts as a precursor for N_2O and NO production when emitted from animal excreta [110] and thus, any approach mitigating NH_3 will also reduce N_2O emissions. Brink et al. [111] reported that NH_3 abatement may have a contrasting effect on N_2O emissions, while abatement of N_2O results in a net decrease in NH_3 volatilization.

Application of swine slurry to crop/pasture field resulted 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 the 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 denitrification 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 N_2O emissions from both treatments [112]. Addition of available C to soil was previously found to increase denitrification and also the ratio of N_2:N_2O produced [113]. Therefore, higher C in the low-CP diet would have favored the production of N_2 rather than N_2O as the product of denitrification [43]. Therefore, the abatement strategies to reduce gaseous emissions of NH_3, N_2O, and CH_4 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 CH_4 emissions of 30 -50% [117]. In animal houses the volatilization of NH_4^+ 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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